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*Synthesis*

# Sensitivity of the Colorado Plateau to Change: Climate, Ecosystems, and Society

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## ABSTRACT

The Colorado Plateau is located in the interior, dry end of two moisture trajectories coming from opposite directions, which have made this region a target for unusual climate fluctuations. A multi-decadal drought event some 850 years ago may have eliminated maize cultivation by the first human settlers of the Colorado Plateau, the Fremont and Anasazi people, and contributed to the abandonment of their settlements. Even today, ranching and farming are vulnerable to drought and struggle to persist. The recent use of the Colorado Plateau primarily as rangeland has made this region less tolerant to drought due to unprecedented levels of surface disturbances that destroy biological crusts, reduce soil carbon and nitrogen stocks, and increase rates of soil erosion. The most recent drought of 2002 demonstrated the vulnerability of the Colorado Plateau in its currently depleted state and the associated costs to the local economies. New climate predictions for the southwestern United States include the possibility of a long-term shift to warmer, more arid conditions, punctuated by megadroughts not seen since medieval times. It remains to be seen whether the present-day extractive industries, aided by external subsidies, can persist in a climate regime that apparently exceeded the adaptive capacities of the Colorado Plateau's prehistoric agriculturalists.

*Key words: biological crust; climate change; megadrought; ranching*

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# INTRODUCTION: THE COLORADO PLATEAU AS A MODEL FOR GLOBAL CHANGE STUDIES

Located between the Rocky Mountains of Colorado and New Mexico on the east and the high Sierra Nevada of California on the west, from about 34° to 40°N and 107° to 114° W, are the driest regions of the United States (Fig.1). The Colorado Plateau, with its mountains, desert basins, and arid woodlands and grasslands, encompasses the southern Four Corners region of the Intermountain West (southeast Utah, southwest Colorado, northwest New Mexico, and northeast Arizona; West 1988). Known also as cowboy country, this vast region of more than 340 000 km$^2$ is used primarily as rangeland, whereas 15 438 km$^2$ of National Park lands showcase some of the most spectacular geologic formations of North America. Humans have lived on the Colorado Plateau for nearly 10 000 years (Powell and Smiley 2002), but occupation has not been continuous and it is now believed that climate instabilities may have contributed to this interruption in human settlement (Larson and Michaelsen 1990, Axtell et al. 2002, Coltrain and Leavitt 2002). We suggest that the unique location of the Colorado Plateau at the boundaries of two climate zones may have exposed this region to more extreme fluctuations in climate than other arid regions nearby.

The Colorado Plateau is situated in the interior, dry end of two moisture trajectories of opposite direction (Fig. 2). In summer, this region receives limited moisture primarily as intense, local convective storms. In fall and winter, limited moisture arrives as large frontal systems. Over the last 10 000 years, the Colorado Plateau has frequently experienced shifts in the abundance of summer and winter precipitation, likely caused by the geographic displacement of these climate boundaries. These changes have altered vegetation growth patterns and are correlated with both the rise and ultimate disappearance of prehistoric Americans from much of the region (Betancourt et al. 1990, Petersen 1994, Cayan et al. 1999, Salzer and Kipfmueller 2005).

Vegetation patterns on the Colorado Plateau not only follow climatic drivers but also reflect a dominant human impact on the landscape through grazing over the past two centuries (Cole et al. 1997). Even today, Colorado Plateau ecosystems are changing as humans increase use of the most remote regions and invasive species continue to replace native vegetation, altering both fire regimes and the nitrogen (N) and carbon (C) cycles. Here, we describe the sensitivity of this ecological community to change and suggest that, due to its unique location, it may be among the most sensitive of ecosystems to past and current drivers of global change. The Colorado Plateau may, therefore, serve as a model for investigating some of the most challenging questions about the relationships between global change and human affairs: What changes are driven by climate, human land use, and their interactions? How do ecological change and climate variability affect human affairs? How can society build capacity to buffer against the short-term instability and preserve long-term stability of ecological communities?

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# PAST, PRESENT, AND FUTURE CLIMATE OF THE COLORADO PLATEAU

The climate of most of the Colorado Plateau is classified as semi-arid, with total annual precipitation approximately 250 mm yr$^{-1}$, with drier areas in the southern portion receiving as little as 130 mm yr$^{-1}$, and higher elevations around the perimeter of the Plateau receiving as much as 670 mm yr$^{-1}$ (Hereford et al. 2002). Winters are cold, with moisture coming from the Gulf of Alaska and Pacific Ocean to the north and west. Summers are warm to hot, with a distinct wet period characterized by intermittent but often intense monsoonal storms coming from the eastern tropical Pacific, the Gulf of California, and the Gulf of Mexico (Adams and Comrie 1997, Barlow et al. 1998). Near Moab, Utah, in the heart of the Colorado Plateau, total average annual precipitation is approximately equal to the Great Basin desert to the north and the Chihuahuan and Sonoran deserts to the south. However, the Great Basin desert largely lacks summer moisture, whereas the Chihuahuan and Sonoran deserts have a greater and more reliable summer rainfall component than the Colorado Plateau (Fig. 2). Both winter and summer precipitation are highly variable from year to year. This variability has been linked to El Niño and La Niña patterns (Harrington et al. 1992, Hereford and Webb 1992, Cayan et al. 1999, Cook et al. 2007), although the correlations are modest. La Niña is generally associated with dry winters in the southwest U.S., and El Niño usually (but not always) leads to increased cool-season precipitation (Hereford et al. 2002).

As we explain in more detail below, the current climate of the Colorado Plateau changed about 10 000 years ago from a previously wetter state. The Holocene has been a period of unparalleled climate variability, with frequent multi-decadal excursions from the precipitation means, but no overall trends in aridity. In the last 30 years, the Colorado Plateau has experienced a 0.2 to 0.5°C increase, particularly in cold-season temperatures, as typical for the mid-latitudes (http://data.giss.nasa.gov), and climate models forecast the continuation of these patterns, but with periodic droughts that will be more severe and possibly longer, due to an overall increase in aridity (Seager et al. 2007).

Paleobotanical evidence from packrat middens on the Colorado Plateau indicates that the current climate pattern has been dominant for at least 10 000 years (Betancourt 1990). However, multi-decadal excursions from the climatic mean have been a common occurrence. Climate reconstructions based on tree ring chronologies (D'Arrigo and Jacoby 1991, Meko et al. 1995, Ni et al. 2002, Gray et al. 2003, Salzer and Kipfmueller 2005, Cook et al. 2007) show that the region experienced several sustained drought periods between the late 10th and mid-13th centuries, dubbed the medieval megadroughts. Anomalously wet periods were apparent in the early 14th, 17th, 19th, and 20th centuries. The latter period was one of the wettest in recorded history. The 1905–1922 period had the highest long-term annual Colorado River flow volume in the 20th century and, coincidentally, served as the basis for estimating water production allocated under the Colorado River Compact (http://pubs.water.usgs.gov/fs20043062).

Multi-decadal drought cycles (Stahle et al. 2000, Gray et al. 2003, Kitzberger et al. 2007) are linked to the Pacific Decadal Oscillation (PDO; Hereford et al. 2002), which is an index of sea surface temperatures (SST) in the northern Pacific (Mantua and Hare 2002). As SST and related properties (surface winds and air pressure) change in the north Pacific, climate transitions lasting 2–3 decades are observed in North America (Zhang et al. 1997). The PDO, tree-ring records, and measured precipitation are correlated on the Colorado Plateau, suggesting that the geographic locations of climatic boundaries in the American Northwest are influenced by multi-decadal global climate cycles, with the Colorado Plateau at the center of the instability.

Five severe, multi-year drought events have been recorded in the western United States in the 20th century: the 1930s' "Dust Bowl," the 1950s' Southwest drought, and the 1980, 1988, and 2002 droughts, the latter having shown signs of ending in 2005 (Cook et al. 2007). All could be attributed to variations of SSTs in the tropics and persistent La Niña-like patterns (Seager et al. 2007). According to the latest climate forecasts, SST fluctuations will continue to cause drought cycles, but against a background of generally increased aridity caused by changes in atmospheric circulation patterns (Seager et al. 2007). A 19-model consensus forecast for the western United States includes the possibilities of megadrought events by the second half of the 21st century. These droughts are predicted to far exceed the severity of any drought in living memory, including the "Dust Bowl," and may be more akin to the medieval megadroughts. The new predictions suggest that the "Dust Bowl" years may better reflect the base condition of the 21st century climate state in the American Southwest, with La Niña patterns creating periodically far drier conditions.

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# VEGETATION AND ECOSYSTEMS

Although summer and winter precipitation amounts may be similar, winter precipitation largely determines total annual primary productivity among perennials (Caldwell 1985). Winter precipitation accumulates in the soil due to the low evaporative demand and low transpiration rates at that time of the year. Plants begin to use these soil moisture reserves in early March, when plant growth is triggered by warmer temperatures. By late June, the driest month of the year, reserves of winter soil moisture near the soil surface are largely depleted and shallow-rooted perennial plants go dormant, whereas deeper-rooted plants take up residual water from deeper in the soil profile (Fernandez and Caldwell 1975, Ehleringer et al. 1991). Summer precipitation from late July to mid-September usually creates short pulses of soil moisture that can only be used by plant species that maintain an active shallow-root system, chiefly grasses and late-summer annuals (Ehleringer et al. 1991, Lin et al. 1996, Schwinning et al. 2003).

The vegetation of this region has been previously reviewed (Caldwell 1985, West 1988, Comstock and Ehleringer 1992). In the subalpine regions of the tallest peaks, Engelmann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*) are common. With decreasing elevation, these communities transition into a mixed-conifer forest with Douglas fir (*Pseudotsuga menziesii*), white fir (*Abies concolor*), Colorado blue spruce (*Picea pungens*), and finally ponderosa pine (*Pinus ponderosa*). Between 2100 and 1600 m, pinyon/juniper woodlands dominate (*Pinus edulis*, *Juniperus osteosperma*, and on the southern edge of the Plateau, *J. monosperma*). Lower in elevation, and occupying most of the Colorado Plateau, is desert scrub, a mixed community of mostly low evergreen or winter-deciduous shrubs and perennial grasses. Sagebrush species (*Artemisia tridentata, A. tridentata* ssp. *vaseyena, A. wyomingensis*) are found at the higher ranges of the shrubland. At lower elevations, the dominant plants include blackbrush (*Coleogyne ramosissima*), Mormon tea (*Ephedra viridis* and *E. torreyana*), saltbush (*Atriplex canescens, A. confertifolia* and *A. garrettii*), and winterfat (*Ceratoides lanata*). Many of these shrubs were already present during the last glacial cycle, although displaced to lower elevations (Betancourt 1990). The many grass species of the Colorado Plateau include the $C_3$ Indian ricegrass (*Stipa hymenoides*) and needle-and-thread grass (*S. comata*) and the $C_4$ *Bouteloua gracilis, B. eriopoda*, *Sporobolus cryptandrus S. airoides*, and galleta (*Hilaria jamesii*).

Notable changes in species composition occurred 200–150 years ago, coincidental with the introduction of sheep and cattle grazing (Cole et al. 1997). Especially palatable species, such as winterfat and ricegrass, declined, whereas species previously absent from packrat midden records and indicative of overgrazed range, increased (Cole et al. 1997). These included snakeweed (*Gutierrezia sarothrae*), viscid rabbitbrush (*Chrysothamnus nauseosus*), Russian thistle (*Salsola tragus*), and cheatgrass (*Bromus tectorum*).

The landscapes of the Colorado Plateau evolved under low or sporadic grazing pressure. Evidence for this includes the lack of dung beetles, a dominance of grasses that are intolerant of grazing, and soils that are highly vulnerable to erosion when trampled (Mack and Thompson 1982). Bison (*Bison bison*), although present in surrounding areas during the Holocene, were absent from the Colorado Plateau (Mack and Thompson 1982, van Vuren and Dietz 1993). Limited surface water and forage kept native ungulates such as pronghorn (*Antilocapra americana*) and mule deer (*Odocoileus hemionus*) numbers low and at higher elevations, except during some winters when snow provided a water source at lower elevations.

Repeat photographs comparing today’s landscapes with those of 100 years ago show that most native ecosystems did not change during this interval and many individual shrubs and grasses photographed in 1898 are still alive and the same size today (Hart and Laycock 1996, Webb et al. 2004). Mortality rates estimated from repeat photographs indicate that many woody shrubs can probably live well over 100 years, and some perennials considered short lived under range conditions can live 70–100 years (Bowers et al. 1995).

The presence of $C_4$ grasses on the Colorado Plateau scrublands is of particular interest, as these species tend to be found where summer rains are reliable. During the last glacial cycle, $C_4$ plants were absent from the Colorado Plateau, but present in the adjacent warm Chihuahuan and Sonoran Deserts (Connin et al. 1998). This has led to speculation that monsoonal flow of summer precipitation to the Colorado Plateau may have been suppressed during the last Ice Age (Anderson et al. 2000). Subsequently, a period of particularly strong summer rains that peaked 9000 years ago may have been responsible for the rapid expansion of $C_4$ grasses in the lowlands and, in the higher elevations, of ponderosa pine (Betancourt 1990).

Multi-decadal patterns of drought and high-moisture periods (Gray et al. 2003, Hereford et al. 2002) may have been associated with the waxing and waning of particular vegetation components and overall plant cover, but did not fundamentally reorganize this community. Decadal droughts have influenced fire frequencies, insect outbreaks, woody encroachment, and plant mortality (Swetnam and Betancourt 1998). The 2002 drought has resulted in mass die-off of pinyon and ponderosa pine, as well as high mortality in shallow-rooted species such as yucca and cactus (Breshears 2006, Belnap pers. obs.). Historic droughts may have also led to shifts in ecosystem boundaries (Allen and Breshears 1998). Anomalously wet periods, such as the post-1976 period, have promoted seedling establishment and the vigorous growth of existing shrubs (Swetnam and Betancourt 1998).

Plants of the Colorado Plateau all respond to winter precipitation, but responses to summer precipitation vary (Ehleringer et al. 1991, Flanagan et al. 1992, Lin et al. 1996, Gebauer and Ehleringer 2000, Schwinning et al. 2002, 2003, West et al. 2007a, 2008). In general, the most deep-rooted species are least responsive to summer rain, among them viscid rabbitbrush, winterfat, and shadscale (*Atriplex confertifolia*). A widely distributed class of vegetation with dimorphic root systems (a shallow, spreading root system combined with a moderately deep taproot)—including blackbrush, snakeweed, and sand sagebrush (*Artemisia filifolia*)—have intermediate use of summer rain. It has been suggested that this root distribution optimizes the plant’s ability to rapidly switch between deeper and shallower water sources according to availability (Schwinning and Ehleringer 2001). Grasses use the most summer rain, but where grass cover is sparse, particularly where livestock grazing occurs, shrubs and subshrubs with dimorphic root systems are probably the greatest consumers of summer rain water at the ecosystem level, provided rain events exceed 10 mm (Schwinning et al. 2003).

Trees to the south of the monsoon boundary tend to have higher capacities for the uptake and utilization of summer precipitation than conspecifics to the north (Fig. 2 inset). This suggests that there may be a minimum amount of summer rain required for changing rooting patterns to occur, whether phenotypically or genetically, and that the minimum for pinyon pine and juniper is between 100 and 150 mm (Williams and Ehleringer 2000, West et al. 2007a, 2007b, 2008). Below this threshold, surface roots may be too costly to maintain through dry periods and plants may lose more C than they would gain on average if they responded to summer rainfall (Schwinning and Sala 2004).

