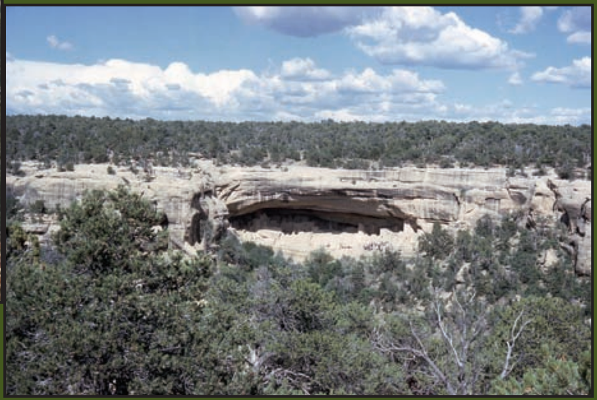



Ecology, Management, and Restoration of Piñon-Juniper and Ponderosa Pine Ecosystems:

Combined Proceedings of the 2005 St. George, Utah and 2006 Albuquerque, New Mexico Workshops

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Abstract

Southwestern piñon-juniper and juniper woodlands cover large areas of Arizona, New Mexico, Utah, and adjacent Colorado. Ponderosa pine forests are the most common timberland in the Southwest. All three ecosystems provide a variety of natural resources and economic benefits to the region. There are different perceptions of desired conditions. Public and private land managers have adapted research results and their observations and experiences to manage these ecosystems for multi-resource benefits. Ways to mitigate the threat of wildfires is a major management issue for these ecosystems, and the wide-spread piñon mortality related to drought and the bark beetle infestation has heightened concerns among managers and the general public. In addition, the impacts of climate change on these ecosystems are a growing concern. As a step in bringing research and management together to answer some of these questions, workshops concerned with the ecology, management, and restoration of piñon-juniper and ponderosa pine ecosystems were held in St. George, Utah in 2005 and in Albuquerque, New Mexico in 2006. The combined proceedings from these two workshops contain papers, extended abstracts, and abstracts based on oral and poster presentations. Some topics included forest and woodland restoration treatments and their impacts on fuels, wildlife, and other ecosystem components, watershed management, insect infestations and drought, wood utilization, landscape changes, basic ecology, and more.

Keywords: Piñon-juniper and juniper woodlands, ponderosa pine forests, ecology, management, restoration, southwestern United States

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Ecology, Management, and Restoration of Piñon-Juniper and Ponderosa Pine Ecosystems:

Combined Proceedings of the 2005 St. George, Utah and 2006 Albuquerque, New Mexico Workshops

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Preface

Southwestern pinyon-juniper and juniper woodlands cover large areas of Arizona, New Mexico, Utah, and adjacent Colorado. Management of these lands has varied from efforts to eradicate the woodlands to favor herbaceous species to efforts at management for multiple resource benefits. Ponderosa pine forests are the most common timberland in the Southwest and provide a variety of natural resources and economic benefits to the region. The Rocky Mountain Research Station, its predecessors and numerous university cooperators have conducted research in Arizona's ponderosa pine forests since 1908 and in woodlands since the 1930s. Public and private land managers have adapted this information and their observations and experiences to manage these important ecosystems. Land managers still have numerous questions concerning the southwestern woodlands and forests and their wise management. There are different perceptions of desired conditions. Ways to mitigate the threat of wildfires is a major management issue for both ecosystems, and the wide-spread pinyon mortality related to drought and the bark beetle infestation has heightened concerns among managers and the general public.

As a step in bringing research and management together to answer some of these questions, workshops concerned with the ecology, management, and restoration of pinyon-juniper and ponderosa pine ecosystems were held in St. George, Utah in 2005 and in Albuquerque, New Mexico in 2006. The first workshop, titled "Ecological Restoration of Southwest Ponderosa Pine and Pinyon-Juniper Ecosystems," was held on May 11 to 13, 2005. The meeting included oral and poster presentations. Some topics included forest and woodland restoration treatments and their impacts on fuels, wildlife, and other resources, insect infestations and drought, wood utilization, landscape changes, basic ecology, and more. The workshop was attended by 110 managers and scientists from the Society of American Foresters, the Arizona Game and Fish Department, Bureau of Land Management, Forest Service, and Utah Department of Wildlife and faculty and students from Northern Arizona University, New Mexico State University, and other institutions. John Helms, the Society of American Forester's President, was the luncheon speaker. The workshop was sponsored by the Southwest and Intermountain Societies of American Foresters, Arizona Game and Fish Department, Bureau of Land Management, Ecological Restoration Institute of Northern Arizona University, and the U.S. Forest Service, Rocky Mountain Research Station. The workshop included a field trip to Mt. Trumbull where pinyon-juniper and ponderosa pine restoration and fire issues were discussed. The organizing committee included David Borland, Doug Page, Aaron Wilkerson, and Ken Moore (all from the Bureau of Land Management), and Pete Fulé (Northern Arizona University). Borland also served as the liaison for the Southwest Section to the Intermountain Section. Personnel from the Ecological Restoration Institute assisted with the field trip.

A second workshop "Ecology and Management of Piñon-Juniper Ecosystems" was held in Albuquerque on September 13 to 15, 2006. It brought together 147 foresters, range managers, and other resource managers from federal, state, and tribal agencies and college and university students and faculty. The Southwestern Society of American Foresters was the lead sponsor. Other sponsors of the workshop included the Bureau of Indian Affairs, Bureau of Land Management, Society for Range Management, U.S. Forest Service, Rocky Mountain Research Station and Southwestern Region and the Water Resources Research Institute of New Mexico State University. Marlin A. Johnson (Forest Service, Southwestern Region) and David Borland (Bureau of Land Management) were the chairpersons. The workshop committee included Shannon Atencio, George Duda, and Kim Paul (all from the New Mexico State Forestry Department), Dennis Dwyer and Jim Youtz (from the Forest Service, Southwestern Region), Gerald Gottfried (Forest Service, Rocky Mountain Research Station), and Robert Partido (Society of American Foresters). The efforts of session moderators and student assistants are greatly appreciated.

The Albuquerque meeting was opened with a presentation, "Mimicking Nature's Fire" co-presented by Carl Fiedler (University of Montana) and Steve Arno (Rocky Mountain Research Station, Missoula, retired), which was based on their popular book. Other oral and poster presentations covered pinyon-juniper ecology, insects and diseases, hydrology, silviculture, growth and yield, impacts of invasive species, restoration techniques, grazing and wildlife management, products and markets, and more. E. Hollis Fuchs (NRCS) gave the banquet talk about 150 years of vegetative change in Lincoln County, NM. The workshop included a field trip to the East Mountains outside of Albuquerque led by Todd Haines (New Mexico State Forestry Department). It included two stops, one at the Starfire Day Camp to see the New Mexico Tree Farm Demonstration Forest where various pinyon-juniper prescriptions and actual treatment demonstrations were observed. The second stop was at Carlito Springs, which is owned by Bernalillo County, where erosion control, mulch utilization, and defensible space fuels projects were presented.

The papers and extended abstracts have received technical reviews; however, the views expressed in each presentation are those of the author(s) and not necessarily those of the sponsoring organizations or the U.S. Forest Service. The use of trade and company names is for the benefit of the reader; such use does not constitute an official endorsement or approval of any service or product by the U.S. Department of Agriculture to the exclusion of others that may be suitable.

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Pinyon-Juniper Woodlands

Pinyon-Juniper Woodland Ecology

Photo by Gerald Gottfried

Ecology of Piñon-Juniper Vegetation in the Southwest and Great Basin

Rex D. Pieper[1]

Abstract—*Piñon-juniper vegetation is conspicuous in foothills surrounding most mountain ranges in the Great Basin and the Southwest. Utah has the largest percentage of piñon-juniper vegetation, followed by New Mexico, Nevada, Arizona, and Colorado. Although piñon-juniper stands may appear to be similar, the vegetation component varies. The most abundant junipers are* Juniperus deppeana, J. monosperma, J. osteosperma, *and* J. scopulorum. *The piñons are* Pinus edulis *in the Southwest and* P. monophylla *in the Great Basin. At most locations the tree layer has 1 to 3 species while the understory is also composed of only a few species. Heavy livestock grazing, tree cutting, reduction of fire frequency and intensity, large-scale control programs, and periodic drought have influenced these woodlands over the past 150 years. Generally woodlands have increased at the expense of grasslands, but there is some debate about the nature of the increase—whether it represents encroachment into grasslands or reoccupation of former woodland sites. Several successional models may be applied to the piñon-juniper woodlands, including Clementsian linear succession, state and transition approaches, and cusp models.*

Introduction

Piñon-juniper vegetation is widely distributed in the West and easily recognized by the size of the tree layer. Utah has the highest percentage of woodlands followed by New Mexico, Nevada, Arizona, and Colorado (figure 1). Woodlands in the northwestern U.S. are represented by juniper woodlands and lack pine representatives.

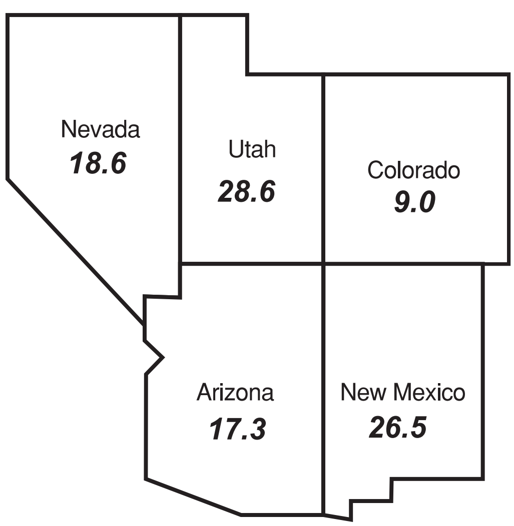

Figure 1—Percent of state occupied by piñon–juniper vegetation (from West and others 1975).

In: Gottfried, Gerald J.; Shaw, John D.; Ford, Paulette L., compilers. 2008. Ecology, management, and restoration of piñon-juniper and ponderosa pine ecosystems: combined proceedings of the 2005 St. George, Utah and 2006 Albuquerque, New Mexico workshops. Proceedings RMRS-P-51. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

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Several classifications of these pigmy forests have been published. Donart and others (1978) recognized a piñon-juniper series as well as a juniper-piñon series; Dick-Peddie (1993) listed a juniper savanna with three series: a oneseed juniper series, a oneseed / Rocky Mountain juniper series, and a Utah juniper series. West and Young (2000) lumped all these types into piñon-juniper woodlands within nine ecological provinces.

Interest in the piñon-juniper woodlands has increased substantially in the last 36 years as evidenced by major symposia presented and proceedings published during that period (table 1).

Vegetational Composition

Ecological provinces listed by West and Young (2000) for piñon-juniper woodlands in the Western United States include Colorado Plateau, Great Basin, northern Rockies, southern Rockies and the Mogollon Rim. Within these woodlands, the following juniper species are common: western juniper (*J. occidentalis*), Utah juniper (*J. osteosperma*), Rocky Mountain juniper (*J. scopulorum*), oneseed juniper (*J. monosperma*), alligator juniper (*J. deppeana*) and redberry juniper (*J. coahuilensis*). The pine species are Rocky Mountain pine (*P. edulis*), singleleaf pine (*P. monophylla*), and Mexican pinyon pine (*P. cembroides*). Many of these tree species overlap in distribution, but form distinct components on a regional basis.

Shrubby and herbaceous layers within the piñon-juniper woodland are more diverse than the tree layer. West and others (1975) listed 31 species in the shrub layer at 20 locations while Gottfried and Pieper (2000) listed 50 species of shrubs in 19 locations (table 2). Forbs were more specific for each location and were more numerous than shrubs and grasses with many annuals (Gottfried and Pieper 2000; West and others 1975). For example, Gottfried and Pieper (2000) listed nearly 200 species of forbs on 19 locations (table 2). Of these only three occurred on more than one location: Louisiana sagewort (*Artemisia ludoviciana*), wholeleaf Indian paintbrush (*Castilleja integra*), and scarlet globemallow (*Sphaeralcea coccinea*). Locations cited by West and others (1975) came mainly from the Great Basin and Intermountain Region while those from Gottfried and Pieper (2000) were mainly from southwestern locations.

Spatial Variation

Fine Scale

Within piñon-juniper woodlands, vegetational patterns are conspicuous under trees as well as in the surrounding open spaces. Arnold (1964) described three distinct zones associated with an individual *J. monosperma* tree in Arizona. Blue

Table 1—Symposia on piñon–juniper ecology.

Location and Host	Year	Pages	Articles	Citation
Logan, UT, Utah State University	1975	196	18	Gifford and Busby (1975)
Santa Fe, NM, U.S. Forest Service	1977	48	12	Aldon and Loring (1977)
Reno, NV, U.S. Forest Service	1987	581	91	Everett (1987)
Santa Fe, NM, N.M. State Land Office	1993	168	38	Aldon and Shaw (1993)
Flagstaff, AZ, U.S. Forest Service	1994	226	35	Shaw and others (1994)
Provo, UT, U.S. Forest Service	1997	441	78	Monsen and Stevens (1999)

Table 2—Common understory shrubs and grasses listed for selected locations in the Western United States

Common Name	Scientific Name	Percent of locations (West and others 1975)	Percent of locations (Gottfried and Pieper 2000)
Shrubs			
Big sagebrush	*Artemisia tridentata*	55	37
Antelope bitterbrush	*Purshia tridentata*	40	32
Gambel oak	*Quercus gambelii*	15	21
Yellow rabbitbrush	*Chrysothamnus viscidiflorus*	30	32
Mormon tea	*Ephedra viridis*	15	16
Grasses			
Squirreltail	*Elymus elymoides*	45	37
Indian ricegrass	*Achnatherum hymenoides*	35	37
Sandberg bluegrass	*Poa secunda*	30	16
Hairy grama	*Bouteloua hirsuta*	15	21
Blue grama	*Bouteloua gracilis*	15	21
Bluebunch wheatgrass	*Pseudoroegneria spicatum*	25	5
Prairie junegrass	*Koeleria macrantha*	15	0

grama was not present next to the bole of the tree, but contributed 0.6% basal cover in the canopy zone and 1.4% in the open space. Similar patterns were noted by Armentrout and Pieper (1988) in New Mexico.

The C_3 grass patterns around individual trees contrast with the pattern for blue grama. Clary and Morrison (1973) found that the biomass of cool-season (C_3) grasses was enhanced by the canopy of alligator juniper (*J. deppeana*) in Arizona (table 3). Similar results were reported by Schott and Pieper (1985) (table 4) and

Table 3—Total herbage and cool-season grass biomass (lb/acre) associated with alligator juniper trees in eastern Arizona (from Clary and Morrison 1973).

Type	Crown zone	Root zone	Open zone
	- - - - - - - - - - - - Biomass- - - - - - - - - - -		
Total herbage	377	121	147
Grass	350	73	78

Table 4—Percent basal cover of three grass species in relation to position under oneseed juniper trees in the Sacramento Mountains in New Mexico (from Schott and Pieper 1982).

Species	Edge	Middle	Center
Bouteloua gracilis			
North	23.8	16.7	1.4
South	23.1	9.6	1.4
Lycurus setosus			
North	2.2	0.4	0
South	2.4	0.2	0
Pipptochaetium fimbriatum			
North	0	4.0	2.3
South	0	0.3	1.3

Pieper (1990) (figure 2). Direction from the tree bole also influenced cover of blue grama (*Bouteloua gracilis*), piñon ricegrass (*Piptochaetium fimbriatum*), and New Mexico feathergrass (*Muhlenbergia pauciflora*) (table 4) (Schott and Pieper 1985).

Biomass of blue grama is generally negatively associated with tree canopy cover in piñon-juniper woodlands in the Sacramento Mountains of New Mexico. Biomass of C_3 generally increased as canopy cover increased (Pieper 1990).

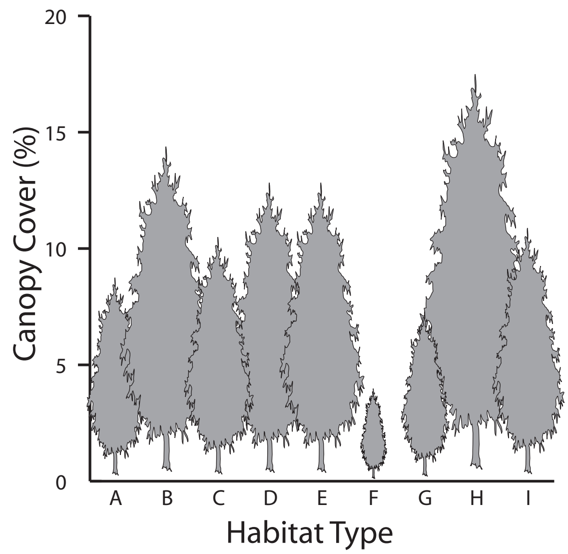

Figure 2—Canopy cover of alligator juniper on different habitat types in western New Mexico (from Hill and others 1990).

Intermediate Scales: Elevation and Aspect

Elevation

Within the woodland, as elevation increases pine abundance generally increases and juniper generally decreases, a pattern exemplified in early studies in northern New Mexico (Woodin and Lindsey 1954). Similar patterns were also shown for the Great Basin (Tueller and others 1979). However, later studies indicate that these patterns may not hold under all circumstances. For example, Kennedy (1983) showed that density of oneseed juniper peaked at intermediate elevations in the Sacramento Mountains in New Mexico. Hill (1990) found that alligator juniper density generally declined with elevation, but tended to level off at intermediate elevations before declining at the lowest elevations.

In southeastern Arizona and northern Mexico, Whittaker and Niering (1965) and Perez (1979) found similar patterns. Mexican pine and alligator juniper increased with elevation from lower positions to intermediate elevation and then declined slightly from intermediate to higher elevations. Undoubtedly many other factors mediate the influence of elevation on the composition of these woodlands as well.

Aspect

Lymbery and Pieper (1983) found that the three tree species in the Sacramento Mountains of New Mexico were distributed on all aspects. Oneseed juniper had a slightly lower canopy cover on xeric southwestern slopes while the highest cover for Rocky Mountain piñon occurred on mesic northwestern aspects. Alligator juniper grew on all aspects, but had slightly higher canopy cover on southwestern slopes.

Regional Scales

West and Young (2000) showed regional differences in tree species within the piñon-juniper woodland, which are also shown by comparisons among various individual studies. Hill and others (1992) identified 10 distinct woodland habitat types in the Gila National Forest in western New Mexico (figure 2), and that the canopy cover of alligator juniper varied from 4 to 18% in these habitat types. Oneseed juniper is distributed largely in central New Mexico with canopy cover as high as 85%, while alligator juniper is more abundant in western New Mexico and Arizona.

Variation in Time

Similar successional patterns were described and diagramed for woodland locations in Colorado, Arizona, and Utah (Arnold and others 1964, Barney and Frischknecht 1974, and Erdman 1976). These diagrams showed that climax woodland communities could be replaced by skeleton forests with sparse understory following fire. Eventually the woodlands would pass through successional stages: annual plants, perennial grass—forbs, shrubby plants, and finally climax juniper or piñon-juniper woodland. Individual species were different for each stage, but they were similar in overall stages. These stages appear similar to many Clementsian linear models, with fire likely to enter at any point in the sequence. An example from Fort Stanton in the central Sacramento Mountains in New Mexico indicated that blue grama grassland became reestablished following severe droughts of the early 1970s, and by 1984 both biomass and composition of blue grama were at pre-drought levels (Pieper and others 1991).

Tress and Klopatek (1987) presented longer time scales for development of piñon-juniper woodlands based on modeling approaches. They showed that understory grasses reached a peak after about 100 years, then declined and plateaued at about 150 years. Shrubs, on the other hand, peaked at about 40 years, then gradually declined and were only minor components after 150 years. Tree canopy cover gradually increased and maintained a maximum canopy cover of about 35% after 180 years. However, these time frames were estimated without major disturbances such as intense fires. Arnold and others (1964) found similar changes following mechanical control treatments in northern Arizona.

Later studies have emphasized the importance of altered fire regimes as mediated by heavy livestock grazing that reduced fine fuels necessary to carry fires. Diagrams showing historical events since European settlement in the 1850s were presented by Allen (1989), Gottfried and others (1995), and West and Van Pelt (1987). These diagrams emphasized the role of heavy livestock grazing that reduced fine fuels and reduced the incidence and intensity of fires. Gottfried and others (1995) stated that "… today most local PJ woodland ecosystems are unstable from a soil perspective, with many moving towards PJ rocklands."

These later analyses suggest that state and transition models (Westoby and others 1989) might be appropriate. However, the review by Iglesias and Kothmann (1997) showed no examples of state and transition diagrams for piñon-juniper woodlands. Figure 3 shows a tentative state and transition model for loamy upland sites with residual soils in the central Sacramento Mountains in New Mexico. In this model, piñon-juniper woodland could be converted to grassland through cutting, other control methods (mechanical, chemical), or intense fire. Grassland dominated by blue grama and Carruth's sagewort (*Artemisia carruthii*) could be changed to herbland dominated by wolfstail (*Lycurus setosus*), broom snakeweed (*Gutierrezia sarothrae*), and showy goldeneye (*Heliomeris multiflora*). Recovery from drought could lead to grassland dominated by blue grama and Carruth sagewort along with many other grasses and forbs, while heavy grazing could tend to increase basal cover of blue grama, creeping muhly, and ring muhly (*Muhlenbergia repens* and *M. torreyi*).

The increase in woody components at the expense of grassland has been noted by several investigators, but interpretations of this change have varied. Krenetsky (1974) resampled several plots in 1964 that had been sampled in 1943 and 1953, and found that canopy cover of oneseed junipers increased under grazed and ungrazed conditions at the expense of grassland in central New Mexico. Miller (1999) used aerial photos to show that woodlands increased in area from 1935 to 1991 at the expense of grasslands on Negrito Creek in western New Mexico. Allred (1996) reported that "…researchers from out of state have suggested a pattern. Ecologists generally assert that piñon and juniper species have either expanded in range or increased in density throughout the West, usually transgressing into adjacent grassland."

Sallach (1986) used photo time sequences to suggest that most of the expansion has been reoccupation of former piñon-juniper sites: "In piñon-juniper woodland, *Pinus edulis* and *Juniperus monosperma* have reestablished on former sites of piñon-juniper woodland. There has not been an extension in area of piñon-juniper woodlands."

It appears that both types of changes have occurred: (1) encroachment onto grassland sites where historic fires tended to limit woody components, and (2) reoccupation of former woodland sites where the trees were removed.

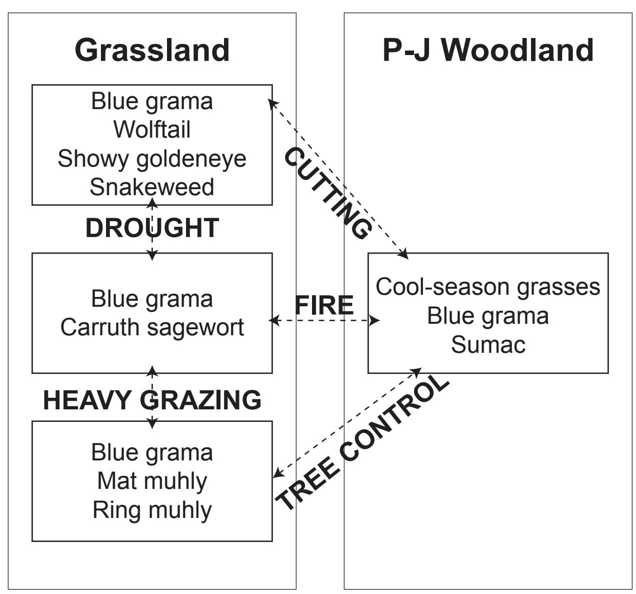

Figure 3—State and transition model for loamy upland ecological sites with residual soils within the central Sacramento Mountains in New Mexico.

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Southwestern U.S. Juniper Savanna and Piñon-Juniper Woodland Communities:

Ecological History and Natural Range of Variability

Brian F. Jacobs[1]

Abstract—*Juniper savanna and piñon-juniper woodland communities collectively represent a widespread and diverse vegetation type that occupies foothill and mesa landforms at middle elevations in semi-arid portions of the American Southwest. Ecological understanding and proper management of these juniper and piñon types requires local knowledge of component species, site history and potential, set within a regional floristic and climatic context. The wide distribution and broad ecological amplitude of this vegetation type across a six-state area of the southwestern United States is best appreciated as the sum of the individual ranges of the component piñon and juniper species, and their environmental tolerances. Key environmental controls of juniper and piñon occurrence, stand age-structure, and composition, include the interaction of climate, topography, soils, and disturbance processes, in combination with biotic interactions, that occur over various spatial and temporal scales.*

Introduction

Juniper savanna and piñon-juniper (P-J) woodlands in the American Southwest have often been viewed by researchers and land managers alike as an ecological unit that can be understood and managed as a single, if variable, entity. The consequences of this approach include confusing and contradictory research findings, ongoing controversy, inappropriate management, and potentially undesirable outcomes. In reality, the P-J type is a simplistic grouping of many different species distributed across diverse climatic and topographic settings, each species with a unique history and range of environmental tolerances. While ecological amplitude of the component species and regional climate set broad limits on potential woodland distributional limits, smaller scale competitive and disturbance factors may ultimately constrain the extent and expression of species occurrence on local landscapes. Recent, historical changes in woodland occurrence and stand density have been variously interpreted as continued adjustment to Holocene climate, recovery from harvest (historic and pre-historic), succession after fire, drought, insect, and/or disease induced mortality events, or response to grazing practices, altered fire regimes, elevated temperatures and CO_2 levels. The following sections provide an overview of the distribution, ecology, environmental controls, disturbance regimes, and historical land use impacts relevant to an understanding of extant southwestern U.S. woodlands.

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Distribution

Juniper savanna and piñon-juniper woodlands collectively constitute one of the most widespread vegetation types in the American Southwest. These plant communities typically occupy foothill, mesa, and mountain slope positions, at middle elevations within a semi-arid climatic zone and between desert grass- and shrub-lands and upland coniferous forests. Within the American Southwest, defined here by New Mexico (NM), Arizona (AZ), Utah (UT), Colorado (CO), Nevada (NV), and eastern California (e. CA), there are five common species of juniper, Utah (*Juniperus osteosperma*), oneseed (*J. monosperma*), Rocky Mountain (*J. scopulorum*), alligator (*J. deppeana*), and western juniper, (*J. occidentalis*), and two of piñon, Colorado (*Pinus edulis*), and singleleaf (*P. monophylla*), which alone and in various assemblages, account for the majority of extant P-J types. An overview of component piñon and juniper species and their respective distributions was obtained by referencing distribution maps originally prepared by Little (1971), subsequently digitized by the USGS-BRD, with taxonomy and nomenclature following Flora North America (Flora of North America Editorial Committee, eds., 1993+).

Colorado piñon has a distribution centered on the Four Corners state area of UT, CO, NM and AZ, while the closely related singleleaf piñon is found to the west in NV, northwest AZ, southeast CA and southwest UT. Piñon typically dominates the upper, or more mesic, end of the woodland zone (although Utah and Rocky Mountain junipers may exhibit greater cold tolerance), while Utah and oneseed junipers gain importance at the lower, or more xeric end. Of the common junipers, Rocky Mountain is the most mesic species with a range extending north to British Columbia, Canada, while alligator juniper, also more mesic than Utah or oneseed junipers, gains importance in warmer areas to the south with a range extending into Mexico. Western juniper is both drought and cold tolerant, and its range, unlike the other four junipers considered, is not associated with any of the piñon species (Miller and others 2005). Within the Four Corners state area, oneseed and alligator junipers predominate in locations under the seasonal influence of the Arizona summer monsoon, defined by Mitchell's climate zone VI (Mitchell 1976), with alligator juniper more common in southern New Mexico and east-central Arizona, while oneseed juniper dominates woodlands in the rest of New Mexico. Utah juniper, conversely, is more common in winter moisture areas, defined by Mitchell's climate zone V, (Mitchell 1976) north and west of the monsoon boundary, and is the most widespread juniper in the American Southwest, forming associations with both Colorado and singleleaf piñon. The ranges of Colorado piñon and Rocky Mountain juniper span the monsoon boundary (although piñon distribution is bounded to the north by the position of the winter polar front, defined by the northern boundary of Mitchell's climate zone V), perhaps in part because moisture patterns tend to lose strong seasonality with increasing elevation.

Looking around the six-state area, and noting piñon and juniper species whose ranges bound the American Southwest, we find California piñon (*Pinus quadrifolia*), Mexican piñon (*P. cembroides*) and redberry juniper (*Juniperus coahuilensis*, sensu ~*J. erythrocarpa*) (Little 1971) just entering our area, but with ranges mostly to the south in Mexico, western juniper, (*J. occidentalis*) occurring in eastern Californa and northern Nevada, but with a range extending northwest to Oregon, Washington and Idaho, and Pinchot's juniper (*J. pinchotii*), ashe juniper (*J. ashei*) and eastern red-cedar, (*J. virginiana*) to the east in Texas and Oklahoma.

While each of these P-J species are distinctive enough to be afforded specific taxonomic status, and even retain integrity as a distinct taxon in the paleo-record, modern distributions are relatively recent and extant populations representing recognized taxa likely represent or express only a portion of the underlying genetic diversity present; further, many or most of the piñon and juniper species have some level of gene flow between related species, which presents additional opportunities from both ecological and evolutionary perspectives. For example, western juniper reportedly hybridizes with Utah juniper, Utah with oneseed, oneseed with alligator, redberry with Pinchot's, and Rocky Mountain with eastern red-cedar (*J. scopulorum* is sometimes classified as a variety of *J. virginiana*); the sprouting ability of redberry and Pinchot's junipers, and small, oneseeded cone features, may suggest relationships between these taxa and *J. deppeana* and *J. monosperma*. *Pinus edulis* can reportedly hybridize with *P. monophylla* (~*P. edulis var. fallax*) and *P. cembroides* (~*P. remota*), while *P. quadrifolia* was formerly recognized as a variety of *P. cembroides*.

From an evolutionary perspective, closely related piñon taxa that maintain capacity for genetic exchange, and whose shifting ranges both maintain intermittent contact while promoting expression of discrete entities, might be more productively viewed as components of larger meta-populations. Long-lived perennials like piñon and juniper, which have potential maximum lifespans exceeding 500 years, are buffered against shorter-term fluctuations in climate, requiring only occasional favorable windows for successful establishment. In contrast, climatic requirements for persistence of mature P-J individuals are often minimal. Long-lived, wind pollinated, out-crossing, perennials could be expected to maintain high levels of genetic diversity in a population, while also being a conservative force mitigating rapid shifts in allele frequency.

Ecology

Southwestern P-J types viewed collectively span an impressive range of environmental settings and present challenges to traditional ways of categorizing vegetation assemblages and interpreting ecological processes. Piñon dominated stands with multi-layered and nearly closed canopies, can occur at the moist, upper elevation end of the woodland zone, while at the interface with desert grasslands, it is common to observe open stands of juniper interspersed with grasses, forbs, and shrubs. In between these extremes, and depending on a variety of local site conditions and histories, and within the context of regional biogeography, one can delineate a great variety of juniper and piñon-juniper types in association with various shrub, grass, and forb understories. Recent vegetation mapping efforts as part of the Southwest Regional Gap Analysis Project, SWReGAP (Lowry and others 2005) with a five-state area (Four Corners state area plus NV) of the American Southwest, and following community classifications prepared by NatureServe (Comer and others 2004), circumscribe four major P&J categories, primarily as groupings of the major piñon and/or juniper species: Great Basin (*P. monophylla* and *J. osteosperma*), Colorado Plateau (*P. edulis* and *J. osteosperma*) , Southern Rocky Mountain (*P. edulis* and *J. monosperma*), and Madrean (*P. edulis*, *P. cembroides*, *J. deppeana*, *J. monosperma*, *J. coahuilensis*, and *J. pinchotii*). These four major groupings are further sub-divided on basis of structure and composition (that is, piñon-juniper woodland versus juniper savanna) with additional community assemblages noted as having a juniper and/or piñon component. Finer grained community and habitat typing in juniper and piñon-juniper types have typically subdivided major tree overstory groupings by dominant shrub-grass-forb understory. Understory composition can be an important indicator of site history and site potential, particularly when the tree overstory is of relatively recent origin; Tausch (1999) suggests understory composition can be key to understanding the potential of particular P-J sites, providing insight on available soil and water resources, and presumably this information would be critical to management at local scales.

Climate, modified by local topographic and soil factors, provides fundamental control over potential species distributions; while disturbance regimes and stochastic events help to shape actual occurrence patterns. A range of potential vegetation types is possible for most locations, and extant vegetation may or may not represent a balanced or optimal state from natural or human perspectives. Extant plant communities should be viewed as the cumulative outcome of multiple interacting factors, and over shorter temporal and smaller spatial scales there appear to be repeating patterns and a sense of dynamic stasis; however, paleo-vegetation reconstructions reinforce the idea that species assemblages are neither prescribed nor static at longer or larger scales (Betancourt and others 1993). Still, ecological concepts such as succession and restoration are still meaningful and useful within the limited spatial and temporal scales that land managers (and—researchers) typically operate. More problematic is how to—integrate rare, episodic, and/or extreme events into our ecological understanding and short term, local management of vegetation systems, especially when these low frequency events have large and long term consequences on community structure, composition and function.

Disturbance

Interpreting the relative importance of disturbance processes and episodic events—like fire, wet and dry climate patterns, and insect or disease outbreaks—in controlling spatial pattern and structure of vegetation is challenging, especially when the current vegetation on a site is, in part, an outcome of earlier disturbances or events that occurred within the context of a different vegetation assemblage. For example, fire disturbance is possible or likely given suitable fuel structures, which are produced by particular vegetation assemblages, and which in turn depend on favorable suites of climate, topographic, and soil factors; fire events and patterns may be strongly associated with particular vegetation assemblages to the extent we can recognize recurring burn patterns (intensity and frequency) and/or infer meaningful relationships between vegetation composition, structure, and life history. Southwest ponderosa pine, tall grass prairie, or northern Rocky Mountain lodgepole pine communities can be somewhat easily assigned to fire regime categories, and there are often meaningful synergies that exist between these vegetation types, life history attributes of dominant species, and recurring fire disturbance.

In contrast, Baker and Shinneman (2004) review a number of fire history studies and note that evidence to substantiate spreading surface fire behavior in woodlands is generally lacking; fire scars on living trees are usually infrequent, and often found at what could be interpreted as ecotonal boundaries (such as rocky outcrops, or an interface with Ponderosa pine savanna) or woodland burn patch edges. Thus, although fire histories have been reconstructed for selected P-J sites where abundant fire evidence is available, this fire evidence may be reflective of historic upper and lower ecotonal boundaries where woodlands abutted high fire frequency forest and grassland systems, or locations where fine scale woodland mosaics (superimposed on topo-edaphic patterns) formerly intermingled with fire prone, non-woodland types, than of the general P-J type in a larger sense. The historic role of surface fire disturbance in maintaining stand structure and composition in pre-settlement P-J types then, is problematic; for example, most of the piñon and juniper species are fire sensitive and easily killed by even moderate fire intensity and the species as a group generally lack life history attributes that can be easily associated with recurrent fire disturbance (although several juniper species can resprout after burning, and one can infer possible mechanisms, such as dense litter mats or suppressed herbaceous, for mitigating fire mortality and scarring).

Observations by the author at numerous field locations in the Four Corners state area, in connection with an effort to model occurrence of pre-settlement woodlands relative to topo-climatic factors, suggest fire disturbance in pre-settlement Colorado Plateau and South Rocky Mountain P-J types was at best uneven. It appears more opportunistic than inevitable, largely dependent on local site conditions, and not obviously essential to maintenance of 'normal' system structure and function. Historic evidence of fire, when present, often suggests a patchy crown fire behavior, with charred stumps, logs, and snags, as might be expected with the discontinuous fuel structure (surface and crown) associated with this vegetation type (Muldavin and others 2003).

Water

Water is a major limiting resource in semi-arid systems, and it is reasonable to infer that extant P-J types are largely responsive to and shaped by (spatial and temporal) variability in available soil moisture. For example, it is widely reported that there is an inverse relationship between increasing density of overstory in P-J types and decreasing understory cover (interpreted as a response to limited soil water); conversely, it has also been demonstrated that mechanical thinning, fire treatment, and drought-insect induced mortality of overstory, will often yield increases in understory cover. Available soil moisture then is an important, perhaps central, environmental control in P-J systems, affecting where they can occur, which species can be present, influencing stand structure by mediating episodic establishment and mortality, and potential for fire or drought-insect disturbance. Eisenhart (2004) proposes density dependent regulation of stand density in *P. edulis – J. osteosperma* types (that is, self-thinning) and periodic drought-insect mortality as viable mechanisms for maintenance of stand structure in P-J types, particularly in the absence of any fire evidence. Savanna structure in low end juniper dominated types could also be interpreted as a density dependent response to limited soil moisture, particularly on shallow substrates where trees are primarily accessing deeper water stored in fractured bedrock. Site productivity and potential in semi-arid settings is largely a function of (spatial and temporal availability of) soil moisture (McAuliffe 2003), which is controlled by the interactions of climate, topography, and soil.

Different growth forms and species employ a variety of strategies for extracting available soil moisture. A site may be productive for deep rooted trees, if available water is mostly at depth, either due to a deep, well drained soil or a shallow soil with fractured bedrock; conversely, a site may be productive for shallow rooted, herbaceous species, if available water is primarily in upper 0-30 cm due to fine textured soils, high clay or organic content or presence of shallow argillic (that is, water perching) horizons. Some, or most, sites can support a mixture of both shallow and deeper rooted species, and many species (including piñon and juniper) have dimorphic root morphologies and flexible strategies that allow them to opportunistically (and temporally) harvest water from both shallow and deep sources, as well as from wide horizontal extents encompassing adjacent intercanopy locations (McAuliffe 2003). Thus, even with a uniform climate context, extant vegetation and site potential can be strongly influenced by local topographic setting and soil properties.

For example, at Bandelier National Monument, NM, pumice soils can strongly influence local vegetation patterns and associated disturbance processes though enhanced water capture and storage. Julius (1999) documented differences in piñon-juniper age-class and density, as well as in composition and cover of associated understory, across three soil types within a 100-acre study area at Bandelier. Woodlands on pumice soils, with an argillic horizon, had both the lowest tree densities and youngest age-class, relative to pumice and non-pumice soils, without an argillic horizon; non-pumice, non-argillic soils had the highest densities and oldest age-class, while pumice, non-argillic soils were intermediate for both density and age-class. In addition, pumice argillic soils supported the highest understory cover, with a composition dominated by grasses, (such as *Schizachyrium scoparium*), while non-argillic, pumice soils had forb dominated understories, and non-pumice, non-argillic soils were characterized by only sparse understory cover (Julius 1999).

Germination and successful establishment are critical life stages for all plants, but in semi-arid climates, proper timing is especially important. Successful establishment of piñon and juniper individuals is enhanced by sufficient moisture during the time period between germination and establishment of a secondary root system, below the average depth of the herbaceous rooting zone; Johnsen (1960, 1962) reported that seedlings of *J. monosperma* were very vulnerable while in direct competition for water with shallow and fibrous rooted herbaceous species, and could only successfully establish during years when soil water was effectively not a limiting resource. As noted by Neilson (2003), the effective window for successful tree establishment may have been enhanced by the reduction of herbaceous competition through sustained grazing.

Grazing

Considerable attention has been focused on the presumed effects of historic grazing, both in altering the structure and composition of pre-settlement P-J types (that is, infill and thickening), as well as in promoting tree encroachment into formerly non-woodland (including forest, shrub- and grass-land) communities. Effects of long-term, sustained grazing in semi-arid systems, particularly during drought episodes, can include reduced herbaceous cover, vigor, and (above and below ground) biomass, increased runoff and sediment transport, and initiation and facilitation of desertification processes (that is, re-allocation of limited nutrient and water resources to shrub and tree 'islands'). Alternatively, simultaneous reduction of understory competition and associated interruption of surface fire regimes, during favorable climatic intervals, would appear to be plausible effects of historic grazing in facilitating P-J encroachment into non-woodland areas. Both of these mechanisms, acting in concert, are likely important factors mediating recent 'invasion' of western rangelands (cool and warm season respectively), by western juniper in OR (Miller and others 2005) and oneseed juniper in AZ and NM (Johnsen 1960, 1962). However, within extant, pre-settlement, savanna and woodland communities (where evidence to support a role for recurrent surface fire in maintaining stand structure is absent), it may be reasonable to conclude that historic grazing effects alone would have been sufficient to alter the competitive environment and facilitate establishment of tree seedlings, by reducing herbaceous competition for water, focusing runoff and enhancing deeper infiltration through reduction of effective ground cover. Recent attention has also been given to the idea of CO_2 facilitated enhancement of tree growth through increased water use efficiency, but this proposed effect may be largely offset by increased evaporative demand and thermal stress from warmer temperatures associated with increased levels of CO_2.

It has also been noted that grazing effects can be extremely variable across different soil types within the same climatic zone, suggesting some sites and soils are more tolerant of grazing, while conversely, other sites and soils are more susceptible to desertification (that is, shrub and tree encroachment). McAuliffe (2003) notes grazed soil types, with shallow argillic horizons, are much more resistant to woody plant encroachment than sites that promote deeper infiltration. As Nielson (2003) suggests, recent and widespread encroachment of woody plants into many western rangelands (and thickening of savanna types) is probably best interpreted as a synergistic interaction of climate and grazing, on susceptible soil sites, and where woody plant populations are proximate.

In some areas of the American Southwest, particularly on portions of the Colorado Plateau characterized by winter moisture patterns, the paradigm of a pervasive and ongoing, grazing induced, western woodland invasion is overstated, or at best mis-applied. For example, Floyd and others (2003) report extant stand densities at Mesa Verde National Park, CO, are generally within the range of historical variability, while Eisenhart (2004) suggests that reports of 'thickened' woodlands in west-central CO woodlands may actually be normal stages in stand development prior to onset of density dependent thinning as trees mature.

Summary

Recent field reconnaissance in southwestern U.S. woodlands, suggests that there is an abundance of pre-settlement status woodland on the Colorado Plateau, where precipitation patterns are either winter dominated or lack strong seasonality. These observations are consistent with the results of detailed stand reconstructions by Floyd-Hanna and others (2003) and Eisenhart (2004) in woodlands of Colorado. Conversely, occurrence of post-settlement status woodland becomes increasingly frequent in rangeland areas of northern New Mexico and east-central Arizona, where they occupy lower gradient basin and valley landform settings, adjacent slopes and rolling hills, in locations characterized by wet summer and comparatively dry winter-spring precipitation patterns. The transition zone, those areas where occurrence of post-settlement woodland becomes increasingly frequent, also roughly corresponds to the distributional limits of oneseed juniper (although both piñon and Rocky Mountain juniper are also commonly present in post-settlement stands). Pre-settlement woodland in the monsoon climate area becomes increasingly restricted to a narrower range of settings, including steeper gradient landforms, with shallow rocky substrate, and/or isolated or broken topographic settings, presumably reflecting locations that promote deeper water infiltration and/or limit fire effects.

Interpreting regional patterns of pre- versus post-settlement woodland occurrence in relation to climatic, topographic, and edaphic variables, can be complicated by associated changes in both woodland species assemblages and geomorphic settings. However, seasonality of moisture appears to be an important determinant of woodland patterns in both areas, but perhaps for different reasons. In winter moisture dominated areas, woodland can successfully occupy a wider range of landform settings, since moisture is available both at greater depth and during the early spring season, promoting woody growth, limiting herbaceous competition, production of fine fuels, and potential for surface fire. Conversely, in summer moisture dominated areas, shallow moisture enhances herbaceous competition, fine fuel production, and the potential for surface fire, effectively restricting woody vegetation to coarser textured soil settings where moisture can infiltrate to depth (on steep slopes, and fractured rocky substrates) or locations where grass production is otherwise limited (reducing potential for surface fire); even discontinuous topographic settings with adequate soils could still be expected to have relatively high potentials for surface fire (in absence of grazing), given the increased incidence of lightning ignition associated with a summer monsoonal precipitation pattern.

Whatever the mechanisms (such as, favorable climatic patterns, grazing effects on herbaceous competition and fire disturbance) responsible for the initial establishment of woodland species onto a new site, persistence of the tree component can be enhanced by positive feedback on local environmental conditions; for example, suppression of herbaceous vegetation by maturing piñon and juniper overstory, and associated reductions in intercanopy cover, changes in soil texture and runoff that promote deeper infiltration, and reduced potential for surface fire, tend to reinforce conditions favorable to woody plant establishment and persistence. We can think of these woodland influences on local site conditions in terms of moisture and fire shadow effects, which in the absence of a disturbance, allow woodland to establish into, persist on, and eventually dominate a wide range of settings.

From a landscape perspective, infilling and thickening of patchy, pre-settlement woodland mosaic patterns by post-settlement woodland, was likely facilitated by regional, synchronous and/or synergistic, effects of climate and grazing. Some reports of woodland thickening may also be a function of the relative spatial and temporal perspective in sampling or observation. As woodland patches expand and merge, ground fuels become limiting while canopy fuel structure becomes more continuous across larger areas; under this scenario, the probability of fire spread from a point ignition can be expected to change, along with the potential frequency, nature, and extent of fire events. Discerning patterns of recent woodland expansion from longer term migrational dynamics, may be possible by comparing the range of environmental settings associated with pre- versus post-settlement stands. For example, while relatively few new northerly locations appear to have been successfully colonized in response to migrational dynamics of piñon pine during the last 1,000 years (Jackson and others 2005), the extent of woodland occurrence across its range has apparently increased several fold since 1850 (West 1999).

Management of the P-J type then, need not be strictly bound by idealistic notions of pre-settlement ecological status and dynamics—which we struggle to reconstruct in any case—but rather informed by ecological knowledge of site potential and vegetation dynamics, in support of the application of sustainable and appropriate management practices that attempt to balance stated societal needs and desires.

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Customized displays of piñon and juniper distributions and climate data can be created and viewed within a GIS environment. USGS digitized shapefiles of individual tree species based on Little (1971) can be downloaded at (http://esp.cr.usgs.gov/data/atlas/little/) and PRISM climate data (4km and 800m resolution) from (http://www.ocs.orst.edu/prism/).

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Growth and Yield of Southwest Pinyon-Juniper Woodlands: Modeling Growth and Drought Effects

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Abstract—*A complex of drought, insects, and disease caused widespread mortality in the pinyon-juniper forest types of the American Southwest in recent years. Most public and scientific attention has been given to the extent of drought-related mortality and causal factors. At the same time, there has been relatively little attention given to non-lethal drought effects. As part of standard data collection protocol, the Interior West Forest Inventory and Analysis program measures radial increment on a portion of tally tree species. Between 1981 and 2005, data were collected on 14,929 plots with pinyon or juniper species present. These data included 10-year increment measurements on 23,080 pinyon and juniper trees. These data were used to characterize the "typical" growth patterns of pinyon and juniper species, and to analyze the effects of drought on radial increment. In addition, these measured increments were compared to those projected by the Forest Vegetation Simulator (FVS). Most species showed a reduction in diameter increment when data from pre-drought years were compared to post-drought years. FVS produced diameter increments that were within the range found in the data, but height growth may be over-predicted in some cases.*

Keywords: *diameter increment, height growth, Forest Inventory and Analysis (FIA), mortality, Forest Vegetation Simulator*

Introduction

Recent mortality in pinyon-juniper woodlands has been caused by a complex of drought, insects, and disease. The dramatic appearance of dying pinyons across the landscape brought much public and scientific attention to the episode, and there have been many attempts to assess the extent and severity of the mortality (Breshears and others 2005, Shaw and others 2005, Shaw 2006a, 2006b). However, there has been little attention given to the non-lethal effects of the drought. Tree rings are well-known as indicators of precipitation in dendrochronological and dendroecological studies (e.g., Baisan and Swetnam 1990, Swetnam and Betancourt 1990). Most of these studies look at long-term trends, commonly for the purpose of climatic reconstructions. Ogle and others (2000) found that variations in growth increment preceded mortality in pinyons that eventually succumbed to drought.

In this paper, increment data are used to make a preliminary assessment of growth patterns in pinyon and juniper species of the Southwest. Growth is evaluated from three perspectives: 1) typical radial growth patterns, 2) yield expectations from typical stands (i.e., growth-growing stock relationships), and 3) factors, including drought, that affect growth patterns. In addition, the diameter growth modeling approach used by the Forest Vegetation Simulator is examined.

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Methods

Data used in this analysis come from the Interior West Forest Inventory and Analysis (IW-FIA) program of the U.S. Department of Agriculture Forest Service. The national FIA program conducts inventory on all forested lands of the U.S. using a nationally standardized plot design at an intensity of approximately 1 field plot per 6000 acres. IW-FIA is responsible for lands in Arizona, Colorado, Idaho, Montana, Nevada, New Mexico, Utah, and Wyoming. These states contain over 85% of the pinyon-juniper forest in the Western United States.

FIA field staff documented and measured 14,929 plots that included pinyon or juniper species between 1981 and 2005. Radial increment was measured on 23,080 pinyon and juniper trees (table 1). Increment from the 10 years prior to plot visit was measured from a core and recorded to the nearest 1/20 inch. This value was transformed to the 10-year diameter increment as a percentage of starting diameter (i.e., diameter 10 years prior to measurement). Diameter and increment are measured at the root collar for all pinyon and juniper species, except for western juniper, which is measured at breast height.

Several other variables were measured or calculated at the plot level, including elevation and basal area. Data were aggregated for the purpose of comparing pre-drought and post-drought conditions. Based on analyses of mortality on FIA plots (Shaw and others 2005, Shaw 2006b), data from 1981-2002 are considered to be pre-drought and data from 2003-2005 are post-drought.

A flexible nonlinear function (equation [1]) was fitted to the transformed increment data in order to explore growth sensitivity to different factors. For the purpose of evaluating the effect of different factors (e.g., stand basal area), the factors were grouped into classes and the model was fitted to each class separately.

$$diapctgrow - a(b - \exp(-c \cdot dia10ago)) \qquad [1]$$

where *diapctgrow* is diameter increment expressed as a percentage of starting period diameter,

*dia*10*ago* is stem diameter at beginning of increment period,
and *a*, *b*, and *c* are parameters to be estimated.

Using the modeled increment curve and a generalized height growth curve, 10-year periodic growth and yield were simulated for a hypothetical "average" stand of common pinyon. In the hypothetical stand, all trees are of identical diameter and height. The stand was initiated using a starting diameter of 3.8 inches and

Table 1—Number of records including increment data for pinyon and juniper species in the IW-FIA database.

Species	Common name (FIA code)	Number of increments
Juniperus coahuilensis (Martinez) Gaussen	redberry juniper (59)	116
Juniperus deppeana Steud.	alligator juniper (63)	1436
Juniperus monosperma (Engelm.) Sarg.	oneseed juniper (69)	2291
Juniperus occidentalis Hook.	western juniper (64)	143
Juniperus osteosperma (Torr.) Little	Utah juniper (65)	6498
Juniperus scopulorum Sarg.	Rocky Mountain juniper (66)	1908
Pinus discolor Bailey & Hawksworth	border pinyon (134)	198
Pinus edulis Engelm.	common pinyon (106)	8761
Pinus monophylla Torr. & Frem.	singleleaf pinyon (133)	1569
Pinus monophylla var. *fallax* (Little) Silba	Arizona pinyon pine (143)	160

a height of 11.3 ft. For each 10-year period in the simulation, diameter increment was predicted as a function of starting period diameter, and height was calculated as a function of diameter. Yield was calculated for each period based on stem densities of 50, 100, 150, 200, 250, 300, and 400 trees per acre.

In addition to data-based projections, the Forest Vegetation Simulator (FVS) (Johnson 1997) was used to project a number of pinyon-juniper stands using FIA data collected in New Mexico. These simulations were not a formal test of FVS performance, but rather a comparative check of FVS output in terms of diameter and height growth.

Results

General Growth Patterns

Growth varied somewhat among species, but followed general patterns; results are shown for representative species. Detailed statistical evaluations are not included because the analyses presented here are exploratory in nature and only presented for illustration of general relationships.

All species showed similar growth patterns: high variability in early growth followed by a decrease that follows a negative exponential pattern (figure 1). In general, starting diameter alone predicted less than 50% of the variation in diameter increment, using equation [1] or related equations. While these equations were sufficiently flexible to capture the basic relationship and produce unbiased fits, the regression models could not account for the high variability of increment in small trees.

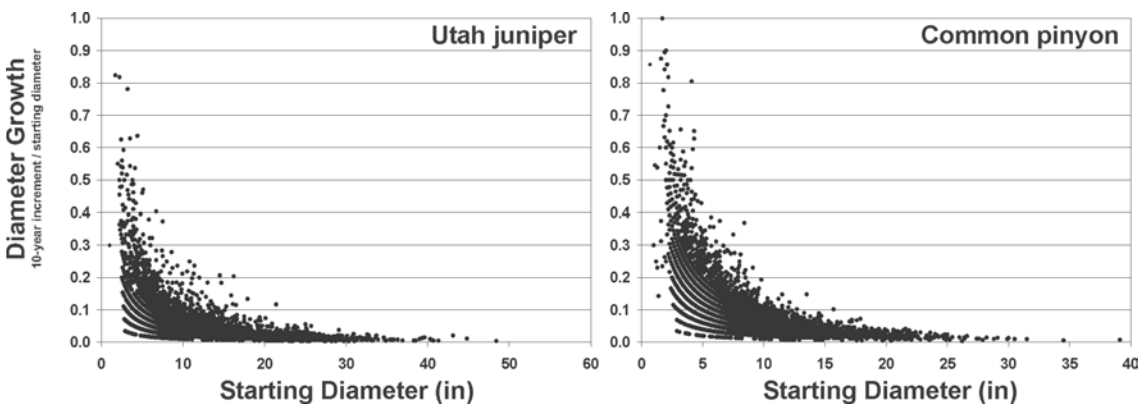

Figure 1— Ten-year radial increment data for Utah juniper (n = 6498) and common pinyon (n = 8761). Radial increments have been transformed to diameter growth as a fraction of starting diameter (i.e., diameter at the start of the 10-year growth period).

Growth and yield of the hypothetical average pinyon stand was consistent with expectations (table 2). Periodic annual increment at most combinations of age and density was generally lower than commonly considered as productive forest land (a minimum of about 10 ft^3/ac/yr). Yields higher than 10 ft^3/ac/yr could be achieved in highly-stocked conditions, but there was an apparent stocking-limited ceiling on yield at approximately 14 ft^3/ac/yr. This situation arises because combinations of size and density needed to achieve higher yield are at the limits of possible relative density—i.e., stand density index greater than about 350. As a result, limits of age and stocking appear to keep mean annual increment below 10 ft^3/ac/yr (figure 2). However, it should be noted that this simulation represents an "average" condition, and that yield could be substantially higher on better sites.

Table 2—Mean annual increment (cubic ft per acre per year) for a hypothetical pinyon stand for stand densities of 50 to 400 trees per acre. Diameter growth is based on average modeled 10-year radial increment of common pinyon. Light shading indicates combinations of age and density that achieve yield considered to be in the commercial range. Dark shading indicates yields that are not likely to be achievable because of limitations on relative density.