## THE SPECIAL ROLE OF BIOLOGICAL SOIL CRUSTS

Soils between the widely spaced vascular plants on the Colorado Plateau are generally dominated by a community of cyanobacteria, microfungi, lichens, and mosses collectively known as biological soil crusts (BSCs; Fig. 1, inset). Biological soil crusts dominate the landscape in this region, can represent up to 70% of the living cover (Belnap 1995), and can heavily influence system function in most ecosystems (Fig. 3). The presence of BSCs is critical for soil fertility, moisture, and stability in this region. Like the higher plants of the Colorado Plateau, the timing and rates of physiological activity of the BSC are governed by rainfall inputs and temperature. Biological soil crusts are active whenever wet, and unlike plants, can respond to even 1-mm rain events.

Biological soil crusts are an important source of C for subsurface soil organisms in these ecosystems, as soil C is limited in desert soils (Belnap 2003a). Whenever dry crusts are wetted, there is an immediate respiratory burst of carbon dioxide ($CO_2$) as well as a 30–60 min time lag before respired C can be replaced by photosynthesis (Lange 2003; Lange et al. 1996, 1997). To compensate for this initial loss, and thus maintain a positive C balance, the soil surface must remain wet for an extended period of time during the day, a condition most likely to be satisfied in late fall through spring when air temperatures are cool.

Summer rains can be problematic for BSCs. Whereas higher plants can control their physiological response to rain in summer to avoid possible C losses, BSCs have no such control and summer rains usually result in rapid wetting and drying cycles. If BSCs dry too soon, they cannot replace the C lost during wetting, and BSC organisms suffer C deficits as a consequence (Belnap et al. 2004, 2006). Net C gain in summer, given high air temperatures, requires precipitation events greater than 3 mm, yet more than 70% of all rainfall events in summer are below this threshold (data courtesy of U.S. Geological Survey Clim-Met, specifically Site #8 at Corral Pocket, southern Utah, 38° 09.687' N, 109° 39.568' W). Furthermore, rainfall events at night will result only in respiratory C losses and no C gains, and up to 70% of rainfall events in summer occur at night.

Free-living and lichenized cyanobacteria (e.g., *Nostoc* in soil and *Nostoc* in the lichen *Collema*) provide the largest N input for Colorado Plateau soils (Evans and Ehleringer 1993, Evans and Belnap 1999). Nitrogen fixation only occurs when the cyanobacteria or lichens are wetted and have sufficient C compounds to act as electron donors (Paul and Clark 1996). Thus, the rainfall requirement for N fixation exceeds the requirements of net C fixation.

With sufficient moisture and C stores, N fixation rates are controlled by temperature. As with C, most N fixation occurs between -5ºC and 30ºC, with the optimum at 20–25ºC (Belnap 2003a). Estimates of annual N fixed on the Colorado Plateau range from 1–9 kg/ha/yr, depending on BSC composition (Belnap 2002). Up to 70% of the newly fixed N is released from the BSC organisms and is readily taken up by surrounding organisms, including vascular plants, fungi, actinomycetes, and bacteria, as well as reabsorbed by the releasing cyanobacteria or lichen (reviewed in Belnap 2003b).

Like C, most N is fixed September–May. Winter fixation rates are aided by the dark color of BSCs, which by enhancing absorption of solar energy can increase their temperature as much as 14ºC on sunny days (Belnap 1995). Due to cold subsurface soils, gaseous N losses are minimal during this time (Barger et al. 2005). Thus, soil N levels build up over fall and winter and peak in spring, when the demand and uptake capacity of native vascular plants and microbial populations is maximal (Gebauer and Ehleringer 2000, Schwinning et al. 2005). In summer, high soil temperatures preclude N fixation and facilitate gaseous loss, although losses are still relatively low (<1 kg/ha/yr; Barger et al. 2005). Biological soil crusts affect the availability of soil nutrients in other ways as well (Belnap 2003b). They secrete powerful metal chelators and other substances that help maintain nutrients in plant-available forms, important in these high pH desert soils. They facilitate the formation of soil aggregates, and soil faunal abundance and richness is greater under crusted surfaces than non-crusted surfaces. Thus, their presence enhances C and nutrient transformation rates. They greatly roughen the soil surface (Fig. 1, inset), increasing the retention of seeds, organic matter, and nutrient-laden dust (Belnap et al. 2003). As a result, plants growing in crusted soils often have greater productivity and higher leaf nutrient concentrations than plants growing in adjacent, uncrusted soils.

Although BSCs are critically important in C and N cycles in this region, their greatest contribution to ecosystem health and the maintenance of soil fertility may be due to their impact on soil physical properties. By binding together soil particles and increasing the formation rate and size of soil aggregates, they increase soil aeration and porosity while drastically reducing both wind and water erosion rates (Belnap and Gardner 1993). Their roughened surface slows runoff of precipitation and thus increases infiltration into the soils (Warren 2003). They also “cap” the soil surface, retarding evaporation and thus increasing soil moisture retention times (George et al. 2003).

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# HUMAN IMPACTS

In prehistoric times, the agricultural settlements of the ancestral Pueblo people on the Colorado Plateau coincided with the Medieval Warm Period (800–1300 AD), a warm period that, during its height, extended the growing season and increased both winter and summer precipitation (Petersen 1994). This climate opened opportunities for dry farming of maize at elevations between 1600 and 2300 m. Reconstructed household numbers in the Long House Valley in the Black Mesa area demonstrate 300 years of population increases, followed by 200 years of relative stasis, and then a sudden collapse at the height of the Little Ice Age around 1300 (Axtell et al. 2002), which would have closed the dry farming belt in upland areas (Petersen 1994). The Medieval Warm Period was also associated with several megadrought events (Cook et al. 2007), the last one peaking in 1253, shortly before the collapse of the Pueblo culture on the Colorado Plateau.

The Medieval Warm Period was followed by the Little Ice Age, and when it ended in the 1800s, human settlers returned to the Colorado Plateau (Petersen 1994). The new settlers were Anglo farmers, who brought sheep and cattle and began dry-farming beans, winter wheat, and, to a lesser degree, fruit in the upland belt (2000–2300 m). Mining of ore deposits occurred where local geology allowed it.

When the Mormon pioneers arrived in Utah in 1847, they encountered a perennial grass-dominated landscape with few native herbivores. Within a short time after their arrival, grazing by their livestock converted the areas surrounding and connecting settlements into shrublands generally dominated by sagebrush (Cottam 1948). The anomalous wet period in the early 1900s coincided with, and likely aided, an unprecedented expansion of ranching and farming throughout the western U.S.

Grasslands recover from overgrazing slowly and incompletely. In a field study at the Desert Experimental Range in southwestern Utah, vegetation took 59 years to recover from a shrub-dominated, heavily overgrazed state in 1935 to a state where grass cover only slightly exceeded shrub cover in 1994 (Alzerreca-Angelo et al. 1998). Reduced grazing pressure and a wetter climate led to consistently increasing grass cover over the entire 59-year study period, but shrub cover changed only slowly with an initial increase and a slow decline only after 25 years of decreased grazing pressure. Currently, almost 90% of the Colorado Plateau is used by the ranching industry, but cattle densities are low compared with the early 20th century, as it frequently takes up to 60.7 ha (150 acres) or more to support one animal. The current year-round human population size on the Colorado Plateau is about 2.1 million (U.S. Census Bureau, 2005 estimate), and in 2001, there were 733 000 head of cattle (USDA National Agricultural Statistics Service), reduced by the 2002 drought to 498 000. The Colorado Plateau now also supports vast deer herds. Whereas in 1900, the deer population was an estimated 8000 animals (Cottam 1948), in the 2002 drought year, the Utah Division of Wildlife estimated the population at 425 000 animals with a "target" population of 450 000 animals (Utah Division of Wildlife, pers. communication to Belnap, 2002). Deer numbers have increased dramatically due the development of surface water sources, the invasion of sagebrush, the removal of natural predators, and a demand from the hunting public for high hunting success rates. To increase hunting opportunities, the Utah Division of Wildlife also transplanted Rocky Mountain elk into Utah. The current herd size of 60 000 animals (Adams 2005) further contributes to the grazing and trampling pressure on these landscapes.

High animal numbers and chronic trampling and herbivory have had profound impacts on Colorado Plateau ecosystems (Fig. 3). Grasslands have been converted into sagebrush-dominated landscapes and BSCs have been severely impacted. Although BSCs have high tensile strength, they have little resistance to the shear and compressional forces generated by trampling or vehicle use. Lichens and mosses, the BSC components that contribute the most to soil fertility and stability, are also the first BSC components to be lost (Belnap and Eldridge 2003). The subsequent loss of C and N inputs, resource capture, and soil stability has reduced soil fertility. Studies show a consistent 40%–70% reduction in soil C and N when low-elevation areas currently grazed by livestock are compared with nearby areas that have never been used by livestock (e.g., Neff et al. 2005, Fernandez et al. 2006). Grazed sites have up to 40% less silt (and thus less water-holding capacity) and up to 50% less total elemental soil magnesium (Mg), sodium (Na), phosphorus (P), and manganese (Mn) contents relative to soils never exposed to livestock disturbance. In addition, soil organic matter in this region has a fast turnover time. Soil C disappears quickly when plant biomass is reduced by heavy grazing.

Other human activities in this region have exacerbated the effects of livestock grazing and high deer populations. Energy exploration and recreation (e.g., hiking, camping, mountain biking, off-road vehicle recreation) are increasing exponentially, and often occur in areas not previously impacted by livestock, as humans are not limited by naturally occurring food and water sources. This has resulted in large amounts of additional soil disturbance, further reducing BSC cover in this region, with an attendant reduction in soil fertility and stability. Areas heavily used by recreationists for 5–10 years often show greater resource loss than landscapes grazed by livestock for over 100 years.

Thus, the widespread surface disturbance in this region will continue to reduce the ability of these lands to support current plant communities and the wildlife and livestock that depend on them. As BSC cover declines, N inputs into the soils decline and loss rates increase, thus reducing the N content of both soil and plants (Evans and Belnap 1999). These losses are long term (Evans and Belnap 1999) and, even though arid and semi-arid lands are primarily limited by water, rangeland forage production is often limited by soil nutrients, particularly in higher than average rainfall years (Guevara et al. 2000). Because of the slow recovery of lichens and mosses in this region (>250 years; Belnap et al. 2003) and soil formation rates (5000–10 000 years; (Dregne 1983), areas receiving intense or chronic soil disturbance should be considered more or less permanently impaired.

In addition to surface disturbance, human-related activities have other consequences as well. Seeds of exotic invasive annual grasses are spread by livestock and vehicles, and their germination is enhanced by soil surface disturbance (e.g., Belsky and Gelbard 2000). In addition, livestock herbivory reduces the ability of native plants to compete with the less-preferred exotic annual grasses, resulting in the loss of the perennial component. During drought years, these annual grasses often fail to germinate or do not grow well (Belnap et al. 2006), leaving the soil surfaces highly vulnerable to erosion. Dust collection networks show that if annualized areas are also grazed, soil loss via wind erosion can increase several orders of magnitude over non-invaded, non-grazed areas (Belnap, unpublished). Conversely, in wet years, these annual grasses create continuous fuels that carry fire. Despite examination of hundreds of soil pits scattered throughout the landscape, we have not found charcoal or any evidence of widespread fires occurring in the low-elevation communities of this region. Present-day fires generally result in the widespread mortality of native perennials. Periods of sustained drought can lead to widespread wildfires at mid-elevation in this region that supports pinyon, juniper, and pondersosa pine, generally following the death of the trees from insect infestation (Cook et al. 2004, Kitzberger et al. 2007). The combination of increased temperatures and earlier springs has also increased fire frequencies in this region (Westerling et al. 2006).

## ECONOMIC AND ECOLOGIC BUFFERING AGAINST CLIMATE INSTABILITIES

The traditional industries of the Colorado Plateau are extractive, and with the exception of mining, depend directly on adequate rainfall. One of the strongest constraints on these industries is the occurrence of drought years, which push agriculture to the brink of bankruptcy and often beyond. Farming economies have developed mechanisms to buffer against this inevitable production variability and uncertainty, allowing them to store the advantages gained in good years to sustain themselves through bad years. This dependence on rainfall was also experienced by the prehistoric occupants of the Colorado Plateau, although they used different mechanisms to buffer against the consequences of a variable year-to-year climate.

The earlier settlers of the Colorado Plateau lived in family and small village units that were probably self-sustaining. Their increasing wealth and population expansion were associated with successful and expanding maize cultivation. During drought periods, people relied on stored grain, and may have increased consumption of wild-plant products, such as rice grass seed, pine nuts, acorns, and tubers, and animals. Isotopic evidence from analysis of human tooth enamel suggests that by the end of the prehistoric occupation of the Colorado Plateau, near the peak of the last medieval megadrought around 1250, cultivated maize had dropped entirely from the diet and people ate predominantly wild plants (Coltrain and Leavitt 2002). The absence of external subsidies did not allow the earlier agriculturalists to extend extractive efforts beyond the level supported by the local environmental conditions. Although irrigation agriculture was practiced, it too depended on the flow-regime of local rivers and streams, which would have declined under persistent drought conditions. In addition, some argue that centuries of maize cultivation and deforestation may have compromised the productivity of the region and with it the abundance of wild foods (Redman 1999). With limited ability to store grains and find wild foods, the buffering capacities of the early Pueblo people may have become overwhelmed, ushering in the abandonment of the Colorado Plateau region and supposed southward migration into more mesic regions in southern Arizona and along the Rio Grande (Nelson and Schachner 2002).

The Anglo settlers of the Colorado Plateau were no less challenged by climate variability and weather extremes. The 20th century has been marked by five extreme droughts and many locally devastating floods (Wilkowske et al. 2003). Although periodic droughts on the Colorado Plateau were not as devastating as the famous Dust Bowl years in the Great Plains (in fact some Dust Bowl refugees resettled on the Colorado Plateau; McPherson 1995), history books provide a view of the early 20th century replete with stories of crop failures, livestock loss due to drought, fire, or cold weather, and the abandonment of whole towns (Firmage 1996). For example, when sheep production failed due to drought and range degradation, the town of Cisco, Grand County, Utah, went from a population size of 323 in 1910 to a population of 95 in 1920. In 1910, Grand County had 172 farms with 62 089 cultivated acres; by 1920, only 114 farms were left with 42 656 cultivated acres, and the numbers continued to decline (Firmage 1996).

Irrigation agriculture was often compromised not only by drought, which failed to fill reservoirs and reduced stream flows, but also by flood years, which destroyed irrigation infrastructure. In the 1930s, government policies encouraged homesteading, and many Dust Bowl refugees started dry wheat farming in SE Utah. However, in 1934, the worst recorded drought hit the Colorado Plateau, leading to total wheat crop failure and many homesteads were abandoned. The period from 1905 to 1941 was followed by an even drier period with a new record drought in 1956 (Hereford et al. 2002).