Age	DRC	Height	TPA50	TPA100	TPA150	TPA200	TPA250	TPA300	TPA400
40	3.8	11.3							
50	5.0	12.9	1.5	2.9	4.4	5.8	7.3	8.7	11.6
60	5.7	13.9	1.2	2.4	3.6	4.8	6.1	7.3	9.7
70	6.4	14.6	1.3	2.6	3.8	5.1	6.4	7.7	10.2
80	7.0	15.3	1.4	2.7	4.1	5.5	6.9	8.2	11.0
90	7.6	15.9	1.5	3.0	4.5	5.9	7.4	8.9	11.9
100	8.1	16.5	1.6	3.2	4.8	6.4	8.0	9.7	12.9
110	8.7	17.1	1.7	3.5	5.2	7.0	8.7	10.4	13.9
120	9.2	17.6	1.9	3.8	5.6	7.5	9.4	11.3	15.0
130	9.7	18.1	2.0	4.0	6.1	8.1	10.1	12.1	16.2
140	10.2	18.6	2.2	4.3	6.5	8.7	10.9	13.0	17.4
150	10.8	19.1	2.3	4.7	7.0	9.3	11.7	14.0	18.7
160	11.3	19.5	2.5	5.0	7.5	10.0	12.5	15.0	20.0
170	11.8	20.0	2.7	5.3	8.0	10.7	13.4	16.0	21.4
180	12.3	20.4	2.9	5.7	8.6	11.4	14.3	17.1	22.8
190	12.8	20.8	3.0	6.1	9.1	12.2	15.2	18.3	24.3
200	13.3	21.2	3.2	6.5	9.7	13.0	16.2	19.4	25.9
210	13.9	21.7	3.4	6.9	10.3	13.8	17.2	20.7	27.6
220	14.4	22.1	3.7	7.3	11.0	14.6	18.3	22.0	29.3
230	14.9	22.5	3.9	7.8	11.7	15.5	19.4	23.3	31.1
240	15.4	22.9	4.1	8.2	12.4	16.5	20.6	24.7	32.9
250	16.0	23.3	4.4	8.7	13.1	17.4	21.8	26.2	34.9

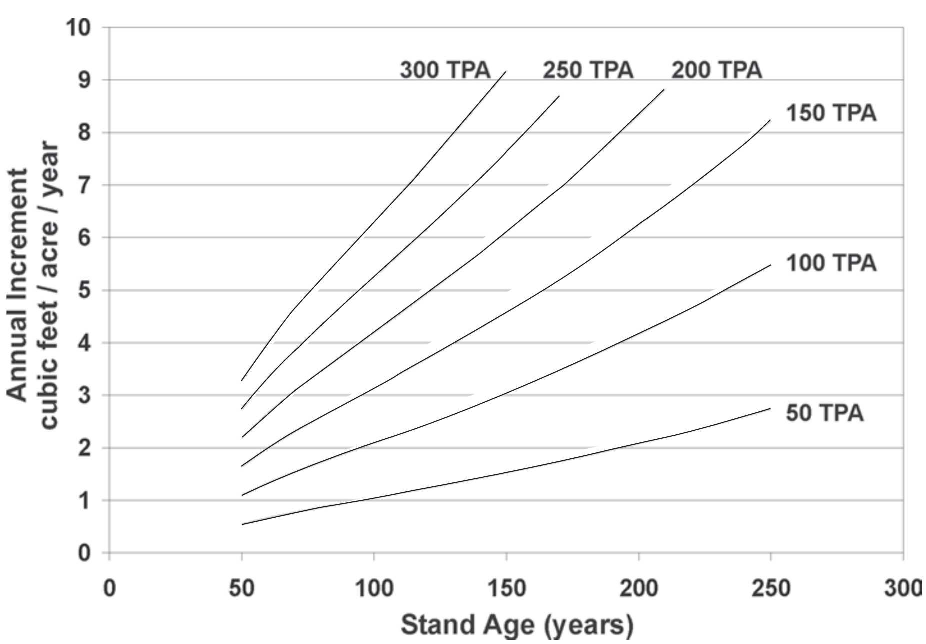

Figure 2—Mean annual increment in cubic feet per acre per year for hypothetical stands of common pinyon at stem densities of 50 to 300 trees per acre. Diameter growth is modeled from 10-year increment data (see figure 1).

Effects of Stand and Site Factors

In most cases, diameter growth responded to stand and site factors as expected. However, there were some exceptions. To evaluate the effect of elevation, plots were grouped into four 2000-ft elevation classes and fitted growth curves to each class separately. Utah juniper showed a clear negative relationship between elevation and diameter increment (figure 3a). The results for common pinyon were mixed, with the 2000-ft class having the smallest increment; diameter growth was higher in the other 3 elevation classes, but there was no clear pattern among those (figure 3b). To evaluate the effect of stand density, classes of 50, 100, 150, and 250 trees per acre were used. As expected, there was a negative relationship between growth and stand density (figure 3c,d). The effect of stand density on diameter growth was clearest when starting diameter was less than 18 inches. In larger trees, there appeared to be little to no effect of density on diameter increment.

Effects of Drought

In general, drought had a negative effect on increment, although the magnitude and range of diameters over which it appeared strongest varied by species. Increment reductions were relatively small for most species, and in the case of common pinyon, larger trees had slightly better growth during drought years (figure 4). This situation may be attributable to the number of large pinyons that succumbed to drought. Average growth of residual trees may have been effectively increased by removal of mortality trees, which tend to have smaller increments

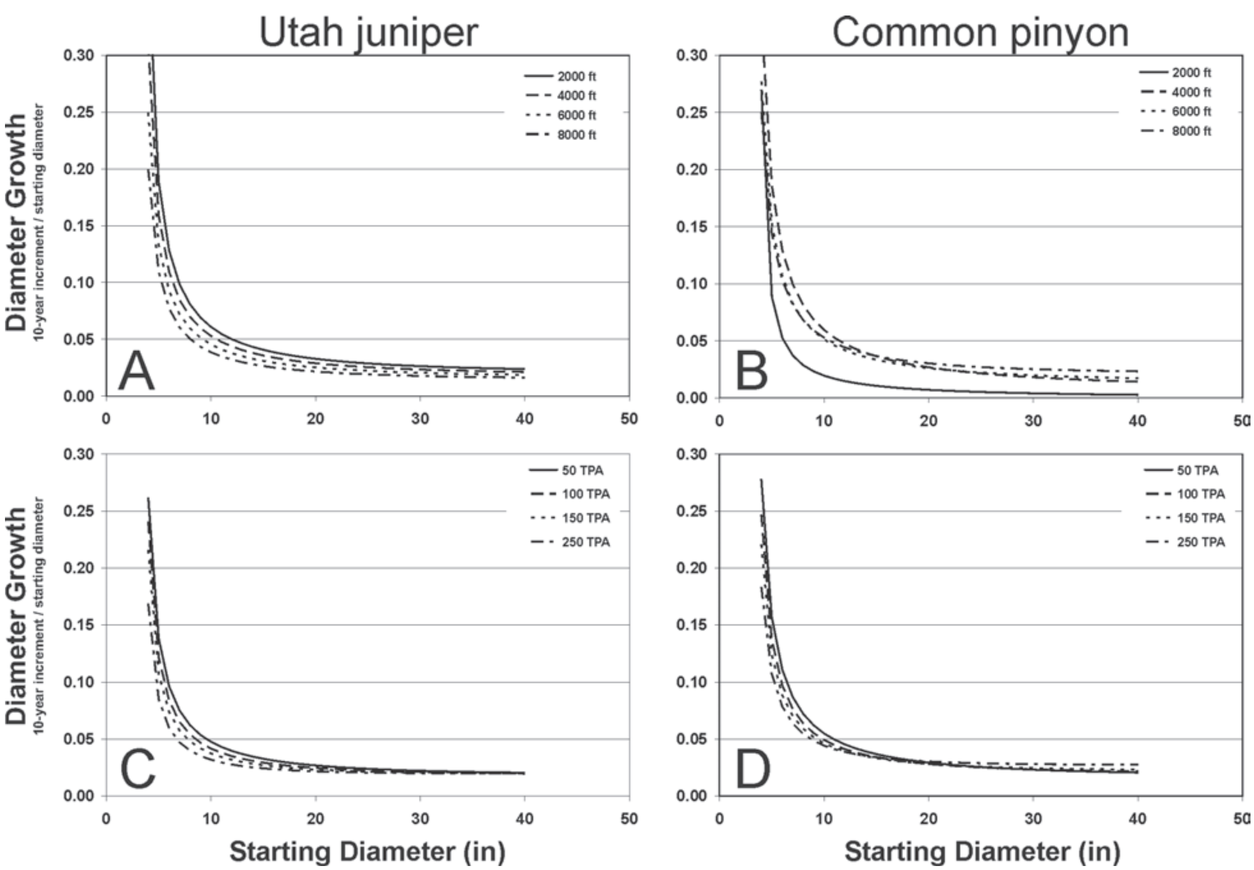

Figure 3—Variation in diameter growth of Utah juniper and common pinyon as a response to elevation (A and B) and stand density (C and D).

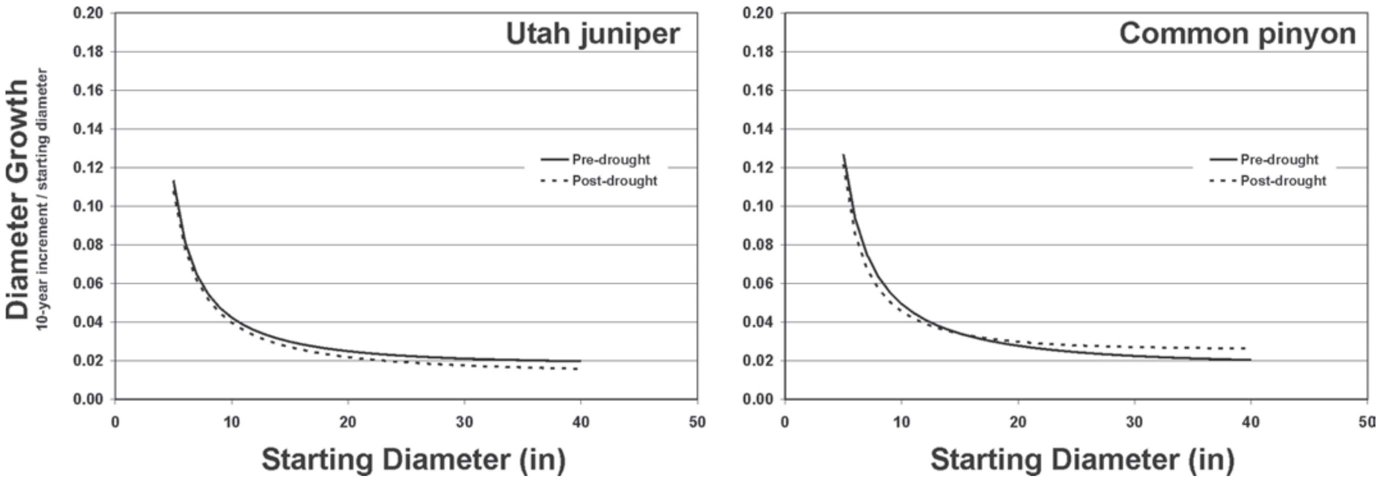

Figure 4—Variation in diameter growth of Utah juniper and common pinyon as a response to drought. Pre-drought data include all increment observations recorded 1981-2002 and post-drought data include increment measurements record 2003-2005.

during drought than similar-sized trees that ultimately survive the drought (Ogle and others 2000). However, for some species there were substantial reductions in increment. At 12 inches diameter, alligator juniper showed a 9 percent reduction and singleleaf pinyon showed a 17 percent reduction.

Forest Vegetation Simulator Projections

In our simulations, FVS appeared to project diameter growth without bias, but only captured a fraction of the natural range of variability found in young trees. However, the limited range of projected growth could be a function of the geographic limits of the source data. Height growth was somewhat over-projected, resulting in slightly higher yield projections.

Discussion

Growth patterns of pinyon-juniper woodlands have been studied in the past (Howell 1941, Reveal 1944, Daniel and others 1966, Chojnacky 1987, Little 1987, Smith and Schuler 1987), but not to the extent of commercially important species. Interest in pinyon-juniper woodlands has been cyclic, with high points of interest based on changing management needs (Van Hooser and Casey 1987). Currently interest is at a high point due, at least in part, to the recent drought and associated mortality of pinyon across the landscape. Some interest in pinyon-juniper woodland may also be attributable to the increased attention given to climate change, carbon sequestration, and wildfire. The recent increase in interest has been shown by an increase in ecological studies, but little in the way of basic productivity research.

Although pinyon-juniper woodlands are only marginally important from a wood productivity standpoint, basic growth information is an important element of ecological studies. Knowledge of growth and yield is essential to understanding how long it will take woodlands to recover from drought- or fire-related mortality, and how carbon balances might change over time. It is also important to understand the effects of climate on growth. The data show relatively predictable growth patterns in pinyon-juniper woodland, but to accurately project growth it will be necessary to account for the variation observed in smaller trees. Much of this variation can be accounted for by various stand and site factors, which will be investigated further in ongoing studies. Additional variation, found in all size classes, is apparently a result of climatic factors—most likely precipitation, temperature, or both. The recent drought had a demonstrable impact on growth and mortality, and only the latter has been taken into account in any detail. Better prediction of future stand growth should be possible by taking climatic factors into account.

The Forest Vegetation Simulator (Johnson 1997) predicts growth reasonably well according to practitioners who have used it to project pinyon-juniper stands. The limited testing done here generally supports their claims, although some improvements can be made. The current FVS diameter growth model has a linear, 13-parameter form and accounts for several stand and site factors, in addition to diameter-based variables. Although this model works reasonably well, there may be limitations based on model form (Shaw and others 2006).

The results presented here suggest that a modeling approach in which the basic, nonlinear size-increment model can be built upon using stand and site factors. Using sensitivity analysis, Vacchiano and others (in press) have shown differential contributions of variables in the current diameter growth model used in the Southern Variant of FVS. Further exploration of the relative importance of stand and site variables may lead to development of a new model form in which variables are added by importance rank. Such an approach may produce a parsimonious model that captures the maximum amount of variability that is practical.

Finally, existing variants of FVS do not account for climate, although development of climate-based variants is under investigation (Crookston 2007). Although the emphasis of this work is on long-term climate change, analysis of recent increment data indicate that incorporation of a climate component would benefit even short-term growth simulations.

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Description and Prediction of Individual Tree Biomass on Piñon (*Pinus edulis*) in Northern New Mexico

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Abstract*—The purpose of this study was to gain reliable information on the distribution of aboveground biomass of an important component of the woodlands of north-central New Mexico, and to develop prediction equations that may be used to quickly compute biomass from relatively simple field measurements. Improved understanding of and ability to predict aboveground biomass distribution in open-grown piñon pine will provide a framework of information that will allow improved understanding of ecosystem processes.*

Introduction

The piñon-juniper woodland forest type encompasses over 22.5 million hectares in the Western United States (Mitchell and Roberts 1999). In New Mexico, the piñon-juniper woodland type encompasses over five million hectares (Mitchell and Roberts 1999). The piñon component in this woodland type in New Mexico is predominately *Pinus edulis* Engelm. (Chojnacky 1994). Since the mid-1800s woody plant encroachment into former grasslands has increased in the southwestern United States (Van Auken 2000). Recent concerns of global warming, more specifically, the potential role of semi-arid woodlands in carbon sequestration, demands a more comprehensive understanding of aboveground biomass distribution in piñon within its natural range. Carbon fixed (sequestered) during the process of photosynthesis has the potential to undergo many fates within a plant. In woody plants, such as piñon, two of these fates are considered long-duration carbon pools—woody tissue and soil carbon pools. While these two fates are not entirely independent, a large portion of the carbon that ends up in the soil carbon pools is originally in the foliage.

Previously published work on the above ground biomass distribution of this species of piñon has primarily been focused on the merchantable (woody) components (Chojnacky 1985, 1988, 1994). However, others have examined both merchantable and non-merchantable aboveground components of the related species, singleleaf piñon (*P. monophylla* Torr. & Frem.) (Miller and others 1981).

The objectives of this study were two-fold; one was to describe the distribution of biomass allocation within individual, open-grown *P. edulis* trees. The second objective of this study was to develop a series of equations that predict component and total tree biomass of open-grown *P. edulis* trees, utilizing efficient field measurements.

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Materials and Methods

Trees used in this study were collected from a private ranch near the town of Guadalupita in Mora County, New Mexico (Lat. N 35.91358º; Lon. W 105.16884º). Trees were growing in a piñon–juniper woodland-grassland interface site. Principle tree species in this woodland type are piñon, oneseed juniper (*Juniperus monosperma* Engelm.), and Rocky Mountain juniper (*J. scopulorum* Sarg.). At lower elevations, the woodland is bordered by a blue grama (*Bouteloua gracilis*) dominated short grass prairie. At higher elevations the woodland is bordered by ponderosa pine (*Pinus ponderosa* Laws.) forest. The project site is situated at an elevation of 2100-2150 m on a sloping (30-40%), predominately southeast-facing slope. Soils found at the study site are classified as a Haplustolls-Rock outcrop complex (Sellnow 1985). The soils are derived from sandstone and shale, which produce a very stony, sandy loam across the site. Depth of soil within the site is variable, ranging from shallow to deep. Exposure of the underlying sandstone is evident throughout the area, and suggests high runoff and water caused erosion potentials.

The climate is semi-arid and annual precipitation is variable, averaging 446 mm annually (Western Regional Climate Center, Ocate, NM; <http://www.wrcc.dri.edu/cgi-bin/cliMAIN.pl?nmocat>). The majority of the precipitation occurs as rainfall in July and August. Mean annual snowfall is 940 mm occurring between October and May with the highest amounts in December and March. Mean annual temperature is 7.9 °C with mean January and July temperatures of –1.1 °C and 18.3 °C, respectively.

Eighteen open-grown piñon trees, age 14 to 33 years old, were selected for inclusion in this study. Open-grown was defined as no other tree canopy within two meters of the select tree. Other tree selection criteria included no observable physical damage, and free of insects and disease. Lastly, selected trees could not be forked within 30 cm of the ground line. Prior to harvesting, several tree attributes were measured for use in model development. These included: root collar diameter outside bark (0.1 cm); mean stem diameter outside bark at 30 cm above ground line (0.1 cm); mean stem diameter outside bark at 100 cm above ground line (0.1 cm); total crown height (0.1 m); height to first fork (if present) (cm); stem diameter outside bark 10 cm above and below base of fork (0.1 cm); crown width north-south axis (0.1 m); and, crown width east-west axis (0.1 m). Observed values are summarized in table 1.

Table 1—Median and range for attributes of 18 open-grown piñon used in this study. (n = 18 except for * where n = 13).

Attribute	Median	Range
Age (years)	18	14 – 33
Height (m)	1.56	0.4 – 3.8
Root Collar Diameter (mm)	65.85	18.9 – 174.6
Diameter @ 30 cm (mm)	43.35	6.0 – 139.8
Diameter @ 100 cm* (mm)	45.15	4.4 – 97.3
Crown Width (m)	1.25	0.3 – 3.5
Fresh Weight (kg)	5.94	0.2 – 118.3
Oven-dry Weight (kg)	3.36	0.1 – 75.6

Mean fresh = 20.24
Mean dry = 12.36

Harvesting was conducted from December 2000 through February 2001 while the trees were dormant. Tissues were separated into the following categories: foliage (needles); branches with diameters less than 0.50 cm; branches with diameters between 0.51 and 2.50 cm; branches with diameters between 2.51 and 5.00 cm; branches with diameters between 5.01 and 10.00 cm; and, branches with diameters greater than 10.01 cm.

Following separation, fresh and oven-dry weights were recorded for each tissue type. Oven-dry weights were obtained by placing tissues into large drying ovens at 120 °C for 24 to 72 hours until a constant weight was achieved. Drying time varied by tissue type.

Prediction Equation Construction

The relationship between field measurements and biomass data was nonlinear. Therefore, variables were transformed logarithmically (base *e*). The transformed variables were then evaluated through the use of forward, reverse, and stepwise regression techniques (PROC REG) (SAS Institute, 1999). The most important predictive variables were found to be total height, diameter at 30 cm, mean crown width, and their interactions. Stem diameter at 1 m was not considered, since more than 25% of the observations were less than 1 m in height.

Equations for each biomass fraction were evaluated for fit based on coefficient of determination (R^2), error mean square, and the *Cp* statistic (Mallows 1972). The precision of the developed equations is measured by percent bias, which is the deviation between observed and predicted observation expressed as a percentage (Faurot 1977). To evaluate the performance of the equations and compare predicted to actual values, predicted values were retransformed into arithmetic units. This process of retransformation may produce bias (Smith 1993), and several suggested correction factors were considered. However, similar to a study by Miller and others (1981), all suggested correction factors increased the bias in all predictions, and were dropped.

Results

The quantities of oven-dry biomass contained in various categories, from foliage to branches greater than 10 cm in diameter, provide a detailed description of structural biomass allocation in piñon trees (table 2). Across all trees sampled,

Table 2—Proportion of above ground biomass allocated to various tissue types and their corresponding moisture contents for eighteen open-grown piñon from northern New Mexico. SE = one standard error of the mean.

Biomass Fraction	Proportion of Total Tree Biomass		Moisture Content	
	(%)	*SE*	*(%)*	*SE*
Foliage	31.6	1.9	48.7	0.6
Branches				
< 0.5 cm	17.7	1.2	41.0	1.2
0.5 – 2.5 cm	31.1	0.9	43.0	0.9
2.5 – 5.0 cm	9.7	0.9	41.8	1.7
5.0 – 10.0 cm	12.4	1.6	38.3	2.4
> 10 cm	10.4	3.2	30.8	1.7
Dead	1.0	0.2	9.1	<0.1

foliage and branches between 0.5 and 2.5 cm in diameter make up the majority of above ground biomass on a dry weight basis constituting 32 percent and 31 percent of the above ground biomass, respectively (table 2). The smallest branch diameter class in this study, less than 0.5 cm, constitutes 18 percent of the above ground biomass, while the three largest branch sizes each constitute between 10 and 13 percent of the above ground biomass. Several of the smaller trees in the study did not contain any branch material greater than 10 cm in diameter, and this contributed to the larger standard error for this category of material (table 2). As trees became larger, the relative proportions of biomass allocated to the different categories became relatively stable (fig. 1). Two exceptions to this was the relationship between foliage biomass and branch biomass for branches between 0.5 and 2.5 cm. Only in the four largest trees did these support branches consistently comprise a larger proportion of aboveground biomass than did foliage. Other exceptions were to be expected, specifically that larger trees had a greater proportion of large branches as well as dead branches than did the smaller trees in the study (fig. 1).

Slightly less than half of the fresh weight of piñon foliage is water (table 2). The three smallest branch classes had similar moisture contents, ranging from 41 percent to 43 percent. As branch diameter increased, percent moisture content decreased to a low of 31 percent for branches greater than 10 cm in diameter (table 2). As expected, deadwood had the lowest moisture content, averaging nine percent moisture content.

All prediction equations included stem diameter at 30 cm and tree height as independent variables, and accounted for a substantial portion of the observed variability found within the data, evidenced by high R^2 values (table 3). All prediction equations developed, except for the prediction equation associated with foliage, also used mean crown width as an independent variable.

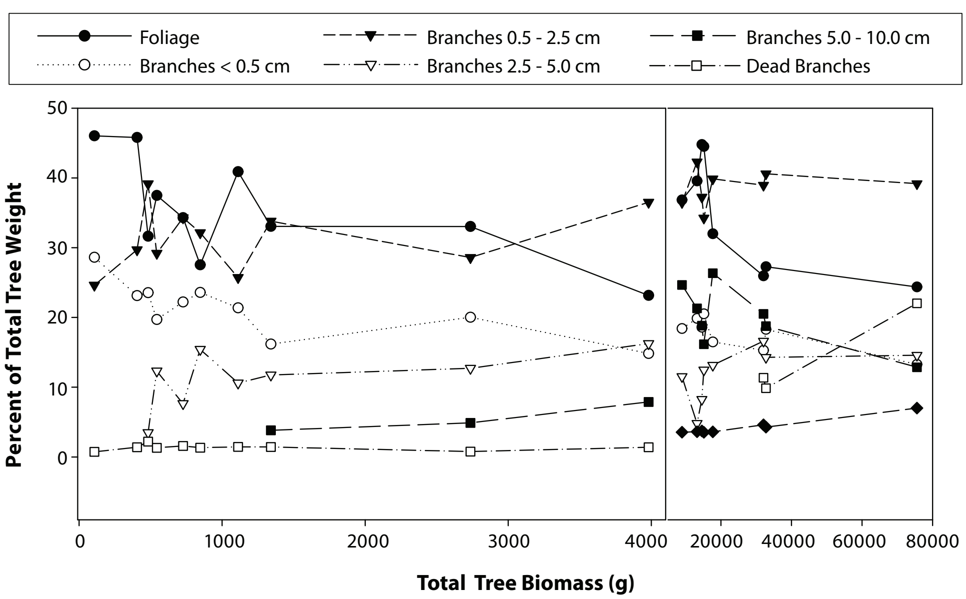

Figure 1—Allocation of above ground oven dry biomass into foliage and four living branch categories based on diameter and dead wood for eighteen open-grown piñon growing in northern New Mexico. (Note: vertical line between total tree biomass values of 4000 and 20000 reflects a gap in the data between the ten smallest and eight largest trees used in this study.)

Table 3—Prediction equations for estimating oven dry weight of the aboveground components of open grown piñon trees in north central New Mexico.

Biomass Fraction	Equation	Adj. R^2	Root MSE	Percent Bias
Total Biomass	WT = 0.179+ 2.111 (D30) + 0.485 (HT * CW)	0.984	0.240	0.334
Foliage	WT = 2.070 + 1.179 (D30) + 0.285 (HT * D30)	0.960	0.335	0.290
< 0.5 cm	WT = 1.594 + 1.214 (D30) + 0.098 (D30 * HT) + 0.144 (D30 * CW)	0.978	0.236	0.317
0.5 to 2.5 cm	WT = -1.122 + 2.152 (D30) + 0.412 (HT * CW)	0.985	0.232	0.373
2.5 to 5.0 cm	WT = 3.136 + 8.863 (HT) + 0.431 (D30) – 4.140 (CW) – 1.717 (HT * D30) + 1.332 (D30 * CW)	0.911	0.512	-0.050
> 5 cm	WT = -10.642 + 4.290 (D30) + 0.026 (HT * D30) – 0.098 (D30 * CW)	0.963	0.346	-0.235

WT = predicted component oven-dry weight (grams) in natural log scale, D30 = natural log of tree diameter at 30 cm on the bole, in mm, HT = natural log of tree height, in meters, and CW = natural log of mean crown width, in meters.

The Root Mean Square Error (MSE) associated with each equation is an indicator of the variability within each sample. The equation with the greatest amount of unexplained variability ($R^2 = 0.911$) is for branches 2.5 to 5.0 cm in diameter. This equation has the highest MSE, slightly greater than 0.5. The next highest MSE values were associated with equations predicting biomass found in the largest branches (> 5 cm diameter) and foliage, 0.346 and 0.335, respectively. The equations for the remaining two fractions and total tree biomass had the lowest MSE values. Percent bias was less than 1% for all equations, with negative values showing a tendency to under predict biomass. Forking was introduced as a qualitative variable and was found to be insignificant in all fractions.

A graphic expression of prediction model accuracy involves plotting the response surface generated against the raw data. This method provides a mechanism to evaluate model applicability across the range of values used to develop the model. As an example, Figure 2 illustrates the response surface of foliage biomass. At low diameter values, increases in height result in only slight increases in predicted biomass, but at larger diameters, increasing height substantially increases predicted biomass.

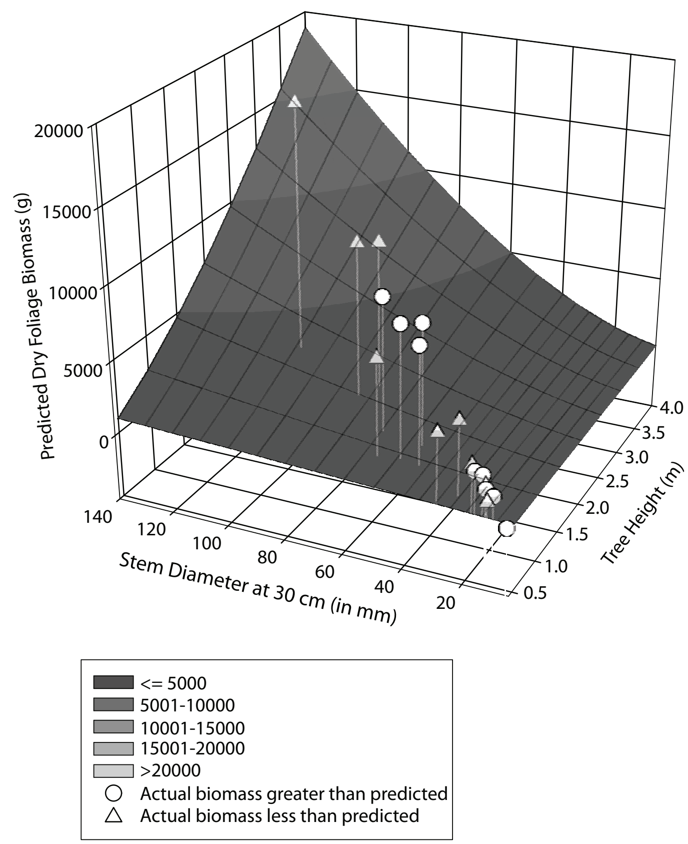

Figure 2—Prediction equation response surface for dry foliage biomass and actual individual tree foliage biomass measures.

Discussion and Conclusions

Study-wide, foliage biomass represented the largest component of above ground biomass (32 percent); however, foliage amounts became proportionally less (46 percent to 22 percent) as trees became larger. This trend was also seen in a similar component analysis of singleleaf piñon (Miller and others 1981). A possible explanation for the foliage component in the present study accounting for a greater amount of the above ground biomass compared to other published reports, may be that only open grown trees were sampled. In the Miller and others (1981) study, no such constraint was required for tree selection. The open grown selection criteria resulted in the live crown of the sample trees being lower to the ground than would be found in trees growing in a denser stand. In denser stands, self-pruning and short needle retention duration occur. The estimate of the proportion of above ground biomass allocated to foliage in the current study is also considerably higher than that published for piñon growing in piñon-juniper stands (*P. edulis-Juniperus monosperma*; Grier and others 1992).

The smallest wood category partitioned in this study, branches with diameters less than 0.5 cm, constitute a small portion of the above ground biomass despite being the point of contact from which the foliage arises. Branches in the next larger size class (0.5 to 2.5 cm diameter) provide the support and arrangement for the photosynthetic tissue. Despite using different branch diameter categories, the proportion of biomass in the two smallest branch classifications (< 2.5 cm) presented in the current study are slightly higher (48 percent versus 32 percent) than previously published values for singleleaf piñon (Miller and others 1981 [smallest two tree categories only]). Again, the sampling criteria of open grown trees, as well as the higher proportion of smaller trees in the current study may be contributing to the higher proportion of biomass allocated to tissues associated with supporting foliage.

Management Implications

Improved understanding of the above ground biomass distribution in open-grown piñon, such as that generated in this report, will provide necessary information that will allow a better understanding of ecosystem function in semi-arid woodlands. The information generated in this study in addition to information on other above and below ground biomass pools should lead to both an improved understanding of energy and carbon flows in these ecosystems.

Estimations of volume and biomass in piñon have historically been difficult. Chojnacky's (1985) equations for estimating volume in singleleaf piñon (*P. monophylla* Torr. & Frem.) in the central Rocky Mountain States contained error rates well above 25 percent for specific size classes found in certain regions. It is the great physical variability exhibited by this species throughout the Western United States, which results in the dilemma of whether to create wide-ranging equations with unsatisfactory error rates, or localized equations of greater accuracy that may prove even less accurate when applied to other areas.

In 1994, Chojnacky published total volume equations for piñon in New Mexico. Compared to his 1985 study, the New Mexico equations exhibited good fit, with high R^2 values slightly greater than 0.9. However, it was reported that the New Mexico equations predicted substantially greater volume than those developed for Colorado and Arizona.

In this study, percent bias in all equations was less than 1 percent. The smallest bias was found within the two largest biomass fractions (2.5 to 5.0 cm and > 5.0 cm). There was less variability for observations within these fractions, as only the largest trees within the sample contained biomass of these diameters. Similar equation development by Miller and others (1981) yielded bias estimates ranging from 1 to 5.3 percent. The trees within that study were singleleaf piñon growing in Utah, with a larger range of heights compared to the current study (from a mean of 2.0 m in the smallest diameter class, to over 9.0 meters tall in the largest class).

Overall, the prediction equations perform well and describe a large proportion of the variability within the data. These equations are reliable predictors of biomass for piñon trees within this and similar regions. It should be noted that the equations should only be utilized on piñon within the same size range as the study, mainly trees approximately 4 m in height. Trees larger than 4 m will likely have their biomass overestimated by the equations, as the response surface shows a rapid increase.

Acknowledgments

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A Native Plant Development Program for the Colorado Plateau

Stephen B. Monsen[1]

Abstract—*Revegetation programs instigated in the Intermountain West have relied on the use of introduced perennial grasses, with attention focused upon improving the agronomic and forage attributes of these species. Currently, a wide number of site adapted native species are required to restore the extensive disturbances throughout the West including the Colorado Plateau. Through cooperation with Federal and State agencies and Universities in Colorado and Utah studies have recently been initiated to identify the principal species associated within pinyon/juniper and sagebrush communities of this region. Studies are designed to evaluate ecotypic variability and ecological adaptation of individual taxa. Concurrently, seed production studies are being established to determine culture requirements to produce, harvest, and process seed for large-scale restoration projects. In addition, seed germination and seedbed ecology studies are being developed to reassemble multiple species including interseeding into areas where retention of some species is desired. The progress and status of individual studies will be reported.*

In: Gottfried, Gerald J.; Shaw, John D.; Ford, Paulette L., compilers. 2008. Ecology, management, and restoration of piñon-juniper and ponderosa pine ecosystems: combined proceedings of the 2005 St. George, Utah and 2006 Albuquerque, New Mexico workshops. Proceedings RMRS-P-51. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

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Historic Vegetation Changes in Lincoln County, New Mexico: The Albuquerque Banquet Presentation

E. Hollis Fuchs[1]

***Abstract**—Repeat photography will demonstrate that since European settlement commenced, the native vegetation of Lincoln County, New Mexico has dramatically changed. Numerous historic photographs have been re-taken, demonstrating how landscapes and ecosystems have changed, not just between early European settlement until the present, but also in the intervening years. Photographs will be seen from as early as 1868, as recently as September 2006, and the years in-between. While the program will focus on ecosystems dominated by piñon and juniper and the implications of the increase of these species, other vegetation types will also be examined. An unusual feature of the repeat photography is that some of the specific photo sites include images from the intervening years, with photo sets that are not limited to a single pair of comparison images. Thus, changes in vegetation over time can be observed in order to better understand the timing and possible influences that have resulted in unmistakable changes in the appearance of Lincoln County landscapes.*

In: Gottfried, Gerald J.; Shaw, John D.; Ford, Paulette L., compilers. 2008. Ecology, management, and restoration of piñon-juniper and ponderosa pine ecosystems: combined proceedings of the 2005 St. George, Utah and 2006 Albuquerque, New Mexico workshops. Proceedings RMRS-P-51. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

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Insects and Diseases

Photo by Neil Cobb

Drought Induced Tree Mortality and Ensuing Bark Beetle Outbreaks in Southwestern Pinyon-Juniper Woodlands

Michael J. Clifford[1], Monique E. Rocca[2], Robert Delph[1], Paulette L. Ford[3], and Neil S. Cobb[1]

Abstract—*The current drought and ensuing bark beetle outbreaks during 2002 to 2004 in the Southwest have greatly increased tree mortality in pinyon-juniper woodlands. We studied causes and consequences of the drought-induced mortality. First, we tested the paradigm that high stand densities in pinyon-juniper woodlands would increase tree mortality. Stand densities did not impact mortality levels for either tree species, which does not support tree thinning to reduce susceptibility to drought-bark beetle outbreaks. Second, we monitored changes in stand structure and dead woody biomass to test whether altered fuel loads might affect potential fire behavior. Mortality can significantly affect torching index but has little effect on crowning index. Finally, we predicted that ground-dwelling arthropods would be highly responsive to habitat changes resulting from dead trees. Although we found significant responses, they were not as strong as predicted. Together, these results suggest that impacts of drought-induced tree mortality may not appear for years or decades after a major mortality event.*

Introduction

The southwestern United States has experienced drought in 9 of the past 10 years, with an extreme drought occurring in 2002 (NOAA 2002). Impacts of these drought conditions have been magnified in the semi-arid areas. Current and recent extreme drought conditions, coupled with increased regional temperatures, lead to stressed pinyon pines (*Pinus edulis*), which were then more susceptible to ips beetles (*Ips confusus*) (Breshears and others 2005). The combination of these three factors caused mortality among pinyons as high as 80 percent in many areas throughout the region (Breshears and others 2005; Shaw and others 2005; fig. 1). Drought and water-stressed host trees have been shown to lead to eruptive insect herbivore outbreaks in the past (Waring and Cobb 1992). With the relatively high susceptibility to climatic variations, pinyons have acted as "barometers of change" in Southwest ecosystems (Gitlin and others 2006; Ruel and Whitham 2002). Previous droughts in Southwest ecosystems have resulted in vegetation changes and shifts of ecotones (Allan and Breshears 1998). Outside the Southwest, climate change effects on vegetation have also been documented (Bigler and others 2006, Fensham and Holman 1999). Current global climate projections indicate further increases in temperatures and more extreme climatic events, such as drought (Easterling and others 2000; Hoerling and Kumar 2004; Intergovernmental Panel on Climate Change 2001). Likely scenarios include increased insect outbreaks (Logan and others 2003), elevational up-shifts of vegetation (Dullinger and others 2004), and increased frequency of extreme climatic events (Easterling and others 2000).

In: Gottfried, Gerald J.; Shaw, John D.; Ford, Paulette L., compilers. 2008. Ecology, management, and restoration of piñon-juniper and ponderosa pine ecosystems: combined proceedings of the 2005 St. George, Utah and 2006 Albuquerque, New Mexico workshops. Proceedings RMRS-P-51. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

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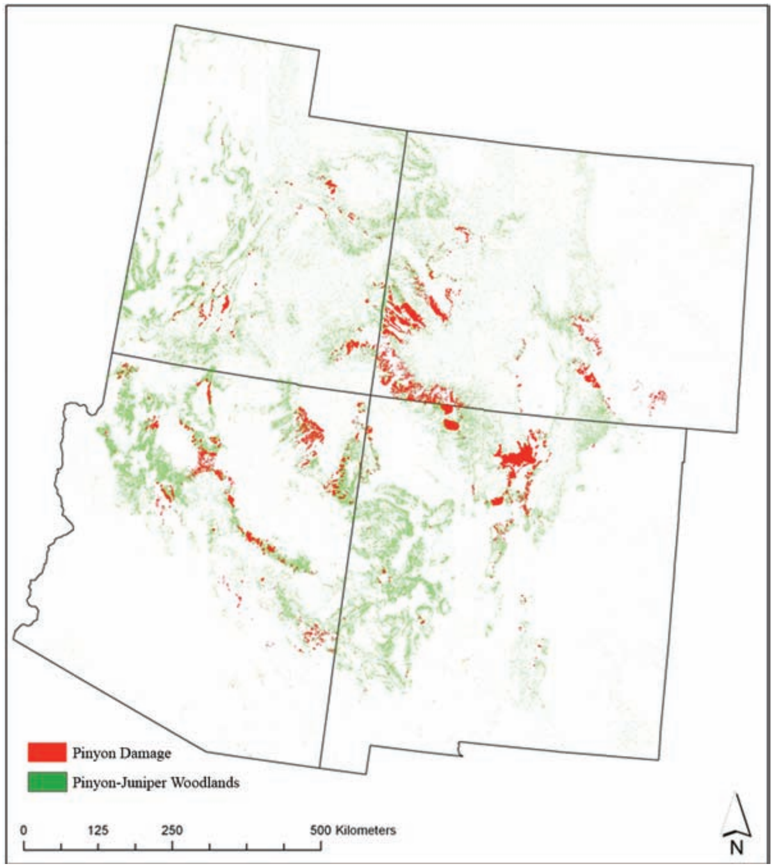

Figure 1—Forest Health Enterprise Team, USDA Forest Service aerial survey of pinyon damage in the Southwest from 2000-2005.

Pinyon-juniper woodlands are the third largest vegetation type in United States and cover over 19 million ha in the Southwest (Evans 1988). These woodlands are used for livestock grazing and wood harvest, and are culturally important to many Native American tribes. Current woodlands in many areas are denser than they were prior to Euro-American settlement. High density woodlands can increase tree stress from competition between trees, thus promoting increased disease susceptibility (Negron and Wilson 2003). Several studies examining pinyon stress and insect herbivore responses show that increases in environmental stressors alter tree-herbivore dynamics (Brown and others 2001; Cobb and others 1997; Waring and Cobb 1992). Expansion of pinyon-juniper woodlands since Euro-American settlement has produced efforts to reduce or remove pinyons and juniper in order to restore woodlands back to pre-settlement conditions (Brockway and others 2002; Ffolliott and Gottfried 2002).

Pinyon-juniper woodlands in northern Arizona and north-central New Mexico were surveyed. In each area, the majority of pinyon mortality occurred from the drought of 2002 (Breshears and others 2005). In these areas, we focused on several issues associated with causes and consequences of tree die-off. First, shifts in stand structures, needle retention, and tree fall of dead pinyons were documented while testing the paradigm that increased stand density will promote tree mortality. Negron and Wilson (2003) showed a positive relationship between pinyon death and an increased stand density index at a smaller, local scale before the major ips beetle outbreak of 2002. This paradigm was examined at the landscape scale after the major drought and pinyon die-off. Second, with changes in stand structure and increased dead woody debris throughout the woodlands, we hypothesized that fire dynamics would be altered in areas of high pinyon mortality. Third, after mortality of pinyons, ground-dwelling arthropod communities were expected to respond to altered habitat at both the population and community levels. By examining these hypotheses, we expect to quantify the response of several key woodland characteristics to a major co-dominant vegetation die-off. These results will assist in evaluating alternative management scenarios and will help to understand ecosystem responses to a future climate.

Methods

Study Sites

New Mexico—Study areas were located in the Middle Rio Grande Basin (MRGB) in north-central New Mexico. We created two types of sites in New Mexico: 1) intensive uniform study sites located in the north, central, and southern areas of the MRGB, and 2) random vegetation sites located throughout the MRGB. The intensive uniform sites were located in a grid pattern and separated into high and low pinyon mortality sites for each area. High and low mortality sites were relative to the mortality throughout the area. High mortality sites were determined to have visually higher mortality than low mortality sites. Each grid within the high and low mortality sites was 100 x 200 m, which was originally designed for 32 evenly spaced ground-dwelling arthropod pitfall traps. These sites were surveyed in 2005, tree status (in other words, alive, dead, standing, downed) and fuels data were collected in a 100 m^2 area around 16 of the 32 pitfall traps at each site. The random vegetation sites were located throughout the MRGB and were surveyed in 2005 and 2006. Each site was placed >5 km from another site. A site consisted of three 100 m^2 plots placed 75 m apart forming a triangular shape.

Arizona—The Arizona study area consisted of 52 sites located in northern Arizona. These sites were 200 x 10 m and split into 100 m^2 plots. Half of the plots were randomly selected for survey and were surveyed annually from 1998 to 2004. Sites were in pinyon-juniper woodlands at varying elevations, ranging from the upper to lower ecotones, and are representative of woodlands throughout the Region (Floyd and others, submitted).

Stand Structures

At each site in New Mexico (n = 53), all trees were measured for basal trunk diameter (BTD, also called root collar diameter), crown height and width, crown base height, and tree status. The documentation of spatial extent and intensity of pinyon die-off in the MRGB was determined by the random vegetation sites. These sites were also used to determine altered stand structures and shifts in age structures of pinyons, and the projected fall of pinyon snags. To test the effect of stand density on pinyon mortality, we used data from high and low mortality areas of the uniform plots. For this comparison, a one-way ANOVA was performed in SPSS (Version 14.0, 2005).

Arizona sites were surveyed from 1998 to 2004. The status of each tree was collected and the percent of needles remaining on dead pinyons was estimated. Estimates of percent crown foliage remaining were used to determine needle retention of dead trees.

Fire Model

Dead and down woody debris (in other words, fuel load measurements) were taken on all intensive uniform sites in each region of the MRGB, New Mexico, using a transect running north to south (see Brown 1974). Estimates of dead and down woody fuel loads were made for high and low mortality stands according to the procedure outlined in Brown and others (1982). Custom fuel models were developed based on average fuel loads for the high mortality and low mortality stands. Percentile weather scenarios were generated based on weather records from the Pecos RAWS, which were analyzed using the FireFamilyPlus software package to generate local estimates for a range of percentile weather conditions. Using the fire behavior program NEXUS (Scott 1999), fire behavior was simulated at the stand scale for high and low mortality stands under a range of percentile weather scenarios.

Arthropod Sampling

To test differences in the ground-dwelling arthropod community between high and low levels of pinyon mortality, pit-fall traps were used to capture arthropods. Pitfall traps (25 mm diameter) included Brosilicate glass test tubes encased by SDR 35 material PVC pipe, complete with a PVC lid to detour rain and debris from falling in the trap. Each test tube was filled with a 1:1 dilution of water and propylene glycol. We established 100 x 200-m grids within each sample area where 32 small pitfall traps were placed on a uniform grid at 35 m intervals.

Arthropod communities were analyzed using nonparametric multi-dimensional scaling (NMDS). This ordination shows differences or similarities between community structures in high and low mortality areas of the uniform sites in New Mexico. Indicator species analysis was also used to determine whether any arthropod species or groups showed a positive association with high or low pinyon mortality areas.

Results

Spatial Patterns of Mortality

Using random vegetation sites throughout the MRGB, we mapped the spatial extent and severity of tree mortality. The current patterns of pinyon mortality in the MRGB show a latitudinal gradient of mortality (fig. 2). In the southern portion (for example, Manzano Mountains) of the study area, pinyon mortality was relatively low (1.2 percent) compared to the central and northern areas (47 percent and 62 percent, respectively). The cause of this latitudinal gradient is unknown, as stand structures throughout the MRGB were similar. Perhaps this pattern can be attributed to associated susceptibility and origin of the initial ips outbreak. If the northern and central areas had a naturally higher ips population before 2002 than the southern area, there may not have been as eruptive outbreak in this area.

Stand Structures

Stand density did not significantly affect pinyon mortality across sites in New Mexico ($P = 0.281$). The high and low mortality sites did not have significantly different stand densities, but did have significantly different levels of pinyon mortality ($P < 0.0001$) (fig. 3) validating that visual observations were correct in classifying high and low mortality sites. This is evidence that increased stand densities do not predispose woodlands to tree mortality and therefore does not support the common paradigm at these scales. These results support the findings of Floyd and others (submitted),

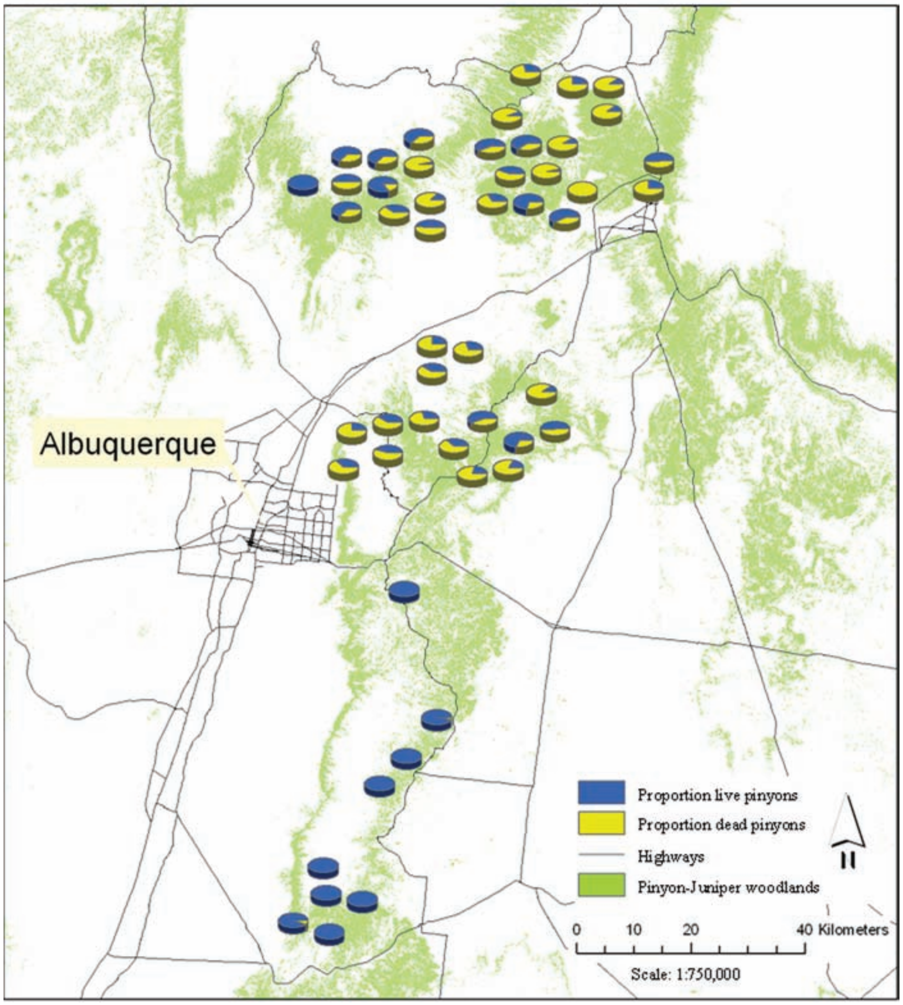

Figure 2—Pinyon mortality in the Middle Rio Grande Basin, New Mexico. The northern (62 percent) and central (47 percent) areas have much higher mortality of pinyons when compared the southern areas (1.2 percent).

which used linear regressions to explore the relationship of stand density and pinyon mortality. Perhaps, under outbreak conditions, traditional stressors (in other words, increased stand densities) are less important in tree mortality than insect population dynamics or landscape scale processes.

The stand structures of the pinyon-juniper woodlands were altered with the high mortality of pinyons. There was an overall shift toward a younger population of pinyons, as the larger, mature pinyons sustained a large loss (fig. 4A). The overall stand structures of the woodlands were altered with the reductions in basal area of pinyons. There remains a component of mature trees in these woodlands due to the retention of junipers (fig. 4B). Reductions in pinyon populations will alter future stand structures, and the current snags and dead trees will continue to alter and shape future stand structures. After 4 years, the majority of dead trees are still standing. It is expected that it will take an additional 7 years for all snags to fall (fig. 5). The effect of the falling snags will likely continue to alter fire regimes and ground-dwelling arthropod population dynamics. We also found that it takes an average of 9 months after mortality for pinyons to lose their needles (fig. 6). This confirms the observations of many others in the Southwest and indicates that for pinyon-juniper woodlands, extreme fire danger from dead retained needles occurs at approximately 9 months after tree death.

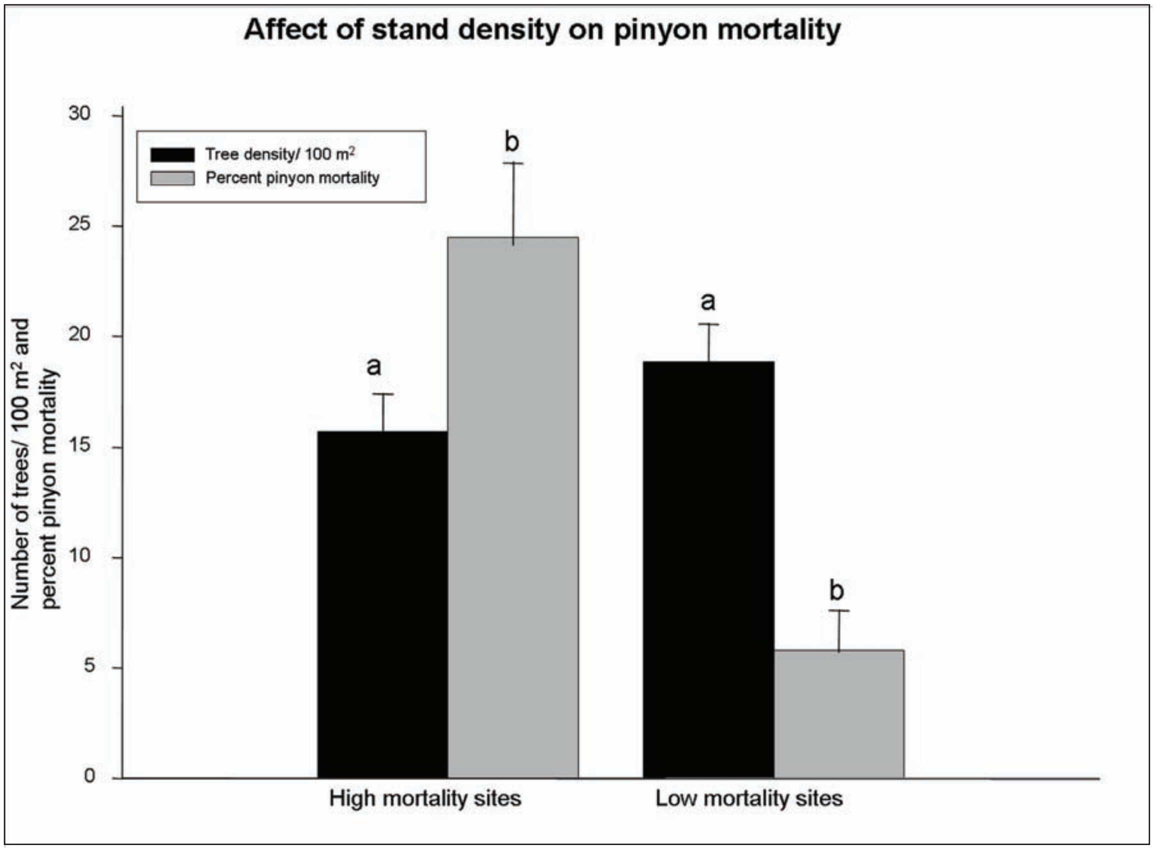

Figure 3—Stand density did not significantly affect pinyon mortality ($P = 0.281$). The high mortality area had significantly more pinyon die-off than the low mortality area ($P < 0.0001$). An "a" indicates non-significance, while a "b" is significant.

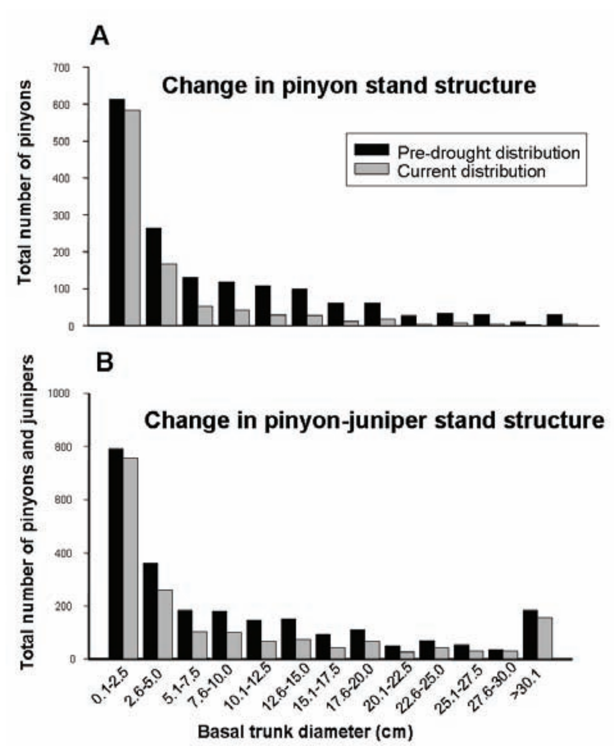

Figure 4—Pinyon stand structure, based on basal trunk diameters, showed severe reduction in larger, reproductive trees (A). When compared to total stand changes (which include junipers), there was still a large loss of larger trees in the population (B).