Today, dry crop productivity still follows rainfall (Fig. 4), but there is no close correlation between precipitation and cattle numbers, suggesting that today's ranchers use economic reserves (e.g., bank accounts) and government subsidies to keep cattle numbers constant and within a profitable range. Government assistance during drought years includes support for fencing, water developments, roads, and occasionally, animal feed. Figure 4 also demonstrates the negative correlation between farm subsidies and precipitation, along with an overall increasing trend in subsidies since the mid-1960s. Government payments reached an unprecedented high during the 2002 drought, the worst drought year since the mid-1970s. Although cattle numbers declined after 2002, the 17% decline relative to the previous 5-year average for southern Utah (33% across the entire Colorado Plateau) is small compared with the severity of the drought, which rivals the drought of 1934 (Wilkowske et al. 2003). Thus, government subsidies and economic reserves appear to be used to stabilize herd sizes, and thus income, against precipitation variability.

Although it is clear that both ancient and modern farmers share the need to buffer against primary productivity cycles by building economic (or food) reserves during "good" years, it is often overlooked that ecosystems need to do the same by building ecological reserves (Fig. 5). Extractive uses of primary production compete with what Noy-Meir (1973) coined "plough-back," or the reinvestment of primary productivity into ecosystem reserves, which enables ecosystems to respond strongly to the next rainy season. The ability of an ecosystem to respond can be expressed as the ratio of primary productivity to rainfall input, the so called "rain use efficiency" (RUE, Le Houérou 1984). Reserves that can affect RUE include soil nutrients, soil organic matter, belowground C storage in plant roots and tubers, meristems, and especially BSCs, which greatly reduce erosion potential during drought or flood years. All of these forms of reserves are negatively impacted either directly or indirectly by extractive land uses. Without careful management of extractive effort in relation to drought cycles, ecological reserves decline and RUEs will drop further, ultimately reducing both economic and ecological reserves.

Much depends on the use of economic reserves. The western United States has had a long history of policies that encourage the maintenance of excessively high extractive effort (Hess and Holechek 1995). Government subsidies have boosted economic reserves and allowed the maintenance of extractive efforts (e.g., livestock herd sizes; oil, gas, and mining exploration and development; water developments, such as dams and canals) beyond ecologically sustainable levels. Particularly damaging may be the speed with which farms return to "normal" activity (e.g., herd sizes) after drought through improved transportation, larger economic reserves, and the ability to import livestock or other goods from afar through the North American Free Trade Agreement (Thurow and Taylor 1999). Both, a delayed management response at the onset of drought and the accelerated return to "normal," take away from ecological reserve building.

The evidence that high and chronic levels of herbivory and surface-disturbing activities have negatively impacted the ecological reserve status of the Colorado Plateau, through a reduction in plant and soil storage of C and nutrients, is strongly supported by many studies (e.g., Neff et al. 2005, Evans and Belnap 1999, Belnap and Eldridge 2003). Extractive industries on the Colorado Plateau, especially those that rely on rainfall, have always been a marginal enterprise (including in prehistoric times), prone to overuse, and in the long-run, unsustainable. Even with the benefit of ever-increasing external subsidies, farming and ranching are currently declining in this region (Fig. 6). By contrast, industries largely independent of regional rainfall, such as the service and tourist sectors, have greatly expanded, both in relative and absolute terms.

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# THE FUTURE OF THE COLORADO PLATEAU

The uncertainty surrounding future climate scenarios is of great concern in this region. The expected increase in both average and extreme temperatures will likely result in less soil moisture and thus greater plant stress, even if precipitation patterns remained constant. Warming at night will increase respiration losses of plants (Alward et al. 1999), which may increase rates of mortality. For example, using warming lamps, we have seen 20% mortality in the summer-active $C_4$ galleta grass (Belnap, unpublished). Drought combined with increased temperatures results in far greater and more widespread tree mortality than drought alone (Breshears 2006). The long-term effects of climate change on the ecosystems of the Colorado Plateau will also depend on the process of drought recovery, in which BSCs play a major role.

Unfortunately, BSCs may be the biggest losers in this climate-change century: a recent field study showed that the dominant BSC lichen *Collema tenax* (the main source of N for this ecosystem) declined from 19% in 1996 to 2% after 6 years of record-breaking high summer temperatures (Belnap et al. 2006).

Although the 2002 drought has waned, it is possible that the Colorado Plateau is experiencing a shift into a negative cycle of the PDO, which would continue to reduce winter precipitation for 20–30 years (Hereford et al. 2002). This will result in sustained losses in primary productivity and diminished ability of BSCs to recover from the stresses of summer (Belnap et al. 2006). Particularly damaging would be a shift toward less winter and relatively more summer precipitation. Experiments showed a significant decline in photosynthetic rates and pigment production in *Collema* and cyanobacterial crusts when exposed to an increased frequency of small rainfall events during summer (Belnap et al. 2004). An ongoing precipitation manipulation study shows an increase in mortality of the dominant moss *Syntrichia caninervis* under these conditions as well (Belnap, unpublished). If both summer and winter precipitation are reduced, the loss of lichens and mosses would likely be exacerbated. These potentially highly negative impacts of climate changes on BSCs, intensified by increasing levels of human use, will likely further diminish BSC cover, slow ecosystem recovery from drought, increase soil erosion rates, and generally accelerate the desertification of the Colorado Plateau.

# CONCLUSION

The Colorado Plateau ecosystems developed under conditions of low total and highly variable precipitation, combined with low disturbance frequencies either from grazing or fire in the lower elevation scrublands. Due to the low productivity of this region, recovery from disturbance is slow, probably in the order of several hundred years. This has resulted in ecosystems being highly vulnerable to changes in land use, and in turn, has left land users with a low intrinsic buffering capacity for climate-dependent fluctuations in yield. In medieval times, a decadal megadrought, perhaps combined with overuse of natural resources, led to the abandonment of the Colorado Plateau by the first human settlers. The comparatively wet and stable conditions that ushered in the Anglo occupation of the Colorado Plateau, together with the introduction of external subsidies, including water subsidies and the development of climate-independent industries, have allowed more humans than ever before to live on the Colorado Plateau. Nevertheless, economic statistics indicate that the many livelihoods that still depend on rainfall are in decline. Now, climate models predict a new cycle of megadroughts to commence in the second half of this century. If realized, it is unlikely that even today’s economic buffering capacities will allow the long-term persistence of extractive industries on the Colorado Plateau, as ecosystems are already challenged by historic overuse. The alternative is an increased reliance on other forms of income, which are less dependent on rainfall inputs. This change is already occurring, as many families that traditionally relied solely on livestock or farming for income have begun to explore other sources of income.

# RESPONSES TO THIS ARTICLE

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# Impact of disturbed desert soils on duration of mountain snow cover

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[1] Snow cover duration in a seasonally snow covered mountain range (San Juan Mountains, USA) was found to be shortened by 18 to 35 days during ablation through surface shortwave radiative forcing by deposition of disturbed desert dust. Frequency of dust deposition and radiative forcing doubled when the Colorado Plateau, the dust source region, experienced intense drought (8 events and 39–59 Watts per square meter in 2006) versus a year with near normal precipitation (4 events and 17–34 Watts per square meter in 2005). It is likely that the current duration of snow cover and surface radiation budget represent a dramatic change from those before the widespread soil disturbance of the western US in the late 1800s that resulted in enhanced dust emission. Moreover, the projected increases in drought intensity and frequency and associated increases in dust emission from the desert southwest US may further reduce snow cover duration.

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## 1. Introduction

[2] Dust is commonly found in the surface layer of late season snow and on glacier surfaces in the world's mountain ranges [*Franzén et al.*, 1994; *Schwikowski et al.*, 1995; *Wake and Mayewski*, 1994]. Dust deposition on mountain snow cover has occurred throughout much of recent history as demonstrated by annual dust layers in high elevation ice cores [*Thompson et al.*, 2000], increasing with prolonged or intense drought and land disturbance in source regions. It is well known that dust in snow enhances absorbed solar radiation and melt rates [*Conway et al.*, 1996; *Warren and Wiscombe*, 1980], but the degree to which dust influences radiative forcing and snow cover duration in a natural system is not quantified. Early studies [*de Quervain*, 1947; *Jones*, 1913] suggested from simple observations that dust may shorten snow cover duration by as much as a month. However, unlike soot, which has received much attention as a potential climate forcing in snow cover [*Hansen and Nazarenko*, 2004; *Hansen et al.*, 2005], the influence of dust on surface shortwave radiative forcing and snow cover duration in mountain regions has been relatively ignored [*Hansen and Nazarenko*, 2004]. Until now, the detailed radiation and energy balance measurements needed to partition dust's influence on radiation and snow cover duration have not been available.

[3] Snow has the highest albedo of any naturally occurring surface on Earth. However, when impurities such as dust or soot are present, snow albedo decreases (particularly in visible wavelengths) [*Conway et al.*, 1996; *Warren and Wiscombe*, 1980] (Figure 1). With enhanced absorption by dust, grain growth rates increase and further depress snow albedo. Dust has high potential to sustain shortwave radiative forcing after deposition because particles tend to accumulate near the snow surface as ablation advances [*Conway et al.*, 1996]. Deposition in mountain ranges comes primarily in the spring when frontal systems entrain dust particles from disturbed and loose soils [*Wake and Mayewski*, 1994], coinciding with solar irradiance approaching its annual maximum.

[4] Mountain snow cover is a critical resource as these high elevation mountain regions provide the majority of fresh water supply in arid and semi-arid environments to more than a billion of the Earth's population [*Bales et al.*, 2006]. The duration of snow pack in mountain regions critically controls the timing and magnitude of water supplies, power generation, agriculture timing, and forest fire regimes [*Westerling et al.*, 2006], as well as the duration over which glacial ice is exposed to absorption and enhanced ablation. Some studies already suggest that climate change has induced earlier snowmelt-fed runoff [*Mote*, 2003; *Stewart et al.*, 2005].

[5] The radiative forcing of dust in snow is considered in terms of its direct effect (absorption by dust, Figure 1a), 1st indirect effect (enhanced absorption by larger grain size due to accelerated grain growth from direct effect), and the 2nd indirect effect (enhanced absorption by darker substrate exposed earlier due to direct and 1st indirect effect) [*Hansen and Nazarenko*, 2004]. In this work, we use the term 'radiative forcing' to mean the instantaneous surface enhanced absorption due to dust through these effects. Here we perform the first coupled determination of the radiative forcing of dust in mountain snow and its impact on snow cover duration through detailed radiation measurements that isolate the effects of dust from other controls.

[6] The simulations show that radiative forcing by desert dust deposits shorten snow cover duration by order 1 month in the San Juan Mountains. That the dust originates in disturbed desert sources rather than locally suggests that this mechanism of increasing radiative forcing and shortened snow cover duration is widely active where the world's mountains receive dust from disturbed lands. Moreover,

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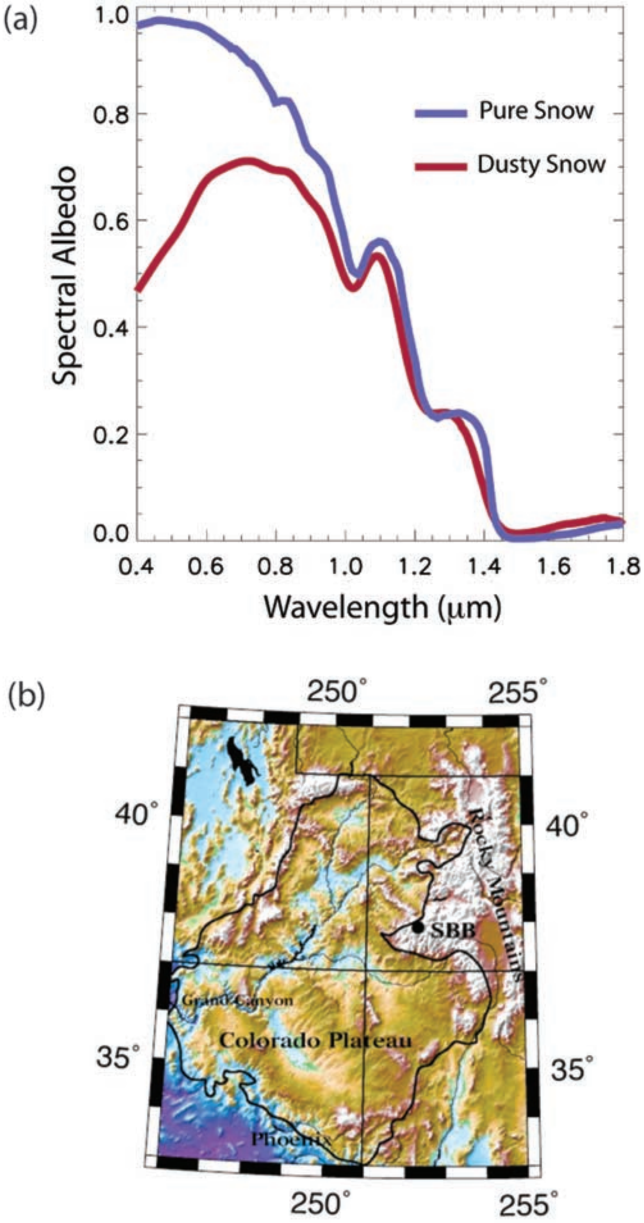

**Figure 1.** Spectral albedo of dust-laden snow and Colorado Plateau/San Juan Mountains. (a) Spectral albedo of pure snow (modeled with same grain size as that measured in the field) and snow with concentrated dust in surface layer (0.37 mg dust per g snow water equivalent) measured with an Analytical Spectral Devices FieldSpec FR spectroradiometer. (b) Colorado Plateau in southwestern US and Senator Beck study site (SBB) indicated in San Juan Mountains of southwest Colorado.

under global warming and associated desertification, the process could threaten snowmelt-fed water resources to arid and semi-arid regions.

## 2. Site and Methods

[7] We estimate radiative effects of dust and changes in snow cover duration in the San Juan Mountains, Colorado (Figure 1b) in two dramatically different dust deposition years, 2005 (4 dust events) and 2006 (8 dust events). The San Juan Mountains cover 32,000 km$^2$ (6900 km$^2$ above tree line at ~3400 m) and have a strong continental climatic regime. Headwaters for major western US rivers including the Rio Grande and the Colorado lie in the range. According to anecdotal evidence, historical research, and recent observations presented here, the San Juan Mountains receive multiple dust deposition events annually in February through May, arriving before and during the snowmelt period (Auxiliary Material Table S1).

### 2.1. Radiative Forcing

[8] Incident and reflected broadband shortwave and near-infrared/shortwave infrared (NIR/SWIR) radiation, along with standard meteorological variables, were measured at two towers, one in the alpine zone at 3719 m and one in the subalpine zone at 3368 m of the Senator Beck Basin. From these measurements, we determine the range of potential radiative forcings due only to dust. Minimum surface radiative forcing $F_{d\min}$ (W m$^{-2}$) is calculated as

$$F_{d\min} = E_{VIS}\Delta_{VIS} \tag{1}$$

where $E_{VIS}$ is the visible irradiance (W m$^{-2}$) determined from the difference between the broadband and NIR/SWIR irradiances, $\Delta_{VIS} = 0.92 - \alpha_{VIS}$, $\alpha_{VIS}$ is calculated visible albedo and 0.92 is the mean visible albedo for dust-event-free snow.