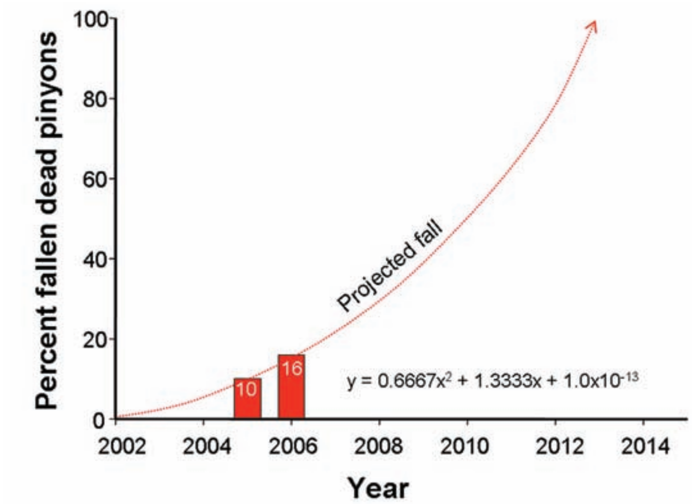

Figure 5—Projected fall of pinyon snags in the MRGB, New Mexico.

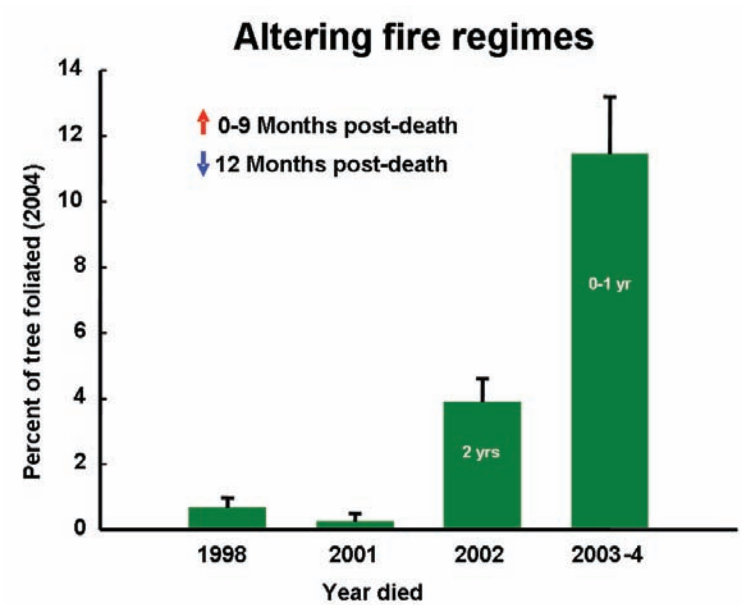

Figure 6—Time of needle retention for dead pinyons in Arizona. Needles were retained for an average of nine months.

Fire Modeling

The fire dynamics in pinyon-juniper woodlands were altered in areas of high pinyon mortality (fig. 7), supporting our hypothesis. Torching and crowning indices capture the severity of weather conditions required to initiate and propagate crown fire, respectively. The torching index is the windspeed (as measured 20 ft above the ground) that will cause the fire to be intense enough to begin to torch individual trees from below (passive crown fire). The crowning index is the windspeed that will cause the fire to spread from tree to tree (active crown fire). Moisture conditions below the 9th percentile will not carry fire (fig. 7).

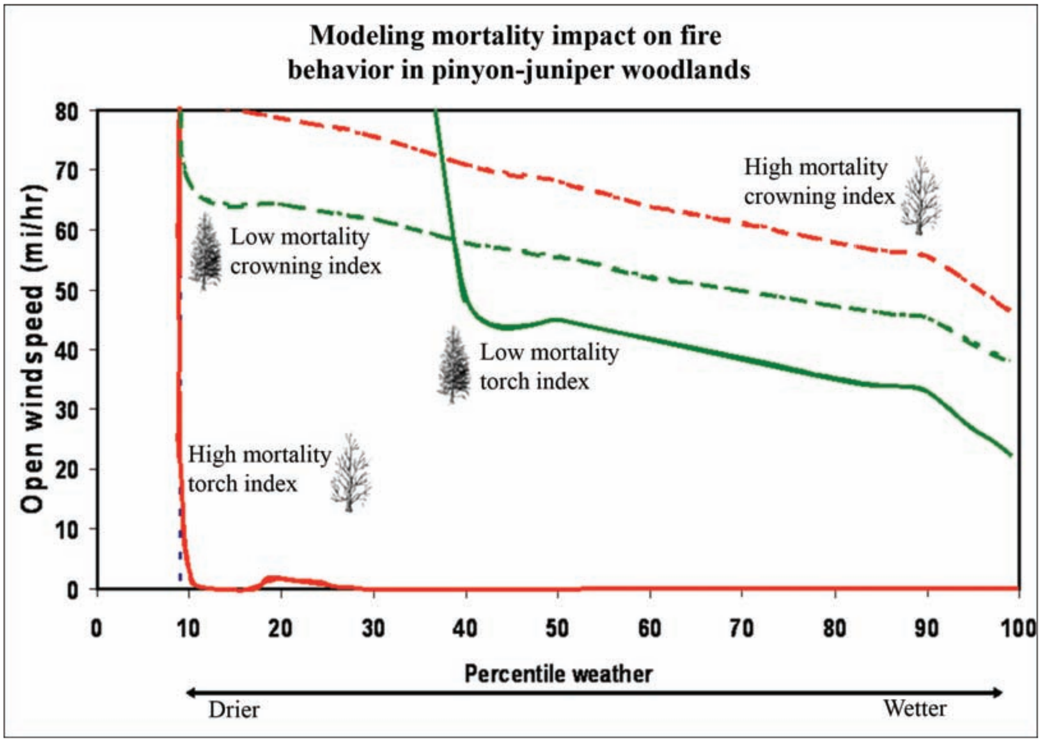

Figure 7—Model showing probability of fire start (torching index) and probability of a crown fire (crowning index) in high and low mortality sites of New Mexico.

Our models predict that fire ignitions will lead to surface fire behavior for the low tree mortality stands under most moisture conditions and for wind speeds up to 22 mph on the driest days. Higher winds will cause tree torching (passive crown fire), which will easily transition to active crown fire due to a relatively dense canopy structure. Due to higher surface fuel loads, high mortality stands cannot carry a surface fire under any weather conditions (fig. 7) without exhibiting torching fire behavior. However, the loss of much of the available canopy fuel (needles and twigs on tree canopies) has raised the crowning index.

Arthropod Indicators

It was hypothesized there would be a strong community response in areas of high pinyon mortality. Data indicate there are differences at the community level between high and low areas of pinyon mortality on the north and central sites, but not at the south site (fig. 8). The southern site had relatively low mortality, even in the "high" mortality areas, when compared to the north and central sites. This lack of high mortality, relative to other sites, may help to explain the lack of community response of ground-dwelling arthropods at this site. These community responses are not as strong as predicted, which may be due to a high number of pinyon snags that have not fallen. There were 15 indicator species (table 1), in high and low mortality areas, supporting our hypothesis. This shows there are responses at the species level and not just an overall community response to increased pinyon mortality.

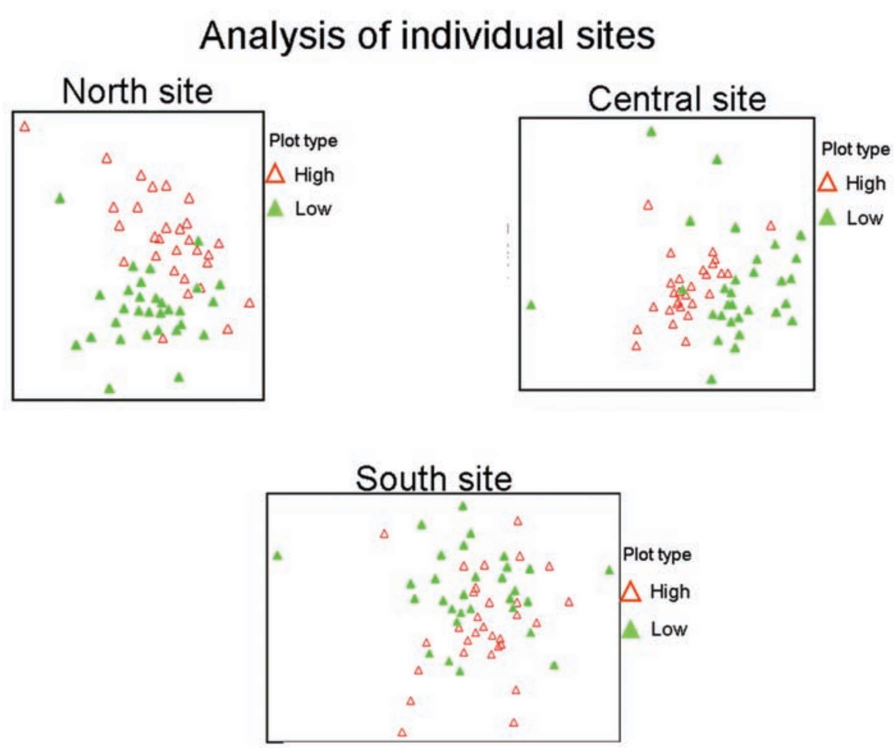

Figure 8—The scatter plots represent patterns of similarities in community structure. The affects of high and low tree mortality are the environmental variables influencing the community structure based on arthropod species abundance. Both north and central sites show strong grouping, more similar within groups than between groups of arthropod communities specific for high or low pinyon mortality areas. The south site did not show any groupings favorable for high or low pinyon mortality areas.

Table 1—List of arthropod indicator species for high and low areas of pinyon morality. Six taxa favored high pinyon mortality areas while 9 taxa favored low pinyon mortality areas. A "1" indicates presence.

			Central		North		South		Significance (α = 0.05)	
Order	Family	Species	High	Low	High	Low	High	Low	High	Low
Collembola	Hypogastruridae	COLE HYPO 001	1	1		1	1	1		0.002
Collembola	Entomobryidae	COLE ENTO 001	1	1	1	1	1	1	0.009	
Orthoptera	Gryllidae	*Gryllus* sp.		1	1	1	1	1		0.001
Coleoptera	Carabidae	*Pasimachus obsoletus*		1		1				0.001
Coleoptera	Tenebrionidae	*Eleodes obscurus*						1		0.007
Hymenoptera	Sphecidae	*Ammophila* sp.	1	1	1	1	1	1		0.042
Hymenoptera	Mutillidae	*Dasymutilla vestita*	1	1	1	1	1	1	0.001	
Hymenoptera	Formicidae	*Crematogaster depilis*	1	1	1	1	1	1	0.031	
Hymenoptera	Formicidae	*Monomorium cyaneum*	1	1	1	1	1	1	0.018	
Hymenoptera	Formicidae	*Solenopsis molesta*	1	1			1	1		0.003
Hymenoptera	Formicidae	*Liometopum apiculatum*	1				1	1	0.002	
Hymenoptera	Formicidae	*Camponotus acutirostris*	1	1	1	1	1			0.001
Aranea	Lycosidae	*Paradosa orophila*	1	1	1	1	1	1	0.025	
Acari	Erythraeidae	ACAR ERYT 001	1			1	1	1		0.047
Acari	Erythraeidae	ACAR ERYT 002	1	1	1	1	1	1		0.001

Discussion

Several aspects of pinyon mortality were documented, from potential causes of mortality to effects of mortality. First, stand densities did not influence pinyon mortality. Second, changes in stand structure altered fire dynamics and ground-dwelling arthropod communities. The die-off of a co-dominant woodland species throughout the Southwest has and will continue to alter and affect many ecological processes (in other words, fire cycle, carbon cycle, etc). Stand structures throughout the pinyon-juniper woodlands have become denser since European settlement (Ffolliott and Gottfried 2002). After 2002, these woodlands have become less dense with respect to living trees. This ips beetle-induced thinning of pinyon-juniper woodlands may indicate climatic changes or extreme events, serving as a natural thinner in a natural oscillation of woodland structures and vegetation dynamics (Allen and Breshears 1998).

Stand Structure

Regional die-off of pinyons has affected stand structure and age structure of pinyons, shifting the population toward younger, non-reproductive trees (fig. 5). The loss of many reproducing trees and reduced pinyon nut crops will likely affect granivores such as pinyon jays (*Gymnorhinus cyanocephalus*), pinyon mice (*Peromyscus truei*), and other fauna associated with pinyon-juniper woodlands. There will also be general changes in woodland ecosystem dynamics with upwards of 80 percent tree mortality. These include responses of understory plant species to the loss of canopy cover and shading with increased solar radiation and altered spatial distributions of soil moisture (Breshears and others 1997; Breshears 2006).

In both high and low mortality areas, stand density did not affect pinyon mortality. These results contradict the current paradigm that suggests higher density stands are more stressed and are therefore predisposed to increased mortality (Negron and Wilson 2003). The temporal context of this study differed from that of previous studies, which were conducted before the majority of pinyon die-off. The large spatial extent of our study may also explain our failure to detect a density effect on mortality. The paradigm of Negron and Wilson (2003), who's study focused on scales <10 km^2, could account for local patterns, but may not explain regional patterns of tree mortality. The severity, longevity, and increased temperatures of the current southwestern drought may have stressed trees to the point where nearly any susceptible tree, specifically pinyon, was attacked by ips beetles.

Fire Modeling

With shifts in stand structures and increases in dead woody fuel load, potential fire dynamics have changed 2 years post mortality. Less severe wind and moisture conditions are needed for crown fire initiation in high mortality areas whereas, in low mortality areas, active crown fire occurs at a lower windspeed (fig. 3). In low mortality areas, the live tree canopy has greater density of available fuel as compared to the high mortality areas. In other words, 2 years after pinyon mortality, the high mortality stands have been effectively thinned.

Associated with this "canopy thinning" are 1) a re-distribution of canopy fuels to the forest floor, and 2) a lowering of the average canopy base height of remaining live trees. A lower canopy base height is most likely an artifact of the increased relative abundance of junipers compared to pinyons after the beetle-kill event. Both factors will lower the torching index within the model framework. Yet the lower canopy base height is not "real" in the sense that the smaller surviving trees have not changed their canopy structure as individuals even though the average canopy base height has lowered. The increased surface fuel loads, however, would be expected to cause a more intense surface fire that will make torching behavior more likely for any canopy base height.

Pinyon mortality has altered potential fire dynamics in the Southwest, but the probability of a catastrophic fire due to high levels of dead trees is unlikely. For a crown fire to occur under the driest conditions (100 percentile), it would take an open wind speed of 50 mph in an area of high mortality for crown fire to be sustained.

There have been mixed results of bark beetle affects on fire regimes in many areas (Kulakowski and others 2003; Bigler and others 2005), but the mortality of pinyons was as high as 80 percent in some areas (Breshears and others 2005). The fire dynamics in pinyon-juniper woodlands have changed with the loss of canopy overstory and the addition of dead woody material on the woodland floor. Four years after the majority of the mortality event, most dead pinyons are still standing, creating a partial overstory of dead material. These new and changing characteristics have altered the probability that a fire is started and the way it will be carried in pinyon-juniper woodlands.

Arthropod Indicators

Many arthropod species (n = 15) showed either a positive or negative response to pinyon mortality (table 1). As indicator species, these arthropods will show a shift at the species level in areas of high and low pinyon mortality. Arthropod population dynamics are important for many ecosystem processes, and in current areas with high mortality there have already been responses. These responses were weaker than the predicted response, but can possibly be explained by the high amount of dead pinyons that remain standing (approximately 84 percent) (fig. 5). If these trees do not fall, then potential habitat for ground-dwelling arthropods will not be created. At the community level, we begin to show a shift in arthropod communities between differing areas of mortality (fig. 8). This community difference should increase as dead trees fall to the ground; however, the duration of this event may take another 6 years to complete (fig. 5).

Management Implications

Mortality of dominant species in major vegetation types has occurred throughout the world and continues to occur in many regions (Breshears 2006; Bigler and others 2006; Gitlin and others 2006; Logan and others 2003; Shaw and others 2005). Mortality of regionally dominant species, such as pinyon pine, suggests that climate change through increasing temperatures and frequent drought events, coupled with increased insect herbivore populations, will increase the probability of future mortality events (Breshears and others 2005). These continuing climatic conditions may alter regional vegetation and promote ecotone shifts (Allen and Breshears 1998; Risser 1995). With changes in dominant vegetation, many ecosystem processes are altered (in other words, biogeochemical processes, fire regime, ecohydrology) from local to regional scales (Breshears 2006), but the extent and intensity of these climatic driven events is still poorly understood.

In the pinyon-juniper woodlands of the Southwest, the full extent and impacts of this ongoing drought have yet to be quantified, but the effects are expected to last for decades or centuries. With the changes in stand structures and ips-induced thinning of Southwestern pinyon-juniper woodlands, mechanical treatment of woodlands considered "unhealthy" because of high tree densities will likely not be necessary. Bark beetles have shown spatially explicit patterns at the stand level and landscape level (Negron and others 2001), suggesting different spatial scales should be used to assess bark beetle induced tree mortality. The rearrangement of fuel loads may warrant management consideration, but the rapid rearrangement of canopy fuels to the forest floor appears more likely to reduce the likelihood of rapidly moving, wind-driven canopy fire rather than exacerbating it.

For management purposes, the frequency and severity of drought are more important than long-term average climate conditions. Too often land managers plan for average climate conditions rather than the climatic extremes that can be expected (Potter and Ford 2004). The future management of pinyon-juniper woodlands must take into consideration predicted extreme climatic events. Southwestern ecosystems, especially the semi-arid woodlands, are susceptible to these extreme events (Allen and Breshears 1998; Breshears and others 1997; Breshears and others 2005; Ruel and Whitham 2002; Swetnam and Betancourt 1998) and may act a barometer for climatic change.

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Piñon Pine Mortality Event in the Southwest: An Update for 2005

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Abstract*—Drought conditions in the Southwest have persisted for a number of years resulting in large areas of piñon pine mortality. In 2002 drought conditions became extreme, facilitating an outbreak of piñon ips beetles (*Ips confusus*, Coleoptera: Scolytidae) that killed many millions of piñon pines over a six-state region by 2003. In response to this unprecedented mortality, Forest Health Monitoring provided funds to Forest Health Protection units to measure the extent and intensity of the event. These impacts are being documented through an extensive network of permanent plots that were installed throughout much of the range of piñon in the southwestern United States in 2003 and 2004.*

The results from this project reveal that the situation on the ground is a quickly changing phenomenon. While the huge amounts of mortality seen in 2003 were caused by the aforementioned piñon ips, mortality throughout much of the affected area decreased sharply in 2004. In addition, a number of other damaging agents played a more prominent role in the damage that did occur. Because conditions varied widely throughout the range of piñon pine, brief summaries from each of the participating Forest Health Protection units are presented.

Attributes Associated With Probability of Infestation by the Piñon Ips, *Ips confusus*, (Coleoptera: Scolytidae) in Piñon Pine, *Pinus edulis*

José F. Negrón[1] and Jill L. Wilson[2]

Abstract*—We examined attributes associated with the probability of infestation by piñon ips* (Ips confusus)*, in piñon pine (*Pinus edulis*), in an outbreak in the Coconino National Forest, Arizona. We used data collected from 87 plots, 59 infested and 28 uninfested, and a logistic regression approach to estimate the probability of infestation based on plot- and tree-level attributes. Piñon pine stand density index was a good predictor of the likelihood of infestation by piñon ips at the plot level and a cross-validation analysis indicated that the model correctly classified 82% of the cases. Diameter at root collar and piñon dwarf mistletoe infestation level were good predictors of individual tree infestation and a cross-validation analysis indicated that the model correctly classified 72% of the cases. Results suggest that the occurrence of piñon ips infestations may be related to stress factors associated with increased stocking and piñon dwarf mistletoe infestations.*

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Piñon Mortality from 2001 to 2005: Causes and Management Strategies

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Abstract*—Piñon mortality in the piñon-juniper and piñon-sage types of the Southwest peaked in 2003 following several years of winter drought. The majority of the drought-weakened trees died from piñon ips bark beetle attacks, but twig beetles also played a role. Forest Service aerial surveyors estimate more than 50 million piñon trees died in New Mexico alone from 2001-2005, most likely a conservative estimate since understory trees cannot be seen from the air. Unprecedented densities, advancement of piñon into drier sites at the low end of its elevation range, and multiple years of drought have made this period of die-off particularly notable. There is considerable evidence from the paleobotanic record that the range of piñon had reached its maximum extent following a comparatively wet period in the 1960s and 1970s, but there is also information that suggests piñon has always colonized and retreated from large areas over time. While natural events such as drought cannot be controlled, some management strategies can ease the adverse impacts on lands where the piñon component is valued. These management strategies must be tailored to address the specific damaging agent from the guild of organisms that are associated with piñon and juniper.*

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Do Bark Beetle Sprays Prevent *Phloeosinus* species from Attacking Cypress and Juniper?

Chris Hayes[1], Tom DeGomez[1], Karen Clancy[2], Joel McMillin[3], John Anhold[3]

Abstract—Phloeosinus-*caused mortality of Arizona cypress,* (Cupressus arizonica), *oneseed juniper,* (Juniperus monosperma) *and alligator juniper,* (J. deppeana) *has been observed at high levels in Arizona during the past 3 years. Currently, there are limited preventative measures to protect high-value cypress and juniper trees against* Phloeosinus *attack. Insecticides that are being used to prevent bark beetle attack of individual high value pine trees may provide one method of protection.*

*This study was designed to determine the efficacy of Sevin SL (carbaryl), Permethrin Plus C (permethrin), and Biflex (bifenthrin) in protecting Arizona cypress and oneseed juniper from attack and colonization by cypress and cedar bark beetles (*Phloeosinus *spp.). Freshly cut bolts were used in the treatment as a surrogate to live trees. Treatments where 1.0% and 2.0% Sevin SL (carbaryl), 0.2% Pemethrin Plus C (permethrin), and 0.03% and 0.06% Biflex (bifenthrin).*

Preliminary results using Arizona cypress indicate both the bifenthrin treatments and the permethrin treatment passed the binomial test (Null: successful defense = 90%), carbaryl did not. The bark beetle species attacking Arizona cypress was identified as Phloeosinus cristatus. *Results from the oneseed juniper treatments were not conclusive since only 50% of the control bolts experienced beetle attacks; causing this experiment to fail to meet the criteria set for a rigorous test of the treatments. Attacking beetles were identified as* Phloeosinus scopulorum neomexicanus.

The preliminary conclusions indicate Sevin SL (carbaryl) was not effective at a 2% formulation in preventing Phloeosinus cristatus *attacks on Arizona cypress bolts, but was effective for protecting oneseed juniper at a 1% formulation. Both the 0.03% and 0.06% Biflex (bifenthrin) formulations and the 0.2% Permethrin Plus C (permethrin) treatment were effective at preventing successful* Phloeosinus *attacks on Arizona cypress and oneseed juniper. These single tree protection tests will be repeated in 2005 to draw more conclusive results, and will be extended to testing on alligator juniper.*

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Hydrology and Soils

Photo by Gerald Gottfried

Photo by George Robertson

Impacts of Pinyon-Juniper Treatments on Water Yields: A Historical Perspective

Peter F. Ffolliott[1] and Cody Stropki[1]

Abstract*—Pinyon-juniper woodlands are not normally considered a high water-yielding type largely because of the low precipitation amounts and high evapotranspiration rates encountered. Nevertheless, a recommendation was made in the 1950s to evaluate the effectiveness of increasing water yields by converting pinyon-juniper overstories to herbaceous covers. A series of process and plot studies and watershed-level experiments were carried out in acting upon this recommendation. It was concluded, however, that the potentials for increasing water yields through conversion treatments is "poor" in the region. A summary of these research findings is presented in this paper.*

Introduction

Pinyon-juniper woodlands, the largest "forest type" in the southwestern United States, lie adjacent to and surround the montane forests in the region. Occurring at lower elevation and generally with less annual precipitation than the forests, these woodlands possess a lower water-yield improvement potential compared to these forests. Because of their extensive distribution, however, early investigators felt that water-yield improvement practices in pinyon-juniper woodlands could conceivably affect the availability of water supplies (Barr 1956, Dortignac 1960). The water-yield improvement practices to be implemented were conversions of comparatively high water-demanding pinyon-juniper overstories to lower water-demanding herbaceous covers by mechanical, chemical, and burning treatments. It was anticipated that the reductions in water consumption by plants might become recoverable water. A historical summary of the effects of these treatments on water yields is presented in this paper.

Hydrologic Characteristics

Pinyon-juniper woodlands are not normally considered a high water-yielding type largely because of the low precipitation amounts and high evapotranspiration rates encountered. Streamflow is largely ephemeral, with no permanent streams originating in the area. While runoff events can coincide with the two seasonal precipitation patterns encountered, most of the annual water flows are associated with the winter period, primarily the result of heavy rainfall, rapid snowmelt-runoff, or rain-on-snow events. Over 70 percent of the annual streamflow originating in the pinyon-juniper woodlands on the Beaver Creek watersheds in north-central Arizona occurs in this winter period (Clary and others 1974, Baker 1984). When they occur, runoff events in the summer are low, variable, and short-lived events following torrential summer thunderstorms.

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Water Yield Improvement Potentials

While there was some evidence in the literature (prior to 1956) that removal of pinyon-juniper overstories with the replacement of a herbaceous cover might not produce additional water, a recommendation of Barr (1956) and others was that evaluations of alternative conversion treatments to achieve water-yield improvement in the type should be made by watershed-level experiments. Before this recommendations could be implemented, however, a justification for considering water-yield improvement in pinyon-juniper woodlands was deemed necessary. This justification was largely attained through process and plot studies.

Process and Plot Studies

In a study of interception, throughfall, and stemflow, Skau (1964a) reported that between 10 to 20 percent of the precipitation falling on pinyon-juniper woodlands on the Beaver Creek watersheds was intercepted by the tree crowns. He felt that these interception losses might be (at least partially) reduced by conversion treatments. A somewhat similar conclusion was also reached by Collings (1966) on the Fort Apache Indian Reservation.

Researchers believed that an increase in soil moisture storage might occur following the removal of pinyon-juniper overstories, reducing infiltration losses and (possibly) increasing water yields. Skau (1964b), in another study, found two percent more soil moisture in the upper two feet of the soil on sites cleared of pinyon-juniper overstories than on uncleared sites at the beginning of the summer and winter precipitation seasons. He implied that this difference might have been greater if it was not for the dense herbaceous vegetation that had invaded the cleared sites.

Decker and Skau (1964) measured transpiration rates of Utah and alligator juniper trees by enclosing sample trees in a ventilated tent of transparent plastic film. Increased humidity of the ventilation stream was a direct index of vapor production and, therefore, transpiration rates through conversion. They observed transpiration rates increasing through the morning, peaking at noon, and remaining at this high rate until the middle of the afternoon when transpiration rapidly decreased into early night. By coupling their results with knowledge of the frequencies of occurrences of trees similar to those sampled, the researchers were able to estimate water losses to the hydrologic cycle by transpiration in woodland communities where juniper trees tend to dominate.

The results of these and earlier process and plot studies formed a basis for subsequent experiments to evaluate the effectiveness of converting pinyon-juniper overstories to herbaceous covers to increase water yields on a watershed-level.

Treatment Methods

A variety of treatments had been used into the early 1960s to increase herbage (forage) production by clearing pinyon-juniper overstories. Researchers suggested that these treatment methods might also be implemented for water-yield improvement purposes. Extensive areas of pinyon-juniper woodlands had been cleared by chaining or cabling, in which a heavy anchor chain or cable was dragged between two tractors (Cotner 1963, Arnold and others 1964). Pushing (bulldozing) had been widely used to remove individual trees. These mechanical treatment methods of conversion pulled trees from the ground, leaving pits or depressions where the trees formerly stood. Parenthetically, Skau (1961) had determined that these pits reduced overland flow on the cleared landscapes. Nevertheless, at the time, these mechanical methods of conversion were considered to be a viable approach to implementing water-yield improvement treatments.

Another approach to clearing pinyon-juniper woodlands was hand-clearing with ax and (or) saw. This treatment method had been used to a lesser extent than chaining, cabling, or pushing treatments, however. Hydrologically speaking, this method of tree removal had a "minimal impact" on overland flow because pitting was eliminated. Recurring problems of increased soil erosion that occurred with chaining, cabling, or pushing were also minimized with hand-clearing treatments.

Broadcast burning was successful where the pinyon-juniper overstory was dense enough to carry a fire (Arnold and others 1964). Once again, pits were not created by this method and increased soil erosion was minimal when the burning treatment was properly prescribed and carefully implemented. Individual-tree burning was carried out on a mostly limited scale (Jameson 1966).

Killing trees with herbicides was also a possible option, but this kind of treatment had not been operationally applied at the time. Herbicidal treatments left the dead trees standing on the landscape, and the trees influenced less surface-area than if they were uprooted or felled by cutting and left on the ground.

Watershed-Level Experiments

An early test of the effects of pinyon-juniper conversion treatments on water yields was conducted on a large (operational) scale in the basin of the adjacent Corduroy and Carrizo Creeks in east-central Arizona in 1957-59. The pinyon-juniper overstory on Corduroy Creek was cleared on 34,000 acres (25 percent of the basin) by chaining, while the shrubs, litter, and duff beneath ponderosa pine stands on 18,000 acres (13 percent of the basin) were burned. Carrizo Creek was left undisturbed to serve as a control. Because the evaluation of these two treatments considered only their "total composite effect" on water yields (Collings and Myrick 1966), conversion of the pinyon-juniper overstory to a herbaceous cover on Corduroy Creek could not be isolated as the "sole influence" on subsequent water yields. Nevertheless, these differences were not important because it was concluded that this pinyon-juniper conversion treatment produced no significant changes in water yields in the post-treatment evaluation period.

Two small watersheds less than 100 acres in size were selected to investigate the effects converting pinyon-juniper overstories to herbaceous plants on a smaller-scale on Cibecue Ridge, in the same general area as Corduroy and Carrizo Creeks. Following a calibration period, the overstory on one of the watersheds was cleared by chaining in 1967, with the resulting slash burned and the watershed seeded with a mixture of perennial grasses and fenced to exclude livestock. The other watershed was the control. A "parameter model" to predict how components of the hydrologic cycle changed as a result of the treatment was used to evaluate this experiment (Robinson 1965, Myrick 1971). It was determined that water yields increased on the converted watershed in the initial two post-treatment years, but it then dropped below the expected water yields on the untreated watershed in the following two years. One reason for the subsequent decrease in water yields was the increase in transpiration by the perennial grasses seeded as part of the treatment prescription.

The effectiveness of converting pinyon-juniper overstories to herbaceous covers to increase water yields was also evaluated on three "experimental watersheds" as part of the Beaver Creek watershed program in north-central Arizona (Worley 1965, Price 1967). Varying conversion methods were imposed on the watersheds.

A cabling treatment similar to that extensively used for rangeland improvement was applied to a 323-acre Beaver Creek watershed in 1963. The larger pinyon and juniper trees were uprooted by a cable pulled between two bulldozers and the smaller trees missed by the cable were felled by ax. These larger trees were then burned and the watershed was then seeded with a mixture of forage species.

A hand-clearing treatment with the trees felled by power saw was applied to a second Beaver Creek watershed 104 acres in size in 1965. It was hoped that the problems of pitting and other soil disturbances associated with uprooting trees would be eliminated with this method. Stumps of alligator juniper, the principal overstory species on the watershed, were treated with polychlorinated-benoic acid to reduce subsequent sprouting. Shrub live oak clumps were initially treated with fenuron and later with picloram and Gambel oak sprouts were sprayed with 2,4,5-T in the dormant season to control the occurrence of these species. No seeding of forage species was done.

A herbicidal treatment of a mixture of two and one-half pounds of picloram and five pounds of 2,4-D was applied to a third Beaver Creek watershed of 363 acres. An aerial application of the herbicide was sprayed on 281 acres of the watershed in 1968. The remaining 82 acres were either untreated or individual trees treated with the same herbicide using a backpack mist-blower. This treatment was intended to reduce transpiration losses by killing the trees, while leaving the dead trees standing to reduce windspeeds and solar radiation to control evaporation losses. The treatment also avoided trapping overland flow in pits formed by uprooting trees. A firewood sale removed all of the merchantable wood from the watershed in 1976, eight years after the herbicidal treatment. The resulting slash was piled and burned the following year.

Results of Watershed-Level Experiments

The results of the watershed-level experiments in the basin of Corduroy and Carrizo Creeks, at Cibecue Ridge, and on Beaver Creek indicated that conversions of pinyon-juniper woodlands to herbaceous covers by mechanical treatments has little effect on water yields for varying reasons. Reduced overland flows caused by pitting on the watersheds where chaining or cabling conversion methods were imposed likely compensated for any changes in water yields brought about by reductions in transpiration rates by the tree removals. Elimination of the trees in the largely open stands on the watershed converted to a herbaceous cover by hand-clearing was probably too little to significantly impact on the loss of water to the transpiration process.

Only killing trees with herbicide and leaving them in place to reduce evapotranspiration losses increased water yields on one of the Beaver Creek watersheds. While the increase averaged nearly 160 percent in the 8-year post-treatment period to the time that the dead trees were removed, it amounted to less than an inch in absolute terms (Baker 1984). Furthermore, the increase could only be expected about 1 out of every 2 years, when winter precipitation equals or exceeds the average. That landscapes treated with herbicides might not be aesthetically pleasing to much of the public was also a possible limitation to its widespread application. More importantly, the use of chemicals for natural resources management purposes is opposed by many people and their use limited by environmental regulations.

Conclusion

Overall, the potentials for increasing water yields through conversions of pinyon-juniper woodlands to herbaceous covers is considered "poor" in the region. While this general conclusion contradicted the earlier recommendation of Barr (1956) and others, it should not necessarily be surprising. Water-yield improvements in any vegetative type are largely based on the premise that streamflow and (or) groundwater regimes are increased by an amount that is equal to the net reduction in evapotranspiration (Hibbert 1979). However, there is little opportunity to reduce evapotranspiration on watersheds where precipitation is less than 18 inches and its total is exceeded by potential evapotranspiration (Thornthwaite and Mather 1957), because this amount of precipitation will not penetrate far enough into the soil to influence the storage of moisture in the soil. This latter situation exists throughout much of the pinyon-juniper woodlands in the region.

A similar conclusion that little, if any, increase in recoverable water should be expected through manipulations of pinyon-juniper overstories has also been reported by Ffolliott and Brooks (1988) in their summary of opportunities for enhancing water yields in the Mountain West, by Roundy and Vernon (1999) in their assessment of watershed values in pinyon-juniper woodlands of the Interior West, and by Baker and Ffolliott (2000) in their analysis of opportunities for increasing water yields through vegetative management in the Colorado River Basin.

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Ecohydrology of Piñon-Juniper Woodlands in the Jemez Mountains, New Mexico: Runoff, Erosion, and Restoration

Craig D. Allen[1]

Woodlands of piñon (*Pinus edulis*) and oneseed juniper (*Juniperus monosperma*) in the Jemez Mountains at Bandelier National Monument in northern New Mexico exhibit greatly accelerated rates of soil erosion, triggered by historic land use practices (livestock grazing and fire suppression). This erosion is degrading these woodland ecosystems and damaging thousands of archaeological sites in this national park unit, with similar patterns evident in woodlands across much of the Jemez Mountains.

In 1993 long-term research began on the runoff and erosion dynamics of a piñon-juniper woodland hillslope at Bandelier National Monument in northern New Mexico (USA). In the 1.09 ha Frijolito watershed, erosion has been continuously studied at 3 spatial scales: 1 square meter, about 1000 square meters, and the entire watershed. This site is currently representative of degraded woodlands of piñon and oneseed juniper in this region, exhibiting substantial connectivity of exposed bare soil interspaces between tree canopy patches and obvious geomorphic signs of accelerated soil erosion (e.g., pedestalling, actively expanding rill networks). Ecological and land use histories show that this site has undergone a number of dramatic ecohydrological shifts since ca. 1850, transitioning from: 1) open ponderosa pine (*Pinus ponderosa*) overstory with limited piñon-juniper component and substantial herbaceous understory that supported surface fires and constrained soil erosion, to; 2) ponderosa pine with reduced herbaceous cover due to livestock grazing after ca.1870, resulting in collapse of the surface fire regime and increased establishment of young piñon and juniper trees, to; 3) mortality of all of the ponderosa pine during the extreme drought of the 1950s, leaving eroding pinyon-juniper woodland, to; 4) mortality of all mature piñon at or above sapling size during the 2002-2003 drought, with juniper now the only dominant woody species. Detailed measurements since 1993 document very high rates of soil erosion that are rapidly stripping the local soils. Runoff and erosion show extreme variability at multiple time scales since 1993, reflecting the inherently variable nature of monsoon convective thunderstorms (and occasional multi-day fall rainstorms) that drive the local hydrology. The multi-scale erosion data from the Frijolito watershed reveal surprisingly little drop off in erosion rate between the one meter-square scale and the 1.09 ha scale, in sharp contrast to the expected pattern that is observed at a nearby (7 km distance) relatively stable woodland watershed (cf. Wilcox and others 2003).

Since 1990 researchers and land managers have teamed up to experiment with restoration techniques in local PJ woodlands (Jacobs and others 2002). Our primary restoration treatment (thinning and application of slash mulch) is demonstrated to be an effective remediation technique for increasing herbaceous cover, stabilizing soils, and supporting surface fire. Monitoring shows that the restoration treatment also increases the resiliency of vegetation to drought effects. Long-term monitoring is essential to distinguish short-term variability from longer term trends, particularly in such climatically sensitive and highly dynamic ecosystems.

In: Gottfried, Gerald J.; Shaw, John D.; Ford, Paulette L., compilers. 2008. Ecology, management, and restoration of piñon-juniper and ponderosa pine ecosystems: combined proceedings of the 2005 St. George, Utah and 2006 Albuquerque, New Mexico workshops. Proceedings RMRS-P-51. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

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Rainfall, Soil Moisture, and Runoff Dynamics in New Mexico Piñón-Juniper Woodland Watersheds

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Abstract—*Clearing trees in piñón-juniper woodlands may increase grass cover and infiltration, leading to reduced surface runoff and erosion. This study was conducted to evaluate piñón-juniper hydrology conditions during baseline data collection in a paired watershed study. We instrumented six 1.0 to 1.3 ha experimental watersheds near Santa Fe, NM to collect rainfall, soil moisture, and runoff data. Volumetric water content reflectometers (VWCRs) were used to measure soil moisture on wooded hillslopes, on grass hillslopes, and in valley bottoms. Time domain reflectometry (TDR) probes were installed in two watersheds to measure soil wetting at hillslope, gully headcut and channel locations. Spatial dynamics of rainfall and runoff interactions were evaluated for 9 rainfall events. Runoff was present at all watersheds in 3 of the 9 rainfall events. Rainfall intensity of 5mm/15min was generally the minimum precipitation required to generate channel runoff. During high intensity rain storms, greater soil moisture was observed at valley bottoms when compared to grass hillslopes and wooded hillslopes. At the watershed scale, total runoff as a proportion of precipitation was relatively low. Wetting depth measured with TDR probes was consistent with increases in soil moisture measured with VWCRs. In general, the wetting depth was greater at the channel, followed by the gully headcut and then by the hillslope. Results to date from the baseline data collection period suggest that larger storms that wet the entire watersheds produce most annual runoff, so clearing trees may increase grass cover, but may have little effect on annual runoff.*

Keywords: *New Mexico, piñón-juniper, baseline data, rainfall, soil moisture, runoff, wetting depth, TDR*

Introduction

Piñón-juniper (PJ) woodlands cover a large portion of semi-desert U.S. ecosystems, extending from Texas to California and occupying an area of approximately 24.7 million ha (Burns and Honkala 1990). In New Mexico, PJ woodlands are the most abundant type of forest and cover nearly 53 percent of the total forested area of the state (O'Brien 2003). Temporal variations in precipitation are important for runoff generation in piñón-juniper woodlands of northern New Mexico. Wilcox (1994) reported two typical precipitation seasons that produce runoff in a steep mountainous area of this region: a mid-summer season, with convective storms generating most of the runoff; and a mid-to-late winter season, with snow melt producing some runoff. Amount and intensity of precipitation is also important for the generation of runoff. Reid and others (1999) reported that convective storms characterized by high-intensity short-duration precipitation events generated most of the runoff and that frontal storms also produced considerable runoff in a 26-month study done in north-central New Mexico. Soil moisture dynamics in piñón-juniper woodlands, especially in the shallow soil layer, are important in

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determining water availability for different plant species. Breshears and others (1997) found that both piñón and juniper trees can extract water from the top 30 cm of intercanopy locations and compete with forage species for water.

Piñón-juniper tree control by mechanical removal, treatment with fire, or herbicide, may increase forage response and soil water infiltration and consequently reduce surface runoff and erosion. In order to characterize hydrology effects of PJ control, it is important to understand and evaluate watershed conditions prior to treatment. This study is part of a research effort to evaluate watershed response to piñón-juniper control in northern New Mexico. This study was conducted to collect baseline data and evaluate hydrology conditions before tree clearing planned in 2009. We formulated two study questions:

1. How does channel runoff respond to different precipitation amounts and intensities?
2. What is the soil moisture response to rainfall on wooded hillslopes, on grass hillslopes, and in valley bottoms of the watersheds?

Study Site

The research site is located at the New Mexico State University-Santa Fe Ranch (NMSU-SFR), 16 km northwest of Santa Fe, NM. The vegetation of the area is represented by piñón-juniper woodland on the hillslopes and grassland in the valleys. Oneseed juniper (*Juniperus monosperma* Engelm.) is the dominant tree species, followed by piñón pine (*Pinus edulis* Engelm.). The dominant species in the understory is blue grama (*Bouteloua gracilis.* Willd. ex Kunth), followed by New Mexico feathergrass (*Stipa neomexicana*), sideoats grama (*Bouteloua curtipendula.* Michx), and purple threeawn (*Aristida purpurea.* Nutt). Piñón-juniper woodlands are generally between 1370 and 2440 m. In New Mexico, these woodlands are found in elevation that range from 1520 m to 2130 m (Burns and Honkala 1990). Elevation of the watersheds at the NMSU-SFR study site ranges from 1939 m in the valley bottoms to 1977 m on the ridge tops. Slopes range from 2% to 5% in the valley bottoms and 20% to 50% on the hillslopes. Average annual precipitation is 387 mm. Most of this precipitation falls during the monsoon season from May to October. The average low annual temperature is 4.4 °C and the average high annual temperature is 26.7 °C. The soil type is classified as part of The Santa Fe Group (Tsf) and is mostly characterized by sediments from fluvial deposits of sand, silt, clay and pebbles with basaltic and sedimentary inclusions (Scholle 2003, Till and others 2003).

Methods

Six watersheds (WS-1, WS-2, WS-3, WS-4, WS-5, and WS-6) of 1.0 to 1.3 ha were instrumented to collect precipitation, runoff, and soil moisture data starting in April of 2003 (fig. 1). WS-2 and WS-3 were equipped with Time Domain Reflectometry (TDR) technology to measure soil wetting depth.

Precipitation measurements were taken with four tipping-bucket rain gauges with datalogger enclosed (HOBO rain gauge, manufactured by Onsetcomp). These rain gauges were installed at mid-slope elevation in four watersheds (WS-2, WS-3, WS-5, and WS-6) to collect precipitation data. The rain gauge at WS-2 was installed in April of 2003 and the remaining three were installed between June and July of the same year. The WS-2 rain gauge was used to represent adjacent WS-1, and the WS-3 rain gauge was used to represent adjacent WS-4 (fig. 1).

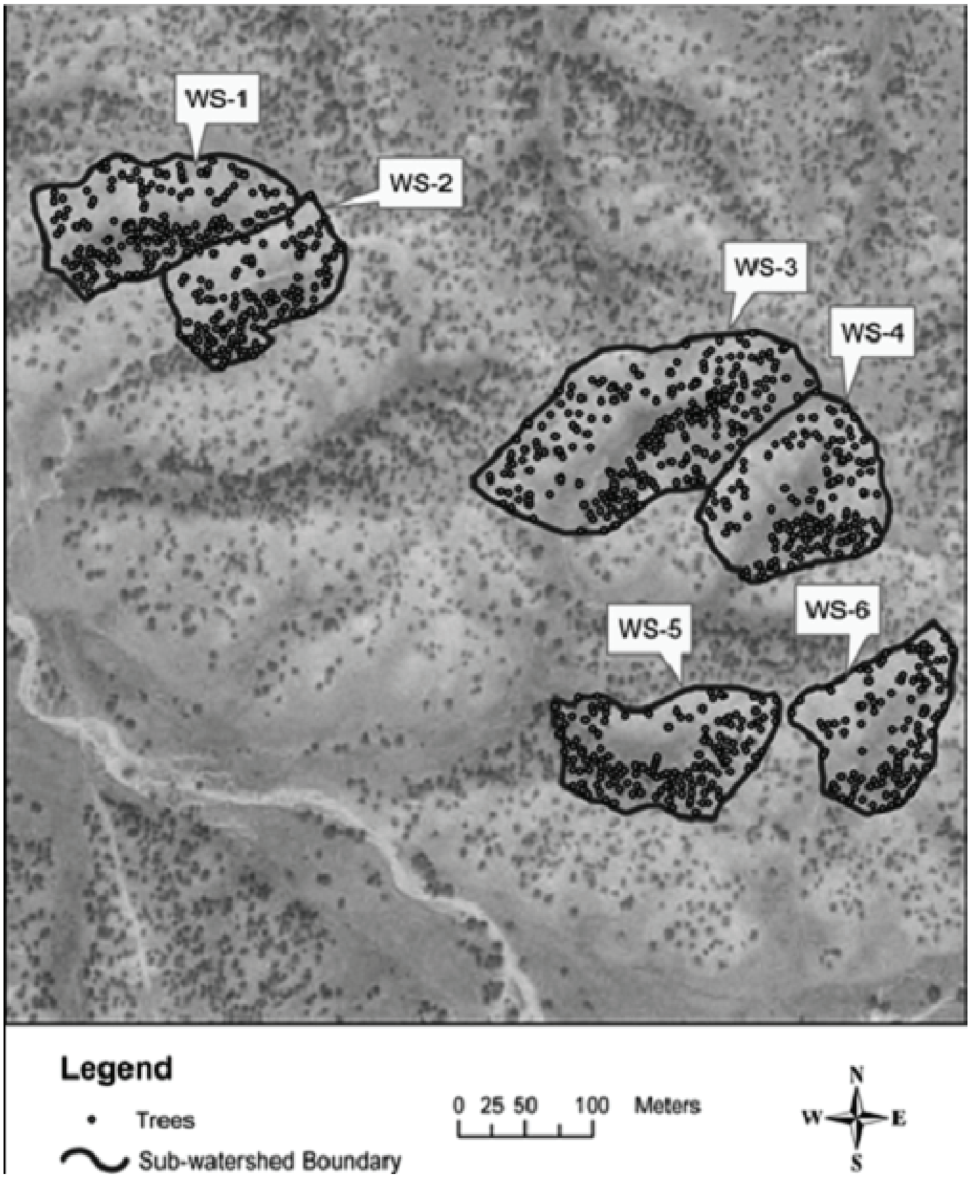

Figure 1—Watersheds of the NMSU-Santa Fe Ranch.

Runoff data were collected at the outlets of the watersheds. Type H flumes equipped with pressure transducers were installed at the outlets to measure water stage. Total amount of runoff per rainfall event was calculated with the low flow discharge equation for type H flumes with floor sloping of 0.003 to 0.03 (Brakensiek and others 1979).

Soil moisture data were collected using six volumetric water content reflectometers (VWCRs) in each watershed. The VWCRs were installed under canopy and in the intercanopy of wooded hillslopes, grass hillslopes, and valley bottoms. The VWCRs were inserted horizontally at 10 cm depth and attached to dataloggers. Soil volumetric water content data were collected every hour. Maximum values of increase in soil moisture measured by all six VWCRs after precipitation events that produced runoff in each watershed were added and divided by the total number of precipitation events to obtain average increase in soil water content for the two-year study period.

Soil wetting depth was calculated using data collected with TDR sensors. A total of sixteen 15 cm-long experimental sensors were installed at WS-2 and WS-3 to collect the TDR trace signals. The TDR sensors were installed vertically, with the tip exposed, at gully headcut, channel bed, channel bank, ridge, and hillslope. Two rain gauges attached to dataloggers were installed in the valley bottoms of WS-2 and WS-3 to activate the TDR systems. During rainfall events, the TDR systems were triggered by the automated rain gauges and data were continuously collected during the precipitation event and until 15 minutes after rainfall stopped in order to capture the total expected time needed for rainfall to either infiltrate or run off. Under dry conditions, TDR data were collected twice a day, at noon and at midnight. TDR data were entered into a physically based multisection model developed by Tidwell and Brainard (2005) to calculate the soil wetting depth. The model uses the S_{11} scatter function and Debeye parameters for dielectric dispersion and loss is used to analyze the TDR traces.

Results

Precipitation

We present precipitation data for nine rainfall events that generated runoff. Two occurred in 2003 and the remaining seven happened in 2004 (table 1). Rainfall was temporally and spatially variable among watersheds. During the two-year study, we observed that precipitation at the study site was characterized by scattered convective storms during the months of May to September and by frontal storms during October and November. Also, it is noteworthy that precipitation varied considerably from one watershed to the other, with differences up to 3.4 mm inter-watershed differences measured in one single rain storm. The year of 2003 was drier than 2004. For example, precipitation recorded at WS-2 from April to December of 2003 was 215.6 mm, and for the same period in 2004 it was 308.6 mm.

Table 1—Runoff response to precipitation events during a two-year study period at the NMSU-Santa Fe Ranch, NM.

	Watersheds and Areas											
	WS-1 1.35 ha		WS-2 1.00 ha		WS-3 1.14 ha		WS-4 1.34 ha		WS-5 1.21 ha		WS-6 1.06 ha	
Date	Rain	Runoff	Rain	Runoff	Rain	Runoff	Rain	Runoff	Rain	Runoff	Rain	Runoff
	(mm)											
5/26/03	14.0	0.175	14.0	0.172	14.0	-	14.0	-	14.0	-	14.0	-
10/4/03	6.0	0.051	6.0	*	5.8	*	5.8	*	5.2	0.025	5.6	*
6/19/04	11.8	-	11.8	0.007	12.4	0.041	12.4	0.004	13.6	-	12.4	0.005
7/12/04	24.6	0.180	24.6	0.019	22.2	0.010	22.2	0.044	24.2	0.977	23.6	0.010
8/18/04	6.6	-	6.6	0.036	5.8	-	5.8	0.005	5.6	-	5.6	0.00002
8/21/04	6.2	-	6.2	0.020	6.8	0.013	6.8	-	6.0	-	6.0	-
10/5/04	12.4	0.953	12.4	0.022	11.3	0.136	10.3	0.229	9.3	0.136	9.0	0.006
10/11/04a	7.6	0.372	7.6	0.027	8.4	0.127	7.6	0.056	7.6	0.184	7.0	*
10/11/04b	17.2	1.534	17.2	0.016	14.2	0.145	13.0	0.402	11.0	0.088	9.0	0.006
Total	**106.4**	**3.3**	**106.4**	**0.3**	**100.9**	**0.5**	**97.9**	**0.7**	**96.5**	**1.4**	**92.2**	**0.03**

- No data
* No runoff

Runoff

The difference in precipitation between years was also reflected in the amount of runoff generated. In 2003, runoff data were collected at two watersheds during a convective, high-intensity short-duration rain storm on 26 May. Also in 2003, runoff data were collected at two watersheds during a low-intensity long-duration frontal storm on 4 October. In 2004 there was more precipitation and runoff than in 2003. A total of seven storms produced runoff in at least two of the watersheds. The highest runoff value of 1.534 mm was calculated from a frontal storm at WS-1 in 11 October 2004, and the lowest runoff value of 0.00002 mm was calculated from a low-intensity rain storm at WS-6 in 18 August 2004. The total amount of single event runoff produced per watershed during the two-year study was relatively low. WS-6 presented the lowest amount of total runoff with 0.03 mm and WS-6 had the highest value with 3.3 mm (table 1).

Soil Moisture

Average increase in soil volumetric water content (VWC) ranged widely depending on the point of measurement, which was dictated by the soil moisture sensor location in the watershed. Calculations of average soil VWC were based on soil moisture increases measured after precipitation events that produced some runoff in the two-year study period. The lowest average value of 0.01 (one percent) was observed in the intercanopy of grass hillslope at WS-6. The highest average value of 0.18 (eighteen percent) was observed in the intercanopy of the valley bottom at WS-4 (table 2).

Table 2—Average soil moisture increase after precipitation events during a two-year study period at the NMSU-Santa Fe Ranch, NM.

	Valley bottom Vol water content (cm^3/cm^3)		Grass hillslope Vol water content (cm^3/cm^3)		Wooded hillslope Vol water content (cm^3/cm^3)	
Watershed	**Under canopy**	**Interspace**	**Under canopy**	**Interspace**	**Under canopy**	**Interspace**
WS-1	0.03	0.06	0.06	0.04	0.02	0.05
WS-2	0.10	0.02	0.12	0.08	0.04	0.09
WS-3	0.07	0.04	0.03	0.03	0.06	0.04
WS-4	0.16	0.18	0.05	0.10	0.04	0.02
WS-5	0.05	0.07	0.05	0.05	0.08	0.08
WS-6	0.01	0.06	0.04	0.01	0.04	0.04
Average	**0.07**	**0.07**	**0.06**	**0.05**	**0.05**	**0.05**

Depth to Wetting Front

Data from TDR sensors in WS-3 were used to calculate soil-wetting depth. Figure 2 shows soil wetting depth calculated using data collected with one TDR sensor and soil volumetric water content data from one VWCR. Both, the TDR sensor and the VWCR are installed within 3 m distance of each other in WS-3. In general, the soil wetting depth measured with the TDR sensor followed the same wetting-drying pattern than soil moisture content measured with the VWCR.

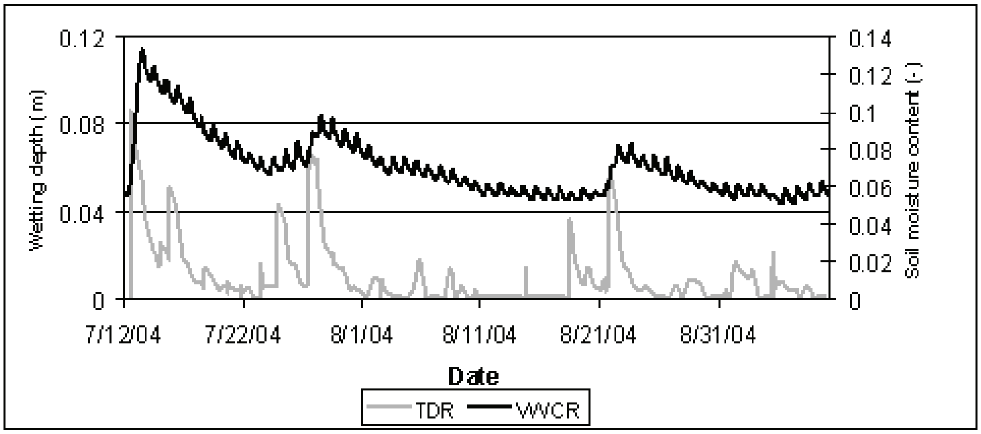

Figure 2—Volumetric water content reflectometer (VWCR) soil moisture and site-adjacent time domain reflectometry sensor (TDR) wetting depth response during the summer of 2004.

Discussion

Precipitation data collected during the two-year study indicated a clear pattern, which showed differences in the spatial distribution of precipitation at the watersheds. The lowest cumulative precipitation occurred in WS-6 and the highest cumulative precipitation occurred in WS-2. Precipitation amount and intensity play an important role in the generation of runoff. We were able to identify the temporal distribution of precipitation that produced runoff in the watersheds. In contrast to results reported by Wilcox (1994) where he stated that runoff in PJ woodlands in northern New Mexico is typically present in mid summer (intense thunderstorms) and mid to late winter (snow melting), we found that runoff at the NMSU-SFR watersheds was present during the summer season when runoff was generated by high intensity convective storms. Runoff was also produced during early to mid fall, when runoff was produced by frontal storms. Rainfall intensity greater than 5 mm/15 min appears to be the minimum amount of precipitation needed before runoff occurred at the watersheds outlet.

It was observed that frontal storms with low-intensity precipitation events, which occurred in the month of October in 2003 and 2004, lasted longer than convective storms with high-intensity precipitation events in May to August of both years. These precipitation events during frontal storms lasted several hours until the soil got wetted and overland flow started to occur. Thus, antecedent soil moisture was important for runoff response at the watersheds. Also, it was noted that sediment movement was greater during these frontal storms than during convective precipitation events, probably due to the moisture regime of the soil.

In general, across all watersheds, greater average soil moisture was observed in the valley bottoms than on the grass hillslopes and wooded hillslopes. However, there were small differences in soil moisture increases between under canopy and intercanopy locations. At the single watershed level, WS-4 presented higher soil moisture in the valley bottom, followed by grass hillslope, and then by wooded hillslope. This difference by location was probably due to the topography of the watershed that includes a greater valley area and less steep hillslopes.

The new application of TDR technology worked well and seems to be a promising tool to accurately calculate soil-wetting depth in real time. In contrast to data obtained with the VWCR, where an average water content value of the soil in contact with the probe waveguides is obtained, the use of this model allowed us to calculate to 1 mm precision the soil water flux in the upper 15 cm of the soil. Also, it was noted that in general the wetting depth was greater at the channel, followed by the gully headcut and then by the hillslope.

Some obstacles and limitations of the study and study site in particular are worth noting. The study site is located on federal land, and it is exposed to the inherent risks of open public access. To protect from vandalism, most of the equipment is inside solid material enclosures. This is the case of the type H flumes that are enclosed in cages built with concrete and steel screen. In some high intensity rain storms, the screen starts accumulating debris, and if not cleaned right after the storm, chances are that accumulated debris will trap sediment, creating a dam and restricting free water flow. For example, it was observed that after a high intensity rainfall in 11 October 2004, the screen that protects the flume got clogged and acted as a dam, accumulating sediment in the upstream arroyo and impeding free water flow through the flume. Thus, for a rain storm that occurred at the same location two days later, data collected was not considered reliable and it was discarded.

Dealing with electronic equipment in remote locations to measure real time natural phenomena is always a challenge. The study site is located 480 km from the investigator offices, and there is to date no remote communication with the sensors installed in the watersheds. Thus, the long distance between the study site and the headquarters and the lack of a telecommunication system that would alert us in real time to precipitation events, made it difficult to perform maintenance needed after each precipitation event. Also, TDR data collected from WS-2 were not considered reliable due to high variability in the TDR signal, probably due to a faulty multiplexer rely or a poor connection that introduced some noise in the signal.

Conclusions

Channel runoff responded differently to different precipitation amount and intensities. Runoff was produced by rainfall events with intensities greater than 5 mm/15 min. Runoff is affected by antecedent soil moisture. Most of the runoff was produced by convective storms during the summer. Some runoff was produced by frontal storms during the fall. Total amount of runoff produced per watershed during the two-year study was relatively low. Soil moisture response to rainfall was higher in the watershed valley bottoms than in grass hillslopes, and wooded hillslopes. The use of TDR technology to measure wetting depth is a potential tool to determine threshold precipitation for runoff generation.

Management Implications

Based on analysis of rainfall, runoff, soil moisture, and channel flow, we feel confident that the main runoff generating mechanism at the NMSU-SFR study site is infiltration excess runoff. This is similar to that observed by Wilcox (2002), who states that overland flow is important and may be the dominant mechanism of runoff in the majority of juniper woodlands. This infiltration excess runoff appeared to occur even during longer-duration frontal storms. Results to date from the baseline data collection period suggest that larger storms that wet the entire watersheds produce most annual runoff, so clearing trees may increase grass cover, but may have little effect on annual runoff. Detailed characterization of actual effects of tree clearing will be improved with continued measurements during baseline data collection and after planned tree clearing in 2009.

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Belowground Carbon Distribution in a Piñon—Juniper / Short Grass Prairie Site

John Harrington[1] and Mary Williams[2]

Abstract*—Piñon-juniper woodlands encompass over 22.5 million hectares in the Western United States. However, little is known about the ability of these ecosystems to sequester carbon. This paper presents the preliminary results of an investigation on the belowground carbon distribution in a piñon-juniper/short grass prarie site in north-central New Mexico. Using a systematic sampling design the influence of tree cover, tree size, and sampling location (under tree canopy, under grass canopy, or under bare soil) were detected. However, these effects were only present in the upper most soil strata (< 20 cm below soil surface).*

Introduction

Piñon-juniper woodlands encompass over 22.5 million hectares in the Western United States (Mitchell and Roberts 1999). In north central New Mexico the dominant woody species of this community are piñon pine (*Pinus edulis* Englem.), Rocky Mountain juniper (*Juniperus scopulorum* Sarg.) and oneseed juniper (*J. monosperma* Sarg.) In the past 150 years, woody encroachment, both the increasing of density of woody plant cover and the encroachment into former grasslands by woody species, has increased in the southwestern United States (Van Auken 2000). Potentially, multiple ecosystem processes can be impacted by the species and structural changes associated with this encroachment. One process of interest are changes associated with carbon. Recent concerns over anthropogenic changes in greenhouse gases have elevated interest in ecosystem processes with particular attention on the carbon cycle.

The positive relationship between soil organic matter (carbon) content and soil quality (site productivity) has been widely accepted (NRC 1993). Improvements in soil quality with increased soil organic carbon have improved soil structure, nutrient availability and resiliency to erosion. Sine the mid-1980s direct and indirect roles of soil carbon pools on the carbon cycle have received attention from both policy makers and the scientific community.[3] In the context of forestry and forest soils much of the investigation on carbon dynamics has focused on boreal, mesic temperate, and tropical forest types. Comparatively, little research has examined carbon dynamics in semi-arid woodlands, such as piñon/juniper woodlands of the Western United States.

Objectives of this study were to examine the influence of vegetation type on soil carbon in a piñon-juniper – short grass prairie community in north central New Mexico. Information presented here is one part of a multi-disciplinary investigation on carbon dynamics in semi-arid and arid woodlands of the southwestern United States.

In: Gottfried, Gerald J.; Shaw, John D.; Ford, Paulette L., compilers. 2008. Ecology, management, and restoration of piñon-juniper and ponderosa pine ecosystems: combined proceedings of the 2005 St. George, Utah and 2006 Albuquerque, New Mexico workshops. Proceedings RMRS-P-51. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

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[3] A good reference on this topic is Soil Processes and the Carbon Cycle. CRC Press 1999.

Materials and Methods

The study site was located in a piñon-oneseed juniper woodland – short grass prairie community in Mora County, New Mexico (35°5'N, 105°10'W). The woodland lies in a transition between a blue grama (*Bouteloua gracilis* (H.B.K.) Lag.) dominated short grass prairie community at lower elevations and ponderosa pine (*P. ponderosa* Laws.) dominated forest at higher elevations. In addition to piñon and oneseed juniper, other woody plants include oak (*Quercus* spp.) and skunkbush sumac (*Rhus trilobata* Nutt.). In addition to blue grama, other ground covers include fringed sage (*Artemesia frigida* Willdenow) and snakeweed (*Gutierrezia sarothrae* Pursh.). The project site is situated at 2100-2150 m on a south-east facing slope of 30-40 perecent. Soils at the site are Haplustolls-Rock outcrop complex (Sellnow 1985). The soils are derived from sandstone and shale, creating a very stony, sandy loam across the study site. Depth of soil within the site is variable, ranging from shallow to deep. Exposure of the underlying sandstone is evident throughout the area, which suggests high runoff and water erosion potentials.

Climate is semi-arid and annual precipitation is variable, averaging 446 mm (Western Regional Climate Center 2004). The majority of precipitation occurs as rainfall in July and August. Mean annual snowfall is 940 mm occurring between October and May with the highest amounts in December and March. Mean annual temperature is 7.9 °C with mean January and July temperatures of –1.11 °C and 18.3 °C, respectively.

Six study plots, approximately 25 by 50 m, were established to represent degrees of piñon/juniper density ranging from 24% to 49% canopy coverage. Percent tree canopy cover and interspace within each plot were determined using digital analysis of a 1997 digital orthophoto quadrat.

Within each plot, 12 trees, six piñon and six oneseed juniper, were selected. Within each species two trees representing three size classes based on height (small (S) < 2.5 m; medium (M) = 2.5 to 4.5 m; and, large (L) > 4.5 m) were selected. Tree selection criteria were based on whether a tree was an isolated tree or a dominant tree on the periphery of a clump of trees.

Three 1-m deep soil cores were taken at each tree using a hydraulic ESP Plus subsoil sampler (Clements Associates, Inc.). Cores were taken along a transect at the mid-point from the stem to the drip line, 2 m from the canopy edge in a bare spot, and 2 m from the canopy edge directly beneath grass. Transect orientation was placed away from nearby trees and into an area of appropriate interspace (a minimum of 2.1 m to the drip line of the nearest tree). Each soil core was divided into 7 depth increments: 0-5, 5-20, 20-40, 40-60, 60-80 and 80-100 cm beneath the soil surface. In some locations depth to bedrock was less than 1m.

Samples were analyzed for total soil carbon content at the Soil Genesis Laboratory at New Mexico State University, Las Cruces, NM. Sample preparation included removal of particles greater than 2 mm and subsequent grinding with a Certiprep Shatterbox to pass a ~0.080 mm screen. Ground samples were weighed to the nearest one-thousandth of a milligram using a Sartorius microbalance. For elemental carbon analysis, each weighed sample was introduced into an elemental analyzer (Eurovector, Milan, Italy) interfaced to an isotope ratio mass spectrometer (Isoprime, Manchester, U.K.). Each sample underwent combustion at 1030 °C by means of a He carrier gas at a flow rate of ~90mL/min with a purge rate of ~50mL/min. A soil reference standard (Eurovector, Milan, Italy) was used to maintain a high level of accuracy and precision. Soil carbon data was analyzed using analysis of variance procedures in SAS (PROC GLM; SAS Institute 1999).

Results

Observed effects on soil carbon were depth dependent with the greatest effects evident only in the upper two soil layers, 0 to 5 cm and 5 to 20 cm beneath the surface (table 1). More subtle differences in soil carbon were observed in the first-order interactions from the 20 - 40 cm depth interval but will not be presented nor discussed here. Therefore, presentation and discussion of results are limited to the upper two soil layers. While percent canopy cover influenced soil carbon

Table 1—Observed significance levels for the influence of tree cover (Cover (C)), tree species (Species (S)), tree size (Size (Z)) and sampling location (Location (L)) on soil carbon percent in a soil profile for a piñon/juniper-short grass prairie community. [*** p≤0.0001; ** 0.0001≤p≤0.001; * 0.001≤p≤0.01; and, X 0.01≤p≤0.05].

	Soil Depth Interval (depth from surface (cm))					
Variable	0 – 5	5 – 20	20 – 40	40 – 60	60 – 80	80 – 100
Cover (C)	***	***	ns	ns	ns	ns
Species (S)	ns	X	ns	ns	ns	ns
Size (Z)	*	ns	ns	ns	ns	ns
Location (L)	***	**	ns	ns	ns	ns
Z * L	***	ns	ns	ns	ns	ns
S * L	ns	ns	ns	ns	ns	ns
C * L	ns	ns	ns	ns	ns	ns
S * Z	ns	ns	X	ns	ns	ns
C * Z	X	ns	X	ns	ns	ns
C * S	ns	ns	X	ns	ns	ns
S * Z * L	ns	ns	ns	ns	ns	ns
C * Z * L	ns	ns	ns	ns	ns	ns
C * S * L	ns	ns	ns	ns	ns	ns
C * S * Z	ns	ns	X	ns	ns	ns
C * S * Z * L	ns	ns	ns	ns	ns	ns

percentage in the upper two soil layers there was no clear response with the 28 and 45 percent woody canopy cover, which have the greatest soil carbon percentages at both upper sampling intervals. In contrast, samples collected from plots in the lowest tree canopy percentage, 24 percent did not differ from those collected in the highest tree canopy percentage, 49% (fig. 1). Soil carbon percentage in the 0 to 5 cm layer was greatest under the canopy of larger trees but did not differ between medium and small trees (fig. 2). Elevated soil carbon percentage was observed under tree canopies relative to beneath grass canopies, which in turn was greater than that observed in the upper sampling layer beneath bare areas (fig. 3). This trend, while not as pronounced, persisted into the lower sampling layers. Influence of tree canopy on soil carbon levels in the uppermost sampling layer was influenced by tree size where soil carbon percentages increased as tree size became larger and, presumably, older (fig. 4).

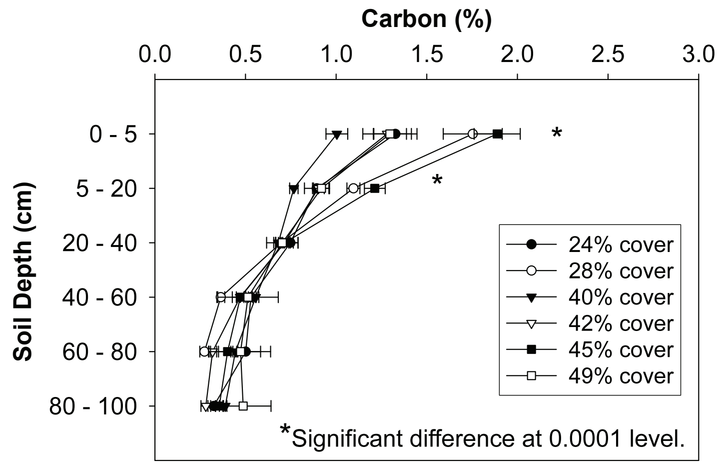

Figure 1—Soil carbon percent through a 1-meter soil profile as influenced by percent tree cover in a piñon/juniper – short grass prairie community in north central New Mexico. Bars reflect ± 1 standard error of the mean.

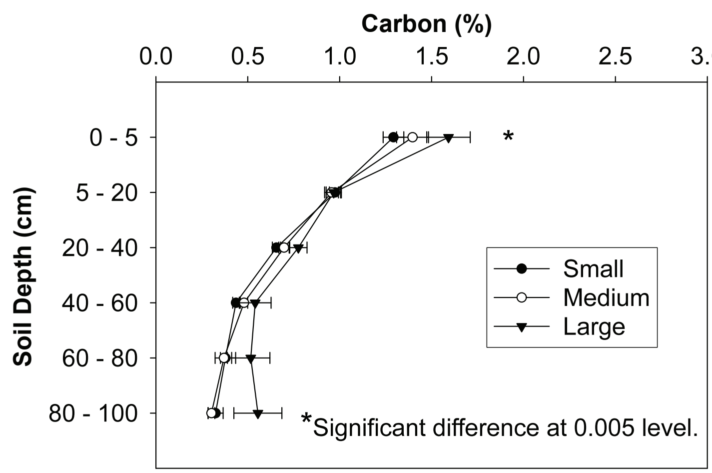

Figure 2—Soil carbon percent through a 1- meter soil profile as influenced by tree size in a piñon/juniper – short grass prairie community in north central New Mexico. Bars reflect ± 1 standard error of the mean.

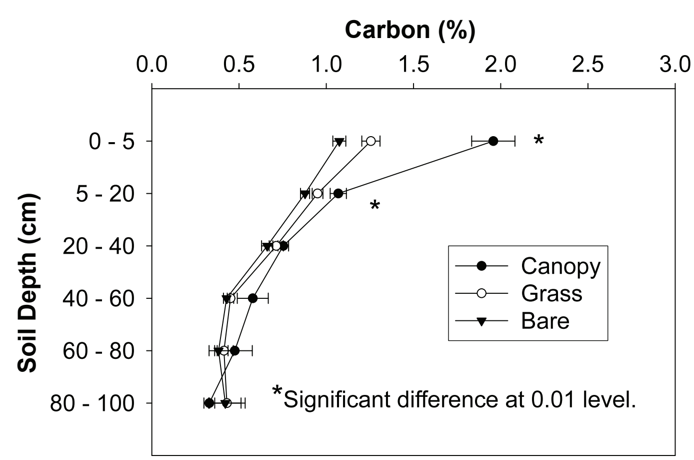

Figure 3—Soil carbon percent through a 1-meter soil profile as influenced by sampling location (tree canopy, grass or bare soil) in a piñon/juniper – short grass prairie community in north central New Mexico. Bars reflect ± 1 standard error of the mean.

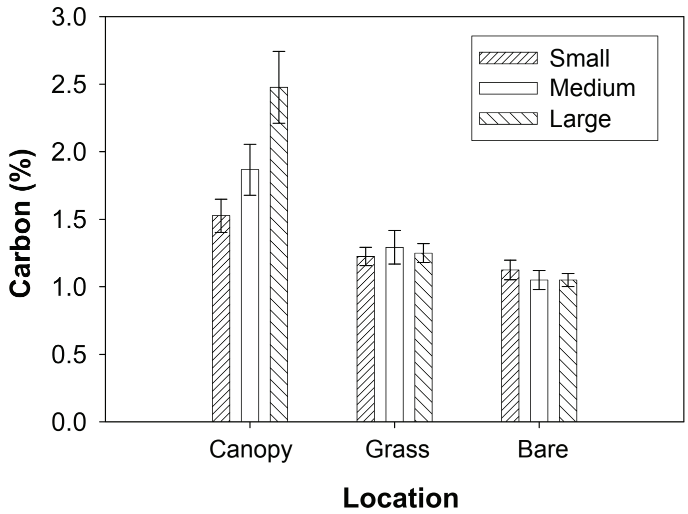

Figure 4—Soil carbon percent through a 1-meter soil profile as influenced by the interaction of tree size and sampling location in a piñon/juniper – short grass prairie community in north central New Mexico. Bars reflect ± 1 standard error of the mean.

Discussion

Increased soil carbon levels under trees relative to soil carbon levels under herbaceous plants and bare soil was also found in a companion study conducted on Mesita del Buey near Los Alamos, New Mexico (Reiley 2003). The trend of elevated soil carbon under tree canopies in piñon-juniper ecosystems has been reported elsewhere (Conant and others 1998; Krammer and Green 2000). Conclusions drawn by Guo and Gifford (2002) indicate that increases in soil carbon accumulation associated with woody vegetation encroachment is influenced by precipitation. Specifically, soil carbon increases are more evident in semi-arid and arid environments. This is in contrast to total system (above and below ground) carbon accumulation in either boreal or tropical forested systems where the majority of accumulation is in above ground or soil surface components. Potentially, enhanced below-ground carbon pools in semi-arid woodlands will be retained longer than in the two former forested types where decomposition rates are slower.

Implications

How do these findings influence woodland management in southwestern piñon/juniper woodlands and interface sites? First, effects of vegetation (tree, grass, bare) on soil carbon are evident only in the upper soil strata, 0 to 20 cm below the surface. Hence, management activities which influence soil stability will have an effect on soil carbon pools. These activities include, but are not limited to, tree thinning/harvesting, grazing, and management activities that alter fire behavior. Secondly, if a management goal is carbon accumulation in the system, including soil carbon strategies should encourage development or protection of larger and, presumably, older trees.

Acknowledgments

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Variation in Herbaceous Vegetation and Soil Moisture Under Treated and Untreated Oneseed Juniper Trees

Hector Ramirez[1], Alexander Fernald[1], Andres Cibils[1], Michelle Morris[1], Shad Cox[2], and Michael Rubio[2]

Abstract—*Clearing oneseed juniper* (Juniperus monosperma) *may make more water available for aquifer recharge or herbaceous vegetation growth, but the effects of tree treatment on soil moisture dynamics are not fully understood. This study investigated juniper treatment effects on understory herbaceous vegetation concurrently with soil moisture dynamics using vegetation sampling, soil sampling, and automated precipitation and soil moisture data collection. The study was conducted at New Mexico State University's Corona Range and Livestock Research Center Corona, NM. We created plots under dead and live juniper trees in three cattle-grazing exclosures (CD, FG, and KI). We applied heavy defoliation clipping treatment and no defoliation in the winter months. This study reports on soil moisture from volumetric water content probes installed at 0-25 cm depth at the drip line or the outside of each plot. Understory herbaceous cover and biomass were significantly higher under dead than under living trees, while volumetric water content was lower under dead than under living trees. Water content was higher on clipped than on unclipped plots for dead and living trees. At this site, water made available by treating oneseed juniper appears to be consumed by additional herbaceous vegetation under dead trees.*

Keywords: *volumetric water content, P-J control, understory defoliation, soil moisture dynamics.*

Introduction

Effects of clearing oneseed juniper woodlands on forage for cattle and big game have been extensively studied (Pieper 1990). However, less is known about the effects of such tree clearing treatments on soil water dynamics. In some locations, juniper clearing may be associated with groundwater recharge, while in others the water not used by trees may simply be consumed by herbaceous vegetation. According to Walter's model cited by Breshears and Barnes (1999), two soil layers may be distinguished on the basis of the rooting depths of plants (herbaceous and woody). Herbaceous plants have a denser root distribution than woody plants and are much more efficient obtaining water from the upper layer. Woody plants have sole access to the lower soil layer. Yet, some studies have shown that root depth to extract water depends on woody species (Montaña and others 1995) and competition for resources with herbaceous plants (Young and Evans 1981), although the distribution of plants varies with plant cover (Joffre and Rambal 1988) and depends on available soil moisture and nutrients (Breshears and Barnes 1999). Extensive research has been conducted on methods to remove piñon-juniper woodlands, yet much less is known about the sustainable management of cleared areas. Where juniper treatment increases understory herbaceous

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vegetation growth, prescribed grazing by goats and sheep may suppress juniper regrowth and improve ongoing recovery of herbaceous vegetation.

A straightforward approach to assessing juniper treatment effects on soil water dynamics is to identify understory herbaceous response and soil moisture content under treated dead and untreated live trees. To improve understanding of how juniper treatments affect understory herbaceous vegetation concurrently with soil moisture dynamics, this study investigated four aspects of vegetation and soil moisture response to juniper treatment: 1) the effect of juniper treatment on understory herbaceous vegetation cover and biomass; 2) the effect of juniper treatment on soil moisture under dead and live trees; 3) the soil moisture response to rainfall under dead and live trees; and 4) the soil moisture response to rainfall under dead and live trees where understory herbaceous vegetation had been subjected to a single high intensity defoliation event during the dormant season.

Materials and Methods

This study was conducted at New Mexico State University's Corona Range and Livestock Research Center also known as the Corona Ranch. Of the Corona Ranch, about half (5797 ha) has been classified as actual or potential piñon-juniper woodland. The herbicide Tebuthiuron was applied aerially to 959 ha in 1995, and tree mortality from the herbicide treatment was readily identifiable by the time of this study ten years later. Three cattle-grazing exclosures (C_t/D_u, F_u/G_t, and K_t/I_u) were selected for this study, with the subscript "t" indicating herbicide treatment and the subscript "u" indicating the untreated half of each exclosure. We created six plots under dead juniper snags and six plots under live juniper trees. Crown dimensions were used to delineate each rectangular plot with a long axis stretching under the largest axis of the crown and a short axis perpendicular to the long axis. All six plots per dead or live treatment were marked within a relatively small (~60 m diameter) area to reduce potential impacts of soil variation and to facilitate installation of an automated soil moisture sensor system.

We applied heavy defoliation (clipping treatment) and no defoliation (unclipped treatment) to aboveground biomass in February 2005. The purpose of the defoliation treatment was to imitate high intensity (>70% utilization) grazing by sheep and goats during the winter months. Six trees were defoliated in each exclosure: three were in the herbicide treated part of the exclosure and three were in untreated part of the exclosure. The remaining six trees were not clipped and left untreated. An additional defoliation treatment will be applied in winter 2006. Basal cover by species was determined prior to the beginning of this study with ten separate ten pin frames in each plot, including percent cover of bare ground, litter, and rock. Aboveground biomass by species was determined at the time of the basal cover measurements in plots that received defoliation. Superficial soil moisture content was determined in spring 2005 from 4x10 cm core samples taken at 1/3, 2/3, and 3/3 of the distance from the tree to the drip line along the short and the long axis of each plot. Each soil sample was stored in sealed plastic bags, weighed, oven dried at 105 °C for 48 hrs, and weighed again to determine water mass. With porosity from soil bulk density measured with a separate core sample, gravimetric water content was converted to volumetric water content (volume water/volume soil). Data were analyzed statistically in SAS using a completely randomized block design, with significant differences determined at $P \leq 0.05$.

For continuous soil moisture measurement, a nest of three CS616 (all Campbell Scientific Inc. equipment) volumetric water content reflectometer soil moisture probes was installed in each plot at the drip line on the outside of each plot at depths of 0 to 25 cm, 25 to 50 cm, and 50 to 75 cm. This paper reports on results from the surface layer, which corresponds to the herbaceous understory rooting depth. The CS616 probes were connected by cable to AM16/32 multiplexers, which in turn were connected to CR10X-2M data loggers powered by SP5-L five watt solar panels and PS100 batteries. In each exclosure, a TE525WS-L tipping bucket rain gage was installed and connected to the data logger to measure precipitation. Soil volumetric water content (volume water/volume soil) data were collected hourly from all locations beginning in September 2005. This study reports on continuous soil moisture in the period from September until November 2005, comparing time series of soil moisture averaged by dead, live, clipped, and unclipped treatments.

Results and Discussion

Juniper treatment significantly affected herbaceous understory vegetation. Basal cover of herbaceous understory was significantly different under live and dead trees ($P \leq 0.05$); it was about three times higher under dead trees (14.60%) than under live trees (4.64%) (table 1). There were also significant differences in vegetation basal cover between exclosures (table 1). Biomass was significantly different under dead and live trees, with much greater vegetation biomass under dead trees (59.52 g•m^{-2}) than under live trees (17.91 g•m^{-2}) (table 2).

The superficial volumetric water content exhibited significant differences between live and dead trees, being greater under live trees (10.36) than under dead trees (8.84) (table 3). There were also significant differences between exclosures

Table 1—Basal cover (%) under dead and live trees in three cattle grazing exclosures at the Corona Ranch. Values in each group with the same superscript letter are not significantly different ($P \leq 0.05$).

	Dead				Live				
Exclosure	Bare ground	Litter	Rock	Veg	Bare ground	Litter	Rock	Veg	Dead+Live Veg
CD	16.50	70.00	0.83	12.60	36.60	61.60	0.83	0.83	6.70^{b}
FG	10.50	70.50	0.00	19.00	9.50	80.10	0.00	10.30	14.60^{a}
KI	17.80	69.30	0.50	12.30	26.50	70.30	0.33	2.80	7.50^{b}
All exclosures	14.93	69.93	0.44	14.60^{a}	24.20	70.67	0.39	4.64^{b}	

Table 2—Biomass g•m^{-2} under dead and live trees in three cattle grazing exclosures at the Corona Ranch. Values in each group with the same superscript letter are not significantly different ($P \leq 0.05$).

	Tree treatment		Average by exclosure
Exclosure	Dead	Live	Average
CD	55.54	7.82	31.68^{b}
FG	83.99	32.82	58.40^{a}
KI	39.06	13.09	26.07^{b}
All exclosures	59.52^{a}	17.91^{b}	

Table 3—Volumetric water content (volume water/volume soil) under dead and live trees in three cattle grazing exclosures at the Corona Ranch.Values in each group with the same superscript letter are not significantly different (P ≤ 0.05).

	Axis of tree canopy		Distance from tree to drip line			Tree treatment		Average by exclosure
Exclosure	**Long**	**Short**	**1/3**	**2/3**	**3/3**	**Dead**	**Live**	**Average**
CD	10.82	12.21	11.79	11.76	11.00	10.90	12.13	11.50[a]
FG	6.84	8.19	7.35	7.97	7.37	7.19	7.89	7.0[c]
KI	10.96	8.52	9.69	9.98	9.54	8.43	11.05	9.70[b]
All exclosures	9.54[a]	9.64[a]	9.61[a]	9.90[a]	9.30[a]	8.84a	10.36[b]	

(table 3). There were not significant differences in soil moisture by long or short plot axis or by distance (1/3, 2/3, or 3/3) from tree to drip line (table 3). Since these preliminary data showed no significant differences in soil moisture content at different locations under the tree canopies, recording soil moisture probes were located under the drip line to be representative of the tree to interspace continuum and to avoid interference with vegetation in each plot under each tree.

The September to November time period was opportune for characterizing soil moisture response to rainfall, because it came after at least four weeks without rain, allowing the herbaceous water consumption to be exhibited, and it included precipitation events then a period of drying into November. Comparing volumetric water content averaged for dead and live trees, the surface soil under dead trees was drier at the start of the period, rose higher than under live trees in response to rainfall, and dried faster than under live trees after rainfall (fig. 1). Comparing volumetric water content under dead trees that were clipped or unclipped, there was a muted response to rainfall, with slightly more wetting and faster drying seen on the clipped plots (fig. 2). Comparing volumetric water content under live trees that were clipped or unclipped, there was visibly greater soil moisture and slightly faster drying under clipped plots (fig. 3).

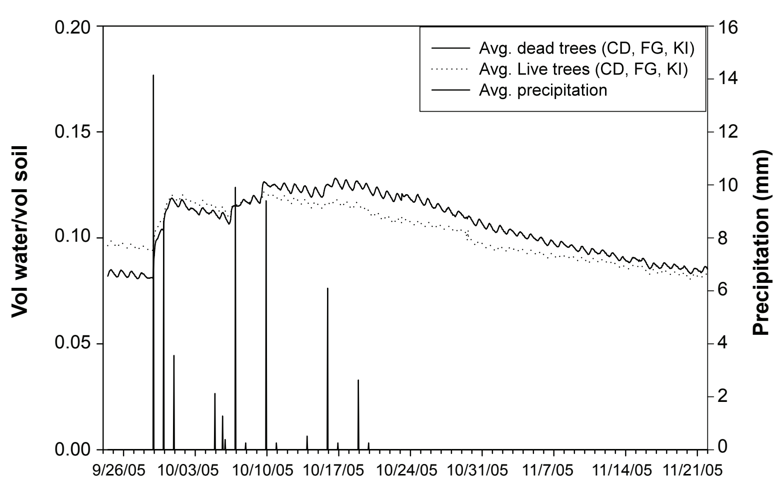

Figure 1—Soil volumetric water content response to rainfall beneath dead and living trees in CD, FG, and KI exclosures at the Corona Ranch.

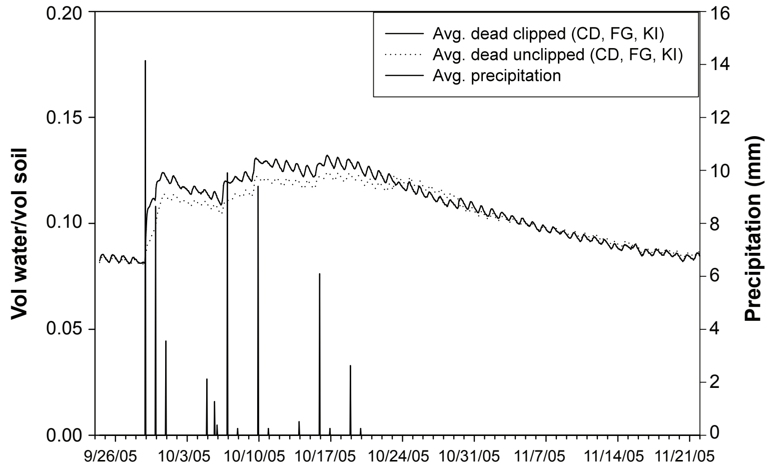

Figure 2—Soil volumetric water content response to rainfall under clipped and unclipped herbaceous vegetation beneath dead trees in CD, FG, and KI exclosures at the Corona Ranch.

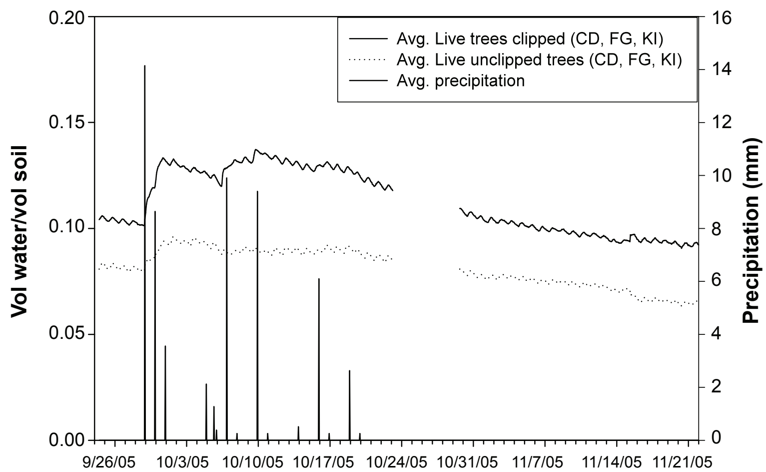

Figure 3—Soil volumetric water content response to rainfall under clipped and unclipped herbaceous vegetation beneath living trees on CD, FG, and KI exclosures at the Corona Ranch.

Understory herbaceous vegetation response to juniper treatment shows more vegetation under dead trees. At the same time, volumetric water content is greater under live trees, likely because there is less vegetative consumptive demand. In the time series comparing soil moisture, there was more rapid drying under dead trees, likely again because of the higher vegetative water consumption. Increased soil moisture after rainfall under dead compared to live trees may be from reduced rainfall interception under dead tree snags. At this site, water made available by treating oneseed juniper appears to be consumed, at least in part, by additional herbaceous vegetation under dead trees.

Clipping treatments in the dormant season did not seem to negatively impact vegetation vigor. Soil moisture time series were consistent with the interpretation that clipped vegetation grows back and consumes water as readily as or more readily than unclipped vegetation. The combination of herbicide treated trees and high intensity dormant season defoliation appeared to result in the most water consumption by herbaceous vegetation.

Conclusion

This study found that herbicide treatment of oneseed juniper resulted in greater understory herbaceous vegetation cover and biomass. While there were no significant differences in soil moisture under the area of the tree canopy, there were soil moisture differences between dead and live trees. Under dead trees, soil moisture increased more after rainfall, dried more quickly, and became drier than under live trees. Intense defoliation of the herbaceous vegetation seemed to follow a muted, but similar pattern as under tree treatment. Removing trees apparently makes more water available for herbaceous vegetation, that in turn consumes more water and with vigor unreduced by intense defoliation. While aquifer recharge may not be increased by juniper control at the Corona Ranch, tree treatment and intense dormant season grazing may result in ongoing improvements in the understory herbaceous vegetation as seen through the window of soil moisture dynamics shown in this study.

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Woodland Stand Management

Photo by Douglas Page

Variation Among Pinyon-Juniper Woodlands: A Cautionary Note

Matthew A. Williamson[1]

Abstract—*The recent emphasis on ecological restoration of forests has forced both scientists and managers to address the idea of the "natural range of variability" of their systems of study. Indeed, much work has been done to identify this range in the ponderosa pine systems of the West and to use that information to develop restoration prescriptions for those forests. The pinyon-juniper woodland, in comparison, has been more intensively managed for a variety of uses yet much of the ecology of the system is relatively unknown. In many instances we lack reliable information on the role of disturbance within the system, the importance of soil erosion potential in determining the response to disturbance, and how species composition may alter these relationships. While the literature outlining the response of pinyon-juniper to treatment is extensive, very rarely are the site descriptions detailed enough to allow the reader to place the results of the study at hand in the context of the woodland that he/she may wish to manage. Thus two identical management strategies may differ drastically in their effects. The purpose of this paper is to highlight the variability among pinyon-juniper systems and to call for increased attention to the effects of this variability when planning treatments aimed at ecological restoration.*

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Silviculture and Multi-Resource Management Case Studies for Southwestern Pinyon-Juniper Woodlands

Gerald J. Gottfried[1]

Abstract*—Southwestern pinyon-juniper and juniper woodlands cover large areas of the Western United States. The woodlands are heterogeneous, consisting of numerous combinations of tree, shrub, and herbaceous species and stand densities that are representative of the wide range of sites and habitat types they occupy. Silvicultural methods can be employed on better sites to meet multi-resource objectives and maintain the health and sustainability of the woodlands. Even-aged, uneven-aged, and coppice regeneration methods can be used in pinyon-juniper woodlands. Thinning operations may have a variety of objectives. Silvicultural prescriptions cannot be applied blindly, but must be based on stand conditions, an understanding of the silvics of the woodlands and their major species, and the biological and economic goals of land managers and owners. Several case studies of silvicultural prescriptions are discussed in detail in this document. These include the single-tree selection and diameter-limit regeneration methods and thinning (an intermediate method). Thinning is used where regeneration is not a primary objective. It is applied to reduce stand cover to improve understory development, improve wildlife habitats, and reduce fuels in wildland-urban-interface areas. Silvopastoral prescriptions, common in agroforestry, are designed to maintain the tree component and provide for increased forage production and improved wildlife habitats, are also discussed in this document.*

Introduction

The concept of applying silviculture techniques to southwestern pinyon-juniper (*Pinus edulis-Juniperus* spp.) woodlands to accomplish multi-resource objectives and maintain the health and sustainability of stands is foreign to many managers. Woodlands have a long history of being ignored or receiving a minimum of active management. Large acreages of pinyon-juniper and juniper woodlands were destroyed in the period following World War II in an attempt to improve forage production for livestock, increase water yields, and improve wildlife habitats. Treatments were applied indiscriminately to old-growth stands and to stands of more recent origin.

There was a shift in attitudes toward pinyon-juniper woodlands after the oil crisis of the mid-1970s when the demands for firewood increased dramatically throughout the Southwest. Managers began to consider woodland management that would sustain healthy stands that could be managed for multiple resources. Silviculture was considered a way to accomplish this goal. More recent concerns include fire suppression and reducing fuel concentrations, especially near towns and exurban developments; improving wildlife habitats by increasing the production of browse and herbaceous species while maintaining sufficient cover; and improving watershed condition. Silviculture can be used to meet these objectives while maintaining healthy and sustainable woodland ecosystems that produce commercial products where markets exist.

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However, not all sites can produce the full range of resource benefits, and this must be considered in land management planning. Silviculture is most successful on highly productive sites that can sustain the production of tree products based on soil properties, slope, and the presence of regeneration (Van Hooser and others 1993). Eighty-six percent of the pinyon-juniper woodlands in New Mexico and 89 percent of these lands in Arizona have been classified as productive sites using this definition (Conner and others 1990, Van Hooser and others 1993). This paper will provide a general discussion of silviculture and of several regeneration methods, intermediate treatments, and silvopastoral prescriptions that are appropriate to southwestern pinyon-juniper woodlands.

A Review of Silviculture

The Society of American Foresters (1958, p. 77) defines silviculture as "The art of producing and tending a forest; the application of the knowledge of silvics in the treatment of a forest; the theory and practice of controlling forest establishment, composition, and growth." Silviculture is directed at creating or maintaining a forest or woodland that meets the objectives of the owner (Smith 1962). Silviculture is divided into two main systems: even–aged and uneven-aged. A system is a comprehensive planned program of silvicultural treatments during the life of a stand (Smith 1962).

In an even-aged stand, the age difference among individual trees is usually between 10 and 20 years, although a difference of 25 percent of rotation age can be applied to a stand with long rotations of 100 to 200 years. Uneven-aged stands contain trees of considerable age differences, where three or more age classes are present. An uneven-aged stand will have a decrease in the number of trees per unit area as the diameter increases. There are progressively fewer trees in the larger size classes than in the smaller size classes. The characteristics of the trees and other vegetation and the site characteristics determine the system and regeneration cutting methods used. Pre- and post-treatment inventories are essential for developing a sound prescription and determining the success of the treatment and any necessary modifications. Seed production, seed dispersal, and seed bed requirements are important if stand sustainability is desired. The coppice-forest method is applied if regeneration of sprouting species, such as alligator juniper (*Juniperus deppeana*) or associated oaks (such as Emory [*Quercus emoryi*] or Gambel oak [*Q. gambelii]*), is important. Light requirements affect tree establishment and initial growth of regeneration and young trees. Although most pinyon seedlings seem to benefit from moderate microclimatic conditions under the shade of nurse trees or inanimate objects, they are shade-intolerant and eventually need light to grow successfully (Gottfried and others 1995). Spacing among residual trees will affect future growth and resistance to damaging agents. Insects and diseases, including dwarf (*Arceuthobium* spp.) and true mistletoes (*Phoradendron* spp.), fire, and wildlife herbivory are important considerations. Prescriptions designed to enhance understory shrubs and herbaceous vegetation must also consider their requirements for establishment and growth.

Silvicultural Methods for Pinyon-Juniper Woodlands

A number of silvicultural regeneration methods can be prescribed for pinyon-juniper woodlands (Bassett 1987) depending on the land manager's desired biological and economic objectives. Most pinyon-juniper stands are uneven-aged.

Single-tree selection, designed to maintain uneven-aged stands, has a number of advantages since it favors the establishment of natural regeneration of the main tree species, protects the site from wind and water erosion, can maximize vertical diversity important for wildlife, manipulates composition easier, and is esthetically pleasing (Bassett 1987). There are also disadvantages to single-tree selection, as it is more difficult to plan and administer wood sales, can result in damage to residual trees, may reduce horizontal diversity over large areas, does not allow for prescribed burning, and will make dwarf mistletoe control difficult. Group selection, another variant where small openings are created in the stand, is designed to maintain an uneven-aged stand.

Other prescriptions designed to create even-aged stands, such as two-step or three-step shelterwood, are used in the Southwest. These shelterwood methods remove the overstory over a short period and are designed to encourage regeneration under the partial shade of the residual trees. Some advantages of these shelterwood methods are that they allow control over site conditions for regeneration, protect the site from erosion, can control dwarf mistletoe, create horizontal diversity, and are esthetically pleasing (Bassett 1987). The disadvantages are similar to those for single-tree selection. Clearcutting, which is the easiest prescription to plan and administer, is discouraged unless the objective is to increase forage and browse for livestock and wildlife, or control dwarf mistletoe (*Arceuthobium divaricatum*). Clearcuts are difficult to regenerate because of poor seed dispersal, except where alligator juniper, which sprouts, is a major stand component. Clearcuts are also the least esthetically pleasing. However, harvesting narrow strips of woodland trees or small openings is beneficial to deer (*Odocoileus* spp.) and elk (*Cervus elaphus*), as large homogeneous landscapes are broken up. This provides food and adjacent hiding-thermal cover. A series of small clearcuts about 2 acres in size, distributed across the landscape, should provide both temporal and spatial diversity. While some private landowners may continue to remove the tree cover, many have recognized the values to their lands and livestock operations of creating mosaics of openings mixed with woodlands, or creating savannas by retaining larger trees or groups of trees.

Artificial regeneration is uncommon in woodlands because of the high expense, but is used to reclaim mining sites and restore vegetation around recreational areas following wildfires. However, artificial regeneration may be necessary if pinyon is to be restored in drought- and insect-impacted woodlands.

Thinning is classified as an intermediate cutting made in immature stands to stimulate the growth of remaining trees and increase the total yield of useful material (Smith 1962). The traditional objective is to redistribute stand growth to the remaining trees and to use all of the merchantable material produced by the stand during the rotation. The term thinning is also commonly used to describe cutting that reduces stand densities when wood production is not a goal. For example, thinning may be used to improve understory herbaceous and shrub production, alternate tree resources (such as production of pinyon nuts), and watershed condition. It is also used to reduce fuel loadings.

Slash is a by-product of silvicultural operations. At least some of the slash should be scattered throughout the site to provide protected regeneration niches for trees and herbaceous species, provide habitats for small mammals, and slow surface runoff and sedimentation. In some rural areas, it is also an important source for firewood (Gottfried and others 1995). An administrative study on an alligator juniper site in central Arizona estimated that forage production increased to 809 lb per acre in openings where harvesting slash had been treated and to 1,366 lb per acre under untreated slash (Soeth and Gottfried 2000). Adjacent untreated openings contained an average of 252 lb per acre and unharvested juniper stands contained an average of 138 lb per acre of forage. Concentrations of slash require treatment. Large piles are not recommended because of damage to soil resources caused if they are burned (Teidemann 1987). Small piles can be burned or left to provide habitats for small mammals that are part of the prey base for some of the Southwest's threatened or endangered avian species. Green slash can attract *Ips*, especially in the spring or summer, and operations may have to be delayed until fall or winter to reduce insect activity.

Case Studies of Silviculture Prescriptions

The following case studies are examples of viable silvicultural methods. It is difficult to say that one method or another should be applied since silvicultural prescriptions must consider the existing stand and physical site characteristics and desired conditions for the future. There are at least 70 pinyon-juniper and juniper plant associations in the Southwest (Moir and Carleton 1987, USDA Forest Service 1997), each with unique features that must be considered. One treatment will not fit all situations and several may be valid within a landscape. Case studies are valuable since they provide information about new or different techniques and allow managers to compare results with their procedures or learn methods that could be appropriate to their situations. New ecological knowledge and management techniques will contribute to future activities within the southwestern pinyon-juniper woodlands.

Regeneration Method Case Studies

An Experimental Single-Tree Selection Prescription—The Rocky Mountain Research Station, in cooperation with the Black Mesa Ranger District of the Apache-Sitgreaves National Forests, Arizona, has been studying several silvicultural treatments for woodlands, including single-tree selection and diameter-limit prescriptions. Results are compared to changes in unharvested control plots. The diameter-limit prescription could also be characterized as the removal harvest of a one-cut shelterwood or an overstory removal, except that an upper diameter for residual trees was specified. The prescriptions were selected because they were being conducted by the District or being considered for future management. The objectives of the study were to evaluate the effects of treatment on overstory characteristics and tree regeneration and to demonstrate the feasibility of these prescriptions for woodland management.

This case study provides results from one of the single-tree selection plots and one of the diameter-limit plots that are part of a larger silvicultural experiment. Prescription planning was coordinated with the forest managers who administered the treatments as commercial fuelwood sales. Considering time and money constraints, treatments had to be practical to be accepted by managers and fuelwood contractors.

The long-term study is located on the Black Mesa Ranger District of the Apache-Sitgreaves National Forest. The study site is northeast of the town of Heber, Arizona. Topography on the study site is relatively flat. Elevation is approximately 6,600 to 6,800 ft. Precipitation occurs during two seasons. Winter precipitation, usually snow, produces about 55 percent of the average annual precipitation (with standard deviation) of 19.0 ± 3.3 inches, as measured at the Ranger District office from 1981 through 2001. The soils are derived from undivided Cretaceous sedimentary rocks, mostly limestone, shale, and sandstone, and most are classified as Lithic Ustochrepts or Udic Haplustalfs and have fine loams in the surface horizon (Laing and others 1987).

The woodlands in the study area consist of Colorado pinyon (*Pinus edulis*), oneseed juniper (*J. monosperma*), alligator juniper, and occasional ponderosa pine (*P. ponderosa*). Pinyon is the most common tree species. Stand conditions in the general area showed an average basal area of 101 ± 23.5 ft^2 per acre and average canopy cover of about 40 percent (Laing and others 1987). The primary plant association is *Pinus edulis*/*Bouteloua gracilis* (USDA Forest Service 1997), which is one of the most common associations in Arizona and New Mexico. Cattle grazed the area during part of the study period, but use was minimal. Local residents had removed some large trees over the years prior to the study.

Preliminary results reported for the single-tree selection and diameter-limit silvicultural treatment are from two 10-acre plots, which are part of a larger replicated study. Each treatment plot contains 12 permanent circular 0.05-acre inventory plots that were located using a stratified random design. Measurements included species, diameter or equivalent diameter at root collar (d.r.c. or e.d.r.c.), height, disease or insect damage, crown characteristics, and tree defects or past wood utilization. Tree seedlings were located within each inventory plot and pinned and numbered for re-identification. The blocks were measured in 1989, prior to treatment; in 1993, after harvesting; and in 2000. Gottfried (2004) discusses the study in more detail. Changes in small mammal populations, understory responses, and soil-plant nutrient dynamics associated with the treatments were studied in some of the silvicultural treatment blocks (Kruse 1999, Kruse and Perry 1995).

Prescription and Sale Administration—The single-tree selection prescription was based on the 1989 pre-treatment inventory that measured a total of 456 trees per acre and 150 ft^2 per acre of basal area. The general objective was to sustain the production of tree products while maintaining the stand's uneven-aged structure, provide micro-sites for tree regeneration, improve stand health, maintain hiding and thermal cover for wildlife, and produce an aesthetically acceptable landscape. The immediate objective was to reduce the basal area of trees greater than 4 inches in diameter by about 60 percent while maintaining the existing structure. The desired maximum diameter for crop trees was 13 to 14 inches; however, some larger junipers were retained for wildlife and aesthetic considerations. These large trees were considered when the inverse-J diameter distribution curve was defined. Regulation was directed to trees that were equal to or greater than 4 inch d.r.c., about 95 percent of the total basal area, because smaller trees do not have an economic value and it would be difficult to justify the tree marking costs to achieve the desired diameter distribution in these smaller trees.

Another objective was to keep the existing distribution of species in the stand. The desired number of trees in each diameter class was calculated using a "*q-value*" of 1.25, and a basal area target of 60 ft^2 per acre. The q is the ratio of the geometric series that defines the number of trees in each successive diameter class (Husch and others 1972).

The desired distribution of size classes is at a stand density index (SDI) of 25 percent of the maximum for pinyon-juniper woodlands (Ellenwood 1995). SDI is the number of trees per acre that a stand would contain at an average basal area as indicated by average diameter (Husch and others 1972). The maximum SDI varies for each species and is measured at a reference diameter (Page, this proceeding). The maximum SDI for pinyon-juniper stands is still being refined; however, values of 240 and 416 can be inferred from the literature (Ellenwood 1995, Page, this proceedings). The advantage of SDI is that it is a measure of use of a site that is unrelated to age and site (Spurr 1952).

The Ranger District marked the residual trees within the harvesting block. The crew consisted of three people: a tally keeper and two measurers/markers. The crew was supplied with the desired stand structure and noted residual trees as they were measured and marked. Leave trees exhibited good vigor, had a potential for seed production, and were free of insect or disease problems. Higher basal areas were allowed in part of the area to keep high-quality trees. The guides also specified that cutting should not create new or enlarged openings of more than 0.25 acre. Markers used a 10 BAF wedge to maintain an average basal area of 60 ft^2 per acre, and they were within 0.9 ft^2 per acre of the target.

Results—The block was harvested in December 1992. Approximately 3 cords per acre were removed. Although the diameter distribution for larger trees was achieved, stand density goals were not achieved because of the reluctance of the harvesters to cut smaller diameter trees. The post-harvest *q-value* met the goal of 1.20, but the harvesting did not achieve the basal area reduction goal for trees equal to or greater than 4 inches in diameter. Only 36 percent of the stand basal area was removed leaving about 90 ft^2 per acre. Figure 1 shows the post-harvest and present stand, including movement of trees among the diameter classes. A future solution is to give greater consideration to market preferences. It may be more realistic to regulate trees in the 7-inch and larger classes than to include the smaller sizes of trees. However, the impacts of dense groups of small trees on residual tree and stand growth still need to be determined. Approximately 678 trees per acre in the regeneration classes (trees less than or equal to 4.5 ft in height), or 85 percent, survived the harvest. This should be more than sufficient to restock the stand. The treatment did achieve the overall goal of retaining tree productivity, wildlife habitats, and aesthetics. The effects of treatment on individual residual tree growth relative to growth on similar sized trees in the control block are being analyzed, as are the impacts of treatment on tree regeneration. However, the number of trees per acre increased in many size classes indicating increased growth of residual trees from 1993 through 2000.

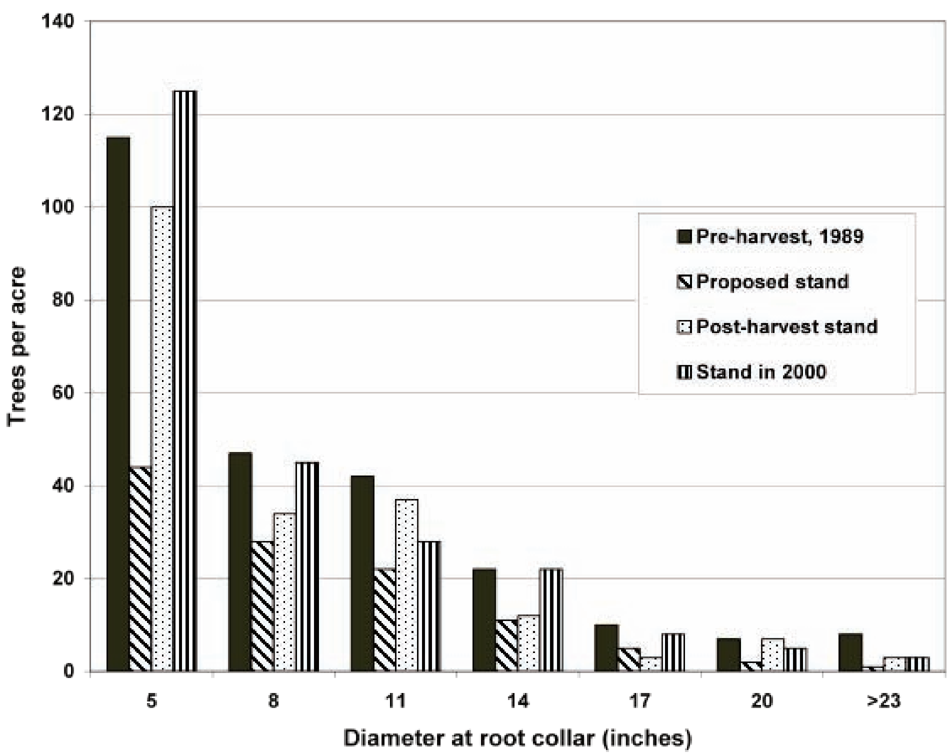

Figure 1—Initial, proposed, and post-harvest stand conditions in 1993 and 2000 for the single-tree selection block (Gottfried 2004). The graph shows the changes related to the treatment and to growth and mortality among the trees. Diameter is measured at the root collar (d.r.c).