[9] Maximum surface radiative forcing $F_{d\max+i1}$ is calculated as

$$F_{d\max+i1} = 0.5\left(E_{VIS}\Delta_{VIS} + E_{NIR}\alpha_{NIR}\left(\frac{1}{\xi} - 1\right)\right) \tag{2}$$

where

$$\begin{array}{ll} \xi = 1 - 1.689\Delta_{VIS}; & \Delta_{VIS} \in [0, 0.17] \\ \xi = 0.67; & \Delta_{VIS} > 0.17 \end{array},$$

$E_{NIR}$ is the NIR/SWIR net shortwave flux, and $\alpha_{NIR}$ is NIR/SWIR albedo. The latter relationship gives the proportion of the change in NIR/SWIR albedo due to presence of dust versus grain coarsening in the absence of dust (see Auxiliary Material).[1]

### 2.2. Snowmelt Modeling

[10] Snow cover duration was simulated for observed net surface shortwave fluxes ($K^*$) and for the observed net surface shortwave fluxes minus $F_{d\min}$ ($K^*_{d\min} = K^* - F_{d\min}$) and $F_{d\max+i1}$ ($K^*_{d\max+i1} = K^* - F_{d\max+i1}$), respectively, using the snow energy balance model SNOBAL [*Marks et al.*, 1998]. SNOBAL uses a two-layer snowmelt approach with an active 25 cm surface layer and the remainder of the snow pack as a second layer. Initial conditions for snow pack properties came from detailed weekly measurements in the field at each site. SNOBAL is driven with hourly averages of observed and estimated dust-free net shortwave, and

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[1] Auxiliary material data sets are available at ftp://ftp.agu.org/apend/gl/2007gl030284. Other auxiliary material files are in the HTML.

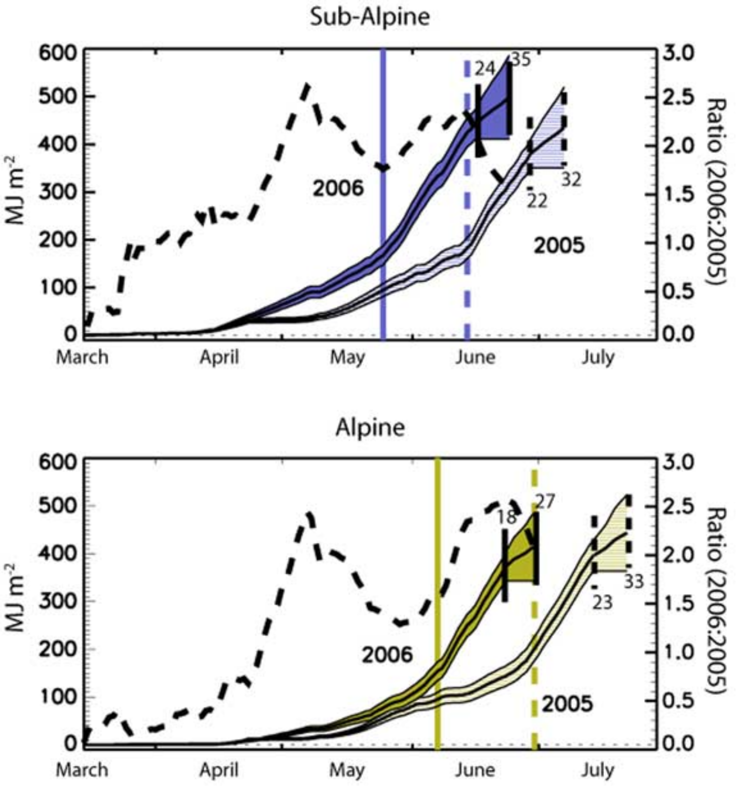

**Figure 2.** Cumulative surface radiative forcing by dust (MJ m$^{-2}$) for subalpine and alpine. The full-length vertical lines indicate observed date of disappearance of snow whereas the shorter vertical lines indicate the modeled date of disappearance of snow in the minimum and maximum forcing cases, with differences from observed date in number of days indicated. Dashed lines show the ratios of daily mean forcing in 2006 with respect to 2005.

observed incident longwave radiation, and air temperature, relative humidity, and wind speed measured at known heights above the snow surface at the respective meteorological towers.

[11] The difference between observed and simulated snow melt out-date for the measured scenario provides a measure of the accuracy of SNOBAL. In all simulations, complete snow ablation occurred within 1 day of observed ablation except in the 2005 subalpine case (2 days), due to discrimination between rain and snow near the end of snow cover.

[12] The second indirect effect for the minimum and maximum cases is the difference between simulated net shortwave radiation for snow free conditions (post ablation) and the net shortwave radiation for snow cover for minimum and maximum forcing scenarios. We add the respective 2nd indirect effect to the minimum and maximum radiative forcings after observed melt-out to obtain total forcings.

### 2.3. Isotopic Analysis

[13] Each dust event was collected from the mountain snow pack by excavating dust layers of snow, melting the snow into HDPE collection vessels and then refreezing the sample until analysis. The samples were freeze-dried to a powder. Resulting solids were dissolved in concentrated HF and HCl$0_4$. Rb and Sr were separated from the solution using SrSpec$^{®}$ resins, while Sm and Nd were obtained using conventional reverse phase chromatographic techniques [*Farmer et al.*, 1991]. Isotope dilution concentration and Sr and Nd isotopic determinations were obtained using a Finnigan-MAT thermal ionization mass spectrometer. $\varepsilon_{\mathrm{Nd}}$ represents a comparison of the $^{143/144}$Nd of the measured sample to a ratio for CHUR (chrondritic uniform reservoir) where

$$\varepsilon_{\mathrm{Nd}}(0) = \left[\left(\left({}^{143/144}\mathrm{Nd}\right)\mathrm{Sample}\left({}^{143/144}\mathrm{Nd}\right)\mathrm{CHUR}\right)/\left({}^{143/144}\mathrm{Nd}\right)\mathrm{CHUR}\right] \quad (3)$$

all at the present time (t = 0).

## 3. Results

[14] Between 2003 and 2005, three to four significant dust deposition events occurred in winter and spring of each year, whereas in 2006, eight significant deposition events occurred (Auxiliary Material Table S1). Broadband albedo ($\alpha_{\mathrm{BB}}$) of new, dust-free snow was 0.85 ± 0.01 but declined between snow events through grain size increase when dust was not exposed (Auxiliary Material Figure S1). Once dust emerged near or at the surface, $\alpha_{\mathrm{BB}}$ decreased more rapidly due to the additional decrease in $\alpha_{\mathrm{VIS}}$ from its dust event-free value of 0.92 ± 0.01.Dust events led to large increases in the daily averaged absorption of incident radiation at the surface in 2005 and 2006 (Figure 2 and Auxiliary Material Figures S2–S3).

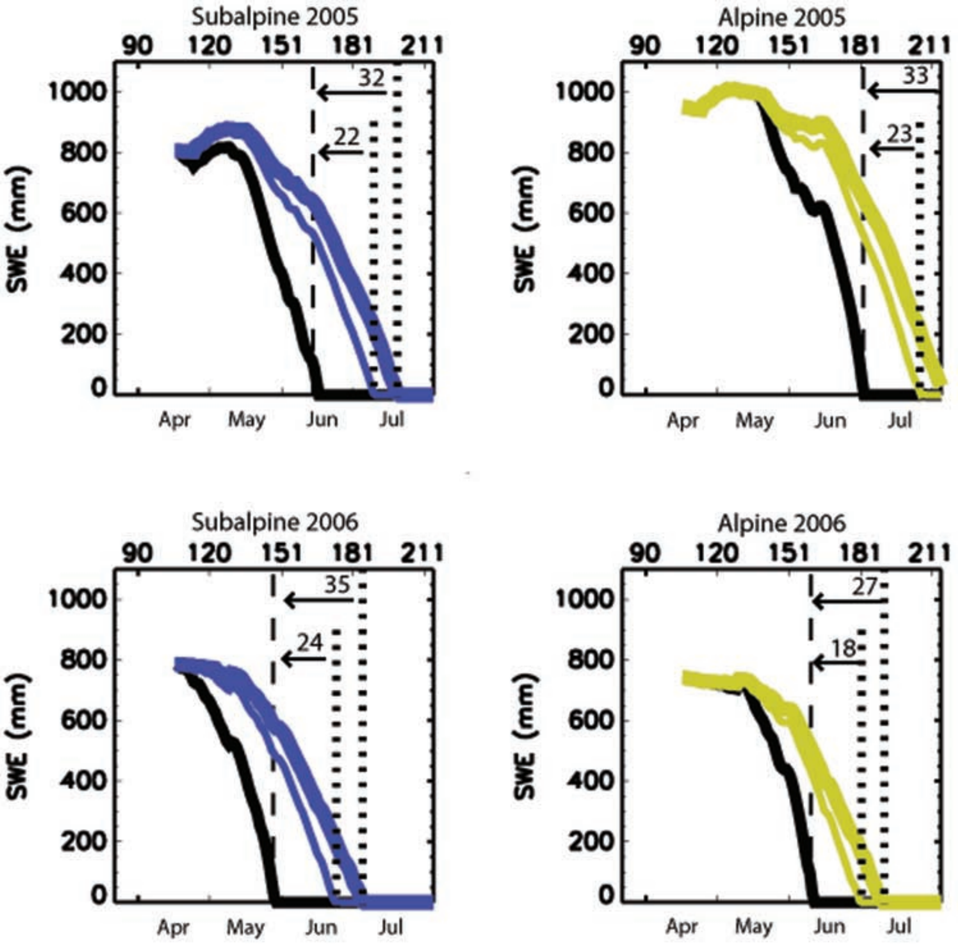

**Figure 3.** Time series of snow water equivalent scenarios with number of days difference from observed indicated: snowmelt model results for subalpine 2005 in measured case (black), dust free $K^*_{d\min}$ (thin), and dust free $K^*_{d\max+i1}$ (thick), where differences in snow cover duration are indicated for respective cases above the horizontal arrows; subalpine snowmelt in 2006; alpine snowmelt in 2005; and alpine snowmelt in 2006.

[15] During the winter/spring 2005, $\alpha_{BB}$ dropped in two periods from 0.85 (subalpine and alpine) to 0.45 (subalpine) and 0.51 (alpine), respectively, and $\alpha_{VIS}$ dropped in the same periods from 0.92 to 0.50 and 0.61, respectively, due to the accumulation of dust at the snow surface (Auxiliary Material Figure S1). In 2006, $\alpha_{BB}$ dropped quasi-monotonically from 0.85 to 0.40 and 0.46, respectively, and $\alpha_{VIS}$ dropped likewise from 0.92 to 0.45 and 0.58, respectively (Auxiliary Material Figure S1).

[16] In 2005, the radiative forcing had two periods of mean forcings of 30–50 (subalpine) and 20–40 (alpine) W m$^{-2}$ as dust layers were exposed at the surface, covered by snowfall and exposed again through melt (Auxiliary Material Figure S2). In 2006, the radiative forcing increased steadily from early April to maxima of 80 (subalpine) and 60 (alpine) W m$^{-2}$ later in ablation despite frequent but small snowfalls (Auxiliary Material Figure S3). For the period March 21 to June 21 (spring), mean daily radiative forcing in 2005 ranged from 31 to 37 W m$^{-2}$ (subalpine) and 14 to 19 W m$^{-2}$ (alpine), respectively, and in 2006 ranged from 56 to 64 W m$^{-2}$ and 36 to 42 W m$^{-2}$, respectively. Radiative forcings in the alpine are generally lower than those in the subalpine because dust concentrations were consistently lower at the wind-exposed alpine tower. However, other slopes and aspects in the alpine collect scoured dust and have higher concentrations.

[17] The combination of more frequent and heavy dust events, less post-event snowfall and thus greater exposure, and clearer skies in 2006 compared with in 2005 resulted in distinctly larger radiative forcing and shorter snow cover duration in 2006 than in 2005. Figure 2 presents the cumulative radiative forcings and the time series of the ratios of daily mean forcings for year 2006 to 2005 for both sites. The ratio generally increased at both sites lying between 1.5 and 2.5 for most of the ablation season, indicating that the net shortwave energy made available by dust for melting was approximately doubled in 2006. Ratios less than 1 result from small but non-zero dust forcing before late April 2005.

[18] SNOBAL snowmelt simulations indicate that in 2005 the subalpine melted out 22 to 32 days earlier relative to the dust free cases and the alpine snow cover melted out 23 to 33 days earlier (Figure 3). In 2006, the subalpine melted out completely 24 to 35 days earlier and the alpine melted out 18 to 27 days earlier. Far smaller peak snow water equivalents in 2006 relative to 2005 (67% in alpine, 85% in subalpine) resulted in the small differences between 2005 and 2006 with respect to snow cover duration despite

the doubling in shortwave radiative forcing. This gave less snow cover duration over which the differences in melt rates could be manifested. Therefore, the associated melt rates were greater in 2006 by up to 40%. In the period after snow cover disappears, the 2nd indirect radiative forcing from enhanced absorption was relatively constant between the two years and sites at 147 ± 8 W $m^{-2}$ in 2006.

[19] Dust is generated by several processes and can originate from a spatial range of sources. We determined provenance of deposited dust through a combination of isotopic analysis, back-trajectory analyses, and remote sensing. For dust from the snow pack, strontium isotope ratios, $^{87/86}$Sr, and a comparison of the neodymium isotope ratios with a ratio for chrondritic uniform reservoir, $\boldsymbol{\varepsilon_{Nd}(0)}$, were compared with the isotopic compositions of exposed surface rocks within the Senator Beck Basin. Sr and Nd isotopic values for the deposited dust were significantly different from the surrounding surface rocks (Auxiliary Material Table S2) and therefore a local, volcanic rock origin for the dust samples taken from the snow pack was highly unlikely.

[20] Precambrian basement rocks in western North America, and sedimentary rocks derived thereof, have geographic variations in their Nd isotopic compositions that constrain the sources of dust found in the San Juan Mountains snow cover. The dust samples have Nd isotopic compositions that overlap those of Paleoproterozoic intermediate to silicic compositions rocks that comprise much of the Mazatzal Province in Arizona and New Mexico (Province 3 of *Bennett and Depaolo* [1987] which lies to the west and south of Senator Beck Basin), indicating that dust deposited in the San Juan Mountains originated in the SW US.

[21] Backtrajectory analysis of atmospheric transport and remotely sensed imagery further narrowed the likely source of the dust. The Stochastic Time-Integrated Lagrangian Transport model (STILT) [*Lin et al.*, 2003] with archived data from the Eta Data Assimilation System (EDAS) 40 km weather analysis produces ensembles of 48-hour back-trajectories from the time of dust deposition. Auxiliary Material Figure S4 shows the mean back-trajectories for the 18 dust events that we have documented since 2003. With the exception of a single event with northwesterly flow (February 22, 2003) across southeast Utah, the remaining events had southwesterly flow across northeastern Arizona and northwestern New Mexico. Clear-sky data from the Geostationary Operational Environmental Satellite 10 (GOES-10) consistently show rising dust plumes from northeastern Arizona and northwestern New Mexico close to the time of dust deposition (Auxiliary Material Figure S5, April 2–3, 2003, as example).

[22] These data are consistent with the combination of trajectories determined by the STILT analyses and isotopic data in showing that dust deposited in the San Juan Mountains comes primarily from the Colorado Plateau. Precipitation in the fall 2005/winter 2006 on the Colorado Plateau was the lowest on record and contributed to the doubling of the number of deposition events over those in 2003–2005. We conclude then that snow cover duration across the San Juan Mountains is reduced by 18–35 days due to the deposition of dust from the disturbed deserts of the Colorado Plateau and not from sources local to the mountain basins.