Another Case Study of a Single-Tree Selection Prescription—Other prescriptions are being evaluated in southwestern pinyon-juniper woodlands. The Bureau of Indian Affairs, Southern Pueblo Agency in New Mexico and southwestern Colorado is testing the impacts of different residual basal areas on tree growth at sites on four pueblos (Schwab 1993). They are evaluating an uneven-aged prescription based on a *q-value* of 1.0 that leaves an equal number of trees in each size class. The *q-value* was selected to concentrate growth on the larger trees. The prescription indicated about 320 residual trees per acre with the largest diameter at 12.9 inches. Each study site has at least four to five plots with residual densities ranging from 20 to 70 square feet of basal area per acre, depending on the site, and a control. The treatment is designed to support a 30-year cutting cycle. Current growth of residual trees is being evaluated.

An Experimental Diameter-Limit Prescription—

Prescription—The diameter-limit prescription was applied to another 10-acre plot at Heber. The stand on an average acre in the block had 438 trees and 142 ft^2 of basal area. The prescription called for the removal of all trees equal to or greater than 7 inches in diameter and the protection of remaining trees and regeneration classes. This prescription was similar to one of the common practices in the area, but one that had not been previously and carefully evaluated. The logging debris was not burned, but left in place or scattered in the interspaces.

Results—The diameter-limit harvest, without debris burning, removed 112 ft^2 per acre of basal area (or 79 percent of the initial overstory cover), retaining 30 ft^2 per acre, and removed 37 percent of the trees per acre, leaving 275 overstory trees per acre. The harvest removed about 5 cords per acre of volume. Approximately 89 percent of the tree seedlings survived harvesting (515 trees per acre). Stand density in the diameter-limit block was similar in 1993 and 2000.

Some of the density reductions in both the diameter-limit and single-tree selection plots can be attributed to attacks and mortality by *Ips.* The infestation that Wilson and Tkacz (1992) described occurred a short distance to the north of the study area. A 1993 inventory of herbaceous vegetation in harvested and un-harvested blocks indicated that harvesting increased the production of blue grama (*Bouteloua gracilis*) (the primary understory species), perennial forbs, and total herbaceous cover (Kruse and Perry 1995). Total production, for example, was 172 lb per acre in the treated blocks and 70 lb per acre in the un-harvested control blocks.

Thinning Method Case Studies

Thinning From Below—Thinning is classified as an intermediate treatment designed to stimulate the growth of residual trees where enhanced regeneration is not a consideration. Fuel managers with the Bureau of Land Management in southern Utah used diameter-limit prescriptions for thinning pinyon-juniper stands by setting a minimum diameter for residual trees (Page 2006). This qualifies as a "thinning from below" treatment. Page (2006) stresses that diameter-limit thinning does not produce sustainable stands since the smaller size classes are removed, and the replacement component is lost and species composition may be altered. Page (this proceedings) suggests using an uneven-aged thinning method based on the stand density index, which is an index of competitive interaction, to guide the prescription. The goal is to achieve an SDI of 5 to 25 percent of maximum SDI, which should provide for satisfactory residual tree growth and provide for the development of an understory herbaceous cover. The number of trees and basal area per acre remaining would depend on the diameter of residual trees. For example, if an even-aged stand is the objective, and an SDI of 104 (25 percent of maximum SDI) is desired, the result would be 49 trees with an average diameter of 16 inches, or 104 trees with a diameter of 10 inches. The residual basal areas would be 68.5 and 56.7 ft^2 per acre, respectively. SDI is also used when multiple size classes are retained.

Nut Production—Another example of thinning from below is the Bureau of Indian Affairs effort to increase pinyon nut production on some of its better sites. Pinyon nuts are an important economic crop for many Native American communities. The prescription calls for the retention of superior nut producing trees while eliminating competition from neighboring trees. At the Southern Ute Reservation near Ignacio, Colorado, the harvest left a residual stand containing about 27 trees per acre with 16.8 ft^2 of basal area per acre. Superior trees are selected because of past evidence of good cone crops as judged by cones in the trees or on the ground surrounding the trees. Residual trees must have a full crown, be free of diseases, and show other signs of good vigor. Tree form is not a criterion for retention. The effects of the treatments have not been evaluated because the current 11-year drought has reduced cone production throughout much of the Southwest.

Wildland-Urban-Interface—Thinning is employed in wildland-urban-interface situations to reduce stand cover and fuel accumulations. Residual tree growth and regeneration are not the basic objectives. Thinning can be conducted by hand crews that often pile the slash for later burning or by mechanical equipment ranging from small vehicles with mounted shears to heavy mastication units. Hand thinning, although more expensive than machine thinning, is favored in visually sensitive areas around recreation areas and home sites and where archeological resources are to be protected. The Dolores Service Center of the San Juan National Forest, which administers woodlands on Forest Service and Bureau of Land Management lands in southwestern Colorado, and the Rocky Mountain Research Station are studying the effects of tree mastication with a hydro-mow machine, and the effects of hand thinning, piling, and burning on the overstory and understory and soil nutrient dynamics in woodlands north of Mesa Verde National Park. Most of the sites are in the wildland-urban-interface, adjacent to, or surrounding, expanding exurban developments. The area has been heavily damaged by the *Ips* infestation. One objective is to create mosaics of different stand conditions on the landscape. Treated plots are being compared to control plots. The invasion of cheatgrass (*Bromus tectorum*) into newly opened stands is a major concern.

Summit II, one of the study sites thinned with chainsaws, contained at least 389 trees per acre-60 percent were pinyon and 40 percent were Utah juniper (*J. osteosperma*)-prior to the infestation. At the time of treatment, 65 percent of the pinyon trees were dead, primarily due to the *Ips* infestation, and the surviving pinyon trees were mostly in the 1- to 6-inch d.r.c. classes. The stand contained junipers that were larger than 20 inches at d.r.c. The Service Center developed the prescription for the Summit Site. It called for cutting 50 percent of the canopy cover of trees up to 8 inches d.b.h. Dead pinyon trees and dense pockets of live pinyon and juniper trees were cut and live trees over 8 inches in diameter and healthy saplings, which were not under the canopy of larger trees, were left. Cutting resulted in mosaics of trees and openings of between one-quarter and two acres in size. Slash piles were restricted to between 6 and 10 ft wide and 4 and 6 ft high and had to be at least 15 ft from residual leave trees. The thinning was conducted in early winter 2005-2006 and the piles were burned in the spring of 2006.

The preliminary post-treatment inventory indicates that the stand now contains 50 live and 38 dead standing pinyon trees per acre and 86 live and 6 dead standing dead junipers per acre. The 136 live trees per acre are about 35 percent of the original density. Future analyses will examine other stand and tree attributes (percent cover, SDI, basal area), the impacts of stand reductions on tree regeneration, shrubs and herbaceous plants, and the effects of treatments on soil nutrient dynamics.

Silvopastoral Prescriptions

The lack of commercial markets for alternative and potentially higher-valued juniper wood products limits management practices (Ffolliott and others 1999). Harvested trees are generally used for firewood, fenceposts, latillas, and vigas, which have relatively low economic value. The Forest Products Laboratory in Madison, Wisconsin, has demonstrated the potential of value-added products from the wood and fiber of oneseed juniper. The use of wood chips and fiber would increase the economic potential of woodlands dominated by smaller trees that are difficult to harvest for traditional products. In February 1999, the U.S. Forest Service's Forest Products Laboratory and Rocky Mountain Research Station received a CROPS (Creative Opportunities) grant for the restoration demonstrations and workshops for management of pinyon-juniper savannas in New Mexico (Gottfried 2004). The grant is part of an effort to develop new products and markets for the juniper resource that could improve the economics of treating these woodlands, not only for range restoration, but also for more intensive management for sustainable tree products.

A manufacturing facility in Mountainair, New Mexico, could influence management on a large part of the 252,402 acres of woodlands in Torrance County (Van Hooser and others 1993) and would have a positive effect on employment and the general economy of Mountainair and Torrance County. Approximately 61 percent of the woodland area and 57 percent of its volume are on private land in Torrance County.

The goal of the project is to demonstrate to the landowners several ecosystem restoration prescriptions with the potential for economic wood products recovery while sustaining livestock production. Silvopastoral prescriptions are common agroforestry practices that favor tree production for wood, nuts, berries, or fodder (coarse feed for livestock), and integrates tree benefits with livestock husbandry goals (Ffolliott and others 1995). Harvested trees provide an additional source of income to ranchers and residual trees left after treatment are a potential economic reserve that can provide funds in hard times. Trees also provide important thermal cover for livestock in the summer and winter. The plan was to use different silvopastoral techniques on three areas and to compare results with an adjacent untreated control site. Although the prescriptions were designed to integrate range and tree production objectives, they could also be useful for treatments in pinyon-juniper dominated wildland-urban-interface areas.

The Demonstration Site

The demonstration was conducted within an area on the Greene Ranch in the Estancia Basin of Torrance County, New Mexico. The site contains sandy soils that are 5 to 6 ft deep and are representative of a band of soil that extends across the county. It is within a mile of the Gran Quivera Unit of the Salinas Missions National Monument and US Highway 55. The site is unique in the number of huge oneseed junipers that it supports- many have straight trunks with large diameters at breast height. This area is considered old-growth by local ecologists. There is little surface erosion on the site that can be related to water movement, probably because of high infiltration capacity of the sands. The area is grazed in winter and appears to have a good cover of grasses, including blue grama, side-oats grama (*B. curtipendula*), and sand bluestem (*Andropogon hallii*). Average annual precipitation at the Gran Quivira National Monument was 15.4 inches between 1938 and 2001. Most of the precipitation occurs during the summer.

Monitoring and Marking

The site was divided into four 20.3-acre treatment blocks, and a tree inventory was conducted in each block prior to marking the residual trees or designating prescriptions. Since the hope was to make pre-harvest inventories practical for ranchers and small acreage landowners, it was decided to arbitrarily limit sampling to 10 randomly located, permanent 0.05-acre fixed plots within each block. It later was apparent that either larger plots or more numerous plots would have given us a better idea of stand conditions because of the variability in each treatment block. The crew measured species and d.r.c. or e.d.r.c. On some plots, total height was measured so that volumes could be determined. The permanent plots will be monitored to provide an indication of post-treatment growth. Range resources were sampled on four transects in each block using a double sampling procedure prior to treatment (Maynard, J. personal correspondence, 2002). The average forage for each plot was: Block I, 260.2 lb per acre; Block II , 373.3 lb per acre; Block III, 585.9 lb per acre; and Block IV, 589.2 lb per acre. Results from the treated blocks will be compared to conditions on a control block.

All residual trees were marked within the blocks to be harvested. The goal was to maintain a relatively uniform crown cover within the limitations of the existing stand; however, groups of trees were retained along water courses and for wildlife cover. Trees that showed signs of wildlife activity, such as bird or rodent nests, were retained. Diameters were measured on all residual trees. The crew consisted of three people: two diameter measurers and one person who calculated and recorded the e.d.r.c. values.

Prescriptions and Results

The specific prescriptions were designed to be general enough to be applied to juniper woodlands in a variety of different sites. The four treatments included a multi-resource production block, a "savannarization" cut, a strip cut for wildlife, and an untreated control block.

The blocks were marked and harvested for firewood during the summer of 2002. A Bobcat equipped with a shear was used to fell trees in the savannarization and strip cut blocks. The trees were bucked for transportation and sale. The sustained multi-resource production block was harvested by chainsaw as there were concerns that the Bobcat would cause excessive damage to residual trees. At this time, all blocks have been harvested, but the wood has not been removed so only the results of the harvesting can be reported. An evaluation of the impacts on forage production will follow wood removal; however, the rancher recently noticed more cattle and deer use in the treated blocks.

Sustained Multi-Resource Production—The Sustained Multi-Resource Production prescription for the first treatment block (Block I) was designed to increase the herbaceous cover but retain sufficient trees of all size classes in order to sustain the tree production option on these productive sites. The prescription was designed to remove approximately 50 percent of the initial basal area but retain the variety of size classes present on the site. The objective was not forcing the residual stand into either an even-aged or uneven-aged structure, although the final result was a relatively all-aged stand with a *q-value* of 1.08. The marking favored healthy trees of all size classes in an attempt to retain younger trees for future harvests or to replace natural losses. Pinyon, which is a minor component of the block, and some snags were retained and protected for wildlife. Slash was left in the channels to slow water movement and trap soil. Groups of trees were retained for wildlife or for esthetic considerations. The final tally indicated that the residual stand contained 30 trees per acre and 29.4 ft^2 per acre of basal area. The residual volume was estimated at 2.9 cords per acre. Preliminary estimates are that about 7 to 10 cords per acre were harvested. Measurements of the inventory plots indicate that the residual basal area is 38 percent and the number of trees per acre is about 21 percent of the original amounts.

Savannarization—The second block (Block II) was treated according to a savannarization prescription. The objective was to restore the range value of the landscape by returning it to the savanna condition that probably existed prior to European settlement. However, the conditions that existed during the period are unknown, so managers must select an option. One option, leaving six trees per acre, had already been applied to an experimental site near the Abo Unit of the Salinas Missions National Monument in the Cibola National Forest (Brockway and others 2002). The current prescription was designed to leave between 15 and 25 large trees or groups of smaller trees per acre. The distribution of trees was not uniform and was considered to have esthetic value. Some areas had no trees and others had more than 25 trees per acre. One recommendation was to retain large trees on 40 to 60 percent of the area (USDA Forest Service 1993). Although this did not occur, the plan was to chip larger slash elements and lop and leave smaller material for soil cover and regeneration protection. Some snags were retained and protected, but were not counted as part of the residual stand.

The final mark indicated that 14 trees per acre in a variety of size classes had been retained on the savanna block. This was 34 percent of the amount indicated by the pre-harvest inventory. The residual basal area was 26.3 ft^2 per acre and the residual trees comprised about 1.2 cords per acre.

Strip Cut—Research and observations throughout the West have indicated that wildlife does not move into large openings even when sufficient forage or browse is available. Animals tend to remain near the edges to take advantage of hiding cover. The general recommendation is that openings be limited to about 600 ft in width (Gottfried and Severson 1994) and that “leave areas” bordering the strip be at least 200 to 330 ft wide (Gottfried and Severson 1994, USDA Forest Service 1993). The leave areas can be thinned, but there should be sufficient residual density so that the animals are not able to see nearby openings through the stand. Very open stands are treated as extensions of the opening and lose their value as hiding and thermal cover.

The actual harvesting created a 13.1-acre strip in the middle of the treatment block; the base was 556 ft wide. The strip was oriented perpendicular to the direction of the prevailing winds to minimize erosion of the sandy soil. The edges were feathered and 3.9 trees per acre were left in the strip to provide additional hiding or thermal cover and to break up and raise the wind flow. Unmerchantable slash was also lopped and left on the ground to reduce wind erosion and provide protected regeneration sites for herbaceous generation. Some snags in the strip cut and borders were retained and protected for wildlife. Critical nesting or birthing sites were identified and not disturbed. An average of 2.9 trees per acre, or 14 percent of the basal area, was harvested in the border strips.

Conclusions

Silvicultural prescriptions can be employed on most productive pinyon-juniper sites to meet multi-resource objectives and to maintain and sustain woodlands. A variety of uneven-aged, even-aged, and coppice regeneration methods and intermediate thinning prescriptions are applicable depending on existing stand and site conditions and on the objectives of the land managers and owners. Several different prescriptions can be applied in a landscape to produce a mosaic of stand conditions. Silviculture can be used to regenerate existing stands for traditional tree products or to enhance other woodland resources such as browse and forage for livestock and wildlife, improve thermal and hiding cover for livestock, large game and other wildlife species, and improve watershed condition. Providing for adequate tree regeneration, whether by creating suitable conditions for new tree establishment and survival or by nurturing existing seedlings and saplings is necessary for sustained ecosystems. Compromises may be necessary among conflicting objectives, for example, near recreation areas and archeological sites, or adjacent to populated areas where aesthetics and fuel reduction objectives may conflict. Silvopastorial prescriptions provide land owners with a compromise between wood production, which can be a potential financial reserve, and traditional livestock production goals. Case studies provide a way for managers to compare their ideas with those of other practitioners.

Silviculture will become even more important as mangers attempt to sustain and improve the health and productivity of the woodlands and their resources after suffering the impacts of the recent drought and insect infestations. Large areas of woodlands have lost their pinyon component and have shifted to a predominance of junipers or brushy species such as Gambel oak. The general public has noticed the changes and is concerned. Although it has not received much past attention, tree planting of pinyon may become necessary in some locations. More intensive, multi-resource management of the pinyon-juniper woodlands depends on the development of economically and technically viable products and markets (Ffolliott and others 1999). Increased demand for manufactured tree products might justify higher stumpage prices and investments in woodland silviculture and management. New silvicultural prescriptions will be developed and traditional ones will be modified over time to achieve the goals of sustainable and productive southwestern pinyon-juniper woodlands.

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Preliminary Thinning Guidelines Using Stand Density Index for the Maintenance of Uneven-aged Pinyon-Juniper Ecosystems

Douglas H. Page[1]

Abstract—*This paper demonstrates how Stand Density Index may be used to guide post-thinning stand structure for the sustainable management of pinyon-juniper ecosystems. The post-thinning residual stand density can be varied to achieve various management objectives. Uneven-aged management is recommended, where possible, as a better approximation of the natural development process of pinyon-juniper stands.*

Introduction

Pinyon-juniper ecosystems cover expansive landscapes in the Western United States. As landscapes are developed, management of these systems is becoming increasingly important for various reasons, including wildland interface fire/fuels, visuals, wildlife habitat, wood products, and pine nuts. Where once most management of these ecosystems focused on removal of trees to favor herbaceous species, land managers today are giving greater consideration to the management of sustainable ecosystems.

Various valid guidelines have been used for thinning pinyon-juniper stands including single-tree selection, thin-from-below, and diameter limit prescriptions. Perhaps the most common has been the specification of a diameter limit, above which trees are to be left uncut. While diameter limit prescriptions can yield acceptable results, done without sufficient pre-treatment stand exam data, diameter limit prescriptions can yield undesirable results. Among these are conversion of a mixed stand into one dominated by a single species or retention of only the older portion of a population that, to be considered healthy and sustainable, should contain young, mid-aged, and old.

The following are preliminary thinning guidelines for forest health based upon research, standard principles of forest ecology, and the ecology of pinyon-juniper ecosystems.

Stand Density Index—Theory and Basics

Stand Density Index (SDI) (Reineke 1933) is an index of competitive interaction. The maximum SDI varies for each tree species and is measured at a given reference diameter. At 25% of maximum SDI, trees begin competing with each other and begin to out-compete understory species (Long 1985). At 35% of maximum SDI, trees fully occupy the site. At higher densities, competition between trees either results in reduced growth and vigor on individual trees or may result in competitive stress and tree mortality, perhaps due in part to secondary agents such as insects that are attracted to stressed trees.

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[1] *Southwest Utah Zone Forester, Bureau of Land Management, Cedar City, UT.*

The maximum SDIs for pinyon and juniper are still being studied,[2] and current literature should be consulted to determine if the numbers presented here should be modified prior to implementing a thinning strategy. SDIs have been developed for Rocky Mountain juniper (*Juniperus scopulorum*) and common pinyon (*Pinus edulis*), but not to date for Utah juniper (*Juniperus osteosperma*) or singleleaf pinyon (*Pinus monophylla*). Research by Schuler and Smith (1988) suggests that the maximum SDI for mixed pinyon-juniper stands is higher than for single-species stands of either species. Speculation is that this may be, in part, a factor of differing rooting depths of the two species. The Central Rockies and Utah variants of the Forest Vegetation Simulator currently use 415 as the maximum SDI values for pinyon and juniper species (Van Dyck 2006). For the purposes of this paper, maximum SDI of 415 for mixed stands was selected. It is recommended that when faced with a choice of differing maximum SDIs, the lower SDI should be selected as the more conservative approach.[3]

SDI can be used as a guideline to develop desired residual stand structure goals. Residual SDI targets may be varied to help achieve various resources objectives. Higher residual SDIs will retain more dense trees, and lower SDIs may be appropriate for projects where more open conditions are desired, such as hazardous fuels reduction projects.

It is my recommendation that thinning leave no more than 25% of maximum SDI after treatment. This will maintain the site in tree cover (providing aerial cover and root mass to protect soils), but still open the canopy sufficiently to allow understory species to increase or become established in the canopy gaps between trees. It will also allow for a reasonable interval of time before retreatment will become necessary to maintain desired densities. The length of time between treatments will vary by site, and might best be estimated using a stand growth simulator program such as the Forest Vegetation Simulator. A residual stand density index of 5% of maximum SDI will leave very open, savanna like conditions. For management goals that are to retain a "woodland" component, it is suggested that residual stand density targets be between 5% and 25% of maximum SDI. When initiating a thinning program, it may be desirable to try several different residual densities, and to monitor the initial treatments over a several year period to determine which residual density seems the best fit for a given site and set of management objectives.

Developing Thinning Guides—Planning

The desired after treatment species mix will be determined by what is available on the pre-treatment site and by management objectives. Managers will want to keep in mind that juniper species tend to regenerate faster than pinyons, and where high proportions of juniper-to-pinyon are retained, this may lead to the need to retreat units earlier to maintain the proper size class mix.

The example below (table 1) illustrates an uneven-aged system. To manintain multiple size classes, the target stand SDI (in this example, 25% of 415, or 105) should be apportioned among size classes. Here, SDI = 26 is allocated to each of 4 diameter classes.

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[2]Sources for information on SDI for pinyon and juniper include: Schuler and Smith, 1988; Shaw, 2004; and the USDA-Forest Service Forest Vegetation Simulator (FVS). The current version of FVS lists the maximum SDI's for pure pinyon or pure juniper as 360 and the maximum for mixed stands as 415.

[3]Personal communication, Dr. James N. Long, Utah State University.

Table 1—Target After-Treatment Stand (at 25% of max SDI)[a].

Size Class (in)[b]	SDI	TPA	BA[c]	Spacing[d]
Regen (<3”)	26	178	8.8	8
Small (3-6”)	26	59	11.6	14
Mid (6-9”)	26	31	13.6	19
Large (>9”)	26	15	16.2	27
Total	*104*		*50.2*	—

[a] Numbers calculated on larger trees in each size class: 3”, 6”, 9”, and 14” DRC, respectively.
[b] Diameter class breaks will be specific to a given site, and it is desirable to use stand examination data to help set these breaks.
[c] Basal area is depicted here for comparative purposes.
[d] It should be noted that the “spacing” column represents space between trees of the same size class. To obtain approximate spacing between trees of different size classes, divide the figure for each size class by two then add these figures together.

Relatively simple mathematical equations may be used to compute the number of residual trees and spacing for any given SDI and size class by manipulating the basic SDI formula: $SDI = N\ (D/10)^{1.6}$, where N = number of trees and D = diameter root collar. The desired number of residual trees then becomes: $N = SDI/(D/10)^{1.6}$. Spacing is a function of the number of size classes and the number of trees per acre by size class: $S = \sqrt{(43560/C/N)}$, where S is the spacing of trees in diameter class in feet, C is the number of diameter classes, and N is the desired number of trees in diameter class, based on SDI allocation.

It might be noted, that to be scientifically accurate, calculations should be done on mid-point diameters, and where woody biomass growth is to be maximized for high value crops, this would be the more appropriate than that presented above. I, however, have chosen to use the largest tree in each size class for the following reasons:

- The smallest size class really does not need to retain more than its upper-end diameter number as there will be advance regeneration and seedling recruitment to more than compensate. Thinning down to the lower number is consistent with pre-commercial thinning philosophy.
- The middle size class(es) may come up somewhat deficit in numbers of trees per acre, depending upon the diameter distribution of acceptable leave trees. However, thinning crews tend to focus on size, and crews have a tendency to cut too few rather than too many trees.
- The largest size class may have more large trees (or at least more SDI) than targeted as there will likely be trees larger than that used to calculate the desired residual number, and larger trees ideally would have even wider spacing than that listed in the table.

Developing Thinning Guides—Implementation

For practical contract application, size classes may be rather broad, with perhaps no more than three to four classes (or implementation may get too complex to be practical). The example in table 2 uses 15% of maximum SDI and three broad size classes. Spacing guides are more important to follow than are the target trees per acre, as not every acre will contain the correct mix of trees to obtain the ideal number of trees. Target SDIs do not need to be in the contract table: SDIs are calculated for planning purposes and are not a field implementable guideline.

Table 2—Contract: Target After-Treatment Stand (at 15% of max SDI).

Diameter Root Collar [a]	TPA	Spacing (clearing radius)
<8”	30	22’ (11’ radius)
8-16”	10	38’ (19’ radius)
>16”	7	46’ (23’ radius)

[a] Numbers calculated on larger trees in each size class: 8”, 16”, and 20” DRC, respectively.

Some will find illustrations easier to understand than tables, thus graphics similar to figure 1 may be an appropriate addition to a contract. Photographs of previous treatments that successfully achieved management goals may also be an aid to contractors.

In additional to spacing guidance, tree selection guidelines similar to those below should be provided.

Trees to be left after thinning should have the following characteristics:

- Pinyon pine are favored over juniper; however, healthy juniper may be retained where there are no suitable pinyon pine.
- Full-crowned trees are preferred over trees with sparse crowns.
- Trees with healthy crowns and free of disease and damage/deformity are preferred over sparse-crowned, diseased, damaged, or deformed trees.

To begin, select a good quality leave tree. Based upon its diameter class, clear other trees around this tree equal to the radial spacing value in the table 2. From the edge of the cleared area, find another quality leave tree that is approximately its radial diameter-spacing from the cleared area and repeat step one around this tree, again clearing the radial spacing guide around this tree based on its size. Vary the size of leave trees when possible based upon the target number of trees to be left and the quality of the on-site trees from which to select. When thinning is complete, the desired condition will be variably spaced trees (based upon the size of the remaining trees) and variably sized trees with only a few large trees, a few more medium sized trees, and most trees in the smallest diameter class.

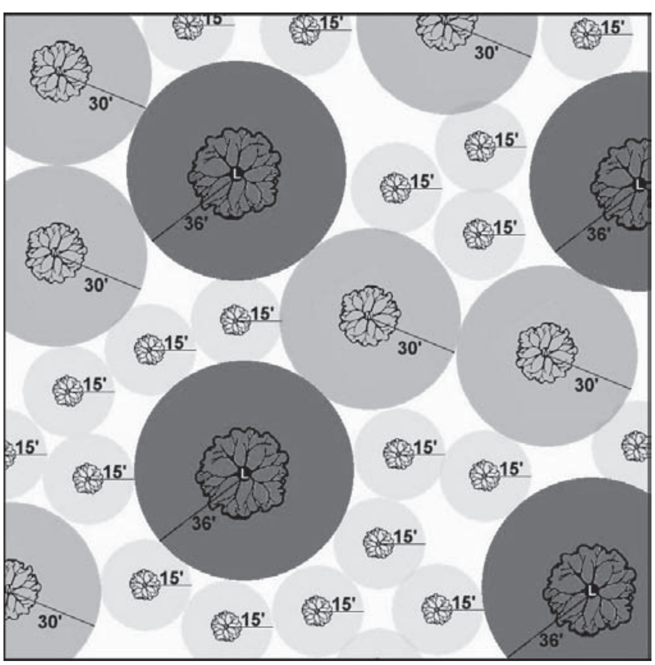

Figure 1—Shaded circles represent the clearing around three size classes of leave trees.

Additional Considerations

Should pinyon *Ips* beetles be a consideration, it may be appropriate to manage for more open canopied stands. Based on research by Negron and Wilson (2003), residual SDIs greater than 5.6% of maximum (where pinyon pine dominates) may leave stands that remain susceptible to attack by the beetle, particularly when high levels of beetle activity are present in the general area. Negron and Wilson studied unmanaged, post-epidemic residual stands and recorded post-epidemic densities at approximately 5.6% of maximum SDI. *Ips* beetles prefer trees with somewhat reduced crown ratios. Pinyon leave trees should be those with the higher percentage of crown-to-height ratio. *Ips* beetles prefer larger diameter pinyon trees, thus it may be desirable to remove most older/larger pinyon trees that show signs of declining vigor. Stand susceptibility to *Ips* may also be influenced by stand composition, and those stands with a higher percentage of pinyon-to-juniper tend to be more susceptible to *Ips*-caused mortality. Thus it is thought to be desirable to maintain a good mix of species.

Timing of implementation and treatment of pinyon slash can be critical factors when *Ips* beetles are present in the general area. Green pinyon slash can serve as an attractant to beetles. Beetles can colonize slash during the spring and summer months and maturing beetles can emerge from this slash seeking new hosts, which will tend to be the nearest available suitable pinyon trees. Even chipped pinyon debris can attract beetles during the beetles' flight periods (*Ips* cannot colonize chips but may attack nearby pinyon trees). If chips or slash are to be left on the site, then treatment is best done in late fall, allowing the winter months for material to dry and become less attractive to beetles. Even then efforts should be made to increase drying rates on any remaining larger green pinyon material. Scattering pieces in sunny locations and damaging the bark to expose the phloem will help dry the phloem layer so it is no longer provides good habitat for bark beetles. If green pinyon material greater than 3" in diameter can be removed from the site within four to six weeks of cutting, then operations may be done at any time without risking increasing the incidence of *Ips* beetles. If neither can be done, then maintaining a "green chain" of freshly cut material throughout the active beetle season will help insure that emerging beetles will attack the fresh cut material and not the standing leave trees.[4] If none of the above can be practically accomplished, then mitigation for increased beetle activity may be either to leave more juniper and fewer pinyon or to leave more trees than the target residual stand, realizing that many of these trees may be subsequently killed by *Ips* beetles. If retention of pinyon trees on the site is of prime concern and *Ips* populations are high in the drainage where the treatment is to take place, it may be best to delay thinning pinyon stands until *Ips* populations subside.

Debris from cut trees may be scattered in created openings to enhance soil protection and provide for microsite protection for establishing vegetation. However, use of green pinyon pine material >3" in diameter should be limited, as noted above.

It may be desirable to vary the spacing (density) within stands through a project area to achieve a mosaic of within-stand conditions, i.e. thin one area to 5% of maximum SDI and another to 25%. This technique is being used to reduce fuels along power line corridors in southern Utah: residual stand densities become progressively lower as one nears the power line.

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[4] Personal communication, Steve Munson, entomologist, USDA-Forest Service, Forest Health Protection, Ogden, Utah.

For wildlife habitat, groups of trees with interlocking crowns may be left interspersed with thinned trees. All trees in the group should have at least one side of the tree free from competition. Use either the largest tree in the group or the largest diameter class for the project as the spacing guide between the group and adjacent trees.

Effectiveness monitoring of the treatment areas will need to be done for a period of years following treatment to help refine future thinning prescriptions. A minimum of five years is suggested. Items to be monitored should be consistent with the project objectives and may include the response of understory species and the comparative incidence of post-treatment *Ips* beetle in treated and untreated areas. Pine nut production and tree vigor, as compared to nearby untreated areas, may also be monitored. These items should be monitored as they relate to residual stand densities and species composition.

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Appendix: Thinning Tables

The following tables may be used to determine the proper spacing between trees of various size classes for selected stand density indices. Guides are applicable to even-aged stands using average or quadratic mead diameter of the stand or they may be applied to uneven-aged stands, using the average or upper diameter limit in each size class. For a contract, numbers should be rounded to whole numbers.

Tables A1–A4 Spacing Guides for Pure Pinyon or Pure Juniper Stands.

(Maximum SDI for pure stands of either species is 360.)

Table A1. 35% of maximum SDI = Site fully occupied by trees leaving limited resources for understory species. Trees are competing with each other.

SDI	D	TPA	BA	Spacing Between Trees of Same Size	Clearing Radius
126	6	285	56.0	12.4	6.2
126	8	180	62.9	15.6	7.8
126	10	126	68.7	18.6	9.3
126	12	94	73.9	21.5	10.8
126	14	74	78.6	24.3	12.2
126	16	59	82.9	27.1	13.5
126	18	49	86.9	29.8	14.9
126	20	42	90.7	32.4	16.2
126	22	36	94.2	34.9	17.5

Table A2. 25% of maximum SDI = Trees on site begin to compete with each other; space is available for understory species to maintain themselves.

SDI	D	TPA	BA	Spacing Between Trees of Same Size	Clearing Radius
90	6	204	40.0	14.6	7.3
90	8	129	44.9	18.4	9.2
90	10	90	49.1	22.0	11.0
90	12	67	52.8	25.5	12.7
90	14	53	56.2	28.8	14.4
90	16	42	59.2	32.0	16.0
90	18	35	62.1	35.2	17.6
90	20	30	64.8	38.3	19.2
90	22	25	67.3	41.3	20.7

Table A3. 15% of maximum SDI = Trees do not generally compete with each other. A substantial amount of site resources is available for understory species.

SDI	D	TPA	BA	Spacing Between Trees of Same Size	Clearing Radius
54	6	122	24.0	18.9	9.4
54	8	77	26.9	23.8	11.9
54	10	54	29.5	28.4	14.2
54	12	40	31.7	32.9	16.4
54	14	32	33.7	37.2	18.6
54	16	25	35.5	41.4	20.7
54	18	21	37.3	45.5	22.7
54	20	18	38.9	49.5	24.7
54	22	15	40.4	53.4	26.7

Table A4. 5% of maximum SDI = Savanna conditions where non-tree species dominate the site.

SDI	D	TPA	BA	Spacing Between Trees of Same Size	Clearing Radius
18	6	41	8.0	32.7	16.3
18	8	26	9.0	41.2	20.6
18	10	18	9.8	49.2	24.6
18	12	13	10.6	56.9	28.5
18	14	11	11.2	64.4	32.2
18	16	8	11.8	71.6	35.8
18	18	7	12.4	78.7	39.4
18	20	6	13.0	85.7	42.8
18	22	5	13.5	92.4	46.2

Tables B1–B4 Spacing Guides For Mixed Pinyon-Juniper Stands.

(Maximum SDI for mixed stands is 415.)

Table B1. 35% of maximum SDI = Site fully occupied by trees leaving limited resources for understory species. Trees are competing with each other.

SDI	D	TPA	BA	Spacing Between Trees of Same Size	Clearing Radius
145	6	328	64.5	11.5	5.8
145	8	207	72.3	14.5	7.2
145	10	145	79.1	17.3	8.7
145	12	108	85.1	20.1	10.0
145	14	85	90.5	22.7	11.3
145	16	68	95.4	25.2	12.6
145	18	57	100.0	27.7	13.9
145	20	48	104.4	30.2	15.1
145	22	41	108.4	32.6	16.3

Table B2. 25% of maximum SDI = Trees on site begin to compete with each other; space is available for understory species to maintain themselves.

SDI	D	TPA	BA	Spacing Between Trees of Same Size	Clearing Radius
104	6	236	46.2	13.6	6.8
104	8	149	51.9	17.1	8.6
104	10	104	56.7	20.5	10.2
104	12	78	61.0	23.7	11.8
104	14	61	64.9	26.8	13.4
104	16	49	68.5	29.8	14.9
104	18	41	71.8	32.8	16.4
104	20	34	74.8	35.6	17.8
104	22	29	77.8	38.5	19.2

Table B3. 15% of maximum SDI = Trees do not generally compete with each other. A substantial amount of site resources is available for understory species.

SDI	D	TPA	BA	Spacing Between Trees of Same Size	Clearing Radius
62	6	140	27.6	17.6	8.8
62	8	89	30.9	22.2	11.1
62	10	62	33.8	26.5	13.3
62	12	46	36.4	30.7	15.3
62	14	36	38.7	34.7	17.3
62	16	29	40.8	38.6	19.3
62	18	24	42.8	42.4	21.2
62	20	20	44.6	46.2	23.1
62	22	18	46.4	49.8	24.9

Table B4. 5% of maximum SDI = Savanna conditions where non-tree species dominate the site.

SDI	D	TPA	BA	Spacing Between Trees of Same Size	Clearing Radius
21	6	48	9.3	30.3	15.1
21	8	30	10.5	38.1	19.0
21	10	21	11.5	45.5	22.8
21	12	16	12.3	52.7	26.3
21	14	12	13.1	59.6	29.8
21	16	10	13.8	66.3	33.2
21	18	8	14.5	72.9	36.4
21	20	7	15.1	79.3	39.6
21	22	6	15.7	85.6	42.8

Effects of Invasive Plants on Public Land Management of Pinyon-Juniper Woodlands in Arizona

Patti Fenner[1]

Abstract—*After a short discussion of terminology used in the fairly new discipline of weed science, specific examples are given to illustrate effects of invasive plants on recreation and scenic values, biodiversity, forage for domestic animals and wildlife, soil stability, fire hazard and frequency, maintenance costs for roads and highways, property values, and funding for other resource management activities.*

Introduction

Weed Science is a new discipline, (the first courses in weed science taught in a university were probably at the University of California at Davis in the late 1930s). One result of this is that basic terminology is still the subject of discussion and disagreement among those who work full-time managing invasive plants. Miscommunication with the public may occur, since many of these terms are commonly used words that have technical meaning for the weed professional. For the purpose of clarity, a few definitions will be given at the outset of this paper.

The definition of an invasive plant by President Clinton's Executive Order 13112, February 1999 (Clinton 1999), while anthropologically-based rather than based on ecological processes, serves the purpose of a widely recognized definition: "An **invasive plant** is an alien species whose introduction does or is likely to cause economic or environmental harm or harm to human health." Further, "**alien species**" means, with respect to a particular ecosystem, any species, including its seeds, eggs, spores, or other biological material capable of propagating that species, that is not native to that ecosystem. **Noxious weeds** are those plants that have been designated by some governing agency as a regulated pest, often due to invasive characteristics. Thus, "noxious weed" is more of a legal than ecological term. **Exotic species** is another term for alien species, except that it normally implies the species has been introduced not only from outside the ecosystem, but outside the country.

Below is a list of invasive plant species that may be found growing in the Pinyon-juniper grassland vegetation type in central Arizona (table 1). Starred species are those discussed in this paper.

Deterioration of Recreation and Scenic Values

Invasive plants such as yellow and Malta starthistle (*Centaurea solstitialis* and *C. melitensis*) and leafy spurge (*Euphorbia esula*) aggressively move into relatively undisturbed plant communities, often in and around sites that have been valued for both dispersed and developed recreation. Objectionable characteristics

In: Gottfried, Gerald J.; Shaw, John D.; Ford, Paulette L., compilers. 2008. Ecology, management, and restoration of piñon-juniper and ponderosa pine ecosystems: combined proceedings of the 2005 St. George, Utah and 2006 Albuquerque, New Mexico workshops. Proceedings RMRS-P-51. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

[1] Invasive Plant Program Manager, USDA Forest Service, Tonto National Forest, Phoenix, AZ.

Table 1—Invasive plants that grow in pinyon-juniper grassland vegetation type in central Arizona[1].

Russian knapweed*	*Acroptilon repens*
Tree of heaven*	*Ailanthus altissima*
Camelthorn*	*Alhagi maurorum* (lower elevations)
Giant reed	*Arundo donax*
Onionweed	*Asphodelus fistulosus*
Wild oats*	*Avena fatua*
Black mustard*	*Brassica nigra*
Red, ripgut, downy, Japanese bromes*	*Bromus rubens, B. rigidus, B. tectorum, B. japonicus*
Lens-podded & globe-podded hoary cress	*Cardaria chalepensis, C.draba*
Hairy white-top	*Cardaria pubescens*
Plumeless thistle	*Carduus acanthoides*
Musk thistle	*Carduus nutans*
Southern & field sandbur	*Cenchrus echinatus, C. spinif*ex
Diffuse knapweed	*Centaurea diffusa*
Malta starthistle*	*Centaurea melitensis*
Yellow starthistle*	*Centaurea solstitialis*
Spotted knapweed	*Centaurea stoebe ssp. micranthus*
Rush skeletonweed	*Chondrilla juncea*
Canada thistle	*Cirsium arvense* (higher elevations)
Bull thistle	*Cirsium vulgare* (higher elevations)
Field bindweed	*Convolvulus arvensis*
Common teasel	*Dipsacus fullonum*
Russian olive	*Eleagnus angustifolia* (lower elevations)
Quackgrass	*Elymus repens* (higher elevations)
Weeping & Lehmann's lovegrass	*Eragrostis curvula & E. Lehmanniana*
Leafy spurge*	*Euphorbia esula*
Euryops subcarnosus*	Sweet resinbush
Dyer's woad	*Isatis tinctoria*
Dalmatian toadflax*	*Linaria dalmatica*
Yellow toadflax*	*Linaria vulgaris*
Purple loosestrife	*Lythrum salicaria*
Yellow sweetclover	*Melilotus officinalis*
Globe chamomile	*Oncosiphon piluliferum*
Scotch thistle*	*Onopordum acanthium* (higher elevations)
Fountain grass	*Pennisetum setaceum* (lower elevations)
Karoo bush	*Pentzia incana*
Japanese knotweed	*Polygonum cuspidatum*
Russian thistle	*Salsola kali*
Wild mustard*	*Sinapis arvensis*
Salt cedar*	*Tamarix pentandra, T. chinensis, T. ramosissima*
Siberian elm	*Ulmus pumila*
Periwinkle	*Vinca major*

[1] Occurrence in pinyon-juniper ecosystems according to the following sources:

USDA Forest Service Fire Effects Information System:
http://www.fs.fed.us/database/feis/plants/index.html

CRISIS (California Regional Invasive Species Information System) database:
cain.ice.ucdavis.edu/crisis/crisismaps/

SWEPIC (Southwest Exotic Plant Information Clearinghouse) database:
http://sbsc.wr.usgs.gov/research/projects/swepic/swemp/sbscmain.asp

University of California Exotic Pest Plant Council
http://ucce.ucdavis.edu/datastore/datareport.cfm?reportnumber=42

such as spines, sap that is extremely irritating to skin, or toxicity to humans and/or domestic animals create environments that people prefer to avoid. The nearly inch-long spines of yellow starthistle plants growing in dense infestations make walking across a meadow a daunting endeavor. Malta starthistle grows in camping areas on the Tonto National Forest that have been abandoned by the public.

Invasive plants are able to out-compete native vegetation in several ways. Many of central Arizona's invasive plants are cool-season annuals. They begin growth in the late fall, overwinter as rosettes, and complete their life cycle in the early spring. They typically germinate and put down roots before native annuals do, so that they are able to monopolize use of water in the top few inches of soil. In addition, they germinate through a continual period of a few months, so that there are roots at varying soil depths, using soil water completely.

Leafy spurge, a perennial, has an extremely fast-growing root system and also reproduces by seeds that are shot out of fruit capsules to a distance of 15 feet when temperatures are high and humidity is low (Simonin 2000). The seeds typically have high viability. Roots are covered by buds that can form new plants if the mother plant is tilled and broken into parts. Root reserves of perennial weeds allow them to sprout earlier than native annual plants growing in the same area. Leafy spurge has milky white sap that may cause severe irritation to human skin and is reported to cause blistering and hair loss on the legs of horses in heavily infested areas.

Salt cedar (*Tamarix* spp.) has invaded floodplains of several rivers in the western U.S. Salt cedar is not restricted to a greenline adjacent to rivers, but can withstand xeric conditions much better than native riparian woody species such as cottonwood (*Populus fremontii*) and willows (*Salix* spp.), and grows in a much broader band along streams. In areas with dense infestations of salt cedar, it may be difficult for river recreationists to find take-out points, or camping spots anywhere on the floodplain or adjacent low terraces.

Salt cedar has rapidly invaded streambanks of the Upper Verde River in recent years. When the season of seed dispersal of salt cedar, as compared to the native cottonwood and willow, is considered, it is easy to understand its competitive advantage even in natural, undisturbed situations. Salt cedar produces seed over a period of many months, from spring through October or later, while cottonwood and willow produce seed for only a period of weeks in the spring (Merkel and Hopkins 1957). Native riparian trees have evolved to produce seed in synchronicity with winter snowmelt/precipitation events in early spring. This climatic pattern produces high stream flows that leave deposits of fine silt as they recede, which are prime locations for germination and establishment of native tree species. Salt cedar uses this scenario to establish, but is also able to establish after summer storms have left similar silt deposits along the river.

Russian knapweed (*Acroptilon repens*) reduces competition from other plants by what is called allelopathy, or the production of biochemicals that inhibit the growth of other plants (Beck 2004). While conversion of a biologically diverse community to a monoculture of Russian knapweed is not aesthetically pleasing, this plant can also have devastating effects on members of the recreating public who enjoy bringing horses onto public lands.

Ingestion of significant quantities of Russian knapweed or yellow starthistle by horses can cause "chewing disease" which is characterized by fatigue, lowered head, an uncontrolled rapid twitching of the lower lip, tongue-flicking, involuntary chewing movements, and an unnatural open position of the mouth. Poisoning occurs after a horse has ingested at least 60 percent of its body weight over a two month period (Panter 1990, 1991). There is no effective treatment for either yellow star thistle or Russian knapweed poisoning because the affected areas in the brain undergo necrosis and do not regenerate.

Opportunities for anglers in streams within the pinyon-juniper vegetation type can be reduced by invasive species. Plants such as bull thistle (*Cirsium vulgare*), Scotch thistle (*Onopordum acanthium*) and yellow starthistle are extremely spiny, changing the activity of walking streambanks from a pleasant stroll to a fight to avoid or pull spines out of gear or clothing. Plants such as Japanese knotweed (*Polygonum cuspidatum*) spread quickly to form dense thickets that exclude native vegetation and greatly alter natural ecosystems. Knotweed is able to rapidly colonize riparian areas after scouring floods, making impenetrable thickets that prevent anglers from accessing the water's edge.

Reduction of Biodiversity

Russian knapweed was mentioned above as producing chemicals that serve to reduce its competition. Leafy spurge also exhibits allelopathy: extracts from the roots of leafy spurge are leached into the soil wherever the weed grows. These extracts inhibit the germination and growth of other plants in the surrounding area (Nova Scotia Department of Agriculture and Fisheries 2003).

The production of toxic chemicals by tree of heaven (*Ailanthus altissima)* may explain the success of this plant. An aqueous extract of *Ailanthus* leaves has been shown to be toxic to 35 species of gymnosperms and 10 species of angiosperms (Mergen 1959).

Other invasive plants accomplish this same result by various means: germination occurring over a several-month period serves the same purpose by depleting the entire soil profile of available water. A sharp demarcation zone at the front of a sweet resinbush (*Euryops subcarnosus*) infestation in southern Arizona was not due to allelopathy, but to extremely efficient uptake of water by the resinbush, leaving none for the native plants. The invasive front of sweet resinbush creates monocultures, excluding normally prevalent half-shrubs like false mesquite (*Calliandra eriophylla*), shrubby buckwheat (*Eriogonum wrightii*), and even snakeweed (*Gutierrezia sarothrae*) (Pierson and McAuliffe 1995).

Extensive root systems of some perennial invasives crowd out other species. What is often left is a monoculture. When vegetation is reduced to this level of sameness, biodiversity in the animal community also is reduced drastically.

Salt cedar is able to use salty water by absorbing the salts through cell membranes. It avoids the toxic effects by using special glands to excrete the salts and by dropping salt-filled leaves. The leaves dropped each fall accumulate to a considerable depth under the canopy. Through this process, salt cedar acts as a salt pump concentrating salts from deep in the ground onto the soil surface. Over time, salts in the mulch layer kill existing plants and prevent others, especially desirable riparian species, from becoming established. As a result, the ground under a salt cedar or within a salt cedar thicket is devoid of plants except, on occasion, another salt tolerant species (Johnson and others 2002).

Black and wild mustards (*Brassica nigra, Sinapis arvensis*) replace other native and invasive species by growing in high densities, using soil nutrients and moisture before other plants. It often causes native annuals to die early in the spring, due to lack of available soil moisture (Brooks 2003).

Toadflaxes *(Linaria* spp.) commonly displace existing plant communities and associated animal life. Both yellow (*L. vulgaris*) and Dalmatian toadflax (*L. dalmatica*) are sold commercially as ornamentals, despite being on noxious weed lists of many states in the United States. These species grow in many different habitat types and climatic regimes. They have become very difficult to manage, as they have now evolved biotypes with variable responses to herbicide treatments (Lajeunesse 1999).

Reduced Forage for Domestic Animals and Wildlife

When leafy spurge infests pastures, herbage production can be reduced by as much as 75 percent (Lym and Messersmith 1985). Cattle will avoid grazing an area with as little as 10 percent cover of leafy spurge (Hein and Miller 1992). If horses are permitted to walk in areas with leafy spurge, the sap will cause severe blistering and hair loss on their feet (Nova Scotia Department of Agriculture and Fisheries 2003). It is reported to cause severe irritation of the mouth and digestive tract in cattle, which may result in death (Whitson 2002).

Dense Dalmatian toadflax infestations adversely impacts livestock and wildlife by replacing vegetation that would normally sustain them (Lajeunesse 1999). Dalmatian toadflax is rarely eaten by wild or domestic animals and has a deep and extensive root system that makes it difficult to control.

Increased Soil Erosion

Elimination of grasses by sweet resinbush has led to a dramatic increase in exposure of bare soil, and increased soil erosion (Pierson and McAuliffe 1995). In a sweet resinbush site in Marijilda Canyon, in southern Arizona, the bare soil created by the dominance of this species resulted in soil that moved more easily. Exposed roots and soil pedestals around bases of the few remaining native grasses were evident in the zone adjacent to the resinbush infestation.

Where Dalmatian toadflax replaces sod-forming or bunch grasses, erosion increases (Lajeunesse 1999). Scotch thistle (*Onopordum acanthium*) also has been documented to replace native bunchgrasses and sod-forming grasses (Beck 1999).

In addition to their above-ground attributes, many invasive weeds such as the biennial thistles have taproots, and replace the more fibrous-rooted streamside sedges and grasses. Fibrous roots give structure to loose stream bank sediments, preventing them from caving off during high flows. Fibrous roots actually create fish habitat by holding upper sediment layers in place as overhanging banks, which fish use as hiding or spawning cover. Soil erodes from around tap-rooted plants, leaving a gradually sloping stream bank that provides no cover for fish.

Increased Fire Hazard and Increased Fire Frequency

Salt cedar disrupts the structure and stability of native plant communities and degrades native wildlife habitat by out-competing and replacing native plant species, monopolizing limited moisture, and increasing the frequency, intensity and effect of fires (Muzika and Swearingen 2006).

Wild oats *(Avena fatua)* is a cool-season annual grass, drying out by late spring, to provide a source of dry standing fuel to carry fires from rights-of-way along highway into adjacent lands. It is frequently a contaminant in seed mixes or in straw mulch used during revegetation of highway construction projects.

A dramatic increase in fire size and frequency has been observed in pinyon-juniper woodlands as cover of nonnative annuals such as cheatgrass (*Bromus tectorum*) increases (Miller and Tausch 2001). Where fires have burned in pinyon-juniper woodland invaded by cheatgrass in Nevada, the woodland is being replaced by great expanses of annual grassland dominated by cheatgrass. (Billings 1994)

Cheatgrass forms a dense, uniform carpet that out-competes native grasses, trees and shrubs. It greens quickly, dries quickly and produces a very flammable cover that often burns completely, without allowing native plants to reestablish. In pinyon-juniper woodlands, the combination of cheatgrass and fire may effectively prevent the re-establishment of the original woodlands (Mitchell 2000).

Red brome (*Bromus rubens*) is a similar cool-season annual exotic grass that dries by early monsoons, and is easily ignited. Dense populations that grow during wet winters fuel fires through the several vegetation types where this species grows, including the pinyon-juniper type (personal observation). Japanese brome (*Bromus japonicus*) typically grows at higher elevations than red brome. The fire hazard it creates is not as significant an effect to the vegetation types in which it grows, which are typically adapted to fire frequencies of 5 to 7 years. However, it moves into forests and rangelands where grazing or other disturbance have reduced the competitiveness of native grasses, and replaces them, changing perennial grasslands to annual grasslands (Howard 1994).

Buffelgrass (*Pennisetum ciliare*) and fountaingrass (*Pennisetum setaceum*) currently grow in Arizona at elevations below the pinyon-juniper vegetation type. They are moving north, and are currently moving into the edges of higher elevation vegetation types (personal observation). New varieties of buffelgrass have recently been developed by the Texas Agricultural Experiment Station and USDA Agricultural Research Service that will be able to withstand lower winter temperatures, giving this plant a new ability to invade higher elevation ecosystems in Arizona (Hussey and Burson 2005). Both species are perennial bunchgrasses that provide enough fuel to create intense fires that can destroy native shrubs, trees and cacti.

Increased Maintenance Costs for Roads and Highways

Camelthorn (*Alhagi maurorum*) can grow through pavement, and the thorns can flatten car tires. Where it grows near highways, it causes extensive cracking in the asphalt, and constant road repairs are necessary (Horsley 2004). Other invasive species, such as knapweeds and starthistles can grow densely along highways, where they are likely to catch fire and burn guardrail support posts (personal observation).

Reduction in Property Values

At the 2005 meeting of the Western Governor's Association, a resolution was passed, recognizing that invasive or undesirable aquatic, riparian and terrestrial species influence the productivity, value, and management of a broad range of land and water resources in the West (Western Governor's Association 2005). They listed "significant negative economic, social and ecological impacts that include, but are not limited to:

a. Reduction of the yield and quality of desirable crop forage plants;
b. Poisoning of livestock;
c. Reduction of native biodiversity resulting in a growing number of threatened, endangered and extinct species;
d. Adverse affects upon human health through allergies, poisoning, and harboring vectors;

e. Degradation of natural aquatic systems including obstruction of water flow in irrigation and drainage systems;
f. Reduction of the value of streams, lakes, reservoirs, oceans, and estuaries for fish and wildlife habitat, and public water supply;
g. High cost of control;
h. Increase in facilities maintenance costs such as power plants, water treatment plants, etc.;
i. Detracting from the aesthetics and recreational value of wildlands, parklands, and other areas;
j. Decreased real estate property value and increased costs of property development; and
k. Competition with or transmission of diseases to wild Pacific salmon or other important marine and aquatic species."

Diversion of Funding from Other Resource Management Activities

An estimated 3.5 million acres of National Forest System lands are infested with invasive weeds, according to the 2000 RPA assessment, which summarized local estimates from individual national forests (USDA Forest Service 2001). However, local estimates vary widely, and the agency lacks a comprehensive inventory. Increasing percentages of budgeted funds must go to inventory and manage invasive weeds on National Forest lands. What makes this one of the Forest Service Chief's "Four Threats" to management of National Forest Lands (Bosworth 2006) is that if adequate funds are not allocated and effectively used to control incipient infestations, our cost to control large infestations will be beyond what we will ever receive from Congress.

Summary

Invasive plants have a profound and far-reaching effect on many aspects of management of public lands, yet their management often receives very little emphasis or funding. Especially in areas where invasive species have not developed large infestations, public land management agencies tend to have a fair degree of apathy regarding their management. It is important to educate the public about invasive plants, but even more important to raise awareness among public land management specialists, including range conservationists, soil scientists, wildlife biologists, and recreation specialists about the effects these plants can have on their resources. And finally, unless land management agencies and policy makers at all levels of government—federal, state, county, and local—recognize this issue and provide funding to deal with it, weed infestations will grow to levels where there simply is not enough funding to control the problem.