## 4. Discussion

[23] Expansion and intensification of grazing, recreational use and agriculture over the past ~140 years has increased the dust emission from the Colorado Plateau and other desert regions of the western US [*Belnap and Gillette*, 1997; *Neff et al.*, 2005; *Reynolds et al.*, 2001]. Therefore it is likely that the above changes in snow cover duration and surface radiative forcing increased significantly with human activity in the late-1800s, as desert surface crusts were disturbed and dust was more freely emitted to the Rocky Mountains and other mountain ranges of the western US that are downwind of disturbed soils. Analyses similar to those performed for the San Juan Mountains and an analysis of time series of dust deposition from mountain lake sediments across the western US will provide a clearer understanding of the spatial and temporal extent of this shortening of snow cover duration.

[24] The phenomenon of increasing dust emission exists beyond the western US and is global in nature with the potential to continue to perturb resource-critical mountain snowmelt systems. Dust emission frequency in China from the Taklimakan and Gobi deserts (proximal to the Tien Shan and Altai ranges) has increased from one event in ~35 years for the period AD85-1949 to annual since 1990 [*Liu and Diamond*, 2005]. A four-fold increase in dust deposition over the previous two centuries was found in the Dasuopu glacier ice core at elevation 7200 m in Tibet [*Thompson et al.*, 2000], with the continuum increase attributed to increased land usage whereas the interannual variability attributed to interannual changes in precipitation. The drying of the Aral Sea has affected enormous increases in dust emission that frequently deposits in the Tien Shan, Pamir, Himalaya, and Altai Mountains [*Waltham and Sholji*, 2001]. Dust deposition to the Antarctic Peninsula has doubled in the 20th Century due to the coupled effects of changing climate and land degradation [*McConnell et al.*, 2007].

[25] Under global warming, increased drought is projected for the southwest US, Middle East, and the expanding Sahel [*Cook et al.*, 2004; *Hansen et al.*, 2005; *Intergovernmental Panel on Climate Change*, 2007]. All of these deserts are known dust sources for winter and spring deposition to mountain snow cover in the Rocky Mountains (this work), Central Asia (Pamir Mountains, Hindu Kush, Karakoram) [*Wake and Mayewski*, 1994], and the Alps [*Franzén et al.*, 1994; *Schwikowski et al.*, 1995], respectively. Future drying in desert regions [*Cook et al.*, 2004; *Hansen et al.*, 2005] and projected expansion and intensification of use of arid and semi-arid lands [*Asner et al.*, 2004] could cause regional dust emission to increase in frequency and magnitude. Therefore, earlier snowmelt and its effects on mountain water resources and glacial extent is a likely scenario in many of the world's mountain ranges under enhanced dust deposition.

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# DISTURBANCE OF BIOLOGICAL SOIL CRUSTS: IMPACTS ON POTENTIAL WIND ERODIBILITY OF SANDY DESERT SOILS IN SOUTHEASTERN UTAH

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## ABSTRACT

Friction threshold velocities (FTVs) were determined for biological soil crusts in different stages of recovery. Particles on the surface of crusts that had been relatively undisturbed for at least 20 years were found to have significantly higher FTVs than those that had been disturbed 5, 10 or 1 years previously (376, 87, and 46 cm sec$^{-1}$, respectively). FTV's for crust breakage was also much higher for undisturbed crusts when compared to the previously disturbed crusts (573, 148, and 88 cm sec$^{-1}$, respectively). All crusted surfaces were more stable than bare sand, which had an FTV of 16 cm sec$^{-1}$. Disturbance treatments were then applied to the three crustal classes. Disturbance significantly reduced the FTVs of all classes by 73–92 per cent. Comparing crustal FTVs with mean and high monthly wind speeds found in this region, it was observed that only crusts that had been undisturbed for approximately 20 years or more were able to protect soil surfaces from wind gusts expected on the average of once a month. Other crustal classes, as well as all disturbance treatments, had FTVs lower or equal to that of commonly occurring winds in this region. Because most of the crustal biomass occurs in the top 0·3 mm of soils, even slight soil loss can negatively influence stability and nutrient inputs to this ecosystem. © 1997 John Wiley & Sons, Ltd.

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KEYWORDS friction threshold velocity; soil loss; cryptogamic; microbiotic; microphytic; cryptobiotic

## INTRODUCTION

Biological soil crusts, also called 'cryptobiotic', 'microbiotic', 'microphytic' or 'cyanobacterial–lichen' soil crusts, are a dominant feature of most semiarid and arid landscapes throughout the world. These crusts differ in species composition and occur on a variety of soils. As a result, crustal function in different geographic regions might vary in regard to ecological processes such as rainfall infiltration and seedling establishment (Harper and Marble, 1989; Johansen, 1993; West, 1990). However, most studies agree that biological soil crusts reduce wind erodibility of soil surfaces (Leys, 1990; MacKenzie and Pearson, 1979; Williams, *et al.*, 1995), although one study found no significant differences (Andrew and Lange, 1986). Scanning electron microscope studies done by Belnap and Gardner (1993) show that the extracellular sheath material of cyanobacteria bind soil particles together, providing soil surface protection.

Biological soil crusts are highly susceptible to disturbance, especially in soils with low aggregate stability such as sands (Belnap and Gardner, 1993; Gillette, *et al.*, 1980; Webb and Wilshire, 1983). Cyanobacterial filaments, lichens and mosses are brittle when dry, and crush easily when subjected to compressional or shear forces by activities such as trampling or vehicular traffic. Because crustal organisms are only metabolically active when wet, re-establishment time is slow in arid systems. While cyanobacteria are mobile, and can often move up through disturbed sediments to reach light levels needed for photosynthesis, lichens and mosses are incapable of such movement and often die as a result. On newly disturbed surfaces, mosses and lichens often have extremely slow colonization and growth rates. Assuming adjoining soils are stable and rainfall is

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average, recovery rates for lichen cover in southern Utah has been most recently estimated at a minimum of 45 years, while recovery of moss cover was estimated at 250 years (Belnap, 1993). Due to this slow recolonization of soil surfaces by the different crustal components, crusts can be found in many stages of development.

Wind is a major erosive force in deserts where there is little organic matter or vegetation cover to protect soil surfaces. Soil deposition by wind often exceeds that of fluvial deposition in these drier regions (Goudie, 1978; Williams, *et al*., 1995). Sediment production from soil surfaces depends on the force of wind needed to detach particles from soil surfaces (threshold friction velocity). Since wind erosion is of major concern both in the western USA and worldwide (Dregne, 1983), it is important to understand how soil surface disturbance affects threshold velocities.

While previous studies have addressed the role soil crusts play in stabilizing desert soil surfaces, none has examined how threshold velocities might vary between stages of crustal development or how disturbance might differentially influence various crustal types. The purpose of this study was to determine typical threshold velocities for different stages of biological soil crust development and to determine the effects of different soil surface disturbances on various stages of crustal development.

## METHODS

The study site was located approximately 16 km south of Moab, Utah, USA, in Rizzo sandy loam soils. The dominant vegetation type is pinyon and juniper at an elevation of 1400 m. Annual precipitation is 250 mm, with 30 per cent of the rainfall occurring as late summer monsoons. Treatments were applied and measurements taken in July 1995 when soils were dry. All areas tested were located within a 300 m circle, with the same substrate type, soil depth and slope.

Soils were collected and analyzed for sand, silt and clay content. Biological soil crust development was placed in one of four time categories, based on previous experiments regarding recovery rates after disturbance from four-wheel vehicles or foot traffic (Belnap, 1993, unpublished data). These included:

(a) Class 0: bare sand, with no visible biological crustal development, indicating very recent disturbance from vehicle or foot traffic.
(b) Class 1: flat crusts, with no visible frost heaving or lichen cover and low cyanobacterial biomass, indicating disturbance from vehicles or foot traffic within one year of observation.
(c) Class 2: moderately bumpy biological crusts with no lichen or moss development and moderate cyanobacterial biomass levels, indicating vehicular or foot traffic disturbances 5–10 years prior to observation.
(d) Class 3: biological crusts were very bumpy, with full lichen and moss development and high cyanobacterial biomass, indicating no vehicular or foot traffic disturbance for at least 20 years.

Friction threshold velocities for movement of loose sand particles on the undisturbed surface (CON in Figure 2), and surface integrity of the crusts (SI in Figure 3) were determined for each crust type at two replicated sites. The FTV for particle movement was defined as the friction velocity at which surface particles were both detached from the soil surface and carried away by the generated wind. The FTV for surface integrity was the friction velocity at which large, intact chunks of the surface were detached and blown away. Because wind stress equals the square of friction velocity times the density of air, relative resistances of the different crustal classes to wind erosion are defined and reported as the square of the ratio of threshold friction velocities between the classes being compared.

Once FTVs were determined for the different undisturbed crustal classes, disturbance treatments were applied to each crust class at each site. These treatments included:

(1) Treatment F1: one pass by walking on crusts with lug-soled boots.
(2) Treatment V1: one pass of a four-wheel drive vehicle with knobbed tires.
(3) Treatment V2: two passes of a four-wheel drive vehicle with knobbed tires.

Figure 1. Portable wind tunnel, assembled in the field.

Comparisons across the three crustal classes were done using a two-way ANOVA and multiple range test. $T$-tests were used to distinguish between disturbance treatments and controls.

### *Wind Tunnel*

A portable, open-bottomed wind tunnel, 150 mm × 150 mm cross-section by 2·4 m length was used so that many wind speeds could be formed over the desert surface (Figure 1: Gillette, 1978). The tunnel used a 5 : 1 contraction section with a honeycomb flow straightener and a roughly conical diffuser attached to the working section. Wind speed data were collected at several heights above the surface midway across the end of the working section. The Pilot tube anemometer was calibrated and corrected for temperature and pressure changes.

To obtain FTVs for the undisturbed control crusts, wind speed in the tunnel was gradually increased until consistent forward sand particle movement was observable across the soil surface. Measurements of airflow velocities were then recorded at the soil surface and 3·2, 6·4, 12·7, 25·4, 38·1, 50·8, 63·5, 76·2, 88·9, 101·6 mm above the soil surface, yielding wind profiles for the controls. To obtain wind profiles for surface integrity values, wind speeds were then increased until chunks of the surface crusts detached. For areas receiving foot or vehicle treatments, FTVs were determined for the undisturbed surface, after which the tunnel was removed and treatments applied. After treatment, the tunnel was replaced and then the same area remeasured. Data for the mean horizontal wind velocity $U$ (cm sec$^{-1}$) versus height $z$ (mm; wind profile data) were fitted to the function for aerodynamically rough flow (Priestley, 1959), using a linear least squares routine:

$$U_{*t} = kz(\mathrm{d}U_t/\mathrm{d}z)$$

where $U_{*t}$ is friction velocity, $kz$ is roughness height characteristic of the surface, $U_t$ is wind speed at the particle movement threshold, and $k$ is Von Karman's constant. Threshold velocities and aerodynamic roughness heights are reported in terms of friction velocity and roughness height. A total of 196 wind profiles were obtained for this study.

Mean, median and maximum wind speeds recorded in this region were converted to rough estimates of friction velocity. This was done using a drag coefficient for 2 m winds of 0·01 which corresponds to a surface roughly similar to the test site. Using similar surfaces as those described by Priestley (1959), the aerodynamic roughness height of this surface was estimated to be 3·7 cm. The conversion factor for wind speeds in meters per second to friction velocities in centimeters per second is 10.

## RESULTS

Characteristics of soils found at the sites are listed in Table I. Friction threshold velocities for soil particle movement in crustal class controls and treatments are presented in Figure 2. Undisturbed class 3 crusts had FTVs of 376 cm $\sec^{-1}$ compared to 16 cm $\sec^{-1}$ for bare sand; consequently, class 3 crusts had 552 times the wind resistance of bare sand. Undisturbed class 1 crusts had average FTVs of 46 cm $\sec^{-1}$, indicating

Table I. Means and standard error for soil texture of different crust types. No statistically significant differences were found between sand, silt or clay content of the three soils ($p > 0.05$)

| Crust class: Age | Sand | Silt | Clay |
|---|---|---|---|
| C1: 1 year | 76 + / − 4·9 | 13 + / − 4·3 | 11 + / − 0·8 |
| C2: 5 year | 66 + / − 2·0 | 20 + / − 0·5 | 14 + / − 1·5 |
| C3: >10 year | 65 + / − 4·4 | 21 + / − 4·2 | 14 + / − 0·9 |

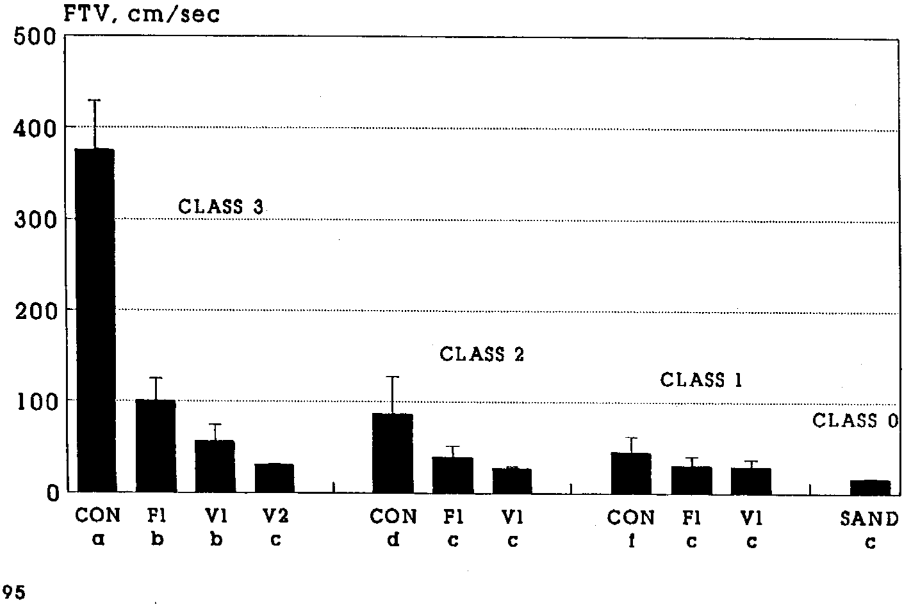


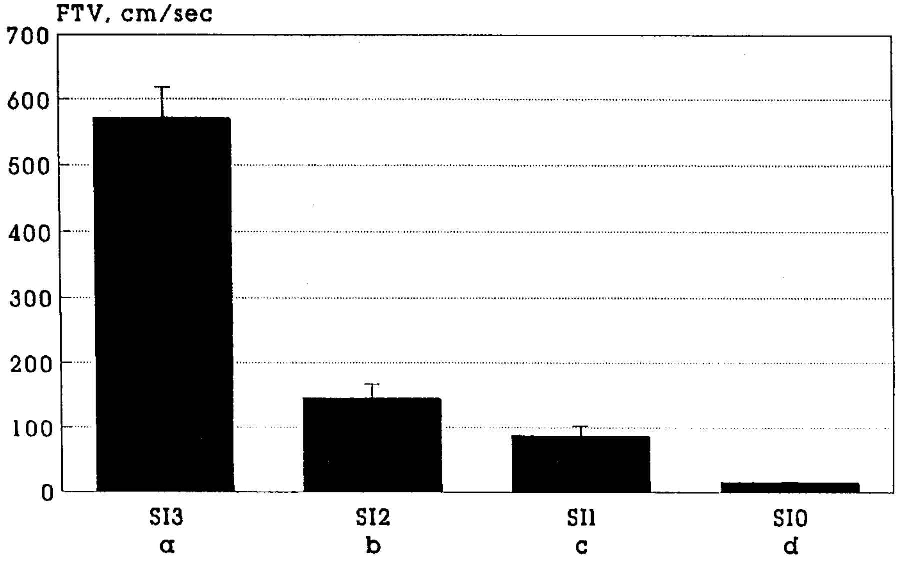
Figure 2. FTVs for individual soil particle detachment in different crustal classes before and after disturbance treatments. CON = control crust; F1 = one pass with lugged boot; V1 = one pass with vehicle; V2 = two passes with vehicle. Within each crustal class, all treatment values were significantly lower than control surfaces of the same class. Within each class, no treatments differed significantly from each other, with the exception of class 3 (V2 was different from F1 and V1). All crustal class controls differed from each other. Categories with different lower case letters differ significantly from each other ($p < 0.05$). Error bars indicate standard error.

that class 3 crusts have 64 times the wind resistance of class 1 crusts. Class 3 crusts showed 19 times the wind resistance of class 2 crusts, whose average FTV was 87 cm $sec^{-1}$. Figure 3 presents the FTVs required to break up the surface integrity of the soil surface. Maximal wind tunnel velocity possible (573 cm $sec^{-1}$) did not break the surface integrity of class 3 crusts; consequently, our maximum FTV underestimates the true FTV for this crust class. Breakage of the surface crust was seen at FTVs of 88 cm $sec^{-1}$ for class 1 crusts and 148 cm $sec^{-1}$ for class 2 crusts. Therefore, class 3 crusts showed more than 15–42 times greater wind resistance than lesser-developed crusts, while class 2 crusts had 2·6 times greater wind resistance than class 1 crusts. All crustal classes had significantly higher resistance than that of bare sand, which had an FTV of 16 cm $sec^{-1}$. Class 3 crusts showed more than 1283 times the wind resistance than that of bare sand, while class 2 and class 1 crusts showed 81 and 30 times, respectively, the wind resistance than that of bare sand.