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A Demonstration Project to Test Ecological Restoration of a Pinyon-Juniper Ecosystem

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Abstract—*To test an approach for restoring historical stand densities and increasing plant species diversity of a pinyon-juniper ecosystem, we implemented a demonstration project at two sites (CR and GP) on the Grand Canyon-Parashant National Monument in northern Arizona. Historical records indicated that livestock grazing was intensive on the sites beginning in the late 1800s and continuing through the mid 1900s. Repeat aerial photographs (1940 and 1992) indicated recent increases in stand density and encroachment of trees into formerly open areas. Age distributions indicated that the majority of pinyon trees at both sites were less than 100 years of age and juniper establishment appeared to peak in the late 1800s to early 1900s, although some junipers had establishment dates as early as 1700-1725. Pretreatment understory communities were sparse (< 7% total herbaceous cover) as were seedling densities in seed banks (151 seedlings per* m^2 *(14 seedlings per* ft^2*) at CR and 192 seedlings per* m^2 *(18 seedlings per* ft^2*) at GP). Before experimental treatments were implemented, a bark beetle outbreak at GP resulted in >50% pinyon mortality, which was positively related to tree size and age. The demonstration treatment consisted of thinning small trees (< 25 cm diameter at root collar (DRC)), lopping and scattering thinned trees, and seeding native understory species. Thinning and mortality reduced overstory density from 638 and 832 trees per hectare pretreatment (258 and 337 trees per acre) to 280 and 251 trees per hectare (113 and 102 per acre) posttreatment at CR and GP, respectively. Posttreatment densities were similar to those suggested for the late 1800s by dendrochronological stand reconstructions. Thinning small diameter pinyon increased residual quadratic mean diameter (QMD) at CR and the relative importance of juniper at both sites. Live canopy fuels were reduced by treatment at CR and by thinning plus beetle-related mortality at GP. Although thinning slash was lopped and scattered, woody surface fuels were not significantly different between treated and control units at either site, perhaps due to the small size of thinned trees and the large interspace areas into which slash was scattered. Treatment had no immediate effects on herbaceous cover or species richness, both of which may take more time to develop. Further monitoring will help to clearly evaluate the effectiveness of this treatment for satisfying restoration and conservation goals.*

Introduction

In this paper, we report preliminary results of an ecological restoration demonstration project we are conducting at two pinyon-juniper sites in northern Arizona. The project was initiated as a response to ecosystem health concerns described for pinyon-juniper woodlands of the Southwest. On many pinyon-juniper sites, recent ecological changes include decreased diversity of grasses and forbs, decreased biotic and genetic diversity, introduction of exotic species, increased soil erosion,

In: Gottfried, Gerald J.; Shaw, John D.; Ford, Paulette L., compilers. 2008. Ecology, management, and restoration of piñon-juniper and ponderosa pine ecosystems: combined proceedings of the 2005 St. George, Utah and 2006 Albuquerque, New Mexico workshops. Proceedings RMRS-P-51. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

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and decreased site productivity (Dahms and Geils 1997, West 1999, Hastings and others 2003). Many of these deleterious effects appear to be linked to intensive livestock grazing, exclusion of naturally occurring fire, and striking increases in tree density commencing in the late 1800s with the arrival of EuroAmerican settlers and industrial land use practices (Burkhardt and Tisdale 1976, Young and Evans 1981, West 1999, Miller and Tausch 2001). One possible solution for improving ecosystem health and sustainability is ecological restoration.

Several papers dealing with ecological restoration are presented in this proceedings and a brief discussion of definitions is warranted. The Society for Ecological Restoration defines restoration as "the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed" (SER 2002). Although this definition implies reparation of an identifiable ecosystem (i.e., not creation of an entirely new system), it does not tell us how to balance strict focus on historical patterns with uncertainty about those patterns and our desires for improved site conditions. Some scholars contend that "good" restoration must represent both an attempt to reestablish ecological integrity as well as a specific focus on addressing social values and concerns (Higgs 1997). Other basic principles of restoration include the following: a) identify and halt causes of degradation; b) set goals for recovery; c) establish monitoring programs, measures of success, and adaptive processes; and d) plan for the long-term. Finally, all principles must be embedded within a collaborative framework. In this demonstration project, we endeavored to restore historical ecosystem characteristics as well as initiate the development of future conditions desired by land managers.

Study Area

We identified two sites for study on Grand Canyon-Parashant National Monument near Mount Trumbull, Arizona (fig. 1). The Craig Ranch site (CR) is located at latitude 36N 26' 01" and longitude 113W 09' 40". The Goose Ponds (GP) site is located at latitude 36N 24' 46", and longitude 113W 12' 15". Elevation of the sites ranges from approximately 1900 m to 1950 m (6270-6435 ft) (fig. 1). Precipitation averages approximately 50 cm (19.7 in) annually and falls during distinct winter and summer periods. Soils at the CR site are shallow to deep gravelly sandy loams to very cobbly clays derived from limestone, basalt, and sandstone alluvium and colluvium. Those at the GP site are shallow to very deep, very cindery loams derived from alluvial and colluvial, scoriaceous basalt and pyroclastics (USDA Soil Conservation Service 1993, 1995a,b). Vegetation at the sites is classified as Great Basin Cold Temperature Woodland (Brown 1994). Overstories are all-aged mixtures of pinyon pine (*Pinus edulis* Engelm.) and juniper (*Juniperus osteosperma* Torr.). Understory communities generally are sparse and include annual forbs: *Descurainia pinnata* and *Draba* spp.; some perennial grasses: *Bouteloua gracilis*, *Bouteloua curtipendula*, and *Aristida purpurea*; perennial forbs: *Eriogonum* spp. and *Chamaesyce fendleri*; shrubs: *Quercus turbinella*, *Purshia mexicana*, and cacti: *Opuntia erinacea*.

Site Assessment

Observations of the two sites suggested ecological degradation in two main forms: (1) low plant species diversity with communities dominated by dense pinyon and juniper overstories; and (2) reduced soil O horizons, particularly in intercanopy openings. Examination of repeat aerial photos showed that both sites had experienced some degree of overstory densification and tree encroachment

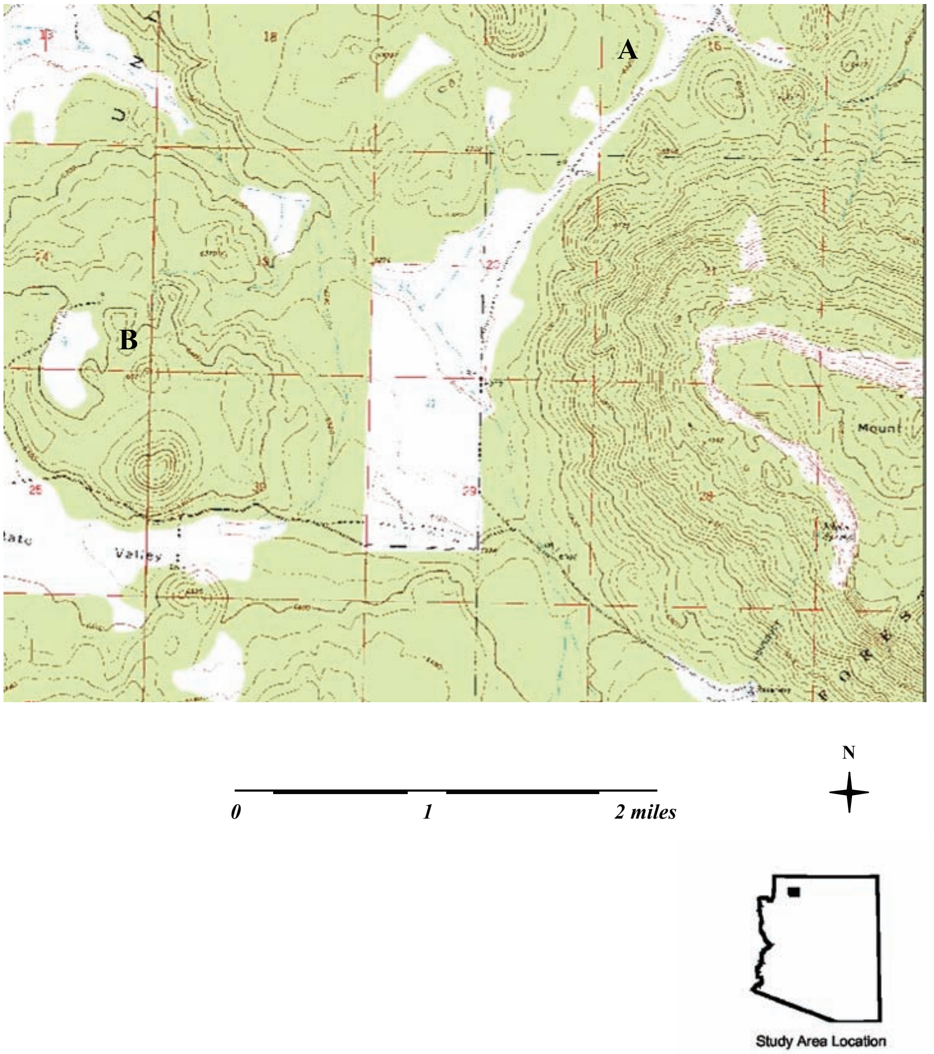

Figure 1—Map of Mount Trumbull area. **A** indicates the location of CR demonstration site and **B** shows the GP site.

from 1940 to 1992. We saw numerous dead shrub remnants on the sites, suggesting recent change in understory community characteristics. We suspected that these conditions may have been in part due to intensive livestock grazing. For example, a 1961 U.S. Forest Service report explained that grass cover, depleted by uncontrolled overgrazing before 1900, had not yet recovered (before 1975 the area was under Forest Service administration) (Unpublished report, BLM District Office, St George, UT). Overuse apparently continued through the 1960s; a range inspection report from 1969 stated that all three allotments in the area were in very poor condition and grass was almost 100% utilized each season (unpublished report, BLM District Office, St George, UT). Repeat aerial photographs indicated that a water catchment was developed at the GP site between 1940 and 1992 (fig. 1) and historical maps indicated that a pipeline was built to provide water to livestock near the CR site. Intensive grazing likely reduced native plant species diversity and impacted soil quality but its effect on natural disturbance patterns is less clear. Field observations at the two study sites revealed some charred wood; however, details regarding the fire history of the two sites are not known.

Measurements

At each site, we used aerial photographs and topographic maps to delineate a 9-ha (22.2-acre) area that was relatively homogenous in terms of overstory density, slope, and aspect. We divided these areas into two 4.5-ha (11.1-acre) units per site and randomly assigned units to receive restoration treatment or remain as a control. We established six 0.04-ha (0.1 acre) circular sampling plots on a 60-m (196.8 ft) grid in each unit. For long-term monitoring purposes, we used steel rebar to mark plot centers and georeferenced these points.

In 2002, we measured overstory, understory, and surface fuels characteristics on each plot. We measured diameter at root collar (DRC) for all live and dead trees and total height for all live and standing dead trees. We collected increment cores from all trees ≥20 cm DRC and from a 20% random subsample of smaller trees (< 20 cm DRC). Increment cores collected in the field were brought back to the laboratory, mounted on wooden slats, sanded, and crossdated (Stokes and Smiley 1968). We tallied dead tree structures (i.e., snags, logs, stumps) by condition class as described by Thomas and others (1979) and Maser and others (1979) for ponderosa pine. Tree seedlings (< 1.37 m (4.5 ft) in height) and shrubs were tallied on subplots (100 m^2; 0.025 acre) within the main plot. We used 10 quadrats (1m^2; 10.8 ft^2) per plot to sample cover of herbaceous species. We measured surface fuels and forest floor depth on planar transects (15.24 m; 50 feet) according to Brown (1974).

In 2002, a severe drought and bark beetle (*Ips confusus*) outbreak occurred. In 2003, before restoration treatments had been implemented, we resampled overstory structure on the plots to assess tree mortality. Signs of beetle presence (e.g., frass or pitch tubes) were noted. After thinning and spring seeding (see **Treatment**), we resampled overstory structure, regeneration, shrubs, fuels, and herbaceous understory (June 2004).

Pretreatment Conditions

We relied on assessment of present stand conditions and historical reconstruction to develop a restoration treatment prescription. Pretreatment measurements in 2002 indicated dense forest conditions at both sites (table 1). At the CR site,

Table 1—Overstory characteristics[1] at Craig Ranch (CR) and Goose Ponds (GP) demonstration sites in 2002 and 2004.

			TPH[2]			BA (m^2/ha)[5]			QMD (cm)[6]	
Site	Unit	Date	JUOS[3]	PIED[4]	Total	JUOS	PIED	Total	JUOS	PIED
CR	Control	2002	580	313	893	33.9	7.2	41.1	26.9	16.6
	Treated		387	251	638	25.9	5.1	31.0	29.4	16.2
	Control	2004	568	304	872	33.5	7.0	40.5	26.9	16.7
	Treated		206**	74**	280**	23.0	3.7	26.7	38.3*	26.8**
GP	Control	2002	144	498	642	14.5	8.3	22.8	35.1	14.7
	Treated		239	593	832	19.5	7.1	26.6	34.5	13.4
	Control	2004	144	267	411	14.5	3.1	17.6	35.1	11.5
	Treated		156	95	251	18.4	1.4	19.8	39.3	13.5

[1] Asterisks denote statistically different means for Control versus Treated conditions in 2004; * $P < 0.05$; ** $P < 0.01$.
[2] Trees per hectare (divide by 2.47 for trees per acre).
[3] Juniper (*Juniperus osteosperma*).
[4] Pinyon (*Pinus edulis*).
[5] Basal area (divide by 0.2296 for ft^2 per acre).
[6] Quadratic mean diameter measured at root collar (divide by 2.54 for inches).

juniper was dominant in the overstory in terms of number of trees (61% of TPH) and BA (83% of BA). At the GP site, juniper trees were outnumbered (29%) by pinyon but made up a greater proportion of the total basal area (73%). Cumulative age distributions derived from increment core analysis showed the majority of pinyon trees at both sites were less than 100 years old and a notable spike in their establishment occurred after 1950, particularly at the GP site (fig. 2). Juniper cores were often difficult to crossdate against known tree-ring chronologies. For such samples, we approximated tree age by conducting ring counts. Juniper at both sites had establishment peaks corresponding to the late 1800s to early 1900s (fig. 2). At the CR site, some junipers established as early as 1700-1725. At the GP site, we found no junipers that had established before 1800. Juniper seedlings (individuals less than 1.37 m (4.5 ft) in height) averaged 117-200 per ha (47-81 per acre) at the CR site and 50-117 per ha (20-47 per acre) at the GP site. Pinyon seedlings averaged 900-967 and 617-1433 per ha (364-391 and 250-580 per acre) at the CR and GP sites, respectively.

We estimated presettlement stand density for each site using age data from increment cores. Trees with center dates less than 1875 were considered presettlement in origin. To account for additional trees that died between 1875 and 2002, we included dead structures (i.e., snags, logs, and stumps) greater that 25 cm DRC in our presettlement density estimates. At the CR site, the number of trees estimated to exist in 1875 was 261 TPH (102.7 TPA) (table 2). Approximately 75% of these presettlement trees (live and dead) were juniper (196 TPH (79 TPA)). At GP, estimated presettlement density was 104 TPH (42 TPA) and approximately 60% of these trees were juniper (table 2).

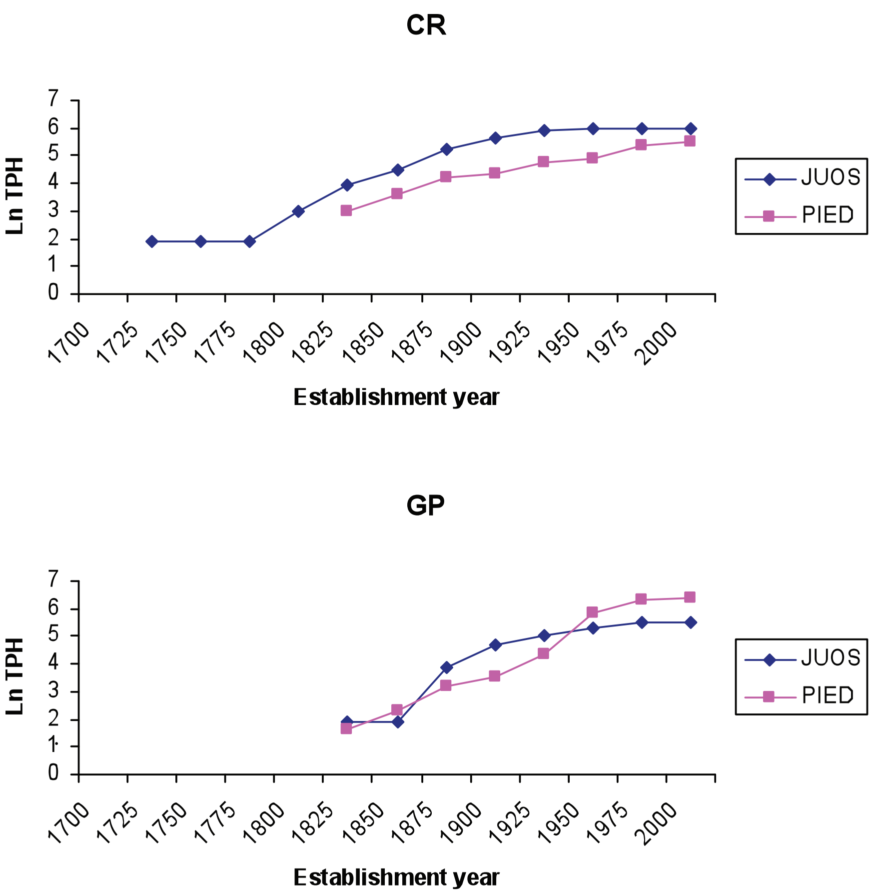

Figure 2—Cumulative establishment distributions for juniper (JUOS) and pinyon (PIED) at the CR and GP demonstration sites. Steeper areas of curves indicate periods of more rapid tree establishment.

Table 2—Number of trees per hectare (TPH) of juniper (JUOS) and pinyon (PIED) at three points in time: reconstructed to 1875, pretreatment in 2002, and posttreatment in 2004.

		Year					
		1875		2002		2004	
Site	Species	TPH	(%)	TPH	(%)	TPH	(%)
CR Treated	JUOS	196	75	387	61	206	73
	PIED	65	25	251	39	74	27
	Total	*261*		*638*		*280*	
GP Treated	JUOS	63	60	239	29	156	62
	PIED	41	40	593	71	95	38
	Total	*104*		*832*		*251*	

Understory vegetation was sparse at both sites (table 3). Mean cover was less than 7% at the CR site and less than 4% at the GP site. Species richness averaged 11.2-12.8 species per m^2 (10.8 ft^2) at CR and 4.2-4.7 at the GP site. Commonly occurring species at CR in 2002 included *Cordylanthus parviflorus, Draba* sp. (annual forbs), *Aristida purpurea*, *Bouteloua curtipendula*, *Bouteloua gracilis* (perennial grasses), *Arabis fendleri*, *Calochortus nuttallii* (perennial forbs), and *Opuntia erinacea* (cactus). Only one exotic species was found; *Lactuca serriola* occurred on ~3% of the quadrats in the treated unit. At GP, perennial forbs were most common, particularly *Chaenactis douglasii*, *Mirablis oxybaphoides*, and the exotic, *Marrubium vulgare*. Two other exotic species were found: *Bromus tectorum* on ~2% of the quadrats in both units and *Salsola tragus* on ~3% of quadrats in the control unit.

Woody surface fuels were minimal at both sites (table 4). Combined 1-hour and 10-hour fuels averaged 1.2 Mg/ha (0.53 T/acre) at the CR site and 1.7 Mg/ha (0.76 T/acre) at the GP site. Total forest floor depth at both sites averaged less than 1.3 cm (0.5 in). Total live canopy fuels averaged near 7.5 Mg/ha (3.3 T/acre) at both sites (table 4).

Table 3—Understory characteristics at Craig Ranch (CR) and Goose Ponds (GP) sites in 2002 and 2004 at Grand Canyon-Parashant National Monument.

Site	Unit	Date	Cover (%)	Richness[1]	Diversity[2]
CR	Control	2002	6.6	12.8	1.28
	Treated		5.1	11.2	1.11
	Control	2004	5.54	12.5	1.55
	Treated		3.68	13.7	1.6
GP	Control	2002	3.46	4.7	0.46
	Treated		1.15	4.2	0.46
	Control	2004	5.18	12	1.49
	Treated		2.85	10.7	1.29

[1] Richness is number of species per m^2 (divide by 0.0929 for number per ft^2).
[2] Shannon-Weiner's index of diversity.

Table 4—Fuels characteristics on control and treated units in 2002 and 2004 at the Craig Ranch (CR) and Goose Ponds (GP) demonstration sites at Grand Canyon-Parashant National Monument. Shown are litter and duff depths, surface fuel weights by moisture timelag class, and live canopy fuels.

			Depth (cm)		Surface Fuels (Mg/ha) [1]						Live Canopy (Mg/ha) [2]		
Site	Unit	Date	Litter	Duff	1H	10H	100H	1000HR	1000HS	Total	JUOS[3]	PIED[4]	Total
CR	Control	2002	0.4	0.4	0.6	1.5	2.9	0.0	0.0	5.0	7.93	2.97	10.9
	Treated		0.3	0.4	0.4	0.8	0.0	1.1	0.0	2.4	5.58	2.11	7.7
	Control	2004	0.2	0.4	0.7	1.2	3.8	0.0	0.0	5.7	7.82*	2.88	10.7*
	Treated		0.4	0.3	0.7	0.3	6.7	0.0	7.8	15.5	4.53	1.39	5.9
GP	Control	2002	0.1	0.8	0.7	1.6	1.4	1.8	2.3	7.8	2.88	3.71	6.6
	Treated		0.2	1.0	0.4	1.3	0.0	6.3	0.6	8.6	3.98	3.32	7.3
	Control	2004	0.2	0.7	0.8	1.9	2.4	3.0	1.8	9.9	2.88	1.51	4.4
	Treated		0.4	0.9	1.0	2.3	4.8	1.4	0.9	10.3	3.46	0.62	4.1

[1] Multiply Mg/ha by 2.4 for approximate tons per acre.
[2] Estimated using allometric equations provided by Grier and others (1992). Asterisks indicate statistically different means between treated and control units at P < 0.05. Biomass is foliage plus fine twigs.
[3] Juniper (*Juniperus osteosperma*).
[4] Pinyon (*Pinus edulis*).

Restoration Goals

Based on preliminary site assessments and analysis of pretreatment stand conditions, we developed the following restoration goals: (1) reduce overstory density to near presettlement levels; (2) reestablish historical overstory species composition; (3) reduce severe fire hazard; (4) reduce bark beetle susceptibility; and (5) reestablish a diverse understory. These goals represented an integration of reestablishing historical patterns of overstory structure and composition, a desire to increase biological diversity in understory communities, and a desire to protect important attributes such as old pinyon and juniper trees.

Treatment Prescription

Our restoration treatment approach entailed thinning trees to low densities, scattering slash, and seeding with native grasses. Specifically, we implemented the following prescription: (1) thin pinyon and juniper trees less than 25 cm (9.8 in) DRC, except for trees retained to replace presettlement evidence (i.e., dead tree structures >25 cm DRC) at a 2:1 ratio by species; (2) lop slash to 1 m (3.3 ft) or less in length and scatter material to cover bare soil; (3) seed with native plant species. Using tree increment cores, linear regression of establishment date and DRC data suggested that pinyon pine trees less than 25 cm DRC were likely to be less than 130 years of age and therefore postsettlement in origin (Establishment Date = 1977.12 - 4.25*(DRC); $R^2 = 0.57$; $P < 0.001$). Age-diameter relationships for juniper were poor ($R^2 < 0.15$). Retaining two postsettlement-aged trees to replace each dead presettlement structure was used as a conservative approach to restoring historical densities and also allowed for posttreatment mortality. Selection of replacement was based on species, size, form, and proximity to structure being replaced. Thinning was completed November 2003.

Selection of native plant species for seeding was based on observations of local occurrence, baseline data from belt transects, and community data reported in relict site literature (Mason and others 1967, Schmutz and others 1967, Thatcher and Hart 1974, Madany and West 1984, Rowlands and Brain 2001). We selected five grasses: *Bouteloua curtipendula*, *B. gracilis*, *Elymus elymoides*, *Oryzopsis hymenoides*, and *Sporobolis cryptandrus*; one forb: *Lupinus argenteus*; and four shrub species: *Amelanchier utahensis*, *Atriplex canescens*, *Ephedra viridis*, and *Rhus trilobata*. We used hand seeders to broadcast at a rate (18 kg/ha (16 lb/acre)) that approximated common standards for range rehabilitation (Clary 1988). However, we chose to seed half the amount in early spring and half in late summer in order accommodate germination and establishment requirements for both cool and warm season species. Using site preparation methods such as plowing or disking before seeding was not feasible. Similarly, we did not harrow or rake the restoration units after the seed was broadcast, but instead utilized thinning slash to provide cover and mulch for the seeds.

Data Analyses

We used logistic regression to test (alpha = 0.05) for relationships between probability of beetle-related pinyon mortality and tree height, DRC, age, and basal area growth at the GP site. We used Student's t-test to compare (alpha = 0.05) forest structure and understory characteristics at each of the two sites for pretreatment (2002) differences. When significant differences were found, we used analysis of covariance (ANCOVA) to test posttreatment differences with pretreatment conditions as a covariate. When no pretreatment differences were found, t-tests were used to compare posttreament means. Tree canopy biomass was estimated using allometric equations for pinyon (*Pinus edulis*) and juniper (*Juniperus monosperma*) provided by Grier and others (1992). We estimated surface fuel loading using equations provided by Brown (1974) and coefficients for ponderosa pine (*Pinus ponderosa*) fuels provided by Sackett (1980); no pinyon or juniper fuels coefficients were available in the published literature. Differences in biomass and fuels means between control and treated units were analyzed as described above for overstory characteristics.

Bark Beetle Effects

Bark beetle-related mortality of pinyon trees in 2003 was significantly related to tree size and growth. Probability increased with increasing height, DRC, and age. Mortality was less likely for trees that showed high relative BA growth. Mean height of beetle-killed trees was 4.2 m (13.8 ft) whereas surviving trees averaged 3.3 m (10.8 ft). Mean DRC was 12.3 cm (4.8 in) and 10.0 cm (3.9 in) for beetle-killed and surviving trees, respectively. Mean age of beetle-killed trees was 89 whereas surviving trees averaged 64 years. Beetle-killed trees showed a mean decrease of 15% in BA increment over the 10 years before mortality.

Treatment Effects

Implementation of the restoration thinning prescription significantly altered overstory structural characteristics at the CR site but did not affect those at the GP site, largely because of the greater impacts of beetle-related mortality (table 1). Thinning trees smaller than 25 cm DRC—while replacing dead structures greater than 25 cm DRC—reduced the number of juniper trees by nearly one-half and the number of pinyon by more than a factor of three at CR. Thinning at GP reduced the mean number of junipers by 83 TPH (33.6 TPA) but this did not result in a significant difference between control and treated conditions (table 1). Bark beetle-related mortality resulted in statistically similar pinyon densities between the control and treated units at GP (table 1). Basal area was not significantly affected by thinning treatment at either site (table 1). Diameter distributions, however, showed that dominance of small pinyon at CR was decreased by thinning. This was expressed as significantly greater QMD of both juniper and pinyon in the treated unit compared with the control at CR (table 1). At the GP site, diameter distributions were affected by both thinning and beetle-related mortality and no significant differences in QMD were found between the control and treated units (table 1).

Stand densities after thinning at both the CR and GP restoration units were similar to presettlement estimates (table 2). The total number of trees remaining after restoration thinning at CR was 280 TPH (113 TPA) and juniper comprised approximately 73%. There were 251 TPH (101.6 TPA) remaining after treatment at GP and juniper made up 62% of the residual number.

No significant differences in herbaceous plant cover, species richness, or diversity were detected between control and treated units at either site in June 2004 (table 3). Species composition was similar to pretreatment conditions. Two exotic species were observed at the CR site (treatment and control): *Lactuca serriola* (3% frequency) and *Tragopogon dubius* (2% frequency). Five exotic species were observed at the GP site. *Bromus tectorum* frequency in the GP treated unit was 1.7% in 2004 whereas it was not observed on these plots in 2002.

Thinning increased surface fuel loading at both sites, particularly for moisture-lag classes greater than 10-hours (table 4). Differences between control and treated units, however, were not statistically significant at either site. Changes in forest floor litter and duff depths due to treatment were minimal and remained low after treatment (table 4). Canopy biomass was significantly reduced by thinning at the CR site (table 4). Due to beetle-related mortality that occurred in the both treated as well as the control units, no significant differences were found in live canopy biomass at GP site in 2004 (table 4).

Discussion

Various lines of evidence, including historical and contemporary aerial photographs, diameter and age distributions, and dendrochronological reconstructions indicated a transition at both study sites from previously more open stand conditions existing in the late 1800s to closed conditions found at the site in 2002. At the CR site, the number of trees in 2002 was more than twice the number estimated to be present in 1875. This difference was even more dramatic at the GP site where 2002 density was greater than the estimated presettlement number of trees by a factor of eight. In addition to changes in overstory density, both sites appeared to be moving toward increased importance of pinyon relative to juniper. Large junipers were present at both sites and this was reflected in greater BA in comparison to pinyon. Age data suggested that peak juniper establishment was around 1875-1900 at CR and 1875-1925 at GP whereas pinyon establishment appeared to peak around 1950 at both sites. It should be noted that precise crossdating of juniper is difficult and these establishment dates are best considered as approximations. Pinyon seedlings outnumbered juniper by a factor of four or more at both sites.

Comparable postsettlement changes have been described on pinyon-juniper sites throughout the Southwest and Great Basin (Blackburn and Tueller 1970, West and others 1975, Tausch and others 1981, Jacobs and Gatewood 1999, West 1999, Romme and others 2003, Landis and Bailey 2005). For example, on four black sagebrush (*Artemisia nova*) sites in Nevada, Blackburn and Tueller (1970) concluded that juniper (*J. osteosperma*) initially invaded open sage communities whereas pinyon became more prevalent as overstory densities increased. Age distributions indicated that juniper trees were present as early as 1725 but establishment began to dramatically increase around 1850 for juniper and 1920 for pinyon. Similarly, Tausch and others (1981) reported that increases in tree dominance since the early 1800s on eastern Nevada and western Utah sites were driven by pinyon establishment. Factors responsible for driving these structural changes include relaxation of interspecific competition due to intensive grazing, increases in woody vegetation ("nurses" for pinyon establishment—see below), fire exclusion due to livestock grazing (removal of fine fuels) and active suppression, warmer, moister climatic patterns of the late 1800s to early 1900s, and recent increases in atmospheric CO_2 (Leopold 1924, Burkhardt and Tisdale 1976, Young and Evans 1981, West 1999, Miller and Tausch 2001, Romme and others 2003).

At the GP site, a bark beetle outbreak reduced overstory density of pinyon and this could be interpreted as a natural disturbance that counteracts recent increases in density and provides restoration benefits. Although this is true in part, at the GP site beetles preferentially attacked larger, older pinyon—elements of the historical stand conditions that are desirable to retain for conservation (e.g., wildlife habitat) and multi-resource reasons. Further, high density of standing dead pinyon may represent increased fire hazard for one or two years while dead needles remain on the tree.

Similar patterns of beetle activity were found by Negrón and Wilson (2003) who reported that tree diameter (DRC) and mistletoe infestation were good predictors (72% correct classification) of beetle attack on pinyon near Flagstaff, Arizona. Stand density was also positively related to beetle attack. In order to reduce the probability of beetle-related mortality in pinyon-juniper woodlands, reducing pinyon stand density index (SDI; Reineke 1933) to values of 50 or less are recommended (Negrón and Wilson 2003). Thinning at the CR site reduced pinyon SDI to 81 from 122 in 2002. Thinning and mortality reduced SDI at the GP site to 34 from 212 in 2002. In the control unit, SDI was 75 in 2005.

The restoration prescription implemented at the two sites appeared to be effective in reduce stand density to levels similar to those suggested by dendrochronological reconstructions. The majority of trees removed from both sites were small pinyon (table 2); this increased the relative importance of juniper and restored overstory composition to characteristics similar to those existing in the late 1800s. Selective tree removal using chainsaws is preferable to indiscriminate techniques such as anchor chaining or cabling that may cause substantial soil disturbance and stimulate regeneration of juniper (Jacobs and Gatewood 1999, Brockway and others 2002). The approach we tested conserves all large trees. Clearer description of historical patterns could be provided by reconstruction models that utilize dendrochronological information, tree death date predictions (i.e., decay rates, harvesting records, insect outbreak dates, etc.), and back-growth equations. Fire history information also would be helpful in developing restoration prescriptions for closely emulating historical conditions. Presettlement fire characteristics in pinyon-juniper woodlands are poorly understood (Romme and others 2003, Baker and Shinneman 2004) and we are currently pursuing fire history research on other northern Arizona and New Mexico sites.

In our study, no significant differences in herbaceous parameters were found between the treatment (slash additions and seeding) and control units. Posttreatment measurements, however, were conducted just four months after seeding was completed. Further monitoring is necessary before conclusions can be made regarding the effectiveness of treatment on increasing understory abundance and species richness.

Thinning significantly reduced live canopy biomass as compared with the control unit at the CR site. This represents a decrease in crown fire hazard, although we did not attempt to model potential fire behavior. At the GP site, bark beetle-related mortality decreased live canopy biomass although trees killed by beetles in the control unit will retain their needles for one or two years and create an increased hazard. Thinning slash lopped and scattered at both sites did not significant affect woody surface fuel loads. This may reflect the combined influences of thinning only small trees and large interspace areas that were targeted for slash dispersal.

Other pinyon-juniper restoration experiments have tested slash dispersal and seeding treatments to promote understory recovery (Jacobs and Gatewood 1999, Brockway and others 2002). Results have been variable. For example, Jacobs and Gatewood (1999) found that lopping and scattering slash into interspaces substantially increased herbaceous cover at two sites in northern New Mexico, although seeding did not significantly contribute to the increases. In contrast, Brockway and others (2002) reported positive effects of tree removal on grass cover but no significant differences between slash removal and slash dispersal treatments at a site in central New Mexico. Stoddard (unpublished) reports finding significant increases in plant cover on plots with slash and seed additions compared with control plots in interspaces adjacent our demonstration units at Mount Trumbull. Thinning slash that is scattered into degraded interspaces may increase rates of seedling establishment by altering microsite conditions. Some of these changes may include dampening of soil temperature fluctuations and extremes, increasing soil moisture content, providing structure that traps seeds, and reducing erosion (Jacobs and Gatewood 1999, Hastings and others 2003, Stoddard and others 2005).

Conclusions

Although there is still much that is not known regarding historical conditions at the two demonstration sites, the restoration treatment implemented appeared to be effective at reestablishing overstory characteristics more similar to those of the late 1800s. Further monitoring of understory changes is needed. The treatment also reduced fuel hazard and conserved large, old pinyon and juniper trees. The permanent plots established will allow further assessment of ecosystem recovery and assist in the adaptive management process.

Acknowledgments

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Removal of Pinyon-Juniper Woodlands on the Colorado Plateau

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***Abstract**—Pinyon-Juniper (PJ) woodland is the 3rd largest vegetation type in the United States, covering 35.5% of the Colorado Plateau, it is the largest vegetation type administered by the Bureau of Land Management (BLM) on the Colorado Plateau. These woodlands have been increasing dramatically in density and extent over the last 100 years. In response to this increase and a concomitant loss of grasslands, the BLM has been actively treating these lands by removing pinyon and juniper and seeding with desired vegetation since the 1950s. To date, over 700 treatments with varying methods have been applied across ~700,000 acres of PJ woodland on BLM lands. Recognizing the value of historical data as a reference for future land management decisions, in 2000 the BLM Colorado Plateau Science Committee adopted a regional Pinyon-Juniper Management Strategy outlining the need for collecting and synthesizing current regional information in order to better understand PJ communities, the dynamics of encroachment, and long-term management. The Merriam-Powell Center for Environmental Research (MPCER) at Northern Arizona University (NAU) collaborated with the BLM to archive all past pinyon-juniper removal and re-vegetation treatment projects the BLM has completed across the Colorado Plateau. The PJWOOD website contains both relational and GIS databases of treatments that will facilitate research, landscape level analyses, and regional land management planning. http://www.mpcer.nau.edu/pj/pjwood/index.htm*

In: Gottfried, Gerald J.; Shaw, John D.; Ford, Paulette L., compilers. 2008. Ecology, management, and restoration of piñon-juniper and ponderosa pine ecosystems: combined proceedings of the 2005 St. George, Utah and 2006 Albuquerque, New Mexico workshops. Proceedings RMRS-P-51. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

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Mesa Prescribed Fire

Scott Glaspie[1] and Erik Rodin[1]

Abstract*—The Mesa project is a landscape scale restoration treatment for a pinyon/juniper woodland. Primarily, the goal of the project is to reduce encroaching woodland on savanna grasslands, thus improving rangeland and wildlife habitat. This project is financed by several non-profit wildlife organizations.*

In: Gottfried, Gerald J.; Shaw, John D.; Ford, Paulette L., compilers. 2008. Ecology, management, and restoration of piñon-juniper and ponderosa pine ecosystems: combined proceedings of the 2005 St. George, Utah and 2006 Albuquerque, New Mexico workshops. Proceedings RMRS-P-51. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

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Mechanical Treatment Methods in the Piñon-Juniper Type

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Abstract—*Mechanical treatments in the piñon-juniper type are driven by the objectives of the treatment. Objectives of restoring meadows or grassland from encroachment or returning to a savanna can require a different method of treatment than thinning piñon-juniper woodlands. Often these types of objectives are intermingled within a treatment area as site characteristics vary across the landscape. Characteristics such as tree/shrub density and size, soils (or the lack of), slope, terrain, species, and selectivity of treatment method can determine the type of mechanical treatment(s) suited for an area. Management of the area following the treatment also can determine the type of mechanical method utilized. Post-treatment management of the site (need for soil/site stabilization, grazing management, wildlife habitat, etc.) and biomass material produced from treatments can define the needs of the mechanical treatment. Examples of variation of treatments are piling/windrowing material for burning, preparation for broadcast burning, lop and scatter, firewood harvesting, biomass harvesting for energy or other products, or maintenance. Treatment methods are presented including positive and negative attributes; selectivity/sensitivity to resource values and objectives; post-treatment management options; and relative or approximate costs.*

Community Forestry and Cultural Resources

Photo by Gerald Gottfried

Ecology and Management of Pinyon-Juniper Ecosystems in the Bureau of Indian Affairs Southwestern Region

John Waconda[1]

***Abstract**—The large acreages of the woodland forest cover type on tribal lands continues to post challenges to this agency's overall management strategies. The development of management plans, evaluation of growth study plots, and anticipated biomass utilization can help resolve some of the challenges.*

Overview of the BIA-SW Region Woodland Management Program

BIA/Tribal Forest Resources in the Southwest Region

The Southwest Regional office of the Bureau of Indian Affairs (BIA) is located in Albuquerque, NM. Twenty-six Native American tribes reside in New Mexico, Southern Colorado and Texas. Total tribal lands encompass 4,832,161 acres, of that 2,805,545 (58%) are designated as forest acres, and 1,946,873 (69% of forest land) acres are considered as woodland forested acres. Tree species include: pinyon pine (*Pinus edulis*), and Utah (*Juniperus osteosperma*), Rocky Mountain (*J. scopulorum*), oneseed (*J. monosperma*), and alligator (*J. deppeana*) junipers. The Bosque riparian zone is also included within our woodland acres; therefore cottonwood (*Populus fremontii*) is part of the species mix.

The management of such vast acreages is accomplished by foresters, technicians, and/or natural resource officers working at BIA facilities located at specific sites within the region. Approximately 1.3 million acres are considered as manageable, accessible acres.

Woodland Management Planning

Currently 11 tribal management plans are either in place or are in the process of being approved. Also, an additional 11 plans are being revised or developed. In order for a tribe to manage woodlands for any set goal, a current management plan is required. Without one, the agencies can only provide minimal assistance towards the management of Native lands, such as, fire management measures, trespass protection and prosecution, insect and disease control, to name a few. Therefore, it is the goal of this Region to update all management plans.

On-going efforts to quantify woodland forest on Native lands have enabled this region to initiate and complete a Woodland Continuous Forest Inventory for one of the larger Native Indian reservations comprised of approximately 126,397 acres of unreserved, accessible, commercial woodland forest. Using ArcGIS, a

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grid was superimposed over the reservation and points were established over commercial stands. Using this information and the CFI field manual for the Southwest Region Reservations, the project is now complete. For analysis purposes the BIA, Branch of Forest Resource Planning (Central Office, Washington, DC) will begin to merge this data with existing data allowing the agency planners to begin modeling for growth, disease and insect infestation, vigor, species numbers, and other information.

Woodland Growth and Yield Studies

About ten years ago, the Southwest Region established the Uneven-aged Woodland Growth Study on four Native Indian reservations within New Mexico and Colorado. The purpose of these plots was to determine the optimal growing stocking levels (in square feet of tree basal area per acre) for different productivity regimes and/or habitat type groups. These growth study plots consist of 2.4 acre (4 X 6 chains) rectangular plots cut to specific residual basal area levels. Within each plot fifteen regeneration fixed plots were also established. These plots were re-measured in 2004 and the data are being processed at our Washington Office. As soon as the data are made available, they will be analyzed with the help of Gerald Gottfried, a research forester with the U.S. Forest Service.

On Zuni tribal lands where one of the growth study plots are located, an ambitious tribal wildlife biologist, Steve Albert (1994), decided to study the use of the plots by deer, small mammals, and songbirds. In summary, a number of species present increased significantly on the treatment plots. Composition of the plant species also changed. Deer use increased in correlation with the amount of trees removed. Overall, small mammal abundance increased on all treated plots. His results indicated that "small thinnings in pinyon-juniper woodlands have less drastic effects on wildlife than chainings and are a viable management tool for multiple-resource managers."

Wood Utilization

For several years, the Division of Forestry and Fire has sought options to deal with challenges in finding viable utilization opportunities for vast amounts of small diameter timber and thinning material generated as a result of forest treatments and fuels management projects on many of the region's tribal forested reservations. Forest stand improvement treatments and fuels reduction projects, taking place across the region, generate vast amounts of nonmerchantable woody material that continue to be a fire hazard problem unless removed from the treated site.

This region is lacking typical value-added forest product processing or utilization industries; the woody material must be removed or disposed of on-site though various secondary processes (i.e. burning, mulching, or trampling). These secondary processes are either environmentally sensitive and/or cost prohibitive which does not lend themselves to reducing treatment costs nor promoting their implementation. In seeking a solution to this large dilemma, one of the options considered has been to remove and utilize the material and capture minimal value to generate heat and/or electricity for local consumption. In essence, this low-value woody material would become the fuel for heat/energy production, a process commonly termed biomass renewable energy.

The production of heat/energy options has the potential to provide other beneficial aspects that promote its study and possible development on tribal lands in the Southwest. Possible benefits include producing heat/energy for local tribal consumptions thereby reducing energy costs, possible heat/energy production for BIA owned and /or managed facilities, and providing biomass fuel material to other entities reducing costs for forest hazard treatment. Accordingly, forest hazard treatments and other processes will create additional jobs and stimulate local tribal economies, and provide opportunities for tribes to become less energy dependent utilizing tribally derived renewable resources. Currently the Southwest Region, Forestry and Fire section is supporting the research of the Forest Service's Southwestern Region's mensurationist to quantify volume inherent in small diameter sized classes. The current direction is to (with help from the Forest Service's Southwestern Region) quantify volume of small diameter trees on tribal lands for anticipation of biomass industries.

Management Projects

Pinyon-juniper health projects can only be accomplished through the intervention of other programs due to the limited funds within the woodland program. Both tribal, Bureau, and other federal programs have allowed for on the ground accomplishments within the woodland type. Most of the tribes within this region are reaching out more to other government agencies for assistance, and to private entities for addition funds. Other federal programs include EPA, U.S. Forest Service; without their connection to the tribes, it would have been difficult to get some of the work done.

Rehabilitation projects, especially along the Bosque would have been minimal to non-existent without the intervention of the Burned Area Emergency Stabilization and Rehabilitation program to reduce the impact of wildfires.

The attitude of stewardship of their lands for present and future members has influenced the Native Indian tribes who are actively pursuing the direction of management of their woodland forests and other natural resources. This can be detected by the reorganization of internal programs (i.e., natural resources and environmental protection agency departments are being staffed by professionals holding degrees in wildlife management, forestry, hydrology, and GIS specialties). These not only aid the tribes but also complement what the BIA is trying to accomplish, especially toward the management of woodland acres. Some of the priorities that the tribes have completed or are near completion pertain to wildlife habitat enhancement, reducing density levels of current woodlands stands from 100 square feet or more of basal area to residuals of 20 to 30 square feet, replanting of native willows along riparian zones, and transporting fuelwood to tribal wood yards.

Hazardous Fuel Reduction Program

The BIA Hazardous Fuels Reduction Program—with its priorities to focus on wildland-urban-interface and hazardous fuels projects—has not only reduced the potential risk from wildfires, but also has helped enhance the woodland forests. When fuels projects are proposed within pinyon-juniper forest types, it is to the advantage of agency personnel to work closely with fuel reduction specialists to foster forest management strategies such as, application of uneven-aged silvicultural systems using individual tree or group selection methods, and providing a means (woodland program funds) to help remove a majority of the fuelwood before the mastication process begins.

Wildfires have had an enormous impact to both standing commercial timber and woodland forests within Native lands. Within dense stands of the pinyon-juniper types, thousands of acres have completely burned leaving "moonscapes" and hydrophobic soils, which may open the area for invasive weed species to invade and dominate these burned over areas. With the aid of the fuels program, this type of scenario will hopefully become less likely to occur.

Insect Mortality Assessment

The mortality of pinyon pine throughout the Southwest by the pinyon ips (*Ips confusus*) is very evident on tribal lands. During the months of March through September 2003, the USFS completed a comprehensive aerial assessment of approximately 6.6 million acres of pinyon-juniper woodlands. Initially, the total tribal acres affected by the *Ips* beetle were 29,265 acres for the State of New Mexico and 285,438 acres for the State of Colorado.

Tribal members were very concerned about the high mortality of pinyon trees. Two of the Northern Pueblo tribes requested funds through the woodland program to establish plots to assess the amount of remaining live trees in the forests. Trees in all size classes from seedlings to larger trees were sampled. On the San Ildefonso and Picuris Indian lands approximately 51 and 95 percent, respectively, of the pinyon were still alive. These two examples depict varying degrees of mortality that occurred within this cover type. There were other instances where only juniper trees are left standing, and places where pinyon pine is still thriving and growing well.

Conclusion

The Native American has always used, and will continue to use, these forest types as a place to acquire food sources and building materials. However, most important are the cultural and traditional values associated with the woodland forests.

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Diablo Trust Piñon-Juniper Restoration Sites: Restoring Structure to Woodlands and Savannas

Andrew Gascho Landis[1] and John Duff Bailey[1]

Abstract*—Piñon-juniper restoration sites are being implemented in northern Arizona on lands managed by the Diablo Trust that have experienced increased piñon and juniper densities. Such land managers want to restore basic ecosystem structure and function to their lands in a way that preserves their livelihoods and open space in the region. The first objective of this project was to create reference conditions by reconstructing age structure and spatial arrangement to 1860, prior to livestock grazing, across three soil types. Stand reconstruction using stem mapping and ring counts revealed pulses of juniper establishment between 1860 and 1880 on both basalt and sandstone-derived soils. Limestone-derived soil showed no increase during this time period, but rather steady increase in tree density since approximately 1700. Juniper trees in 1860 across all soil types were found in clumps of a minimum radius of 15 m. Presettlement diameter distributions and basal area were reconstructed and used to develop structure control (e.g., BDq) prescriptions to best approximate stand conditions of both piñon-juniper savannas and woodlands. After these appropriate density and distributions were determined, the next phase of this project is to test restoration methods, which include burning, seeding, and slash treatments (leave tree, pile, lop and scatter) to determine which treatment fosters greatest understory plant response.*

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Restoration of Juniper Savanna on the Pueblo of Santa Ana, Sandoval County, New Mexico

Glenn Harper[1]

***Abstract**—The Pueblo of Santa Ana (Pueblo) is located in north central New Mexico within southeastern Sandoval County, about 15 miles north of Albuquerque and 45 miles south of Santa Fe. The Pueblo encompasses approximately 79,000 acres of trust lands. Between 1999 and 2001, the Pueblo of Santa Ana Department of Natural Resources, through a United States Environmental Protection Agency grant, collected quantitative and qualitative baseline ecological data from 145 long-term monitoring plots established across the Pueblo. These data were used to develop a plant association and watershed assessment that facilitated the Pueblo's ability to identify priority natural resource concerns and implement ecosystem-based management and restoration practices.*

Predominant plant associations on the Pueblo are oneseed juniper (*Juniperus monosperma*) dominated woodlands (nine juniper plant associations), which vary in both species composition and structure and their position within the landscape. Of particular management concern to the Pueblo are oneseed juniper woodlands with perennial herbaceous understories on well developed soils—Juniper Savannah. Juniper Savannah is a valuable trust asset to the Pueblo because it provides critical habitat for traditionally important wildlife species such as mule deer (*Odocoileus hemionus*), pronghorn antelope (*Antilocapra americana*), Rocky Mountain elk (*Cervus elaphus*), and many species of birds and small mammals. Analysis of aerial photographs from 1935 and 1998 indicate that juniper tree densities have nearly doubled over the last 63 years within Juniper Savannah on the Pueblo. On average, tree densities per acre have increased by 93.2 percent or from 17 to 30 trees per acre. At the current rate of oneseed juniper encroachment, the composition and structure and continued persistence of Juniper Savannah on the Pueblo is threatened. An increase in juniper tree density within Juniper Savannah on the Pueblo is considered a problem because it may result in: 1) reductions in perennial herbaceous vegetation; 2) increases in soil erosion; 3) slower ground water infiltration rates; and 4) loss of quality wildlife habitats. Furthermore, juniper encroachment increases the probability of large, high intensity catastrophic wildfires, which can lead to and compound all of the consequences listed above.

Given the potential negative impacts of an overabundance of oneseed juniper on the Pueblo, the Pueblo developed a project to restore Juniper Savannah to historic conditions. The goals of the project are to reduce the threat of catastrophic wildfire, reinstate a semi-natural fire regime, increase perennial herbaceous cover, and improve wildlife habitat within Juniper Savannah. Using a combination of aerial photography and a vegetation map, the Pueblo delineated 2,700 acres of Juniper Savannah to be restored. A low impact treatment method was adopted to minimize soil and vegetation disturbance. The method consists of manual chainsaw thinning to reduce tree density and lopping and scattering of slash to promote

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perennial herbaceous recruitment. Old growth (single stemmed) trees and trees with bird nests or burrowing activity near the root crown are preserved. In addition to reducing the density of juniper trees, the project involves the extraction of salvageable fuel wood to reduce fuel loads in preparation for implementation of prescribed fire. Furthermore, the project provides valuable fuel wood to community members. Since 2003, the Pueblo has reduced the density of juniper trees from approximately 1,200 acres of Juniper Savannah and extracted and provided 42,400 ft^3 (331 cords) of salvaged fuel wood to community members. Three advancements in methods were made over the first three years of the project. First, a more precise estimate of work effort was identified—it takes a five person crew an average of 1.3 days to reduce the density of juniper trees, lop and scatter slash, and extract and transport salvageable fuel wood from 1 acre of Juniper Savannah on the Pueblo. Second, it was determined that the original treatment of reducing tree density by 75 percent per acre needed to be modified to 100 foot spacing between trees in moderately dense areas and 60 foot spacing between trees in highly dense areas. Thirdly, in order to maximize the amount of time the crew spends working in the field, it was determined that work schedules needed to be adjusted according to seasonal variation in available daylight and remoteness of treatment site. Specifically, the most efficient work schedule involves working four, 10-hour days per week during spring and summer at more remote sites and five, 8-hour days per week during fall and winter at less remote sites.

To evaluate if the project is effectively restoring Juniper Savannah to historic and desired conditions, the Pueblo, Bureau of Indian Affairs-Southern Pueblos Agency (BIA-SPA), local youth groups, and Southwestern Indian Polytechnic Institute (SIPI) and Santa Fe Indian School (SFIS) students will collaborate in collecting and analyzing wildlife, vegetation, fuels reduction, and fuel load monitoring data. The project is a collaborative effort funded by the Pueblo, United States Forest Service Collaborative Forest Restoration Program, and BIA-SPA. Results from this project will have great potential for application to other landscape-scale watershed restoration projects within Juniper Savannah across the Southwest.

Why Is Cultural Resource Site Density High in the Piñon-Juniper Woodland?

Sarah Schlanger[1] and Signa Larralde[2]

Hunter Gatherers

Hunter gatherers relied on healthy piñon-juniper woodland because it supports a wide variety of small game, large game, and bird species that shelter in the trees and forage on piñon nuts, a rich food source for humans as well as game.

Piñon-juniper woodland distribution is a proxy for **Ancestral Puebloan** occupation because: 1) dry-land maize agriculture requires good summer monsoons plus 120 frost free days; 2) piñon pine grows best with good summer monsoons, in the same elevation zone as maize agriculture (Benson 2003, Peterson 1994); and 3) ancestral Puebloans required wood for dwellings, pottery firing, heating and cooking.

Navajo and Apache peoples used piñon-juniper woodlands in much the same way as hunter gatherers, constructing wickiups, hogans, and sweat lodges in a dispersed pattern across the landscape and using downed wood for corrals and temporary shelters.

How Did PJ Woodland Die-Off Affect Human Occupation?

Piñon do not set cones until they are about 25 years old; maximum seed production is reached at 75-100 years. Catastrophic droughts leading to wildland fire and pine beetle epidemics mean decades-long delays before woodlands begin to recover. Archaeologists often propose two options: move somewhere more favorable for farming, or fall back on hunting and gathering. If woodland die-offs hit the prehistoric Southwest on the scale we are seeing today, there would be no forest to fall back to, no woodland to support hunter-gatherers for decades. These scenarios point out something we have overlooked: forests may recover much more slowly than farms and, under this condition, farmers may be more resilient than foragers.

How Can We Address Prehistoric Landscape-Scale Transformations?

We propose three research strategies to investigate the impact of large-scale landscape change in Southwestern prehistory:

- Identify periods of massive landscape-scale woodland die-off. The late 1100s warrant a careful look. Ancestral Pueblo populations reached their greatest geographic extent during this time, adding another stress factor on woodlands. The 1130s drought that followed has been tied to the Chacoan regional system collapse.

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- Re-evaluate "occupation gaps" and reoccupations that lagged far behind resumption of favorable rainfall and growing season regimes. The Mesa Verde region is a prime candidate for re-analysis; Arizona's Black Mesa and New Mexico's Pajarito Plateau are also promising.
- Re-think the relationship between agricultural intensification, population size, and settlement behavior in piñon-juniper-woodland-adapted populations. The propensity of these woodlands to undergo extreme change, their slow recovery rate, and the scale of the impacts merit some thought. Under these conditions, even with harsh droughts, agricultural dependence may have been the most reliable option.

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Range and Wildlife

Photo by Gerald Gottfried

Photo from Rocky Mountain Research Station Archives, Flagstaff, AZ

Wildlife Management in Southwestern Piñon-Juniper Woodlands

Jeffery C. Whitney[1]

Introduction

Piñon-juniper woodlands in the southwestern United States (Arizona and New Mexico) represent approximately 54,000 square miles, equivalent to roughly 20% of the land base for the two states. Within this broad habitat type, there is a high degree of variability of vegetation in terms of species composition, their relative abundance, percent canopy cover, and typically inverse percentages of associated mixed shrub and herbaceous ground cover. This variation is explained by climate, soil type, elevation, slope, aspect, interspecies competition, and past anthropogenic activities. Another important consideration is the extent of individual stands of piñon-juniper, their arrangement on the landscape, and proximity to other general habitat types (i.e. riparian areas, chaparral, pine, oak woodlands, and/or desert grasslands.