All applied disturbances significantly decreased FTV in all crust classes when compared to the controls for each crustal class. However, disturbance affected the well-developed class 3 crusts the least. Treatment F1 (one pass of footprints) reduced class 3 crusts' FTV by 73 per cent; however, this was still well above the FTVs of undisturbed class 2 and class 1 crusts. On class 3 crusts, treatment V1 (one vehicle pass) reduced FTV by 85 per cent, a value just below that for undisturbed class 2 crusts. It took treatment V2 (two vehicle passes) on class 3 crust to reduce the FTV to values comparable to that for all disturbances to other crust classes (a 92 per cent reduction), and to approach values for bare sand. In comparison, class 1 and 2 crusts showed very little resistance to disturbance, with treatments 1 and 2 resulting in FTVs equivalent to bare sand. The sum of the squares for all ANOVA consistently showed a greater difference between crustal classes than between any treatments.

## DISCUSSION

This study demonstrated that the degree of organic soil crust development can be very important in determining FTVs of soil surface. Mean, median and maximum wind speeds for 1989–91 (30 minute averages) at a site in the same region are presented in Table II (as reported by Williams, *et al.*, 1995). Table III shows



7/95; all diff at p<0.01

Figure 3. FTVs required to break up surface integrity (SI) of soils (detachment and loss of large chunks of untreated crustal classes). SI3 = class 3 crust; SI2 = class 2 crust; SI1 = class 1 crust; SI0 = class 0 crust. All values were significantly different ($p < 0{\cdot}01$). Error bars indicate standard error.

Table II. Mean, median and maximum (m $\sec^{-1}$; averaged over 30 minutes) wind speeds and friction velocity estimates recorded for a research site in the region, 1989–91 (as reported in Williams, *et al*., 1995)

| Year | Mean | Median | Maximum | FTV estimate |
|---|---|---|---|---|
| 1989 | 1·11 | 0·58 | 14·81 (June)* | 148 |
| 1990 | 0·97 | 0·50 | 13·42 (July)* | 134 |
| 1991 | 1·03 | 0·53 | 13·67 (March)* | 137 |

*Month when maximum wind speed was recorded.

Table III. Maximum wind speeds (m $\sec^{-1}$) and friction velocity estimates reported for the Moab area, 1982–88 (as reported in Williams, *et al*., 1995)

| Month | Wind speed | Friction velocity | Month | Wind speed | Friction velocity |
|---|---|---|---|---|---|
| Jan. | 33 | 330 | July | 32 | 320 |
| Feb. | 29 | 290 | Aug. | 27 | 270 |
| Mar. | 35 | 350 | Sept. | 34 | 340 |
| Apr. | 32 | 320 | Oct. | 28 | 280 |
| May | 35 | 350 | Nov. | 32 | 320 |
| June | 32 | 320 | Dec. | 28 | 280 |

maximum monthly winds in Moab, Utah, 10 miles north of the research site. Conversion of these wind speeds to rough estimates of friction velocity showed that mean, medium and maximum winds from both sites in this region never exceeded the FTV for undisturbed class 3 crusts. In contrast, FTVs for class 0, 1, and 2 crusts were all below monthly maximum wind speed at both sites. Consequently, in the absence of other protecting elements like vegetation, only relatively undisturbed soil crusts would be capable of protecting soil surfaces from winds that commonly occur in this region. Even after five to ten years recovery, crustal= integrity of class 1 and 2 crusts is still not adequate to protect soil surfaces from monthly maximum wind speeds.

This study also showed that previously disturbed soils had less resistance to additional disturbance than the relatively undisturbed soils, while all soil surface disturbances had negative impacts on the FTVs of all crust classes. Though foot traffic in the class 3 crusts resulted in an FTV still above mean wind speeds of the region, it reduced the ability of this crust to resist monthly maximum winds. Any vehicular disturbance significantly reduced the ability of these surfaces to resist wind erosion by reducing FTVs below average monthly wind speeds. All disturbed crusts were at risk from the monthly high wind speeds reported from this area. In addition, adjoining areas are placed at risk of being 'sand-blasted' by material from disturbed areas, and thus a small disturbance might trigger much larger impacts.

Friction threshold velocities reported in this study are similar to values reported in other work. Selah and Fryrear (1995) reported an FTV of 31 cm $\sec^{-1}$ for dry bare soils, compared to 16 cm $\sec^{-1}$ in this study. Gillette (1988) reported an FTV of 290 cm $\sec^{-1}$ in rain-crusted soils in the Mojave, and 20–60 cm $\sec^{-1}$ for loose sandy soils (Gillette, *et al*., 1980). Gillette, *et al*. (1980) also showed that FTV increases with increasing clay and silt in soils. This is supported by the work of Williams, *et al*. (1995). They reported FTVs of 200 cm $\sec^{-1}$ for alluvial soils with fairly high silt–clay contents that had been fenced off from grazing disturbance for three years. As would be expected for a more recently disturbed crust, these soils had a much lower lichen cover than those in the current study (2 vs. 20 per cent cover), and would be considered class 2 crusts. As suggested by the work of Gillette, *et al*. (1980), the silty soils in the study by Williams, *et al*. (1995) showed higher FTVs than the class 2 sandy soil crusts examined in this study, and lower FTVs than the sandy class 3 crusts.

Decreasing FTVs are directly associated with increased sediment movement (Leys, 1990; Williams, *et al*., 1995). Increased sediment movement can result in many direct and indirect problems. Slow recovery rates recorded for crusts in this region (Belnap, 1993) would result in disturbed soils being exposed to erosion for from 50 to 250 years after disturbance. Soil formation is estimated to take 5000 to 10 000 years (Webb and Wilshire, 1983); therefore, soil loss can have long-term consequences. Work done by Garcia-Pichel and Belnap (1996) has demonstrated that over 75 per cent of the photosynthetic biomass, and almost all photosynthetic productivity is from organisms in the top 0·3 mm of these soils. Therefore, very small soil losses, or burial of photosynthetic organisms by wind or water-borne sediments, can dramatically reduce site fertility. In addition, many plants have relatively inflexible rooting depths, and often cannot adapt to rapidly changing soil depths.

## CONCLUSIONS

This study demonstrated that disturbance to biological soil crusts on sandy soils in southeastern Utah has left soil surfaces susceptible to wind erosion from commonly occurring wind speeds for at least twenty years. In addition, previously disturbed soil crusts were shown to be less resistant to new disturbances than previously undisturbed soil crusts.

Soil erosion in arid lands is a major threat worldwide. Beasley, *et al.* (1984) estimated that in rangeland of the USA alone, 3·6 million hectare has some degree of accelerated wind erosion. Relatively undisturbed biological soil crusts can contribute a great deal of stability to otherwise highly erodible soils. Unlike vascular plant cover, crustal cover is not reduced in drought, and unlike rain crusts, these organic crusts are present year-round. Consequently, they offer stability over time and in adverse conditions that is often lacking in other soil surface protectors. Unfortunately, the more-disturbed class 0, 1 and 2 crusts now cover vast areas in the western USA as a result of the ever-increasing recreational and commercial uses of these semiarid and arid areas. Based on the results of this study, the tremendous land area currently being impacted may lead to significant increases in regional and global wind erosion rates. For these reasons, management policies of arid and semiarid regions should reflect the important role these crusts play in soil surface stability, and should reduce disturbance to these biological crusts whenever possible.

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# Soil carbon storage responses to expanding pinyon–juniper populations in southern Utah

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*Abstract*. Over the past several decades, the expansion and thickening of woodlands in the western United States has caused a range of ecological changes. Woody expansion often leads to increases in soil organic matter (SOM) pools with implications for both biogeochemical cycling and ecological responses to management strategies aimed at restoration of rangeland ecosystems. Here we directly measure C and N stocks and use simple non-steady-state models to quantify the dynamics of soil C accumulation under and around trees of varied ages in southern Utah woodlands. In the two pinyon–juniper forests of Grand Staircase Escalante National Monument studied here, we found ~3 kg C/m$^2$ and ~0.12 kg N/m$^2$ larger C and N stocks in soils under pinyon canopies compared to interspace sites. These apparent increases in soil C and N stocks under woody plant species were dominated by elevated SOM in the surface 10 cm of soil, particularly within non-mineral-associated organic fractions. The most significant accumulation of C was in the >850 µm fraction, which had an estimated C residence time of <20 yr. Rates of carbon accumulation following pinyon–juniper expansion appear to be dominated by changes in this fast-cycling surface soil fraction. In contrast, we found that after separating >850 µm OM from the remaining light fraction (LF), C had residence times of ~400 yr and mineral-associated (MA) soil C had residence times of ~600 yr. As a result, we calculate that input rates to the LF and MA pools to be 10 ± 1 and 0.68 ± 0.15 g·m$^{-2}$·yr$^{-1}$ (mean ± SE), respectively. These findings suggest that one consequence of management activities aimed at the reduction of pinyon–juniper biomass may be a relatively rapid loss of soil C and N pools associated with the >850 µm fraction. The temporal dynamics of the <850 µm pools suggest that carbon and nitrogen continue to accumulate in these fractions, albeit at very slow rates, and suggest that multidecadal storage of C following tree recruitment is limited to relatively small, subsurface fractions of the total soil C pool.

*Key words: arid; carbon; Grand Staircase Escalante National Monument, Utah, USA; grazing; nitrogen; pinyon–juniper forest; soil; woody encroachment.*

## Introduction

The arid and semiarid regions of the intermountain western United States have experienced rapid ecological and human change over the past 100 years. These changes include a widespread expansion of woody plant species into adjacent plant communities (Archer et al. 1995, Van Auken 2000), changes in fire frequency and severity (Baker and Shinneman 2004), and widespread disturbance of soils through land use change (Neff et al. 2008). Arid and semiarid ecosystems are among the most disturbed places on earth (Asner and Martin 2004), and disturbance in these settings can lead to dramatic changes in ecological states (Westoby et al. 1989). In both the context of response to disturbance and long-term sustainability of arid land ecosystems, soil organic matter (SOM) content plays a critical role. As the major repository for C, N, and available P, the organic pools are critical to the maintenance of nutrient cycles and aboveground productivity. In dryland ecosystems, these stocks of organic matter may be particularly important because these settings typically have relatively low concentrations of organic matter compared to more mesic environments (Batjes 1996). With small standing stocks of soil C and N, these ecosystems may be particularly vulnerable to disturbance (Fernandez et al. 2008), but conversely these soils may represent a potential target for activities designed to increase soil carbon storage (Lal 2004).

The response of arid ecosystems to the encroachment of woody species is highly variable, but typically involves a moderate to large increase in aboveground carbon stocks (Gill and Burke 1999, Asner et al. 2003, Hibbard et al. 2003, Hughes et al. 2006), although this is

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not always the case (Schlesinger and Pilmanis 1998, Wilson and Thompson 2005). Changes in aboveground C inputs, whether positive or negative, also impact belowground C and N transformations (Gill and Burke 1999, Jackson et al. 2002, McCulley et al. 2004) and alter the spatial patterning of belowground resources (Schlesinger and Pilmanis 1998). Whereas the trend is for woody encroachment to increase aboveground carbon, the belowground response is substantially more variable. Soil organic matter stocks can increase, decrease, or stay the same after woody encroachment (Connin et al. 1997, Gill and Burke 1999, McCulley et al. 2004, Hughes et al. 2006). The implications of these changes are important because the broad expansion of woody biomass across the western United States and associated changes in carbon stocks may be large enough to impact the continental carbon balance (Birdsey et al. 2006). Estimates of the impact of western U.S. tree and shrub expansion on the U.S. carbon sink range from a negligible effect (Jackson et al. 2002) to a sink of 0.13 Pg C/yr (Pacala et al. 2001). Much of the uncertainty related to the size of this sink is due to the fate of belowground carbon pools following woody encroachment and there is an ongoing debate about the causes for this variation, with evidence that water availability may play an important role in the fate of belowground C following the expansion of woody biomass (Goodale and Davidson 2002, Guo and Gifford 2002, Jackson et al. 2002, Wheeler et al. 2007).

The majority of studies describing the effect of vegetation change on soil carbon stocks in drylands focus on changes in bulk soil carbon pools. Although important, the measurement of change in total soil carbon stocks provides only limited insight into the broader biogeochemical and ecological implications of changes in woody plant communities. Soil organic matter pools have a variety of turnover times, with a capacity for multidecadal carbon storage in only those fractions with slow turnover times. In many ecosystems, physically stabilized carbon pools will drive long-term carbon storage because mineral-stabilized forms of carbon turn over on decadal to century timescales (Trumbore 2000). As woody plant communities expand into grassland settings, the form and degree of stabilization of soil organic matter will play a major role in determining the long-term carbon and nutrient balance implications. If, for example, mineral-stabilized soil carbon stocks are responsible for the majority of soil organic matter changes associated with vegetation change, then the impact on carbon stocks will be long-lived (decades to centuries). If, on the other hand, soil carbon changes are dominantly in the non-mineral-stabilized organic fraction of soils, then these pools will likely be more ephemeral, prone to loss during disturbance, and more likely to respond to specific management activities. Additionally, the stabilization of N into these slower or faster turnover pools will have broad implications for the degree to which woody encroachment leads to increases in fertility under shrubs and trees (Schlesinger et al. 1996) and to the long-term implications of changes in soil nutrient content on restoration or management activities.

The role of physical stabilization in soil organic matter storage in dryland ecosystems is not particularly clear. In one study in a New Mexico desert, there was a relatively rapid incorporation of C from mesquite shrubs into the mineral-associated soil carbon pool (Connin et al. 1997). In other arid settings, however, the role of physically stabilized carbon is less apparent, with little evidence for correlations between soil silt and clay content and total soil C (Fernandez et al. 2008). The possibility of limited soil textural control on carbon suggests that mineralogic control of organic matter storage could be limited in some arid settings. These findings, in combination with evidence that soil $CO_2$ fluxes have nonlinear (e.g., threshold) responses to soil moisture and temperature (Fernandez et al. 2006), suggest that some arid ecosystems have relatively little stable soil organic matter and/or capacity for long-term carbon storage. Without additional evidence for the capacity of arid ecosystems to store carbon in long-term organic matter pools, it is difficult to evaluate the permanence of potential changes in soil organic matter associated with woody encroachment or to evaluate claims of a large carbon sequestration potential in dryland systems (Lal 2004).

The history of land use in the western United States also complicates interpretation of soil organic matter changes in response to tree and shrub expansion. Livestock grazing can have significant and long-lasting impacts on soil carbon stocks (Archer et al. 1995, Hiernaux et al. 1999, Neff et al. 2005) and may directly influence the expansion of woody species (Archer et al. 1995). Western rangelands have been extensively used for livestock grazing, and there is an exceptionally limited number of sites that have not experienced significant land use change during the 20th century (Van Pelt et al. 1992). In this study, we carried out an evaluation of carbon stocks under and between tree canopies in a site that has experienced very limited livestock grazing over the past century and a historically grazed site that has been grazed since the late 1800s in Grand Staircase Escalante National Monument, Utah, USA. Historical aerial photographs and stand age structure analysis from these sites suggest that there has been rapid recruitment in these populations over the last century (Harris et al. 2003; N. N. Barger, H. Adams, and C. Woodhouse, *unpublished manuscript*). We specifically focus on an evaluation of the rates of C and N accumulation under pinyon–juniper canopies to test the hypothesis that tree encroachment has led to increased soil carbon stocks and to evaluate the role of land use in mediating these responses. To further examine the nature of soil carbon, we also measured the distribution of organic matter through soil physical fractions to assess the degree of organic matter stabilization in these desert soils. These measurements were combined with a non-steady-state modeling approach to evaluate the potential for these fractions to accumulate C and N in pinyon–juniper woodland soils following woody expansion.

## METHODS

This study was conducted in Grand Staircase Escalante National Monument (GSENM) in southern Utah. This area of the Colorado Plateau contains a range of ecosystems including grasslands, shrublands, and pinyon–juniper woodlands (Stohlgren et al. 2005). Both sites receive ~360 mm of precipitation each year and are located at an elevation of 2000–2200 m. Mean daily minimum temperature at the site is –8.2°C and mean daily maximum temperature is 30.5°C. No Man's Mesa has not been grazed by cattle and was only grazed by goats for a brief period (two years) in the late 1920s. No Man's Mesa is geographically separated from surrounding areas by high cliffs and steep access trails. In contrast, Deer Spring Point has been extensively used for cattle grazing since the 1880s and is easily accessed from surrounding terrain. Soil for both sites is derived from eolian and alluvial Carmel-Page Formation sandstone. Soils are classified as a Pinepoint soil, which are well-drained sandy soils with sand content ranging from 85% to 90% across our sites. Vegetation structure at these sites can best be described as upland "wooded shrublands," sites where shrubs are dominant but support significant tree cover (see Romme et al. 2008; U.S. Department of Agriculture Natural Resources Conservation Service, *available online*).[6] Pinyon–juniper (P–J) wooded shrublands are intermediate to P–J savanna, which is characterized by a well-developed and often dense herbaceous understory, or old-growth P–J that maintains a dense P–J overstory with a depauperate herbaceous understory (Romme et al. 2008). The dominant shrub at this site was mountain big sagebrush (*Artemisia tridentata* subsp. vaseyana Nutt.(Rydb.) Beetle). Prior work at these two sites indicates that the sites have similar tree and shrub densities (Harris et al. 2003, Guenther et al. 2004), and related work indicates that tree ages are very similar between the two sites and that both sites underwent a large and synchronous expansion of pinyon cover in the time period since 1930 (N. N. Barger, H. Adams, and C. Woodhouse, *unpublished manuscript*). For this study, we controlled for slope angle by sampling on low-angle slopes (<15°). Field tests of soil $CaCO_3$ content was carried out with acid tests and observing soil effervescence. Sites were not located in areas where there was a strong indication of high $CaCO_3$ content.

### *Sampling techniques*

We used bedrock geology and soil classification maps in a geographic information system (GIS) to establish six transects on similar soil and geologic settings on No Man's Mesa and six transects on Deer Spring Point. Transects were 300 m, with sampling sites placed every 25 m. At each transect stop, one tree core was extracted for dating tree ages. Each tree core was collected ~15 cm from the ground with an increment borer. We sampled soils at a midpoint between the tree trunk and the outer canopy edge and in an adjacent interspace without input of organic matter from trees. Two 10 cm deep soil cores were extracted at every point (one interspace and one under-canopy) with a volumetric soil corer (Soil Core Sampler Model 0200; Soil Moisture Equipment, Santa Barbara, California, USA). At every second stop, two additional cores, representing the 10–30 cm deep section of the profile, were extracted. At every third stop, we collected a 0–10 cm, >10–30 cm, and a >30–50 cm sample of the soil profile. With this sampling scheme, we collected 50 samples per transect: 26 with a depth of 0–10 cm, 16 with a depth of >10–30 cm, and 8 representing the >30–50 cm deep section of the soil profile.

The ecosystems in this area do not support a clearly separate litter layer above an O horizon. Rather, there is considerable intermixing of surface mineral soils and high organic content, particularly under tree canopies. In effect, the surface 10-cm samples include what would traditionally be the litter layer and the O horizon; in these ecosystems, the lack of soil layer differentiation makes it difficult to physically separate these layers, and so we used a variety of approaches to differentiate between soil C fractions.

### *Soil carbon analysis*

All soil samples were weighed after oven drying at 60°C for 48 h. Particles larger than 2 mm were then removed by passing the sample through a 2-mm sieve that removed large-size plant materials. The >2-mm size fraction was, on average, ~2% of the mass of the sample and under 1% of total size carbon. Samples were then passed through an 850-μm sieve to obtain two size fractions; a <850-μm size fraction comprised of mineral and amorphous organic soil particles and a >850-μm size fraction comprised largely of recognizable organic particles. We chose the relatively large cutoff size because of the substantial contribution of relatively un-decomposed organic matter from under the tree canopies and because this large size cutoff allowed separation of a fraction that was entirely organic and clearly related to litter deposition. Each size fraction was weighed in order to back-calculate kilograms of C per square meter for each sample as a whole (<850-μm + >850-μm size fraction). To remove carbonates from the soil, each <850-μm size fraction was then split into a 2-g aliquot and acidified with 15% HCL until effervescence was no longer observed. After acidification, samples were re-dried and analyzed for C and N concentration with an EA 1110 CNS combustion analyzer (Thermo Electron, Waltham, Massachusetts, USA) at the Uni-

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[6] ⟨http://www.ut.nrcs.usda.gov/technical/technology/range/mlra35.html⟩

TABLE 1. Estimates of organic and inorganic C stocks and organic N stocks for for pinyon–juniper ecosystems in the No-Man's Mesa and Deer Springs Point sites in Grand Staircase Escalante National Monument, Utah, USA.

| C or N stock | Soil layer 0–10 cm | Soil layer 10–30 cm | Soil layer 30–50 cm |
|---|---|---|---|
| Soil organic C (kg/m$^2$) | | | |
| Canopy | 5.32 (0.56) | 1.59 (0.14) | 1.25 (0.11) |
| Interspace | 1.07 (0.11) | 0.96 (0.09) | 0.83 (0.07) |
| Soil organic N (kg/m$^2$) | | | |
| Canopy | 0.23 (0.03) | 0.12 (0.03) | 0.08 (0.01) |
| Interspace | 0.07 (0.01) | 0.06 (0.005) | 0.05 (0.004) |
| Soil $CO_3$-C (kg/m$^2$) | | | |
| Canopy | 0.51 (0.25) | 1.93 (1.38) | 5.56 (3.66) |
| Interspace | 1.13 (0.59) | 4.96 (3.19) | 5.02 (4.84) |

*Notes:* We sampled soils at a midpoint between the tree trunk and the outer canopy edge and in an adjacent interspace without input of organic matter from trees. Values are means with SE in parentheses.

versity of Colorado, Boulder, Colorado, USA. All C and N concentrations are presented in this paper on a carbonate-free basis; however, estimates of C and N stocks include a correction of measured C and N concentrations to compensate for the removal of carbonates prior to analysis. In the majority of surface soils, this correction is very small (<1% of soil mass), but in some subsurface soils, there are somewhat higher concentrations of carbonates, and these values are presented in the results (Table 1). All surface soil samples (0–10 cm) were sampled volumetrically to allow calculation of bulk densities. Subsurface samples were not sampled volumetrically, and so for calculations of carbon stocks in subsurface horizons, we use an average of 1.6 g soil/cm$^3$ for mineral-dominated soils of low organic content in this area.

To further separate the type of organic matter in these soils, we carried out density separations on a selected subset of surface (0–10 cm), <850-μm samples from both the grazed and ungrazed sites for a total of 46 samples. Only sites located under tree canopies contained appreciable amounts of non-mineral organic matter, so for density separations we selected a range of samples from underneath trees with an age range of 50–400 years. The age range was used to examine the temporal dynamics of C accumulation under trees of varied age in these sites. For this analysis, pre- and post-density separation masses were recorded to determine the C and N fractional contribution from each density. The density separations were carried out in a sodium tungstate solution with a density of 1.68 to separate light (dominantly non-mineral) and heavy (dominantly mineral) soil fractions. These separations were repeated several times by centrifuging and decanting the suspended portion (light fraction) onto a Buchner funnel with a Whatman 50 filter. Each sample was centrifuged until all light-fraction material was removed and only heavy-fraction material remained at the bottom of the centrifuge tube. After centrifuging, each light and heavy fraction was thoroughly rinsed with deionized water to remove residual sodium polytungstate, oven dried at 60°C, acidified to remove carbonate, and analyzed for C and N concentration. The >850-μm size fraction is referred to here as particulate organic matter (POM), whereas the <850-μm low-density fraction is called light-fraction (LF) organic matter, and the <850-μm high-density fraction is designated as mineral-associated (MA) organic matter.

### *Tree age dating*

Tree cores were analyzed at the Institute for Arctic and Alpine Ecology, Dendrochronology Laboratory, University of Colorado, Boulder. Increment cores were mounted (Stokes and Smiley 1968) and progressively sanded with Federation of European Producers of Abrasives (FEPA) 120-, 220-, 320-, and 400-grit (162, ?1 68, 44.7–47.7, and 33.5–36.5 μm, respectively) sandpaper using a Bosch belt sander (Robert Bosch, Farmington Hills, Michigan, USA). Cores were also hand-surfaced using 400- and 1200-grit sandpaper (9 and 30 μm, respectively). Cores were then visually cross-dated by using a previously developed pinyon pine (*Pinus edulis*) chronology for No Man's Mesa (C. Woodhouse, *unpublished data*). For samples difficult to visually cross-date, undated inner sections were run against a previously developed chronology using COFECHA (Holmes 1983, Grissino-Mayer 2001). All matching dates found using the COFECHA program were then visually verified. We recorded the first year of secondary growth to calculate tree age. Although we made three to five attempts in the field to obtain cores that included pith, we were not always successful. We estimated dates for cores without pith by overlaying sets of concentric circles on the inner rings of the core (Applequist 1958). To correct for the time to coring height, we collected pinyon seedlings from a range of sites and soil types across the region. Mean time to a coring height of 15 cm is ~11 yr. Once an inner ring date was obtained from the cores, 11 yr was subtracted from this date to correct for the time to coring height.

### *Radiocarbon analysis*

We measured the $^{14}C/^{12}C$ ratios in a representative set of soils from the LF and MA fractions of the <850-μm soil size class. These samples were separated and then freeze-dried on a Virtis BT6KEL-85 freeze dryer (SP Industries, Gardiner, New York, USA) fitted with an Edwards FL20K foreline trap (Edwards Limited, Crawley, West Sussex, UK) to prevent oil contamination of samples. The soils were analyzed for $^{14}C$ and $^{13}C$ at the Keck Accelerator Mass Spectroscopy facility at the University of California, Irvine, California, USA. All values are reported in $\Delta^{14}C$ and have reported uncertainties of <2‰.

### *Statistical analysis*

It is not possible to replicate entire ungrazed mesa sites, so for our statistical comparisons, we used each 300-m transect as a replicate. We carried out site comparisons using multivariate analysis of variance (MANOVA) and regression analysis. For MANOVA tests, we used a mean value for the interspace or undercanopy and depth soil samples for an entire transect ($n \sim 13$ soil cores per transect) as a single measurement with a true replicate number equal to the number of transects on each mesa top (six on each). All soil samples were analyzed for C and N concentration, affording a full factorial MANOVA using C and N as response variables and site history (grazed vs. ungrazed), vegetation type (canopy vs. interspace), and soil depth as factors. For comparisons against tree age and other continuous variables, we use regression analysis. We also carried out an ANOVA of surface soil carbon and nitrogen stocks with site history, vegetation type, and depth as factors. All carbon turnover model fits were determined using iterative methods in JMP (SAS Institute, Cary, North Carolina, USA) and/or Microsoft Excel.

### *Quantifying soil organic C dynamics*

The combination of C differences among interspace and canopy sites and variation in C stocks with tree age provide an opportunity to examine the non-steady-state dynamics of carbon accumulation in these ecosystems. The perturbation to soil carbon associated with tree recruitment at GSENM is still underway, and so the measurements made here (and in other similar settings) represent carbon stocks that are not in steady state. This aspect of soil carbon dynamics influences estimates of turnover time and accumulation rates, so here we present a non-steady-state approach to quantifying soil organic carbon turnover in the 0–10 cm soil fractions and outline a sequential procedure to estimate soil carbon changes through time. In all cases, we quantify changes in C stocks over "time," using canopy age (determined from tree ring analysis) as a surrogate for the passage of hundreds of years. We first calculate the time dependence of aboveground C inputs to the soil by fitting a logistic model to the accumulation of the most ephemeral soil C pool (>850 μm). We then estimate turnover times by fitting the $\Delta^{14}C$ values and observed C accumulation in soil C fractions. Finally, we obtain an overall equation for the soil C fractions as a function of time by combining observed C accumulation data with the estimated turnover rate to fit the remaining unknown, the rate of C input to the fraction.