Species Compostion

In general terms, piñon-juniper in the Southwest is dominated by two-needle piñon (*Pinus edulis*), singleleaf piñon (*P. monophylla*), oneseed juniper *(Juniperus monosperma),* Utah juniper (*J. oesteosperma*), and alligator juniper (*J. deppeana*).

Understory vegetation is highly variable depending upon geographical location and local environmental conditions. Representative shrubs are generally sagebrush (*Artemisia* spp.), shrub live oak (*Quercus turbinella*), mountain mahogany (*Cercocarpus montana, C. betuloides*), and bitter brush (*Purshia* spp.)

Grasses and forbs vary considerably with generally cool season (C_3) grasses in the northern or upper elevation extents, and warm season (C_4) grasses at the southern or lower elevations.

Geographic Distribution

Piñon-juniper woodlands occur in foothills, mesas, and lower mountain slopes below 7,500 ft in elevation. These woodlands often form a transition zone with more homogeneous cover types, e.g., ponderosa pine (*P. ponderosa*) at higher elevations and oak woodlands at lower elevations.

Wildlife Species

Numerous wildlife species frequent piñon-juniper habitats in the western United States. At least 70 species of birds and 48 species of mammals have been associated with these woodlands (Gottfried and others 1995). In general, they

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include most classes including ungulates, large and small mammals, reptiles, avian species, and invertebrates. Although only a few species rely solely on the woodland habitats, many species utilize the wide array of piñon-juniper habitats for food, shelter, or breeding during part of the year.

Habitat Characteristics

There are four basic components that comprise wildlife habitat: space, cover, forage, and water. Obviously, the latter two generally are the limiting factors. Construction of artificial waters and/or silvicultural treatments can often be undertaken with the management objective of enhancing potential or existing habitat. It should also be noted that there are management issues related to endangered species within piñon-juniper habitats due to the expanse and extent.

Ecological Considerations

Much of the ecological considerations related to piñon-juniper woodlands center around, site capability, soil condition, successional status and state and transition model considerations that identify successional affecters. Past and current management activities and site potential are directly affected by site potential. There are considerable opportunities to meet identified desired conditions and habitat improvement.

Watershed Value

Not only do piñon-juniper woodlands provide habitat *in situ*, but these areas also represent substantial components within many watersheds. Woodland character and condition directly affect habitats within local stands but also can impact associated aquatic and terrestrial habitats "downstream" because of rapid runoff and accelerated erosion that affect channel characteristics.

Management Implications

Due to the large land base represented by piñon-juniper:

- Most wildlife species occur within this habitat type, at least seasonally, or as migratory species, pass through them;
- The high degree of variability within piñon-juniper in the Southwest provides an array of conditions and management opportunities;
- At lower successional states (lower densities), piñon-juniper provides high quality habitat; and
- Opportunities exist to generally improve piñon-juniper stand conditions depending upon management goals and objectives.

Mechanical, chemical, and prescribed burning treatments have been applied most frequently to decrease basal area/canopy cover and to increase production of grasses. Direct effects on target and non-target vegetation depend on how a particular method meets stated resource objectives. Indirect effects on non-target vegetation depend, in part, on the previous use of basic resources (i.e., water, soil nutrients, light) by the target vegetation and their availability to other species.

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Grazing Management for Healthy Watersheds

Karl Wood[1]

Abstract—*New Mexico was historically grazed by many native and introduced ungulates, often called wildlife. Their distribution was limited especially in deserts until domestic animals were introduced and drinking water was provided. Plants respond to grazing with little resistance (black grama), to great resistance (blue grama), and to being stimulated (antelope bitterbrush). A plant species such as blue grama may be in sod form when grazed heavily and in bunch form when grazed more judiciously. Grazing and trampling that reduces plant cover and volume affect the watershed by: (1) decreasing interception and transpiration, (2) decreasing organic matter additions to the soil, which affects soil structure and porosity, (3) decreasing infiltration into the soil surface, (4) increasing runoff and erosion, and (5) losing sustainability. However, grazing can retard the invasion of some undesirable species such as salt cedar. Trampling increases soil roughness and decreases runoff and erosion when soils are wet. The opposite is true when soils are dry. Unfortunately, southwestern soils are usually dry. Animal trails that go up and down slopes increase runoff and erosion. Trails that are across slopes decrease runoff and erosion. Trampling increases bulk density, which decreases porosity and results in increased runoff and erosion. However, increased bulk density also results in increased water holding capacity. Trampling may control pocket gophers, which plow the land and contribute to accelerated runoff and erosion. Grazing and subsequent deposition of animal wastes result in vegetation being returned to the soil, much of which would otherwise be lost to wind or volatilized. Usually, the only animal wastes found in streams are those that are deposited directly. Some streams can benefit from the added nutrients while other streams are adequate in fertility without additions from animals.*

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Piñon-Juniper Management Research at Corona Range and Livestock Research Center in Central New Mexico

Andrés Cibils[1], Mark Petersen[1], Shad Cox[2], and Michael Rubio[2]

***Abstract**—New Mexico State University's Corona Range and Livestock Research Center (CRLRC) is located in a piñon-juniper (PJ)/grassland ecotone in the southern Basin and Range Province in south central New Mexico. A number of research projects conducted at this facility revolve around soil, plant, livestock, and wildlife responses to PJ woodland management. The objective of this paper is to provide an overview of on-going research at CRLRC dealing with sustainable management of PJ ecosystems. This paper is divided into 2 sections; the first deals with mature PJ woodland management research, while the second covers current juniper reinvasion suppression studies. We have compiled contributions from a team of faculty and graduate students that conducts research at CRLRC. Names of investigators are listed after each project title.*

Introduction

Corona Range and Livestock Research Center (CRLRC) is a 11,330-ha research facility operated by New Mexico State University located approximately 22.5 km east of the village of Corona, New Mexico in a piñon-juniper (PJ) /grassland ecotone. Over half of its area has been classified as actual or potential PJ woodland and much of the research conducted at CRLRC, therefore, revolves around soil, plant, livestock, and wildlife responses to PJ woodland management.

Average elevation at CRLRC (Lat 34° 15' 36" N, Long 105° 24' 36" W) is 1,900 m and mean annual precipitation across the facility is 400 mm. Most rainfall occurs during the months of July and August as short duration convectional thunderstorms. Over 20 different soil associations are present on CRLRC and about 301 plant species belonging to 60 families have been identified on the facility (USDA-SCS 1970, 1983; Forbes and Allred 2001). Approximately half the area of CRLRC is covered by oneseed juniper (*Juniperus monosperma*) - piñon pine (*Pinus edulis*) woodlands; the remaining area has shortgrass plant communities dominated by blue grama (*Bouteloua gracilis*), buffalo grass (*Buchloe dactyloides*) and New Mexico feathergrass (*Stipa neomexicana*). Primary productivity of grassland sites ranges from less than 200 Kg/ha/yr during drought years (207 mm annual PPT) to over 1,000 Kg/ha/yr in moist years (455 mm annual PPT) (McDaniel 2002).

Since NMSU began managing this facility in 1989, grazing has been moderate across the ranch. CRLRC is currently grazed by cattle and sheep at a rate of approximately 28 ha/AU/yr. A base population of about 300 female mule deer (*Odocoileus hemionus*) and 100 pronghorn antelope (*Antilocapra americana*) are present on the facility.

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[2] Corona Range and Livestock Research Center, Corona, NM.

The objective of this paper is to provide an overview of on-going research projects at CRLRC that deal with sustainable management of PJ ecosystems. This paper is divided into two sections; the first deals with mature PJ woodland management research, while the second covers juniper reinvasion suppression experiments. We compiled contributions from a team of faculty and graduate students that conducts research at CRLRC. Names of investigators are listed after each project title.

Influence of Piñon-Juniper Clearing on Soil Water Dynamics, Livestock Distribution, and Mule Deer Population Numbers

Approximately 28% of the woodland area at CRLRC was mechanically cleared in the 1980s; an additional 40% was thinned with aerially applied herbicides in the mid 1990s (McDaniel 2002). Chemically cleared pastures have received either winter or summer season-long or rotational grazing since treatments were applied. Understory response to chemical control of adult trees ranged from increases of 0.3 to 3 times the existing herbaceous basal cover under mature woodlands. PJ removal has created a mosaic of treated and intact woodlands across much of the ranch (fig. 1). What follows is a brief description of a number of current research projects that focus on the implications of PJ removal on soil water dynamics, cattle grazing distribution, and mule deer dynamics.

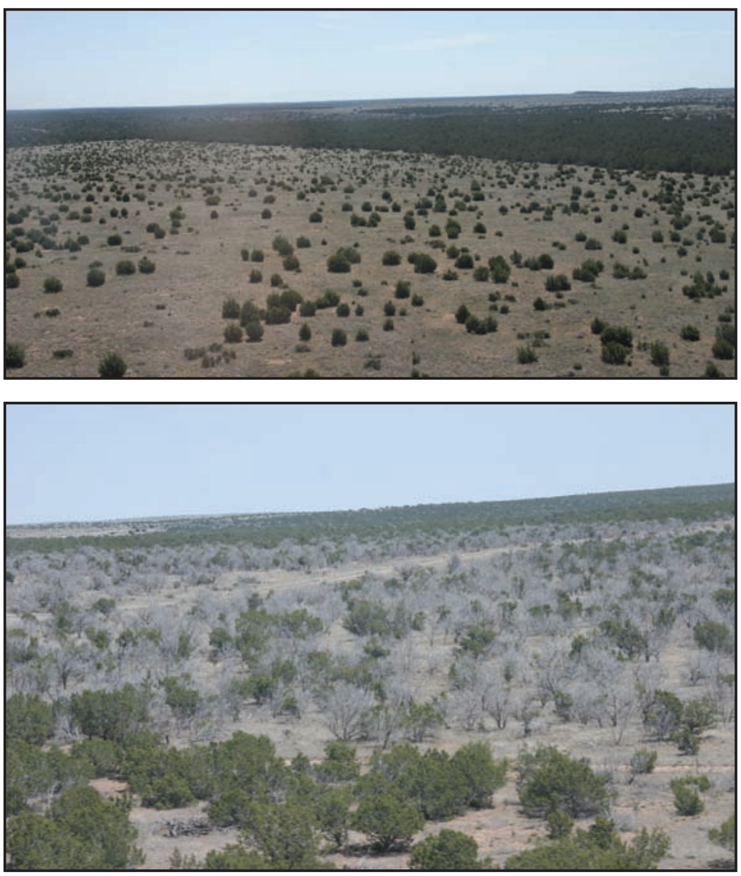

Figure 1—PJ-dominated areas at CRLRC. (Top) Encroaching saplings on areas that were mechanically cleared in the 1980s. (Bottom) Standing dead tree snags in areas that were chemically thinned with aerial application of tebuthiron clay pellets in the 1990s (Photos: J. Boren, 2005.)

Soil moisture dynamics under oneseed juniper trees in relation to chemical tree control and understory defoliation

Héctor Ramirez, Sam Fernald, Michelle Morris, Andrés Cibils, Shad Cox, and Michael Rubio *(Contact: fernald@nmsu.edu)*

Effects of clearing oneseed juniper woodlands on forage yield for cattle and big game has been studied extensively (Pieper 1995). However, less is known about the influence of such treatments on soil water dynamics. It is often assumed that juniper clearing is associated with an increase in groundwater recharge, yet the fate of moisture otherwise used by juniper trees could vary considerably depending on overall rainfall, soil texture, depth to caliche layer, and herbaceous understory cover among other factors (Huxman and others 2005).

A 2-year study is being conducted to monitor soil water dynamics under chemically treated (dead) and live oneseed juniper trees. Interactions among vegetation, soils, and understory defoliation regime are being characterized. CS616 soil moisture probes located at 3 depths beneath the tree canopy drip lines were installed in 24 plots under dead and living trees inside cattle-grazing exclosures (fig. 2). Placement of soil moisture probes was determined in a preliminary study that showed no significant variation in superficial soil moisture content at different

Figure 2—Installing soil moisture probes in plots under live juniper trees and dead snags in a cattle grazing exclosure at CRLRC. (Photos: M. and N. Morris, 2005.)

locations under tree canopies. Each probe is connected to a data logger that also records rainfall data from a gauge placed at a central location in every exclosure. Preliminary data gathered in 2005 suggest that the rate of superficial soil moisture depletion under dead trees was higher than under their live counterparts. Basal cover of herbaceous understory beneath dead juniper snags was up to 3 times higher than under live trees and may have been responsible for the apparent differences observed. Soils under live trees, however, tended to exhibit lower volumetric soil moisture content. Heavy clipping of herbaceous understory was apparently associated with higher levels of superficial soil moisture content. This study will allow initial insights into the effects of tree removal on soil moisture dynamics and will expand current understanding about understory-overstory interactions in PJ ecosystems.

Cattle Feeding Site Selection in Pastures with a Mosaic of Intact and Cleared PJ Woodlands

Christy B. Rubio, Andrés Cibils, Rachel Endecott, Mark Petersen, Shad Cox and Michael Rubio (contact: *acibils@nmsu.edu*)

Development of sustainable management strategies for PJ rangelands requires quantifying the potential trade-offs involved in traditional tree-clearing practices. The well-known benefits of increased grass production after tree removal could at some point be offset by decreased shelter for cattle or cover for mule deer. Since calving is possibly the single most important event affecting the income of a cow-calf operation, a study was set up to determine the importance of PJ woodlands as shelter for pregnant and nursing cattle.

During the first 2 years of this study (2004 and 2005), 8 pregnant and 8 open cross-bred and Angus cows were tracked with GPS collars (Lotek Engineering Inc.) set to record animal locations, neck movement and temperature at 5 minute intervals. Eight animals (4 pregnant and 4 open) from a herd of approximately 80 young mother cows were collared a week before expected calving date each year. Different cows were collared in each year. The herd was first placed in a 138-ha pasture (Horse pasture) for approximately 20 days. After that, they were moved into a 219-ha pasture (Mesa pasture) where they remained until the end of this study. Piñon-juniper had been mechanically cleared from approximately half the area of both pastures during the 1980s.

So far, results show that cattle mostly avoided PJ woodlands except during periods when forage was scarce (less than 280 kg/ha) and wind chill-corrected air temperatures fell below 30 °F for 4 consecutive days. Under such conditions, cattle spent up to 90% of their day in wooded areas (fig. 3). Overall, cattle preferred open to dense PJ and tended to spend more time in PJ during the morning than afternoon hours. Pregnant cows spent considerably more time in PJ than open cows on calving day and on the days immediately before and after calving. Otherwise, both open and nursing cows tended to show similar preference for wooded areas. Compared to open cows, nursing mothers tended to spend less time at water, explore smaller areas on any given day, and spend more time in dense PJ during the afternoons.

PJ woodlands appear to play an important role in providing shelter for cattle during calving in years when forage is scarce. We are planning simple modeling exercises to determine the implications of observed PJ preference patterns on energy status of lactating cows. Because mule deer and cattle have similar standard operative temperatures (animal's thermal environment) under windy conditions, heavy use of PJ by cattle on cold windy days could potentially affect woodland use by mule deer (Beaver and others 1996). Recent collaring of 11 mule deer will help investigate these interactions.

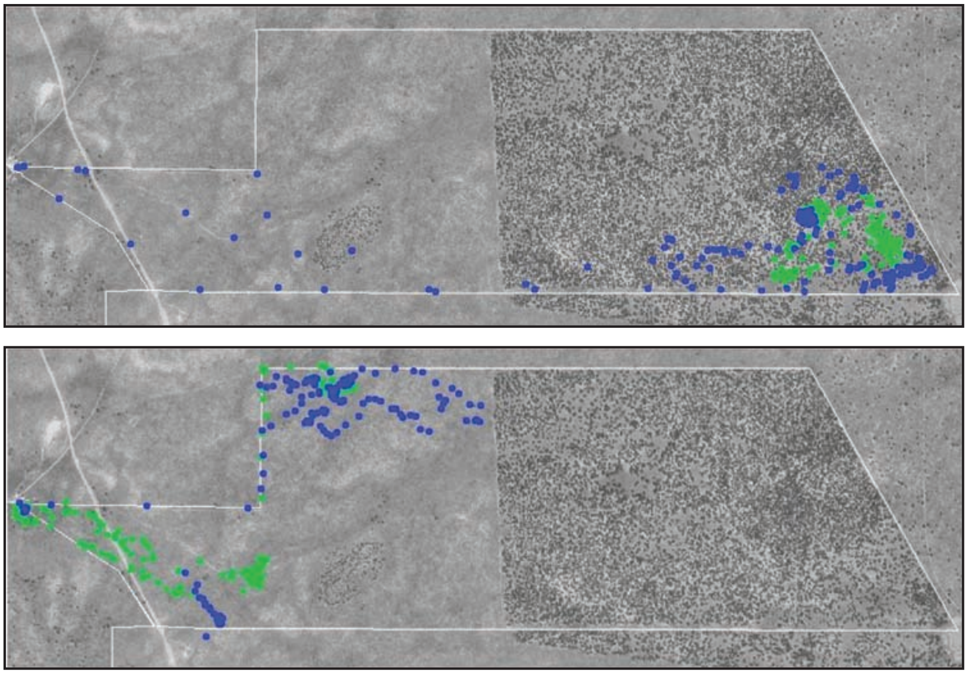

Figure 3—Horse pasture with cleared (left side) and intact PJ woodland (right side). Above: 5-minute interval location positions of one open and one nursing cow on a day with 29 °F wind chill corrected temperatures (March 4, 2004). Below: 5-minute interval location positions of same animals on a day with 47 °F wind chill corrected temperatures (March 17, 2004). The green points indicate 5 min-interval locations of a nursing cow. The blue points indicate the same but of a non-pregnant non-lactating cow.

Population Dynamics and Nutritional Ecology of Mule Deer on CRLRC

Jon Boren, Louis Bender, Brian Hurd, Summer Eaton, Shad Cox and Michael Rubio (*contact:* jboren@nmsu.edu)

Mule deer numbers in the State of New Mexico reached an estimated 302,000 in 1964 and have declined ever since. A combination of several factors may be responsible for the observed decline including local overpopulation, the aging of shrub habitats to less productive stages, drought, loss of quality fawning habitat, and increased predation. The presence of suitable habitat determines where and in what densities mule deer are found. Habitat management involves plant management. The two key points to plant management are: 1) knowing what plants are important for mule deer food and cover; and 2) knowing how to manipulate them. For rangeland and forest habitats in New Mexico, management tools may include livestock grazing practices, timber and brush management, water development, prescribed burning, and reseeding.

Mule deer on the CRLRC were subjected to a fee-hunting operation prior to New Mexico State University's acquisition of the area. The fee-hunting program continues to be operated with approximately 33 mule deer harvested annually. Data are collected from hunter-harvested males to assess trophy quality and body condition. Aerial surveys are used to estimate population size, age ratios, and sex ratios. The timing of these surveys allows determination of numbers of juveniles recruited. In addition, pre- and post-hunt spotlight surveys are conducted to estimate sex and fawn ratios.

Piñon-juniper modification has had a positive effect on mule deer numbers on the CRLRC. Past surveys indicated that mule deer utilize the areas treated with strip tebuthiron and the cleared areas the most. These results also suggested the need to maintain early successional stages in the cleared areas since there has been an overall decline in mule deer use of cleared areas across the facility. Past surveys have also shown that woodland use by mule deer was relatively consistent through time compared to other vegetation types and treatments. This suggests maintaining much of the woody component on the Corona Ranch is important and excessive removal of juniper could be detrimental and cause a decline in deer numbers.

Management of Juniper Reinvasion into Previously Cleared Sites

Suppression of juniper invasion into previously cleared woodland sites has been a central concern of rangeland managers throughout western United States for several decades. Juniper invasion is associated with severe alteration of a number of hydro-ecological processes including reduction in the diversity and biomass of herbaceous plants, decline in habitat quality for wild and domestic ungulates, and major hydrological dysfunction in watersheds. Direct or indirect suppression of fire is thought to be the historical primary cause for this trend (Miller an others, 2005, and references therein). Although reinstatement of historical fire regimes has been proposed as a solution to juniper encroachment problems, controlled burns are often unfeasible for a number of biophysical and regulatory reasons. Alternative control methods such as herbicides or prescribed grazing have been used either in combination with fire or as fire surrogates.

Aggressive encroachment of oneseed juniper saplings can be observed on most cleared areas at CRLRC. Juniper reinvasion in these pastures has been partially suppressed using prescribed burns (in 2002) and herbicides (in 2004 and 2005). Understory vegetation response to chemical thinning of juniper woodlands in some of CRLRC's pastures has not been sufficient to provide necessary fine fuels to carry a controlled fire. Such sites offer unique opportunities to study the influence of fire surrogate treatments.

Prescribed Grazing by Goats and Sheep to Suppress Juniper Sapling Encroachment on Cleared Sites

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If prescribed grazing by sheep and goats is to be used as a successful fire surrogate, two basic conditions must be met: 1) juniper browsing intensity must promote rates of seedling and sapling mortality comparable to those observed after a burn; and 2) timing and duration of prescribed grazing events must allow levels of recovery of non target plant species similar to those observed after a fire. This research project focuses on aspects of the first basic condition.

Preliminary studies showed that oneseed juniper saplings at our research site synthesize over 50 volatile oils many of which are known to deter herbivory. Level of browsing intensity by sheep and goats is therefore, strongly constrained by the detoxification capabilities of their digestive systems. Free ranging ungulates cope with plant toxicity problems by continuously mixing nutrients and toxins, a strategy that allows them to feed on chemically defended plants that would otherwise be excluded from their diets (Provenza and others 2003). The objective of this project is to manipulate nutrient-toxin mixes to increase levels of juniper intake by goats and sheep thus augmenting their ability to suppress juniper sapling growth and recruitment.

Controlled pen trials currently in progress, will determine the effects of supplements on juniper intake (fig. 4). Additional trials with cannulated sheep are addressing the effects of juniper intake on the animal's rumen microbial population and its overall nutritional status. So far, data suggest that protein supplements can increase levels of juniper intake by both sheep and goats significantly and that the magnitude of this response is associated with rumen degradability of the proteins included in the supplement. Data also suggest that juniper ingestion has detrimental effects on most rumen microbial populations of sheep. Results from these pen experiments will be used to calibrate a field study to measure browsing impact on juniper sapling growth and survival.

Figure 4—Ramboullet ewe and Spanish-Boer cross nanny in juniper sapling pen feeding trials. Goats consumed significantly more juniper than sheep and supplements promoted a three-fold increase in juniper intake by both sheep and goats. (Photos: S. Utsumi 2005.)

Controlled Defoliation of Woodland Herbaceous Understory in Cattle Exclosures

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Impact of prescribed grazing on non-target vegetation is of critical importance to the success of a prescribed grazing program. The use of goats and sheep as a fire surrogate to suppress juniper sapling growth and recruitment must insure levels of recovery of non-target plant species similar or better than those obtained after a burn. This study involves simulating a single high intensity defoliation event of understory herbaceous vegetation during the dormant season. Aboveground biomass of grasses and forbs in plots under live and dead juniper trees (described above) is clipped to simulate a single heavy defoliation event (>70% utilization) during the month of February. Thirty-six plots in 3 grazing exclosures are included in this study. Half the plots will receive a single intense defoliation on 2 consecutive years (winters 2005 and 2006). Basal cover by species was determined prior to the beginning of this experiment and will be measured again in fall 2006 to determine treatment effects. The first defoliation event was applied prior to a spring with above average rainfall. Preliminary observations suggest that clipping treatment had no detrimental effects on the vigor of grasses and forbs within experimental plots.

Acknowledgments

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Utilization

Photo by Gerald Gottfried

Photo by Gerald Gottfried

Uses of Pinyon and Juniper

Mark Knaebe[1]

Abstract—*There is an overabundance of pinyon and juniper, especially in the Southwest. Finding value in these undesirable trees will help it pay its way off the land. This presentation provides information on potential uses of pinyon and juniper. The value of wood products made from these trees could range from $10/ton to more than $200/ton. Unfortunately the higher valued products, such as lumber and poles, might be impossible to produce from these trees, given their growth patterns. Methods such as cutting wood apart and putting it back together (for example, finger jointing, laminates) greatly increase the wood's value but at significant expense. Additional mastication can produce fibers useful for water pollution control and be the basis for composites. Composites can be made by blending various forms of wood (for example, dust, excelsior, chips) with recycled plastics or cement-based materials for a variety of products or rubbery material to make wheelchair accessible playgrounds. The advantages of keeping the wood round, if available in straight sections, could result in products such as guardrail posts or elaborate structures. Energy is the lowest valued use for wood; however, its demand exceeds supply so only economics limit the harvestable quantities. Considering the high cost of other forms of energy, the value of biomass markets could drive the value of the wood significantly higher. Given the importance of finding uses for pinyon and juniper, both expertise and some financing can be helpful.*

Introduction

This paper is based on a power point presentation that provides information on the potential uses of pinyon and juniper wood. The presentation contains more than 100 photos and is available from the author. The following text briefly describes or discusses this power point presentation.

Market Value

- The market value of poles posts and lumber is generally more than $200/ton.
- The market value of firewood, chips, and gasifier fuel is generally less than $30/ton.
- The cost to harvest wood is greater than $30/ton and transporting this wood adds $0.20/ton/mile. Transportation costs can be reduced (as shown later) by processing, simply by increasing the density and removing water of a low value material (making pellets from chips), or by increasing the value by secondary manufacturing (making flooring from logs).
- Lower value material can be cut up and reassembled—finger-jointed studs, laminated beams, glulams, laminated veneer lumber. Even very crooked logs can be curve sawed making 1 by 1 material (still crooked) that can be laminated to make flat surfaces (similar to butcher block), which in turn can be cut into boards to make structural header I joists.

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Wood Fibers

Wood can be fiberized, essentially broken into fibers or fiber bundles (for example, larger than flour and smaller than chips). There are several outlets for this material when woven (more like snarled in a diaper machine that has a lot of barbed needles).

- Woven mats can be used as water filters. Depending on the pollutant, modification can be done to optimize pollutant absorption.
- Bulk fiber can also be loaded into flow-through screen baskets in parking lot drainage systems (among others).
- Seed can be incorporated into mats and used for erosion control or even hydroponics (astronauts grew vegetables in space using this technology).
- Spun plastic can be blended into mats and then hot pressed to form composite boards.

Wood Plastic Composites

Processes to make oriented strand board (OSB), particle board, and other hard boards require huge capital, and thus are only suitable when large sums of money are available. Alternatively, making wood plastic composites can be a small operation. In cooperation with a plastic manufacturer, such as one that manufactures back yard plastic chairs, recycled plastic can be blended with wood flour, which is much finer than fiber, and replace about half the plastic, making a potentially better product. This material is currently in use in many interior automotive components. Other products include decking, railing, window and door profiles, roofing, marine structures, siding, and fencing. Potential products include musical instruments, play centers, and shoes (clogs).

Low value high extractive wood, like juniper, can be suitably blended with plastic. There is a slight processing advantage (requires less lubricants) because of higher extractive content, some durability advantages, but the composite has a greater potential for color fade. With appropriate additives (pigments), the fade issue can be controlled, if that is important to the customer. A notable advantage of wood plastic signs is improved resistance to bullets. Accelerated weathering tests of wood plastic roofing shakes, which are very attractive, predict a 50-year life. The extruded decking market is growing fast.

Wood Cement Composites

Inorganic (cement-based) wood composites offer durable products. Examples include highway sound barriers, siding, roofing, and ceiling tiles.

Wood Rubber Composites

Flexible binders (latexes) can solidify wood chip playgrounds and trails enough to permit wheelchair access. There are still plenty of shock-absorbing capabilities. Cost is significantly less than rubber equivalents.

Animal Litter

Chips screened for ideal size can be used as a highly absorbent cat (and other creatures) litter. Superior to pellets that breakdown to (wet) dust, they reduce odor; and because there is no clumping the open spaces permit drying. The only problem is that under normal to light use, these chips appear to last forever.

Keep it Round

Cutting the largest rectangular shape from roundwood reduces its strength and stiffness considerably. The original roundwood is easily three times stiffer and might be as much as 10 times stiffer, depending on wood characteristics such as knots and taper. Reasons that rectangular shapes are weaker, in addition to having less wood, include the exposure of diving grain and juvenile wood. The variability of rectangular wood is considerably greater than round, so groupings of roundwood can have a higher classification. Uses for short irregular pieces include road barrier posts. However, depending on markets, slow-growing trees can be more valuable as lumber.

Structural Uses of Roundwood

Connectors, such as the dowel nut on 6-inch logs, can withstand as much as 40,000 lb before failing. Various arrangements (trusses, space frame) permit long spans, and buildings such as pavilions, bridges, and other structures have been built using roundwood. It is important to realize that building with roundwood is different, not necessarily harder. You need to see the various connections and/or work with an experienced user to decrease the growing pains. Fences, furniture, and erosion control structures (and fish cribs) are other uses.

Lower Cost Structures

A system of widely (32 inch) spaced studs and using only windows that fit between them eliminates the need for headers. Along with not attempting to show off the wood, roundwood can make economical dwellings. For an additional use for roundwood, exterior siding can be made from quartered logs. Residual wood from making siding can be used for nailers (large lath) for interior sheathing (drywall, paneling). Finally, sawdust can be used for insulation (important to treat and/or keep dry).

Energy

The lowest value for wood is energy. However, the market exceeds available wood supplies so it will always have an outlet.

Large-Scale Energy

St. Paul, Minnesota, has a 25 MW electric power generator (steam turbine) that burns urban wood wastes, collected from within 20 miles. This is an ideal size for this part of the country. To improve the economics, downtown St. Paul is heated with the low value steam (combined heat and power). Several other such plants are under construction or in planning stages. Somewhat larger plants would be possible but it is important to make sure the wood supply would be available on a continuous basis.

Small-Scale Energy

At the other extreme, Community Power Corporation in Littleton, Colorado, has a 5 KW generator (down draft gasifier) that burns wood pellets. Their larger units can burn a variety of feed stocks.

Processing and Transportation

There are a variety of wood processors from grinders to chippers with the associated advantages and disadvantages of each (for example, purchase costs, operations, maintenance). Depending on the processing, roll off containers could be the best transportation method.

Pellets

Because of the high cost of transportation, wood pellets become a better alternative because you then haul a less bulky, and less heavier biomass as well as removing much of the water. Considering the value of pellets (they are also easy to handle and burn cleanly), the cost of setting up a pelletizing plant could pay for itself in a relatively short period. This is especially true if the supply of wood is already dry, such as the waste wood from a secondary processing plant. Setting up a pelletizer in the woods for the sole purpose of selling pellets would take a considerably longer period to financially pay back. A new 2.5 ton/hour unit can be purchased for about $300,000 plus the cost of a building. Pellet furnaces and boilers are becoming increasingly available in both small (home) and large (industrial) sizes.

Fuels for Schools

Today, schools and colleges have newer and more efficient heating systems and emit less pollution than those that use coal-burning systems. Examples include Chadron State College for 14 years, University of Idaho for 20+ years, and many other businesses in the wood products industry.

There are three options for handling this fuel:

Fully automated: suited to large facilities.
Surge Bin (smaller, simpler, less expensive, 2-5 day supply): suited for small facilities.
Pellet systems: fuel more costly; storage considerations, smaller sized, less expensive boiler and entire system.

Darby school costs for the winter of 2005/2006 were $25,000, which included a $5,000 test. Fuel oil would have cost $115,000 so savings were $90,000. Construction costs, which were greater than new construction because of retrofits and three new buildings, were about $900,000. As fuel costs increase, the payback will likely be under 10 years.

Project Viability Factors
Community Enthusiasm and Support
Proximity to Biomass Fuel
Processing and Delivery Infrastructure
Fuel type/volume, Use Profile, and Unit Costs
Site Access and Space
Existing System Age, Condition, Adequacy
Construction and Integration Costs
Air Quality Permitting

Fuel Supply Considerations
Sources
Processing, Delivery and Storage—Clustering
Fuel Quality
Moisture Management
Ash—clinkers management
Chips versus pellets

The Good News

Using woody biomass will

- Reduce smoke from disposal burning
  - Human/Enviro Health—SOX, NOX, Green House Gas
  - Airshed Aesthetics—“Smokey Air”
  - Airshed space for Prescribed Burns
- Reduce cost to treat wooded land
- Save on heat and power bills
- Energy independence—Renewable
- Engage communities in solutions by creating jobs and small business opportunities

Finances

Hopefully, using pinyon and juniper will make economic sense. With a broader view of the cost and benefits to the environment, assistance (expertise and money) is available for demonstration projects or just to start projects.

Grants

- Federal, State, Foundations

USDA Rural Development

- Rural Economic Development Loans and Grants (REDLG)
- Community Facilities
- Rural Business Enterprise Grants (RBEG)

Carbon Trading

Municipal Leases

Fuel SAVINGS

Conclusions

Every effort should be made to turn the problems of excess pinyon and juniper into positive assets. Whether this means making valuable products out of it or developing methods to economically harvest it for energy, it is important that the work be done. Not only will this help the environment in ways, such as freeing up groundwater, but jobs will also be created.

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Using Small Diameter Trees for Wood Fiber-Plastic Composites

Phil T. Archuletta[1]

P&M Plastics, Inc. ("P&M" or the "Company" began operation in 1998 as a result of efforts within P&M Signs, a sister company, to develop a new composite material to be used for external signage—one more rugged than wood. The result of these efforts is a 40% woody biomass and a 60% plastic (HDPE) fortified wood composite. The composite was appropriately named Altree™ and was patented. Altree™ was chosen because the woody biomass is made up of small diameter trees and includes the trunk, limbs, bark, needles, and berries. Thus, no additional residue is created during harvest, adding more fuel to an already catastrophic situation.

Rarely do all the necessary ingredients for a successful small business come together at the right time and for the right reason. P&M Plastics is uniquely positioned to:

1. Become a major competitor in the composite manufacturing market space due to its extremely low cost of production and early entry into the marketplace. A board foot of Altree™ costs less than $2.50 to produce.
2. Become a major employer in Torrance County, New Mexico.
3. Become a model business venture supporting the USDA Forest Service initiatives aimed at utilizing pre-commercial, plentiful resources and woody shrubs like juniper and pine in place of old growth forest resources.

P&M Plastics has the commercial viability to become a successful leader in the woody biomass composite manufacturing industry. The Mountainair facility may also become the first of many P&M Plastic's recycling facilities throughout North America.

The economic development impact is one of the primary reasons that this project has received such tremendous support from the Estancia Valley Economic Development Association (EVEDA) and the State of New Mexico Economic Development Department. Initial meetings that included EVEDA and the State Economic Development Department encouraged the company to seek their assistance in getting financial and administrative support.

Of course, the economic benefits will not materialize without a well funded, managed company capable of taking an idea and turning that idea into a commercially viable enterprise. Our research indicates the market for wood plastic composites is saturated in some cases (e.g. patio decking) and without presence in others. For instance, we found only two companies that manufacture wood plastic composite sheets similar to ones P&M Plastics is developing. One of those companies deals exclusively with the automobile industry (door liners) and the other sells wholesale and retail locally and on the internet.

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Identifying Markets for Pinyon Pine in the Four Corners Region

Kurt H. Mackes[1]

***Abstract**—A search for opportunities to use pinyon pine is currently being conducted at Colorado State University by the Colorado Wood Utilization and Marketing Program as part of an effort to improve financial feasibility of forest restoration and hazardous fuel reduction work in pinyon-juniper stands. The properties of pinyon wood reveal that it is suitable for a range of traditional and value-added products. However, significant utilization challenges must be overcome, including the economics of harvesting, transporting, and processing pinyon, supply inconsistencies, lack of market development, and need for additional research into processing pinyon before increased utilization will occur.*

***Keywords**:* Pinus edulis*, pinyon, wood properties, utilization*

Introduction

Pinyon pine (*Pinus edulis*) is distributed throughout the southern Rocky Mountain region, including the foothills of Colorado and Utah, south to central Arizona, and New Mexico. Mature pinyon trees typically reach heights of 10 to 51 feet, with main stem diameter at breast height ranging from 6 to 30 inches (Alden 1997). Although larger trees have been recorded, they are more often small, less than 35 feet tall, with diameters of less then 18 inches. Pinyon trees are relatively slow growing and long lived, with dominant trees growing for up to 400 years or more. Tree stems can exhibit considerable taper and often have numerous large limbs. Pinyon pine continues to be an underutilized species in the region.

The characteristics and properties of pinyon wood, including anatomical structure and characteristics, moisture and shrinkage properties, weight and specific gravity, mechanical properties, and processing characteristics are discussed in this paper. Then a wide range of traditional and potential uses for pinyon wood are considered. Pinyon utilization challenges are then discussed.

Characteristics and Properties

Anatomical

Pinyon is considered to be a resinous softwood, normally containing large, numerous resin canals. The heartwood is yellow. The earlywood to latewood transition is abrupt. Annual growth increments are clearly delineated by dark, dense bands of latewood that transition to lighter earlywood. Rays are extremely fine and hardly visible, even with a hand lens. Pinyon is moderately heavy and relatively strong. Pinyon has a pleasant "piney" odor, especially when green. Pinyon wood often contains numerous knots that can be relatively large.

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Moisture Content and Shrinkage

Moisture content can be defined as the amount of water contained in wood expressed as a percentage of oven-dry wood weight (USDA 2002). Markwardt and Wilson (1935) reported average moisture content of 61% for green pinyon. Moisture content data for green pinyon sapwood and heartwood is currently unavailable.

The percentage of shrinkage from the FSP to OD condition is reported for pinyon in table 1. From the FSP to OD conditions, pinyon shrinks on average 5.2 percent in the tangential direction, 4.6 percent in the radial direction, and the volumetric shrinkage is 9.9 percent (Markwardt and Wilson 1935). While the volumetric shrinkage is comparable to ponderosa pine (also presented in table 1), the ratio of radial-tangential shrinkage for pinyon is relatively low compared to ponderosa pine, which helps reduce drying defects.

Weight and Specific Gravity

Markwardt and Wilson (1935) reported that green pinyon wood has an average weight of 51 pounds per cubic foot. If a typical cord of wood has between 80 to 90 cubic feet of solid wood (Lynch 2005), an average cord of green pinyon will weigh between 4,100 and 4,600 pounds. Pinyon at 12% moisture content was reported to have an average weight of 37 pounds per cubic foot (Markwardt and Wilson 1935). Assuming that 1,000 board feet of lumber is composed of 83.33 cubic feet of solid wood (Lynch 2005), this equates to roughly 3,100 pounds per 1,000 board feet of lumber at 12 percent moisture content. Dry pinyon typically has an average weight of 33 pounds per cubic foot. Weight values for pinyon are compared to ponderosa pine in table 2. Green pinyon is 13.3 percent heavier than ponderosa pine and dry pinyon is 32 percent heavier.

Table 1—Shrinkage Properties of Pinyon Pine and Ponderosa Pine.

	Percentage of Shrinkage (Green to Oven-dry Condition)	
Type of Shrinkage	**Pinyon Pine***	**Ponderosa Pine****
Tangential	5.2	6.2
Radial	4.6	3.9
Volumetric	9.9	9.7

* Markwardt and Wilson 1935
** Alden 1997

Table 2—Weight of Pinyon Pine Compared to Ponderosa Pine.

Characteristic	Pinyon Pine*	Ponderosa Pine**
Weight:		
Green	51 lb/ft3	45 lb/ft3
12 Percent	37 lb/ft3	28 lb/ft3
Specific Gravity:		
Green	0.50	0.38
12 Percent	0.53	0.40
Ovendry	0.57	0.42

* Markwardt and Wilson 1935
** Alden 1997

Specific gravity is expressed as the ratio of oven-dry sample weight to the weight of a volume of water equal to the sample volume at a specified moisture content (USDA 2002). Because specific gravity is a relationship or index, it is expressed as a unit-less number typically based on green volume or volume at 12 percent moisture content. Markwardt and Wilson (1935) reported an average specific gravity for pinyon of 0.50 based on green volume, 0.53 based on volume at 12 percent moisture content, and 0.57 based on oven-dry volume. Specific gravity values for pinyon are compared to specific gravity values for ponderosa pine in table 2. The specific gravity of green pinyon is 31.6 percent greater than that of green ponderosa pine, while specific gravity for oven-dry pinyon is 35.7 percent greater than that of oven-dry ponderosa pine.

Mechanical Properties

Strength and stiffness values are summarized for pinyon in table 3. As expected, oven-dry, clearwood stiffness and strength values are greater than green values for pinyon. In table 4 oven-dry values for pinyon are compared to oven-dry values for ponderosa pine. When compared to ponderosa pine, the modulus of elasticity (MOE) and modulus of rupture (MOR) of pinyon wood are lower, while compression properties are higher for pinyon. Compression perpendicular to grain strength is almost 3 times greater for pinyon. Pinyon is also significantly harder, almost twice as hard as ponderosa pine.

Processing Pinyon

When seasoning pinyon, it is important to dry wood at a high enough temperature to set pitch; otherwise it will bleed. Kiln-drying schedules are currently not available for pinyon. Machining and sanding properties are also not available in the literature. Preliminary results from a study of pinyon working properties

Table 3—Mechanical Properties of Pinyon Pine.

Property	Green*	Dry*
MOE	0.65 x 106 psi	1.14 x 106 psi
MOR	4.80 x 103 psi	7.80 x 103 psi
Compression Parallel-to-grain	2.59 x 103 psi	6.40 x 103 psi
Compression Perpendicular-to-grain	0.48 x 103 psi	1.52 x 103 psi
Shear Parallel-to-grain	0.92 x 103 psi	NA
Hardness	600 lbf	860 lbf

* Markwardt and Wilson 1935

Table 4—Mechanical Properties of Dry Pinyon Pine Compared to Dry Ponderosa Pine.

Property	Pinyon Pine*	Ponderosa Pine**
MOE	1.14 x 106 psi	1.29 x 106 psi
MOR	7.80 x 103 psi	9.40 x 103 psi
Compression Parallel-to-grain	6.40 x 103 psi	5.32 x 103 psi
Compression Perpendicular-to-grain	1.52 x 103 psi	0.58 x 103 psi
Shear Parallel-to-grain	NA	1.13 x 103 psi
Hardness	860 lbf	460 lbf

* Markwardt and Wilson 1935
** Alden 1997

currently being conducted at Colorado State University indicated that generally, pinyon wood machined very well (Bueche 2005). Machining properties evaluated included sawing, planing, shaping, boring, and turning. However, pinyon did not sand well because of the high pitch content, which tended to gum up the sandpaper. The heartwood of pinyon is easily treated with preservatives (USDA 2002). Additional information on pinyon bonding, durability (including finishing), and preservation properties is currently not available. More research is needed to fully understand pinyon processing properties and characteristics.

Pinyon Wood Products

Past inhabitants of southwestern North America used pinyon as a source of food, shelter, firewood, medicinal compounds, and ceremonial materials; however, the importance of pinyon to local inhabitants has declined dramatically (Fogg 1966). There are currently a variety of traditional uses for pinyon wood including Christmas trees, firewood, novelties, mine timbers, railroad ties, pulp, and charcoal (Alden 1997, Garcia 1993, Voorhies 1977). Van Hooser and Casey (1987) concluded that pinyon-juniper can be considered a commercial resource. In addition to pinyon nuts, Christmas trees and firewood were cited as current commercial uses of pinyon wood. Though not a "wood" product, pinyon nuts are mentioned because they are thought by many to be a culinary delicacy.

Wagstaff (1987) concluded that the economics of managing pinyon-juniper lands relies heavily on fuelwood sales, with firewood sales to individuals for personal use or small lot sales dominating the market. Fox (1987) looked at fuelwood opportunities in Arizona pinyon-juniper stands concluding that fuelwood prices would have to increase considerably (up to four times or more) to cover the cost of treating stands. In a subsequent publication, Fox (1990) evaluated standard stumpage rates for commercial pinyon-juniper fuelwood sales that occurred between 1984 and 1988, concluding that the increased number of no-bid and default sales indicate that the standard rate appraisal approach needed to be revised or replaced for some sales to be successful.

Researchers with the Colorado Wood Utilization and Marketing Program at Colorado State University are currently evaluating the economic potential of producing value-added products from pinyon wood. Products being considered include flooring, cabinets, furniture and furniture parts, cut stock, truck beds, and novelty items. Preliminary results indicate that because pinyon wood is aesthetically appealing, relatively hard with good clearwood strength properties, and machines well, it is suitable for producing these value-added products. There are several manufacturers in southwestern Colorado that are currently using blue-stained pinyon to produce furniture and a variety of novelty items (Jennings 2005).

Additional uses for pinyon include particleboard, cement-wood composite boards, and wood-plastic composite boards. Murphy (1987) reported that urea-bonded particleboard produced from pinyon was not recommended for exterior applications, but that a urea-bonded panel produced with a longer flake (1 to 1.5 inches) would likely be suitable for interior applications meeting both strength and stability requirements. Murphy (1987) also reported that a suitable cement-wood composite, comprised of 60 percent cement, 20 percent fibers, and 20 percent fluids (mostly water) can be produced using pinyon fiber. The USDA Forest Products Laboratory (2000) has been investigating the use of wood fiber from various species including pinyon pine to develop wood-plastic composites for use in products such as signs.

Because pinyon is very resinous, with branches and needles having up to up to four times the amount of resin in comparison to Douglas-fir (Murphy 1987), there is potential for using pinyon to produce naval store products. Deaver and Haskell (1955) found that the resin of *Pinus edulis* had some desirable qualities, yielding rosin and valuable volatile oils. However, they concluded that low output per tree and poor access to stands of sufficient density (20 to 25 trees of greater than 6 inches in diameter at breast height per acre) would likely make collection uneconomical.

Pinyon Utilization Challenges

There are many challenges to increased utilization of pinyon. High harvesting and processing costs result in the economics of utilizing pinyon often being unfavorable. Transportation challenges both in terms of accessibility and high hauling costs continue to be a challenge. In the past, supply inconsistencies have occurred. There is currently a lack of merchandizing strategies for marketing pinyon. There is need for more research into drying, machining, bonding, finishing, and adhesive properties of pinyon.

Conclusions

Although properties and characteristics of pinyon wood make it suitable for a variety of products, processing and hauling costs are often too high for this material to be utilized. Currently a high percentage of pinyon is left masticated on the forest floor. For wood processors to consider utilizing more pinyon, economically viable markets for pinyon must be developed and future restoration and fuel mitigation programs must be designed to provide a consistent supply of raw material to processors. At present, economically viable uses for pinyon are limited and there is likely no single product use or market that will utilize all the harvested pinyon. A stable, diverse mix of traditional and value-added uses for pinyon wood appears to be the most desirable outcome for increased utilization in the future.

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Energy from the Woodlands

Jerry Payne[1]

The woodlands offer a significant opportunity for conversion of biomass to energy projects. With the vast acreage in the Southwest in the woodland type, and with the significant soil loss problems prevalent in this area, there is a dire need to treat the woodlands. Since there are limited opportunities for marketable products, the material is a very good source of fuel for bio-energy options.

Wood is a relatively uniform source of energy. Unlike wind that is intermittent, or solar energy on cloudy days or at night, wood is dependable source of energy and can be part of a base load for electricity.

Wood generally has about 8,000 to 9,000 BTU per pound when bone dry. The net yield is approximately 5,400 BTU per pound for 40% moisture content. This works out to about 11,000,000 BTU per ton. Assuming a cord weighs about a ton, then we can figure about 11,000,000 BTU per cord. Energy is bought and sold by the million BTU.

Currently natural gas is selling for about \$6 to \$10/ MMBTU. Wood chips are being bought for \$18/ton in the White Mountains. These are the types of numbers that we need to consider.

In: Gottfried, Gerald J.; Shaw, John D.; Ford, Paulette L., compilers. 2008. Ecology, management, and restoration of piñon-juniper and ponderosa pine ecosystems: combined proceedings of the 2005 St. George, Utah and 2006 Albuquerque, New Mexico workshops. Proceedings RMRS-P-51. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

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Ponderosa Pine Forests

Photo by Gerald Gottfried

Influence of Elevation on Bark Beetle Community Structure in Ponderosa Pine Stands of Northern Arizona

Andrew Miller[1], Kelly Barton[1], Joel McMillin[2], Tom DeGomez[1], Karen Clancy[3], John Anhold[2]

Abstract*—Bark beetles killed more than 20 million ponderosa pine trees in Arizona during 2002-2004. Historically, bark beetle populations remained endemic and ponderosa pine mortality was limited to localized areas in Arizona. Consequently, there is a lack of information on bark beetle community structure in ponderosa pine stands of Arizona. Furthermore, it is unknown how elevation influences the community structure of these bark beetles. Understanding the bark beetle complex at different elevations will enable development of more effective forest management guidelines.*

Ten ponderosa pine stands were selected in each of three elevational zones in north-central Arizona: 1) Low ~5500 ft, 2) Mid~7000 ft, 3) High~8500 ft. Three Lindgren funnel traps were placed at each of the 30 sites. Each trap was baited with a different combination of commercially available lures developed for Ips pini*,* I. lecontei *and* Dendroctonus *spp. Traps catches were collected weekly (April-November) during 2004. Beetles and associated insects (predators and wood borers) were identified and tallied in the lab.*

A total of 31,010 pine bark beetles belonging to 15 species were trapped and identified in 2004. More than 3,000 associated invertebrate predators and woodborers were collected. Preliminary observations indicate that Ips *species in aggregate were most abundant at low elevation sites; however, individual species showed different distribution patterns.* Ips pini *was evenly distribution across elevations while* I. lecontei *and* I. calligraphus *numbers decreased with increasing elevation.* Dendroctonus *species in sum were most abundant at mid elevations; however, again there was considerable variation in distributions on the individual species level.* Dendroctonus frontalis *was much more abundant at low to mid elevations compared with the high elevation sites, while* D. brevicomis *was the most abundant at mid elevations. Numbers of other* Dendroctonus*, such as* D. valens, D. adjunctus, *and* D. approximatus, *increased with increasing elevation. The two most abundant invertebrate predators collected,* Enoclerus *and* Temnochila*, also showed disparate distribution patterns across the elevation gradient.* Enoclerus *species increased with increasing elevation, while* Temnochila *were most abundant at low to mid elevation. The study will be repeated at the same sites in 2005, and will be used to determine seasonal flight periods for each species by elevation.*

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Restoration Treatment Effects

Photo from Rocky Mountain Research Station Archives, Flagstaff, AZ

Forest Restoration and Fuels Reduction in Ponderosa Pine and Dry Mixed Conifer in the Southwest

Marlin Johnson[1]

***Abstract**—Most people agree that ponderosa pine and dry mixed conifer stands need to be thinned and burned to move the stands to within a normal range of variability. Unfortunately, people are in disagreement beyond that point. To some, restoration and fuels reduction means restoring stands to more open, pre-European (pre-1880) conditions. To others, fuels reduction should involve the removal of only the smallest trees (less than 6-9") within the treated stands. The Forest Vegetation Simulator (FVS) stand model, along with the Fire and Fuels Extension (FFE) of FVS, were used to demonstrate the effects on fuel hazard and forest stand structures when various stocking levels and diameter caps are applied. The effectiveness of the various treatments was evaluated based upon resulting crowning and torching index values following various thinning treatments. The results of the stand treatment simulations point out that pine and dry mixed conifer stands need to be fairly open (residual basal areas <80 ft^2) to adequately reduce fuel hazard to reduce the threat of stand-replacement fires in these forest types. Stand simulations also pointed out that when diameter caps are applied that are too small, treatments tend to remove most or all of the smaller trees, reducing within-stand diversity and moving the stands toward more even-aged stand structures. Amended forest plans (1996) in the Southwestern Region specifically call for multi-storied stand structures to increase within-stand diversity to promote better goshawk, Mexican spotted owl, and old growth habitats.*

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The Effects of Hazardous Fuel Reduction Treatments in the Wildland Urban Interface on the Activity of Bark Beetles Infesting Ponderosa Pine

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Abstract—*Selective logging, fire suppression, forest succession, and climatic changes have resulted in high fire hazards over large areas of the western United States. Federal and state hazardous fuel reduction programs have increased accordingly to reduce the risk, extent and severity of these events, particularly in the wildland urban interface. In this study, we examined the effect of mechanical fuel reduction treatments on the activity of conifer-infesting bark beetles in ponderosa pine,* (Pinus ponderosa *Dougl* ex. *Laws.*)*, stands. Treatments were applied in both late spring (April-May) and late summer (August-September) and included: (1) thinned biomass chipped and randomly dispersed within each plot, (2) thinned biomass chipped, randomly dispersed, and raked 2 m from the base of residual trees within each plot, (3) thinned biomass lop-and-scattered within each plot, and (4) an untreated control. The mean percentage of trees attacked by bark beetles ranged from 2.0% (untreated control) to 30.2% (plots thinned in spring with all biomass chipped). A three-fold increase in the proportion of trees attacked by bark beetles was observed in chipped versus lop-and-scattered plots. Higher levels of bark beetle activity were associated with spring treatments, which in general corresponded with periods of peak adult beetle activity. Raking chips away from the base of residual trees did not significantly affect attack rates. Several bark beetle species were present including the roundheaded pine beetle,* (Dendroctonus adjunctus *Blandford*)*, western pine beetle,* (D. brevicomis *LeConte*)*, mountain pine beetle,* (D. ponderosae *Hopkins*)*, red turpentine beetle,* (D. valens *LeConte*)*, Arizona fivespined ips,* (Ips lecontei *Swaine*)*, California fivespined ips,* (I. paraconfusus *Lanier*)*, and pine engraver,* (I. pini (*Say*)). Dendroctonus valens *was the most common bark beetle infesting residual trees. A significant correlation was found between number of trees chipped per plot and the percentage of residual stems with* D. valens *attacks. At present, no significant difference in tree mortality exists among treatments. In a laboratory study, monoterpene elution rates declined sharply over time in chipped treatments, but were relatively constant in lop-and-piled treatments. The quantity of α-pinene, β-pinene, 3-carene, and myrcene eluding from chips exceeded that of lop-and-piled slash during each of 15 sample periods. These laboratory results may, in part, explain the bark beetle response observed in regard to chipping treatments. The implications of these results to sustainable forest management are discussed in detail.*

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Restoration of the Ponderosa Pine Ecosystem and Its Understory

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***Abstract**—Restoration of the Mt. Logan ponderosa pine ecosystem has been on-going since 1995. This effort included tree thinning to a density based on what the tree density was in 1870. The desired plant community objectives from the Mt. Trumbull Resource Conservation Area Plan had a forest objective as 50% trees to be in old-growth — i.e., a diameter class of 20-31.9+ inch diameter at breast height (dbh). The other 50% of the trees were to be spread amongst the diameter classes ranging from openings to 19.9 inch dbh trees. The two Lava Units, on Mt. Logan, were logged, thinned, burned in 1996; one was seeded (13 acres), and the other was not seeded (33 acres). Using plot data, the seeded and unseeded units were compared. The seeding of native grass cultivars and imports increased species diversity of grasses, but seemed to suppress the native grasses (squirreltail, muttongrass, and blue grama) already present in the ecosystem. The other reason for seeding was to prevent annual bromes from becoming a major presence. The data on annual bromes in the Lava seeded unit and unseeded units show annual bromes equally evident in both, to the present.*

Introduction

During the early 1990s, the ponderosa pine ecosystem on the Arizona Strip was termed a forest in poor health by the Bureau of Land Management (BLM). This was largely due to the density of its trees (hundreds per acre). The pre-settlement forest, which existed prior to Europeans settling in Arizona, had in general, 15-40 trees per acre with some patches of denser stands. The latest year fixed for this condition is 1870, as determined from old photographs taken during the 1860s to 1880s in the ponderosa pine areas around Flagstaff, Arizona. Counting annual tree growth rings aided in establishing that date. This, pre-settlement forest density was corroborated by counting old-growth stumps and snags on the Arizona Strip. Currently, the Arizona Strip has around 15,000 acres of ponderosa pine. Approximately 12,000 acres of that is in the Mt. Trumbull-Mt. Logan area. Pre-settlement forests occupied, as best can be determined, about the same acreage. Critics claimed that the decline of the ponderosa pine ecosystem from the pre-settlement time period was largely due to livestock grazing, a lack of logging, and fire suppression efforts.

Regardless of the cause for the decline, the thick forest on the Arizona Strip was already at a high potential for disease and damaging wildfire. Consequently, there were forces to change management habits from within the BLM and by the public. A movement was implemented to balance the Mt. Trumbull-Mt. Logan forest size class and structure.

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Planning the Forest Structure

The Arizona Strip Field Office developed a Resource Conservation Plan (RCA) for the Mt. Trumbull (includes Mt. Logan) area from 1993 to 1995. During development of the plan, objectives were designed to describe quantitatively the diameter size of trees and structure stages desired for the Mt. Trumbull/Mt. Logan forests. From initial inventories the size class/structure of the Mt. Logan forest showed a large quantity of small trees – 90% with a 1-19" diameter at breast height (dbh) size class. Only 6% of the forest had areas dominated by 20" dbh trees or larger size class.

Six tree size diameter classes /structure stages were described in the RCA plan ranging from: openings or meadows, seedling/sapling (1-5" dbh), young trees (5-12" dbh), middle-aged trees (12-20" dbh), mature trees (20-24" dbh), old-growth trees (24"+ at dbh). The Mt. Trumbull interdisciplinary team, which involved BLM, Arizona Fish and Game, affected citizens, and academia, came to a consensus that the Mt. Trumbull landscape would be healthiest and meet multiple use objectives, at 50% old-growth trees (20-32" + dbh) and the remaining 50% of the forest needed to consist of openings and of trees up to middle age (12-20" dbh).

Getting There by Prescription

Early in 1995, Dr. Wallace Covington and a restoration team from the Northern Arizona University (NAU) Ecological Restoration Institute (ERI) visited the Arizona Strip and began a process to provide the BLM with a tree-density, prescription to restore the Mt. Logan forest to its 1870 density levels (15-40 trees per acre). The 1870 level was just prior to European settlements, subsequent fire suppression efforts, and livestock grazing. One of the first steps was to identify spacing of the trees to pre-settlement levels. The process of marking trees to leave began by locating evidence of dead, old-growth trees such as cut stumps and snags. Most were easily found, but on occasion, remnants were merely a woody outline on the ground or a portion of a log. For each old-growth stump, snag, or remnant of old-growth, one and a half leave trees were marked. In other words, two leave trees were marked one time and the next stump had one leave tree marked. That process was then repeated. The leave tree was the tree left after logging and thinning. The leave tree had to be greater than 16 inches in dbh. If only smaller trees were available, three times as many trees were marked. More smaller ones were marked due to their higher mortality from the post-thinning prescribed fire. This then left the desired amount of trees needed as replacements after the restoration. All existing old-growth, yellow-bark ponderosa were marked as leave trees.

The first marking of leave trees started in the fall of 1995. Two small units (both are designated Lava or 96-1) were marked and cruised (volume of timber estimated in board feet) and readied for harvest. One unit was 13 acres and the other 33 acres. They were both commercially logged, after which post-commercial thinning was done. Post-commercial thinning occurred where small trees were left uncut by the logger. Thinning involved cutting the small trees. The slash from the thinning was then burned and the 13-acre unit was seeded. The 33-acre unit was not seeded. The unseeded unit was done to compare unseeded to seeded understory. This completed the restoration effort for the units.

At the same time, the NAU team established fire monitoring plots to monitor vegetation and fuels. The plots were laid out in a 300-yard grid pattern across Mt. Logan's entire 8,500 acres of ponderosa forest. The study was to demonstrate the non-treatment, pre-treatment, and post-treatment impacts to vegetation. Other studies on wildlife were established to study the effects of restoration on wildlife populations.

Planning the Understory Vegetation

No vegetation objectives were set for the ponderosa pine understory in the 1993-1995 planning. A vegetation inventory provided information of understory characteristics. Across the landscape a grid of plots, each 50x20 meters in dimension, was established at 300 meter spacing. Inside the plots herbaceous plants were counted on two point-intercept transects. Each transect had 166 points along each 50 meter side of the inside the plot. Outside the plot, on both 50 meter sides, were belt transects of 10x50 meters and all species are listed that occur within the two belt transects.

The 1996 NAU pretreatment inventory of the ponderosa pine in the Lava Units (the 13 and 33 acre units) showed the top four understory species in frequency were big sage (*Artemesia tridentata*), squirreltail (*Elymus elymoides*), silver lupine (*Lupinus argenteus*) and cheatgrass (*Bromus tectorum*). New Mexico locust (*Robinia neomexicana*), beardlip penstemon (*Penstemon barbatus*), and mullein (*Verbascum thapsus*) were present but less common (table 1).

Seed bank studies were conducted in the late 1990s in a subset of plant communities on Mt. Logan by Judy Springer of NAU. She found that early successional and non-native forbs dominated the seed bank, whereas seeds of shrubs and perennial grasses were scarce. In ponderosa pine plant communities (old-growth and pole-size) squirreltail, a native, had a presence in the seed bank. Mullein and horseweed, however, were the dominant viable seeds by a large margin in the seed-bank. The seed bank and extant vegetation proved to be the best predictor of what the restored plant community would have as vegetation cover in the first few years after treatment.

Table 1—Vegetation trend plot data, lava unit plots, frequency data.

	13-acre Seeded Unit 1384				33-acre Unseeded Unit 1385 and 1435							
Species	**1995***	**1997**	**2001**	**2003**	**1995***	**1997**	**2001**	**2003**	**1995**	**1997**	**2001**	**2003**
	(%)											
Native Perennial (Squirreltail Blue Grama Muttongrass)	13	2	4	0	6	9	41	10	8	10	19	8
Seeded (Wheatgrass Mt. Brome Junegrass)	0	3	62	3	0	0	0	0	0	0	0	0
Forbs (Horseweed chenopods Mullein, etc.)	9	7	108	58	7	35	48	26	2	52	57	29
Shrubs (Locust) Browse	15	0	0	0	7	33	33	14	8	31	44	13
Cheatgrass	4	0	11	60	0	1	0	92	4	0	3	66
Percent Native to Non-Native	85:15	100:0	80:20	17:83	95:5	97:3	93:7	41:59	69:31	92:8	98:2	53:47

1995 is pre-treatment, 1997 is one year after treatment, 2001 is six years after treatment.

Seeding

As a result of the inventories and seed-bank studies, seeding of native grasses appeared necessary for perennial grass diversity. Fifteen species of perennial grass were seeded in designated-to-be-seeded units. Eight species of perennial forbs and five species of shrubs were seeded one time only due to extreme high cost ($15 to $82 per pound). The seeded Lava Unit (13 acres) had to be reseeded the following year, as the first seeding failed to survive a drought.

Four experimental units that were restored outside the Lava units on Mt. Logan were not seeded. This was done to determine the kind of understory plant community that would result after different year intervals in these restored units. Inventories have shown that in old wildfire burns, grasses such as squirreltail, blue grama (*Bouteloua gracilis*), and muttongrass (*Poa fendleriana*) have come in vigorously and abundantly in some locales in the ponderosa zone, without seeding.

In 1989, a 78 acre fire occurred in the wilderness area on the top of Mt. Trumbull. Weight transects, where vegetation is clipped and weighed at points along the transect to determine species composition by weight, were done in 1992 and 2001 in the fire scar and adjacent unburned forest (table 2). The transects in the fire scar had a range of 23-42% squirreltail and 16-35% oak with 18 other species weighing in at much lesser rates. The adjacent unburned forest had trace amounts of squirreltail with upland sedge occurring at up to 24%. Few other species occurred under the pine.

Table 2—Mt. Trumbull 1989 fire scar vegetation, unseeded area. Values are species dry weight as a percentage of total dry weight sampled on the transect.

	Dry weight (%)	
Species or group	**1992**	**2001**
Oak	35	16-28
Ponderosa	11	0-14
Locust	0	2-10
Ribes	0	1-4
Rabbitbrush	0	1
Wormwood	0	1-2
Muttongrass	2	9
Squirreltail	32	23-41
Blue grama	2	0
Carex	0	5-9
Mullein	1	0
Golden rod	0	7
Mahonia	0	1
Daisy	0	5
Lotus	0	2
Penstemon	0	2
Milkweed	0	3
Chenopodia	15	2-4
Aster	0	1-6
Knotweed	0	1

Cost of Going Native

In the first restoration units treated, seeding costs were high due to using very expensive forbs and shrubs, along with the native grasses (table 3). The forbs and shrubs ranged in price from $15 to $82 a pound, with most in the $40 to $60 per pound range. Native grass costs ranged from about $1 to $40 per pound with most in the range of $5 to $20 per pound. The costs shown in table 2 are seed costs, the labor and operating equipment costs only. All seeding was done by using hand powered broadcasters and dragging the area after seeding with all-terrain-vehicles pulling small chains. The seeded and unseeded units were fenced with an electric fence to prevent livestock from grazing the forage prior to good establishment.

Table 3—Cost per acre of seeding on other units.

Year seeded	Logged unit	Seeded	Seed	Labor / Equipment	Total cost
		(acres)	- - - - - - - - - - - - -	*($/ac)*	- - - - - - - - - - - - -
1997	96-1	13	$214	Not figured	
1998	96-1	13	$228		
1998	96-2	156	$73	$34	$107
1999	96-2	80	$79	$38	$117
2000	96-3	110	$103	$38	$141

Conclusion

The forest diameter size class and structure objectives of 50% old-growth and 50% smaller trees and openings, in the higher elevational units were close to being met after the logging, thinning and burning. Reaching stability on the forest floor, however, will take longer, even with seeding, due to successional processes.

Seeding native grasses, shrubs and forbs greatly added grass diversity and cost in the seeded unit. Five years after seeding took place, the understory became co-dominated by seeded grasses (see table 1), but forbs and non-native forbs still were dominant and abundant in the seeded unit. The unseeded understories were dominated by shrubs, annuals, and forbs with less diversity of grasses. Is the seeded native grass diversity worth the cost? Could the understory restoration still be considered restoration with the species that are already present such as blue grama, squirreltail, muttongrass and the numerous native forbs and shrubs shown in the inventories?

The resulting vegetation after logging, post-commercial thinning and burning, has not resulted in a total takeover, as yet, by non-native weeds, which was of some concern. As table 1 shows, it is other natives that remain in dominance until 2001. The 2002-2003 very dry part of the drought killed seeded and native grasses and allowed cheatgrass to increase to dominance in the seeded and unseeded unit. One encouraging item was the native grass in the unseeded unit. The drought did not kill the native squirreltail totally but left a good remnant to grow from after the drought. With rest-rotation grazing and rest from grazing, and maturing of the plant community after treatment, over the years, a stable perennial plant community would probably result without expensive seeding or at least a reduced amount of seeding, as it did on Mt. Trumbull. Need for seeding can be ascertained and done at a later date – for example, after one of the repeat burns that will be conducted at 5-year intervals, as specified by the future the restoration plan to keep tree densities low.

Cheatgrass Encroachment on a Ponderosa Pine Ecological Restoration Project in Northern Arizona, U.S.A.

Christopher M. McGlone[1], Judith D. Springer[1], and W. Wallace Covington[1,2]

Abstract—*Land managers frequently thin small-diameter trees and apply prescribed fire to reduce fuel loads and restore ecosystem structure, function, and process in forested areas. There is increasing concern that disturbances associated with these management practices can facilitate nonnative plant invasions.* Bromus tectorum *is an annual grass from the Mediterranean region. It has invaded large areas of the Interior West and has become the dominant species in many of these areas. In 2003, a ponderosa pine ecological restoration site on Mt. Trumbull in the Uinkaret Mountains of northern Arizona experienced a large increase in* Bromus. *Thinning and burning projects had been conducted on this site since 1996.* Bromus *frequency increased on the thinned and burned plots by six-fold between 1996 and 2003. While* Bromus *also increased on thinned plots that were not burned and the untreated control plots, the frequency of* Bromus *was significantly lower than on the thinned and burned plots. There were two additional factors that may have influenced the* Bromus *invasion. In 2002, the region experienced the most extreme drought recorded in the past 100 years. Substantial rainfall returned to the area in September 2002, coincident with the timing of* Bromus *germination. Additionally, cattle were reintroduced to the study area in July 2002 after a five year hiatus in grazing. We present data that suggest the interaction of prescribed fire, small-diameter tree thinning, cattle grazing, and drought were the primary causes of the spread of* Bromus.

In: Gottfried, Gerald J.; Shaw, John D.; Ford, Paulette L., compilers. 2008. Ecology, management, and restoration of piñon-juniper and ponderosa pine ecosystems: combined proceedings of the 2005 St. George, Utah and 2006 Albuquerque, New Mexico workshops. Proceedings RMRS-P-51. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

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Changes in Canopy Fuels and Fire Behavior After Ponderosa Pine Restoration Treatments: A Landscape Perspective

J. P. Roccaforte[1], and P.Z. Fulé[1]

Abstract—*We modeled crown fire behavior and assessed changes in canopy fuels before and after the implementation of restoration treatments in a ponderosa pine landscape at Mt. Trumbull, Arizona. We measured 117 permanent plots before (1996/1997) and after (2003) thinning and burning treatments. The plots are evenly distributed across the landscape and represent an area of approximately 1,200 ha (2,964 ac), about half of which is an untreated control. Basal area decreased significantly by 42% from 32.6 m² (142.0 ft²) to 18.9 m² (82.3 ft²) in the treated area between 1996 and 2003, while the control did not change significantly over the same time period. Canopy biomass decreased significantly by 50% from 18.3 Mg/ha (8.2 tons/ac) to 9.1 Mg/ha (4.1 tons/ac) and canopy bulk density decreased by 42% from 0.093 kg/m³ (0.006 lb/ft³) to 0.048 kg/m³ (0.003 lb/ft³) in the treated area, while slight increases occurred in the control. Analysis of crown fire behavior using simulation models under extreme drought and wind conditions suggests that the proportion of the landscape susceptible to active crown fire and the mean patch size of these areas were both reduced in the treated area. In contrast, the models suggest little change in active crown fire susceptibility in the control over the same time period.*

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[1] Ecological Restoration Institute, Northern Arizona University, Flagstaff, AZ.

Restoration of Southwestern Ponderosa Pine Forests: Implications and Opportunities for Wildlife

Catherine S. Wightman[1] and Steven S. Rosenstock[1]

Abstract*—After a century of fire suppression, livestock grazing, and even-aged timber harvest practices, forest managers in the Southwest face an enormous challenge. Millions of acres of ponderosa pine forest are extremely susceptible to uncharacteristic, high intensity wildfires, the consequences of which were amply demonstrated by recent mega-fires in Arizona and New Mexico. Current condition of ponderosa pine forests are also atypically homogeneous in structure and composition, which results in reduced habitat biodiversity for wildlife. In response, land managers have begun planning and implementing extensive forest treatment projects along urban interfaces and in wildland areas. Although many of these projects are designed primarily to reduce the risk of high intensity wildfire, these treatments have considerable potential to improve wildlife habitat, by creating diversity at the stand and landscape level and increasing productivity in shrub and understory layers. While there is widespread agreement that restoration of Southwestern forests is needed, the location, scale, and approach of treatment prescriptions remain controversial, especially when addressing wildlife needs. We present a conceptual overview of wildlife responses to treatment-induced changes in ponderosa pine forest structure and summarize results from ongoing studies on the Mt. Trumbull restoration area in northern Arizona.*

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[1] Arizona Game and Fish Department, Research Branch, Phoenix, AZ.

Home Range and Habitat Selection Patterns of Mule Deer in a Restoration-Treated Ponderosa Pine Forest

R. Fenner Yarborough[1] and Catherine S. Wightman[1]

Abstract*—Forest restoration treatments are currently being conducted throughout the state of Arizona. Restoration treatments open the existing forest structure and may improve foraging habitat for mule deer (*Odocoileus hemionus*) but may reduce the suitability of day bed sites or decrease fawn recruitment due to removal of sufficient hiding cover. To evaluate mule deer habitat selection patterns across a restoration-treated ponderosa pine (*Pinus ponderosa*) landscape, we outfitted 15 female mule deer with GPS store-on-board collars in 2003. Our main objectives were to evaluate habitat selection within home ranges and core use areas, to evaluate whether female mule deer select different habitats during the fawning period, and to compare habitat selection patterns among day, night, and crepuscular hours. In 2004, we retrieved 2 collars from mortalities and plotted the location data in a Geographic Information System. Average home range size in summer was 2848.773 km 100% MCP (minimum convex polygon (Hayne (1949)), 1736.526 km 95% MCP, and 1430.019 km 95% kernel estimates. We found mule deer in most habitat types except meadows and shrub lands during the day. We have recovered two collars and found that restoration-treated habitats were used 76% of the time during night hours, 26% during the day and 54% during crepuscular hours for one individual and found that the other deer used the treatment areas 76% during the day, 84% at night, and 73% at dusk. We plan to complete our data analysis after we retrieve all GPS collars in fall 2005.*

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[1] Arizona Game and Fish Department, Research Branch, Phoenix, AZ.

Reference

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The Irrationality of Continued Fire Suppression: A Partial Analysis of the Costs and Benefits of Restoration-Based Fuel Reduction Treatments vs. No Treatment

G.B. Snider[1] and P.J. Daugherty[1]

***Abstract**—In 1905, the Bureau of Forestry became the U.S. Forest Service and was given responsibility for protecting newly designated forest reserves. A critical part of its charge was the prevention and control of fires. In 1908 Congress set up a unique system, like an open checkbook, that assured payment for fire suppression as needed. Since that time the fire suppression checkbook has never run out of blank checks and federal land management agencies continue to spend orders of magnitude more money on suppression than on pre-fire fuel control. We argue that the perpetuation of this pattern represents an irrational investment in the further disruption of fire cycles and the continued depreciation of forest values. We present results of an economic analysis comparing restoration-based fuel treatments to no treatments for areas identified at high risk for crown fire to support our claim. The with and without treatment comparison focuses on ponderosa pine and dry mixed conifer forest ecosystems in Arizona and New Mexico (Forest Service Region 3). The analysis provides a conservative estimate of the potential economic losses due to no action.*

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Topics Common to Pinyon-Juniper and Ponderosa Pine Ecosystems

Photo by Gerald Gottfried

Landscape-Level Changes

A. Joel Frandsen[1]

Abstract—*Since European settlement, Utah's vegetative landscapes have changed. Like other arid states, these wildland systems were depleted and altered. Certain steps were taken through private, community, and finally public efforts, such as establishment of Forest Reserves (National Forests), to stop the slide. Conservation and management actions were taken to restore, rehabilitate and manage these landscapes. Utah has numerous examples where the productive capability of the land has been restored. Unfortunately, in this environmental era, we are again in a downward ecological spiral, and the productivity of these landscapes towards desired objectives is not being met. The action needed to stop this trend is not getting the attention to stabilize and correct the problem. The Healthy Forests Initiative and the Healthy Forests Restoration Act provide some positive steps in this direction, and the knowledge and technology are available and can be expanded upon. The challenge is: can we muster the will and support to reverse the downward spiral?*

Keywords: *landscape-level changes; vegetation Management; forest health; invasive species; conservation; restoration; rehabilitation.*

Introduction

Utah has as much land diversity as any state, ranging from alpine forests to developed, metropolitan areas, from white salt flats to redrock canyons. With the exception of water, which is scarce in this second-driest state in the nation, the State is rich in natural resources, and home to a great diversity of user groups.

Utah is facing significant environmental issues; however, much of our effort is spent on the wrong issues, while other problems are overtaking us. For instance, millions of hours and tons of media effort have been spent on the Bureau of Land Management (BLM) "wilderness issue," but there are bigger problems that remain unaddressed. Natural resource managers and policymakers need to redirect their efforts to the bigger picture.

Certain groups seem to be driving the natural resource agenda, and our resources have been deteriorating. The absence of proper management and lack of necessary tools have tied the hands of natural resource managers; the resulting condition of our lands is obvious to the trained professional.

Utah's wide variety of landscapes and cover types, combined with a complicated land ownership pattern, makes it much more complex and difficult to get coordinated action on natural resource management problems. What one neighbor does affects the others.

Utah's Conservation History

Utah's early pioneers, with their unfamiliarity with dry climates, lacked understanding regarding management of the local natural resources. They came from the east, where precipitation was greater and soils and vegetation responded

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faster. Early day logging and livestock grazing resulted in negative impacts to Utah's forest and rangelands, but after 40 to 50 years, citizens petitioned for Forest Reserves (National Forest) and a conservation effort came forth.

Great strides in land restoration and conservation have taken place on National Forest, BLM, and private lands to mitigate the problems caused by early abuse. Examples of these successful mitigation projects are documented in publications such as "Vegetation Changes on the Manti-La Sal National Forest" (USDA Forest Service 1993). These comparative photographic studies from the early to late 1900s demonstrate that resource managers have the knowledge and experience needed to manage vegetation towards our desired objectives.

Landscape-Level Changes: Utah's Forest Lands

Despite the "environmental era" we live in, landscape-level changes are occurring in our forestlands, rangelands, wetlands, and other open spaces. The condition of Utah's forests is bleak. Insects and disease are ravaging Utah's forests, making them even more susceptible to wildfire. When infestations started, the Forest Service (Dixie National Forest) proposed to remove infested trees, therefore removing the bugs. Their projects were appealed and litigated, delaying their ability to respond to the infestations; the result is wide-spread devastation (fig. 1). Spruce beetle has caused the most noticeable forest die-off in the state since the late 1980s, especially in central and southern Utah; thousands of acres of dead or dying trees are apparent on Cedar Mountain alone, in Southwest Utah. According to data obtained from Forest Health Protection annual insect aerial surveys, spruce beetle has caused up to sixty percent mortality on approximately 278,000 acres over three of Utah's National Forests in the last 10 to 15 years. (fig. 2).

Figure 1—Tree mortality, Mirror Lake Highway, Utah

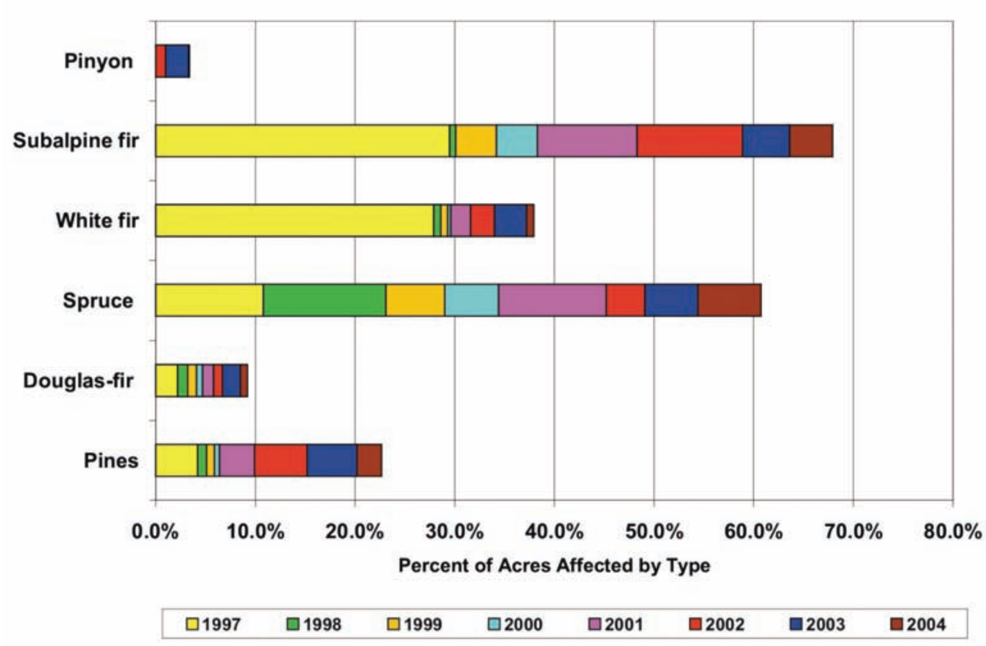

Figure 2—Bark beetle-induced mortality by species or species group in Utah, 1997-2004.

Evergreen Magazine recently reported, "Time is Running Out" for our forests. Petersen (2003) quotes from W.W. Covington's testimony before the congressional Committee on Resources, Subcommittee on Forests and Forest Health, in Flagstaff, Arizona, March 3, 2003:

> "The greatest threat to the sustainability, diversity and social viability of the forests and communities of the West is our failure to restore forest health in the frequent fire forests of the West. Simply installing fuel breaks around our cities and rural developments and forsaking the wildlands would be an abdication of our responsibility to future generations."
>
> "Attention cannot be narrowly focused on a ring around the developed areas. Such actions will fail to address one of the most contentious issues of our time, the protection of endangered species. Severe wildfires in frequent fire forests of the West are the greatest single threat to critical habitat for many of these vulnerable species because they are not adapted to stand-replacing fires."

The absence of fire and other disturbance, including logging, has resulted in the overcrowding of our forests. The monster fires of 2002 – Hayman, Rodeo-Chediski, Biscuit – each burned hundreds of thousands of acres; Utah had its own monster fire – the Rattle Complex – in Southeast Utah, which burned around 100,000 acres in 2002. The 2003 fires in Montana and California were just as devastating.

Our forests across the nation are growing much more product than is being used, building a storehouse of energy. For example, in the revised forest plan for one of the National Forests in Utah, the annual allowable sale quantity (ASQ) for sawtimber is listed at 2 MMBF (million board feet). The net annual growth of sawtimber trees on non-reserved timberland in this national forest is approximately 53 MMBF of all species and 49 MMBF for just softwoods (O'Brien 1999). This indicates a planned harvest level of only four percent of the net annual growth for the forest. Ninety-six percent of the net annual growth is adding additional fuel to the forest each year. This added fuel accumulation (a storehouse of energy) is setting the stage for future catastrophic monster fires.

Another issue in the West is the decline of aspen. There used to be almost nine million acres of aspen; now there are less than four million acres. Utah has lost nearly sixty percent of its native aspen. Colorado has lost fifty percent of its native aspen, and Arizona has lost ninety-six percent. (USDA Forest Service 2000) This is primarily due to the absence of proper management in the face of natural succession, and the absence of productive wildfire in the ecosystem.

Landscape-Level Changes: Utah's Rangelands

There has been an increase of pinyon juniper and a decrease of sagebrush in many areas throughout the state. In some areas, what used to be a sea of sagebrush has become a sea of pinyon-juniper. The loss of sagebrush means loss of habitat for the Greater Sage-Grouse, listed as a sensitive species in Utah, and other species.

In many areas, sagebrush is being replaced by cheatgrass. A Bureau of Land Management report estimates that cheatgrass invades 4,000 acres a day throughout the Great Basin area. The report describes cheatgrass as a volatile fuel that carries fire quickly, and is especially adept at taking over disturbed areas, resulting in a downward ecological spiral. (Bureau of Land Management 2001) Other undesirable invasive species are coming in behind cheatgrass, such as squarrose knapweed, diffuse knapweed, Russian knapweed, medusahead rye, and Scotch thistle.

Invasive, exotic pests have been referred to as "biological pollutants," which "threaten our crops, our forests, and perhaps our very existence…Once biological pollutants are imported, they grow, adapt, and spread on their own unless people take direct, vigorous, and often costly actions to stop them." (Britton 2004) Left to nature, invasive species out-compete the desirable species.

The USDA Forest Service has identified invasive species as one of the four threats to the nation's forests and grasslands. According to the Forest Service website, "of 2,000 nonnative plants found in the United States, 400 are invasive species. The U.S. spends $13 billion per year to prevent and contain the spread of invasives. For all invasives combined, the price tag is $138 billion per year in total economic damages and associated control costs. In addition to nonnative plants, 70 million acres of forest in all ownerships (public and private landholdings) are at risk from 26 different insects and diseases." "With the globalization of commerce and foreign travel to and from the U.S., the number of new invasive species from abroad is growing…" although, sometimes these species spread within the United States itself. Invasive species are disrupting native ecosystems and draining the nation's resources.

The Western Forestry Leadership Coalition issued its own policy statement on invasive species in 2003 that states, "The continued introduction and accelerated spread of invasive species are some of the greatest natural resources concerns in the West – prevention and control are critical."

There are numerous examples in southern Utah where desirable seed was planted by mechanical means, with good success. We can also utilize desirable exotic species to counter the undesirable invasives; one example is *Kochia prostrata*, an exotic that is fire tolerant and good for fall and winter forage use. These desirable species have been instrumental as fuel breaks and in stopping fast-moving cheatgrass fires.

The secret to stopping cheatgrass takeover is mechanical treatment. The main thing preventing mechanical treatment – and allowing cheatgrass and other invasive species to flourish – is that the treated area would then not qualify as "wilderness." But those who defend "wilderness" against mechanical treatment, citing threats to biodiversity, will soon see that there is no biodiversity in the cheatgrass wilderness.

Landscape-Level Changes: Utah's Riparian Areas

Riparian and wetland areas are also at risk of invasive species. The Southwestern Willow Flycatcher, a species federally listed as endangered, has lost its primary habitat due to the infestation of tamarisk or salt cedar, which out-competes and replaces the native willow species. Tamarisk proliferates in most waterways in Utah, monopolizing one of Utah's most valuable resources – water.

Purple loosestrife is another invasive species flourishing in Utah; this species has a wide range of habitats. Of most concern in Utah is the plant's proliferation in waterways, where it forms dense, homogeneous stands that pose a severe threat to waterfowl habitat.

Landscape-Level Changes: Utah's Open Spaces

The West is also losing its open spaces to urban encroachment (fig. 3). More and more people are building in the "wildland-urban interface," putting their lives and property at risk of wildfire. Utah has hundreds of these wildland-urban interface communities, occupying around 137,000 acres throughout the state.

Not only are these communities creating additional challenges for wildland firefighters, but the encroachment of this development into the wildlands is decreasing the percentage of available agricultural land in the state. Because of this loss, as former Commissioner of Agriculture Cary Peterson has noted, Utah is becoming increasingly dependent on food imported from other states. As it has often been said, "Asphalt is the land's last crop."

Figure 3—Urban encroachment, Snyderville, Utah

Taking Action

While everyone is busy debating "wilderness areas," landscape-level changes are occurring and ecological health is at risk. Fortunately, we are making some policy inroads that can help us address our problems, through the Healthy Forests Restoration Act and changes in National Environmental Policy Act (NEPA) requirements. The Healthy Forests Restoration Act provides for expedited hazardous fuel treatments, biomass grants, watershed forestry assistance, insect and disease applied research, the Healthy Forests Reserve Program, and enhanced forest inventory and monitoring authorities.

Another helpful measure is the Good Neighbor Authority, which was authorized by Congress for Utah through an amendment to the Knutson-Vandenberg Act; Colorado has similar authority. This authority provides for the State Forester of Utah to accomplish restoration work on Utah's National Forests by mutual consent and agreement. The state acts as agent for the Forest Service, using the state's purchasing authorities and contractual procedures, while NEPA responsibilities reside with the federal agency. Projects may include treatment of insect infested trees, reduction of hazardous fuels, and other activities to restore or improve forest, rangeland, and watershed health including fish and wildlife habitat.

How can we stop our downward ecological spiral? – through vegetation management. To use a poker analogy, "we have to play the cards we're dealt." We can't do much about precipitation, aspect, soil, or landform/slope, but vegetation management is our "ace in the hole." Depending on how we play that card, we can win or lose the game. What are the stakes in this game? – healthy ecosystems.

We know the outcome when ground cover falls below the site conservation threshold – increased erosion. We know how the degree of site protection impacts changes in plant communities. Resource managers have the knowledge, expertise, and now, available technology to manage vegetation to meet our desired objectives. Figure 4 represents computer-generated simulations taken from a landowner's Forest Stewardship Plan, which demonstrate how a forester can utilize his knowledge and available technology to project treatments that will meet landowner objectives.

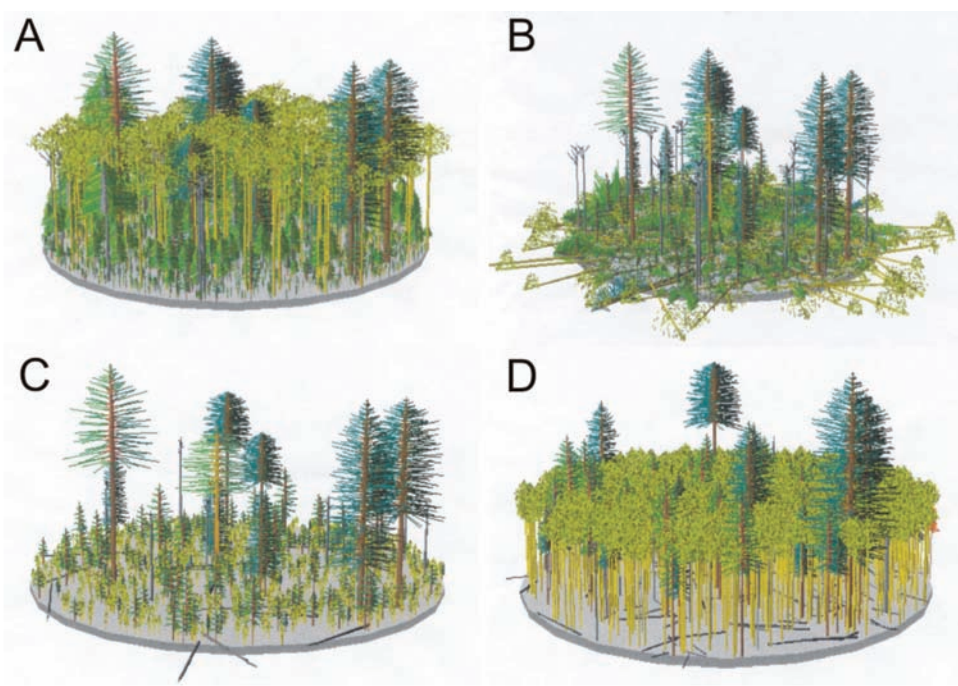

Figure 4—Stand Visualization. A) year 2000 (inventory), B) year 2000 (post-cut, C) year 2010, D) year 2100.

Conclusion / Management Implications

Just like fire, correcting the real problems in our environment will require an ongoing, persistent process of prevention, detection, suppression, rehabilitation/restoration, and monitoring (fig. 5). In a report on Forest Health in Utah (Utah Division of Forestry, Fire, and State Lands 2003) , several recommendations were made to address some of these problems:

1. The report recommended addressing National Environmental Policy Act (NEPA) guidelines and internal review processes to allow for emergency action to address insect and disease infestations and noxious weed invasions. While some changes have occurred, they need to go further for emergency action. Increased flexibility is needed to allow appropriate and timely action to deal with forest health (and other) threats.
2. Land managers need the opportunity to make up-front investments for healthy ecosystems, which will result in reduced expenditures in suppression (such as fire suppression). Examples include green strips and prepared fuel breaks that could resist the invasion of cheatgrass and slow the progression of wildfire.
3. Management direction on federal forest lands should provide for the harvest or management of an amount closer to present net annual growth, possibly 50 to 75 percent in non-reserved areas. This could reduce the accumulation of biomass while contributing to rural economies and sustaining local forest-based businesses, without which we would lose the expertise and equipment needed to accomplish forest management.
4. Private landowners, county weed boards, and land management agencies need to take aggressive action for prevention and suppression of noxious and invasive weeds through proper management of vegetation for healthy ecosystems. Vacant fields and abandoned properties are ripe for takeover.

The question then is, do we have the will to reverse these landscape-level changes? It won't happen unless natural resource professionals have their hands freed so the proper application of vegetation management techniques can take place.

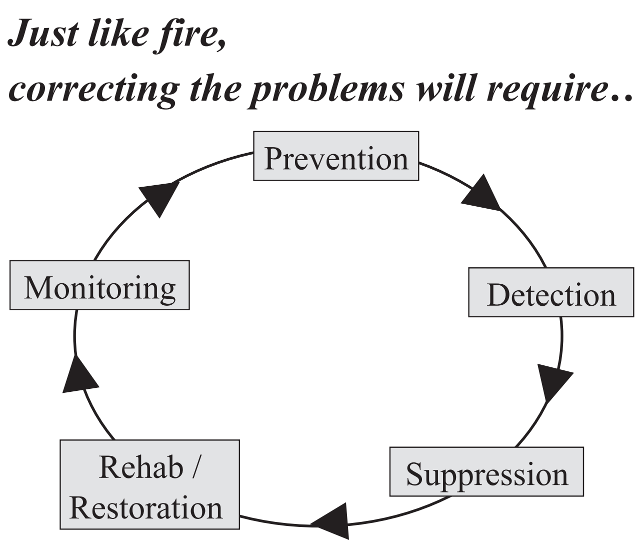

Figure 5—Land management cycle.

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Stand Level Impacts of *Ips* and *Dendroctonus* Bark Beetles in Pine Forest Types of Northern Arizona

Joel McMillin[1], John Anhold[1] and José F. Negrón[2]

Abstract—*Extensive tree mortality occurred in ponderosa pine forests and piñon-juniper woodlands of Arizona from 2001–2004. This mortality has been attributed to a combination of an extensive drought, overstocked stands of pine, and increased bark beetle populations. A complex of* Ips *and* Dendroctonus *species worked in concert to kill ponderosa pine. Piñon pine was attacked primarily by pinyon ips and to a lesser extent twig beetles. Initial tree mortality was associated with poor site quality (i.e., shallow soils, cinder hills, south-facing aspects, and lower elevations) and high tree densities.*

Forest health monitoring, evaluation monitoring, funds were used to: 1) quantify the impact, extent and severity of bark beetles on ponderosa and piñon pine at the stand level through an extensive plot network on a portion of Arizona's northern National Forests, 2) describe the forest conditions in areas that have experienced moderate to high levels of mortality induced by recent drought and bark beetles and 3) look for correlations between stand and site conditions and pine mortality.

A GIS approach was used to populate sample points for each National Forest and forest type. The number of sample points was determined by the amount of area per forest type per Forest. A total of 941 fixed-radius plots were established in 2003 and 2004 across five National Forests in Arizona: Apache-Sitgreaves, Coconino, Kaibab, Prescott, and Tonto. Of these 633 were in ponderosa pine forests and 308 in piñon-juniper woodlands. On the Forest level, ponderosa pine basal area killed ranged from 5 to 23 percent. Ponderosa pine mortality caused by bark beetles was positively correlated with tree density and negatively correlated with elevation (most Forests) and tree diameter (Prescott). Piñon mortality ranged from zero to 48 percent on the Forest level. Piñon mortality was positively correlated with tree density and negatively correlated with elevation on most Forests. In ponderosa pine forests most of the observed mortality was in 10-30 cm diameter classes. In the piñon-juniper woodlands, piñon mortality occurred across all diameter classes with a higher percent of trees killed in the larger diameter classes. Piñon-juniper woodlands have been converted to essentially juniper only in many stands throughout north-central Arizona.

In addition, to this "on the ground" work, we collaborated with Forest Health Technology Enterprise Team, Remote Sensing Application Center and Kodak to analyze different remote sensing applications for the extent and severity of pinyon pine mortality across the Southwest. Satellite and multispectral imagery were collected from the same areas where we installed ground plots.

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Assessment of Drought Related Mortality in Pinyon-Juniper and Ponderosa Pine Forests Using Forest Inventory and Analysis Data

John D. Shaw[1]

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Abstract*—Widespread mortality in several forest types is associated with several years of drought in the Southwest. Implementation of USDA Forest Service Forest Inventory and Analysis (FIA) annual inventory in several states coincided with the onset of elevated mortality rates. Analysis of data collected 2000-2004 reveals the status and trends of mortality in pinyon-juniper and ponderosa pine forests across the Southwest. A complex of drought, insects, and disease is responsible for mortality rates approaching 100 percent in some areas, while other areas have experienced little or no mortality. Drought-related mortality is almost exclusively limited to the pinyon component in pinyon-juniper stands. The proportion of mortality of ponderosa pine varies among the cover types in which it occurs. Elevation and Palmer Drought Severity Index appear to be correlated with observed mortality.*

The Essence of Fire Regime—Condition Class Assessment

McKinley-Ben Miller[1]

Abstract*—The interagency-Fire Regime / Condition Class - assessment process (FRCC) represents a contemporary and effective means of estimating the relative degree of difference or "departure" a subject landscape condition is currently in, as compared to the historic or reference ecological conditions. This process generally applied to fire adapted systems is science-based and adaptive as are the very ecosystems that are being studied. FRCC is also approachable and understandable by citizens participating in an interdisciplinary approach to assessing current ecological conditions.*

Statement

Uncharacteristic catastrophic wildfires that destroy valuable public resources and private property are not beneficial to the citizens of the United States.

Observations

A. There has been an increase in the number and severity of wildfires occurring in the United States.
B. Concerned citizens and our governing bodies are requesting that the potential for uncharacteristic catastrophic wildfires be reduced.
C. It has been, to date, rather difficult to precisely gauge the relative potential of a landscape to support and experience uncharacteristic fire occurrence PRIOR TO a catastrophic fire occurring within a given landscape.

A measure of the relative "combustible condition" or "fire resiliency" a fire adapted landscape of Forest, Shrub or Grassland was in depends upon the degree of deviation the area is experiencing from a combination of ecological factors:

1. Natural or reference fire occurrence (fire frequency)
2. Potential of key ecosystem components to be affected (fire severity)
3. Presence of historic ecosystem components across the landscape (successional presence)
4. Species present within a given landscape (compositional makeup)

These four characteristics, viewed in combination with one another, show a degree of ecological imbalance which can be interpreted as the current "state of health" of a subject landscape.

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Similar interagency assessment protocols are applied to other vegetative types:

1. The Proper Functioning Condition assessment process (PFC) is applied in riparian habitat types.
2. The Indicators of Rangeland Health assessment process (IRH) is applied in Grass/Shrub habitat types.

When considering a larger vegetated landscape, all three assessment protocols can assist the general public and land manager to understand the relative difference or departure to historic conditions that is currently exhibited across the whole subject landscape.

Here are the basic steps to FRCC

1. Select a landscape to evaluate.
2. Determine the Biophysical Settings (Bps) within the selected landscape.
3. Review the historic or reference conditions for each Bps.
4. Ascertain the current fire interval and severity for each Bps.
5. Ascertain the current ecological conditions for each Bps.
6. Compare the "reference" to "current" conditions for each Bps.
7. Assemble the determinations into a landscape result.

The four (4) assessment characteristics—explained:

1. "Natural"/"Reference" fire intervals (occurrence):

There are THREE (3) basic categories of fire intervals.

Forest/Woodland/Shrub/Grassland ecosystems that ignite and support self-sustaining combustion can be placed in one of these three (3) fire return intervals.

1. 0- to 35-year fire return interval (fire frequency)
2. 36- to 199-year fire return interval (fire frequency)
3. 200- + year fire return interval (fire frequency)

2."Natural" / "Reference" fire severity:

There are THREE (3) basic categories of fire severity (the degree to which the aboveground surface vegetation is burned and consumed by fire.)

1. Low severity: surface fires most common, less than 25% of the dominant overstory vegetation replaced.
2. Mixed severity: surface fires quite common, more than 5% but less than 75% of the dominant overstory vegetation replaced.
3. High severity: stand replacement, crown fires are common, greater than 75% of the dominant overstory vegetation replaced.

When combining these two parameters—fire occurrence interval and fire combustion severity—one can categorize vegetated ecosystems into five (5) fire regimes:

I. 0- to 35-year frequency and low to mixed severity (surface fires most common) less than 75% of the dominant overstory vegetation replaced.
II. 0- to 35-year frequency and high severity (stand replacement) greater than 75% of the dominant overstory vegetation replaced.
III. 36- to 199-year frequency and mixed severity (less than 75% of the dominant overstory vegetation replaced).

IV. 36- to 199-year frequency and high severity (stand replacement) greater than 75% of the dominant overstory vegetation replaced.

V. 200 + year frequency and high severity (stand replacement) greater than 75% of the dominant overstory vegetation replaced.

A "natural" or "reference" fire regime is a general classification of the role fire would play across a landscape in the absence of modern human mechanical intervention, but including the influence of aboriginal ignitions.

So… the "FR" in FR/CC is an assessment of the amount of departure from the natural or reference regime.

The four (4) assessment characteristics—continued:

3. Successional presence

All vegetated ecosystems have a centralized tendency to grow in a specific or unique successional pattern. Simply put, each plant must begin life as a seed/spore or immature plant and progress over time—long or short—to an adult mature plant. Then with more time, it progresses to a point where self-sustained respiration ceases and the plant returns to elements from which it was created.

Within a given landscape—some very large, some very small—there is a central tendency to have a specific combination of age groups (structural stages) present at any given time. Obviously, no ecosystem is fixed; therefore natural variability will allow for more or less of this age or that age. This is where the concept of "central tendency" or "expected for this site" applies.

4. Compositional makeup

This attribute describes the "Reference" and/or "Current" ecological components including: species composition, stand age, canopy closure, and size class.

The "current" characteristic is also viewed and interpreted as:

"what is expected for the site."

The data (variables) collected during the FRCC assessment process characterizes the size of the area, geographic location, biophysical conditions, and the fire regime characteristics. This will provide the ecological information that can be used to classify the landscape fire regime and determine the similarity, departure, ecological sustainability risks, abundance of vegetation, fuel classes and the fire regime / condition class.

Ecosystems can occur at any scale, from site to landscape to region. The emphasis in FR/CC is on mid-scale landscapes, because this is broad enough to display the characteristic patterns of a fire regime—the mix of fire frequency, severity and patterns. If the area is too small, a false picture of fire regime and/or condition class is likely. If the area is too broad, we lose the ability to discern meaningful changes in FR/CC.

Effective suggested landscape delineations would be a 5th or 6th code Hydrologic Unit Code (HUC).

In terms of size, a 6th code HUC is generally from 10 to 50 thousand acres, 5th from 25 to 100 thousand acres.

The landscape is then stratified into biophysical settings, project or treatment strata.

The FR/CC Assessment Classes defined:

Condition Class 1:

Fire regimes are within the natural or historical range (+ 33%) and the risk of losing key ecosystem components is considered low.

Vegetative attributes such as composition and structure, are generally intact and, most importantly, functioning.

Condition Class 2:

Fire regimes have been moderately altered (+ 34 to 66%) and the risk of losing key ecosystem components is considered moderate.

(Indicators: Fire frequencies may have departed by one or more return intervals, either increased or decreased. This generally results in moderate changes in fire and vegetative attributes.)

Condition Class 3:

Fire regimes have been substantially altered (+ 67 to 100%) and the risk of losing key ecosystem components is considered high.

(Indicators: Fire frequencies may have departed by multiple return intervals, either increased or decreased. This generally results in dramatic/severe changes in fire size, intensity, severity and landscape patterns. Vegetative attributes have been substantially altered.)

Glossary

Biophysical setting: Biophysical settings are the primary landscape delineations for determination of the natural fire regime and condition class. These units are based on geographic area, physical setting and vegetative community that can occupy the setting. Physical characteristics include climate, geology, geomorphology and soils. Vegetation includes native species and successional stages found under our best understanding of the historic range of variation, including disturbances.

Reference conditions: An estimate of the central tendency of natural or historical (or historic) vegetation-fuel class composition, fire frequency and fire severity for a biophysical unit or landscape area. Reference conditions are the basis for calculating the ecological departure used to determine the Fire Regime/Condition Class.

Similarity: The FRCC methodology compares conditions across a landscape to a central tendency estimate for the natural or historical reference conditions of the potential natural vegetation (PNV).

Abundance: The abundance class is the amount of current vegetation/fuel class compared to the reference condition amount. Classified into Trace, Under-represented, Similar, Over-represented, Abundant.

Fire resilience: "Fire resilience can be defined as a forest stand's ability to survive fires without permanent loss of functional or structural elements. The upper canopy with the oldest and largest trees represents such a structural element. A stand can be considered fire resilient if the probability of a complete loss of the upper canopy is reasonably low.

Sustainable Development through Biomass Utilization: A Practical Approach

Ravi Malhotra[1]

***Abstract**—This paper is for folks involved in community development efforts targeted towards biomass utilization.*

Our approach to evaluate the potential for establishing enterprises that utilize locally available forest resources is tailored specifically to the needs of the local community. We evaluate the:

1. *Technical feasibility and economic viability of the bio-energy and/or wood products manufacturing enterprise.*
2. *Social responsibility of establishing the enterprise in the local community.*

To accomplish our objective, the following tasks are completed:

- *A resource assessment of forest and other biomass sources in the region;*
- *Identification and evaluation of bio-energy and wood product manufacturing technologies;*
- *Recommendations of options for the establishment of biomass-based enterprises;*
- *Business plan development for the recommended options;*
- *Outreach efforts to disseminate information on the business plans developed;*
- *Establishment of the new business enterprise.*

Keywords or phrases: *sustainability; economic development; biomass utilization*

The International Center for Appropriate and Sustainable Technology (iCAST) approach to biomass utilization for the creation of sustainable development for communities is simple. We begin by building awareness and understanding of biomass utilization and land management practices in communities. Next we conduct:

1. A resource assessment of forest and other biomass sources in the region—we evaluate the quantity, quality, and type of biomass available. We estimate the procurement costs to get the biomass to the processing site, we break this down by owner type i.e. USFS, BLM, State forests, private, city, parks and recreation department, etc.
2. Identification and evaluation of bio-energy and wood product manufacturing technologies that match the resource availability – building a 100 MW biomass power plant at a site where 20,000 tons/year of biomass is available is not a viable option because the power plant will need orders of magnitude more biomass. Also, building a 200,000 tons/year wood pellet plant at a location with a demand for 2,000 tons/year of pellets may not be viable in the long term. Finally, building a state-of-the-art ethanol/methanol production facility in the middle of a rural area, with no access to the markets, no infrastructure, and no access to the technical and business expertise needed

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to operate and manage such as facility is not viable either. When evaluating biomass options we look at the technical feasibility i.e. is the technology reliable? And the level of automation i.e. is the technology cost effective and consistent?

3. Recommendations of options for the establishment of biomass-based enterprises—Supply needs to match demand needs to match resource availability for a business to be truly sustainable. The iCAST approach to making recommendations involves using the decision matrix approach where the community is part of the decision making process and the final recommendation is based on their needs. When evaluating biomass options we look at the technical feasibility (i.e., is the technology reliable). And the level of automation (i.e., is the technology cost effective and consistent). We also look at economic viability by developing a business plan for the highest value-add products with a market analysis, financial plan and operations strategy.
4. Establishment of the new business enterprise—After developing a business plan for the recommended option, we conduct an outreach program to disseminate information on the business plans developed. The business plan includes a market analysis, financial plan, and operations strategy. The business plan is selected by prioritizing the options by impact on community. By creating a project plan and schedule for each option, we evaluate the ease of implementation. The optimal pilot project should be the one with the highest chance of success or lowest risk of failure so that it creates a success story for others. We support effective project implementation by identifying local entrepreneurs willing to work on the plan and assisting them in locating the funding and management team to execute on the business plan.

We also provide real-life service learning opportunities to university students. Our approach involves developing a stakeholder-approved action plan that identifies innovative service learning opportunities for Colorado university students. Finally we disseminate the knowledge by conducting workshops in those communities in partnership with local stakeholders.

Management Implications

The beauty of our approach is in its simplicity. By matching local biomass supply to the business capacity and demands of the market, our approach ensures that the business is viable and sustainable for the long-term. Many times, the business capacity is driven by volume efficiencies and after a few years comes the realization that biomass supply is unable to meet demand so procurement costs rise or worse, the business is unable to run at capacity and is highly inefficient. The viability of the business was dependent on cheap biomass supply and so the business fails. Meeting local demand with local supply is a sustainable approach to not only business viability but also forest health, because there is no pressure for additional biomass resources from the utilization business.

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Poster Presentation Titles

Photo by Gerald Gottfried

Poster Presentation Titles

(An asterisk indicates that the abstract or article appears in this proceedings. The authorships and titles could be slightly different. Senior authors or presenters are mentioned.)

St. George, Utah

Presenter	Title
John Anhold	1. *Stand Level Impacts of *Ips* and *Dendroctonus* Bark Beetles in Pine Forest Types of Northern Arizona
	2. *Piñon Pine Mortality Event in the Southwest: An Update for 2005
	3. *Influence of Elevation on Bark Beetle Community Structure in Ponderosa Pine Stands of Northern Arizona
	4. *Do Bark Beetle Community Sprays Prevent *Phloeosinus* Species from Attacking Cypress and Juniper?
Michelle Cattaneo	Small-Scale Variation in Soil Moisture Content Under Canopies of One-Seed Juniper Trees
Andres Cibils	*Pinyon-Juniper Research at New Mexico State University's Corona Ranch
J.M. Dhaemers	Understory Recovery After Spring Prescribed Fire in a Pinyon-Juniper Watershed
Brett Dickson	Spatial Tools for Predicting the Effects of Fuels and Restoration Treatments on Forested Landscapes
F.C. Hassler	Expansion Patterns of *Juniperus monosperma* into a Semi-Arid Grassland at Wupatki National Monument in Northern Arizona, USA
Valerie Horncastle	Tree Roosting Bat Responses to Forest Restoration: Techniques and Preliminary Results
Mark W. Loveall	*Prediction Equations for Aboveground Biomass and Carbon Distribution in *Pinus edulis*
McKinley-Ben Miller	*The essence of Fire Regime Condition Class
Carlos Ochoa	*Rainfall, Soil Moisture, and Runoff Dynamics in New Mexico Piñon-Juniper Woodland Watersheds
Brytten Steed	Piñon Pine Mortality in Utah and Nevada: 2000-2004
Michael Stoddard	Does Slash Help Retain Soil and Increase Grass Cover in a Pinyon-Juniper Woodland?
Robert Sturtevant	Colorado Wood-Utilization and Marketing Program
Joseph Trudeau	Fungal Inoculum and Wood Mulch Application in Restoring Roads in Arizona Ponderosa Pine Forests
Mary Williams	*Carbon Distribution within a *Pinus edulis/Juniperus monosperma-Bouteloua gracilis* Interface Site in North Central New Mexico

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Albuquerque, New Mexico

Presenter	Title
Greg Gallegos	Santa Fe County Open Spaces: Change in P-J in Southern New Mexico
Douglas Page	*Preliminary Thinning Guidelines Using Stand Density Index for the maintenance of Uneven-aged Pinyon-Juniper Ecosystems
Julie Prior-Magee	*Removal of Pinyon-Juniper Woodlands on the Colorado Plateau
Signa Larralde and Sarah Schlanger	*Why is Cultural Resource Site Density High in the Piñon-Juniper Woodland?

Notes

Notes

The Rocky Mountain Research Station develops scientific information and technology to improve management, protection, and use of the forests and rangelands. Research is designed to meet the needs of the National Forest managers, Federal and State agencies, public and private organizations, academic institutions, industry, and individuals. Studies accelerate solutions to problems involving ecosystems, range, forests, water, recreation, fire, resource inventory, land reclamation, community sustainability, forest engineering technology, multiple use economics, wildlife and fish habitat, and forest insects and diseases. Studies are conducted cooperatively, and applications may be found worldwide.

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