Our equations are based on a simple C balance for each fraction, *i*, of the form

$$\frac{dC_i}{dt} = I_i - k_i C_i \tag{1}$$

where $C_i$ is the C stock in the fraction (in grams per square meter), $k_i$ is the turnover rate of the fraction, and $I_i$ represents inputs to the pool. As written, $k_i$ is consistent with all forms of C turnover, including loss as $CO_2$, dissolved organic carbon (DOC), or particulate transfer to subsoil horizons, and erosion. No attempt was made to model C stocks in the subsoil (>10 cm). Carbon inputs were assumed to follow a logistic curve, based on the observation of very low inputs until individual trees emerged from shrub canopies. This slow emergence is followed by a period of rapid canopy development and a subsequent phase in which inputs are assumed to remain at steady state. To obtain a quantitative estimate of these C inputs, we fitted the logistic equation, written as

$$\frac{dC_{(\mathrm{POM})}}{dt} = rC_{(\mathrm{POM})}\left(1 - \frac{C_{(\mathrm{POM})}}{C_{\mathrm{S(POM)}}}\right) \tag{2}$$

to the pattern of accumulation in the >850 μm C fraction ($C_{(\mathrm{POM})}$), yielding the following result:

$$C_{(\mathrm{POM})}(t) = \frac{C_{\mathrm{S(POM)}}}{(1 - e^{-rt}) + \dfrac{1}{C_{0(\mathrm{POM})}} e^{-k_{\mathrm{pom}} t}} \tag{3}$$

where $C_{0(\mathrm{POM})}$ represents the initial >850-μm C stock (0.001 ± 0.0005 kg/m$^2$ [mean ± SE]), $C_\mathrm{S}$ is the steady-state >850-μm C stock (3.0 ± 1.2 kg/m$^2$), $k_{\mathrm{pom}}$ is a rate constant (0.13 ± 0.04 yr$^{-1}$), and $t$ is the tree age as determined by dendrochronology (in years).

For the LF and MA, we estimated C dynamics using the following C accumulation equation, which is a solution to Eq 1: ?2

$$C(\mathrm{age}) = \frac{I}{k}\left(1 - e^{-k(\mathrm{age})} + C_o\right) \tag{4}$$

where *I* is carbon inputs (in grams of C per square meter per year) and age is tree age in years. In this equation *I* represents inputs entering the fraction and *k* is the turnover rate (per year). Turnover rates were estimated by fitting a $\Delta^{14}C$ model and are commonly expressed as turnover times ($1/k$). The $\Delta^{14}C$ model for estimating *k* and remaining parameters in Eq. 4 were fitted using an iterative

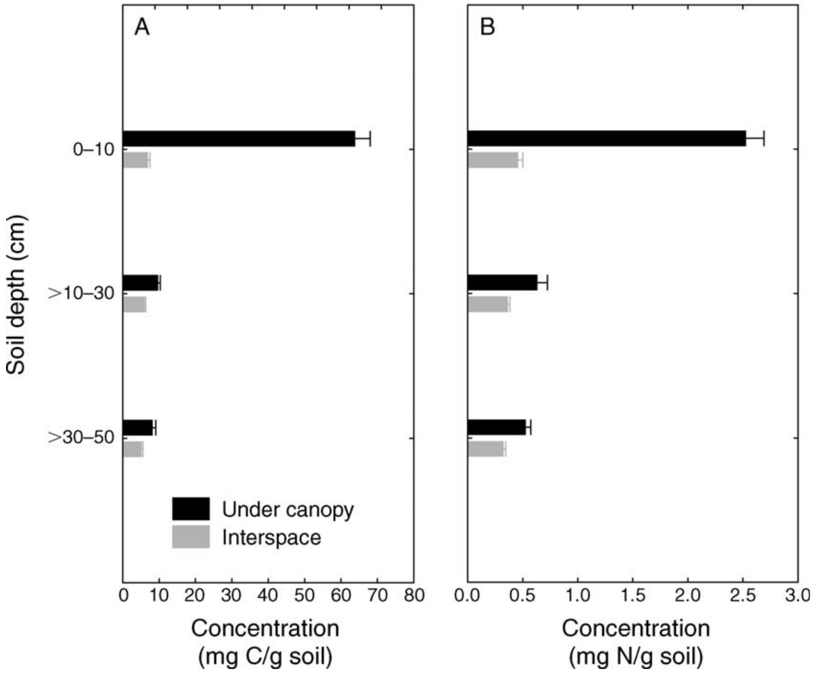

FIG. 1. Distribution of carbon and nitrogen concentrations (mean + SE) with depth and across landscape (pinyon canopies vs. interspace sites) in two pinyon–juniper forests of Grand Staircase Escalante National Monument, Utah, USA. Concentrations are presented on a carbonate-free basis.

procedure, repeated several times until the adjustments were less than analytical uncertainty. The initial stocks $C_{0\mathrm{LF}}$ and $C_{0\mathrm{MA}}$ were estimated using the mean C stocks for those fractions in open (non-canopy) sites.

Changes in $\Delta^{14}$C were modeled using a separate version of Eq. 1 established for $^{14}$C, taking into account the variable $\Delta^{14}$C of inputs driven by changes in atmospheric $\Delta^{14}CO_2$ as well as incorporating radioactive decay in losses. To implement this model of $\Delta^{14}$C in LF and MA, we modified an existing annual-time-step model (Baisden et al. 2002) to account for non-steady-state conditions, specifically, the accumulation of C in the modeled pool following vegetation change. The atmospheric $\Delta^{14}$C record was derived from Hua and Barbetti (2004) for 1955–1997, Baisden et al. (2002) prior to this period, and Levin and Kromer (2004) as well as Turnbull et al. (2007) for 1997–2005. All atmospheric $\Delta^{14}$C data chosen reflect a spring–autumn growing season for mid-latitude regions of the Northern Hemisphere.

## RESULTS

### *Organic matter concentrations and stocks*

Soil carbon and nitrogen are generally low in the study sites in GSENM, with the exception of the surface soils under tree canopies. Grazing history had no impact on soil C or N stocks, but there were significant differences in C and N content of under-canopy and interspace sites ($F_{2,62} = 88.7$, $P < 0.001$), with surface under-canopy sites averaging 60 mg C/g soil and 2.5 mg N/g soil and interspace sites ranging between 10 and 20 mg C/g soil and between 0.3 and 0.5 mg N/g soil on a carbonate-free mass basis (Fig. 1). Soil C and N concentrations also varied with soil depth ($F_{4,124} = 41.4$, $P < 0.001$), but with a more significant proportional reduction in concentrations under canopies than in the interspace (depth × canopy/interspace interaction term $F_{4,124} = 39.0$, $P < 0.001$; Fig. 1). Soil carbonate stocks are typically <10 mg C/g soil in surface soils and increase to an average of 30 mg C/g soil in subsurface soils.

The C and N concentration differences under tree canopies are driven by changes in the accumulation of organic matter under the trees. Overall, the accumulation of organic matter under trees is caused primarily by the accumulation of POM and LF organic matter (Fig. 2). The implication of this trend is evident in the significant negative exponential relationship between soil carbon content and soil bulk density ($P < 0.001$, $r = -0.8$, $\%C = 78.09 \times \exp(-3.05 \times \text{bulk density})$). High values of bulk density in the figure (>1.5 g/cm$^3$) are nearly exclusively mineral, whereas the values below 0.5 g/cm$^3$ (with organic carbon contents of ~10%) are in sites (nearly exclusively in under-canopy settings) where significant amounts of low-density organic material are intermixed with mineral soil.

Soil C and N stocks follow similar patterns to the organic matter concentration data above with higher under-canopy stocks of C and N compared to interspace sites ($F_{2,173} = 22.7$, $P < 0.001$), higher surface organic C and N stocks compared to subsurface soils ($F_{4,346} = 15.1$, $P < 0.001$) and no significant effect of site grazing

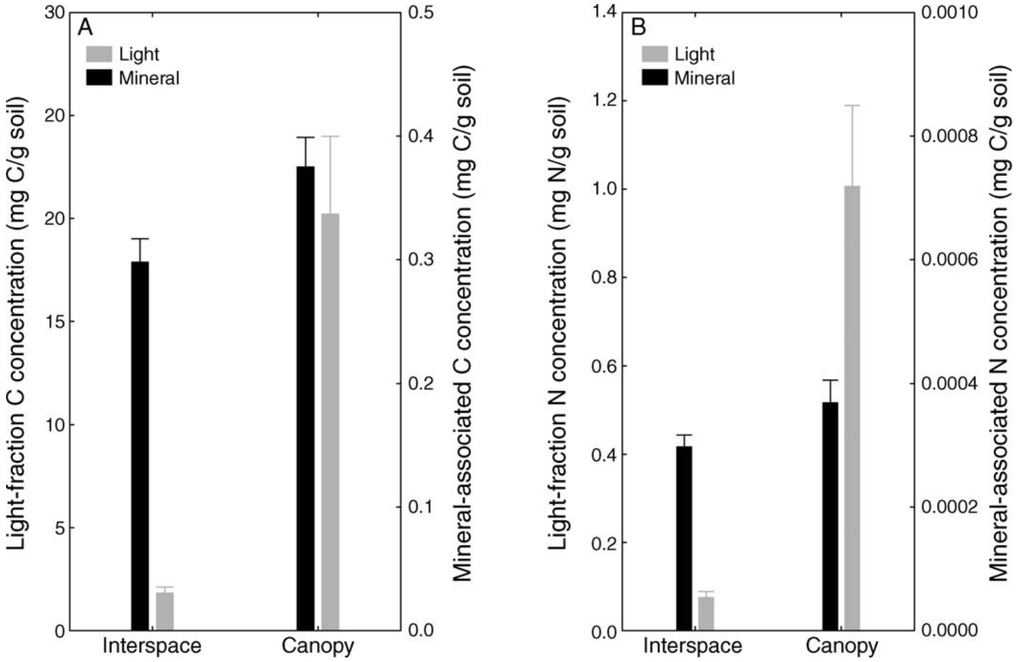

FIG. 2. Distributions of carbon and nitrogen in light ($<1.7$ g/cm$^3$) and mineral-associated ($>1.7$ g/cm$^3$) soil fractions. (A) Concentration of C in interspace and under-canopy soils in light and mineral-associated fractions. (B) Concentration of N for both density fractions in the interspace and under-canopy soils.

history (Table 1). Both the light- and heavy-fraction element pools reflect these patterns in both concentrations (Fig. 2) and in stocks.

### *Soil C dynamics*

Parameters describing soil C dynamics, including turnover, were estimated by fitting the C stocks in each fraction vs. age to Eqs. 3 and 4 (Fig. 3). Fitting the POM fraction to Eq. 3 produces turnover ($k_{\mathrm{POM}}$), $C_{0(\mathrm{POM})}$, and $C_{\mathrm{S(POM)}}$ estimates (Table 2). The primary purpose of fitting Eq. 3 was to quantify changes in the rate of C inputs to the LF and MA pools as a function of tree age. As a result, the values for POM parameters reported in Table 2 should be viewed with some caution. In particular, the values reported for $k_{\mathrm{POM}}$ and $C_{\mathrm{S(POM)}}$ imply a C input rate of 380 ± 190 g C·m$^{-2}$·yr$^{-1}$, which is considerably higher than a three-year estimate of litterfall inputs of ~120 g C·m$^{-2}$·yr$^{-1}$ from a pinyon–juniper ecosystem in Arizona (Grier et al. 1992). Factors that may explain this discrepancy include that the model implicitly includes belowground inputs (e.g., root litter), climate, and spatial variability, as well the fact that the use of Eq. 2 to estimate changes in input rates through time may oversimplify C dynamics. Additionally, these rates of input are for the area immediately under the tree canopy, and spatially integrated landscape-scale means would be considerably lower. Regardless, Fig. 3 shows that the $C_{0(\mathrm{POM})}$, and $C_{\mathrm{S(POM)}}$, as well as our overall estimates of POM C stock vs. age, generate reasonable values given the large amount of variability in the data.

The soil POM accumulates rapidly in these sites and shows signs of reaching steady state (Fig. 3). In contrast, the dynamics of LF and MA accumulation appear much slower and benefit from the use of radiocarbon to constrain the apparent turnover rate. By modeling both total C and $^{14}$C using Eq. 4 in an iterative procedure to constrain both inputs and turnover rate, we calculate the LF turnover rates shown in Table 3. In the six samples with measured $\Delta^{14}$C, calculated residence times occupy a large range, spanning 237 years to 613 years, with an estimate of 433 ± 61 years. This residence time estimate is much longer than many previous estimates (Connin et al. 1997), reflecting the potential importance of correctly representing non-steady-state conditions in the model. The long residence times of LF reported here also demonstrate effectiveness of the 850-μm delineation used to separate POM prior to density separation and highlight the unusual dynamics of this fraction of soil carbon in dryland systems. The sample-to-sample variability in LF turnover estimates may reflect spatial variability, including variability in slow-turnover materials such as black carbon that may contribute to the fraction. The model fit suggests an input rate to the LF pool of 10 ± 1 g C·m$^{-2}$·yr$^{-1}$, indicating a relatively low C accumulation potential for this fraction when compared to site level net primary productivity (NPP).

Using the same procedure, we calculated MA soil C turnover time of 542 and 591 yr for two individual samples, yielding a mean estimate of 567 yr. The fit to Eq. 4 suggests a value for $I$ of 0.68 ± 0.15 g C·m$^{-2}$·yr$^{-1}$, an order of magnitude lower than the estimated C input

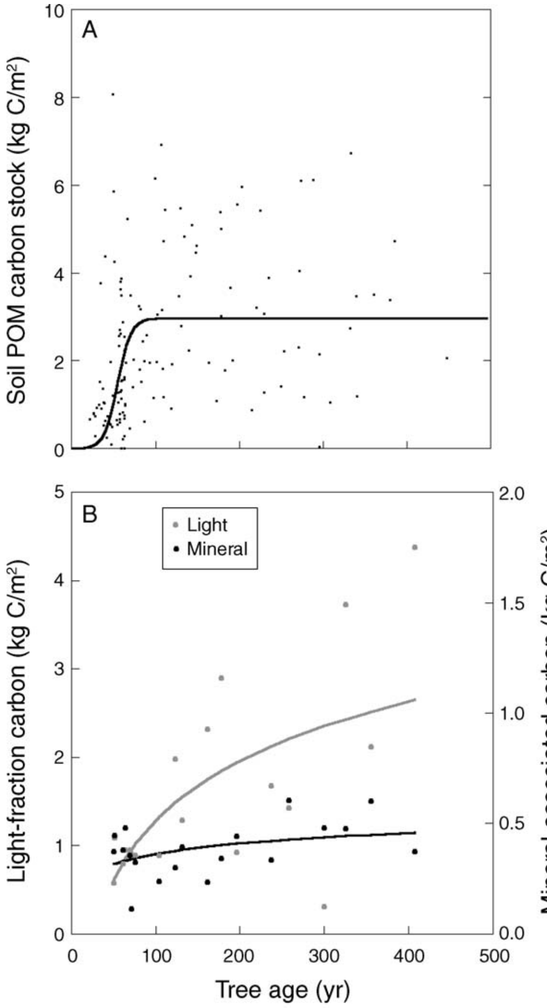

FIG. 3. Comparison of model fits and data for carbon in three soil fractions. (A) The modeled (solid line) and measured particulate organic matter (POM) fraction (>850-μm size class). (B) Fits to the light fraction and the mineral-associated fraction and data for these two fractions. Both graphs use tree age as a surrogate for time.

rate for LF. This $I$ value implies that the accumulation rate of carbon into the physically protected MA forms in these ecosystems is <1% of NPP (Grier et al. 1992).

## DISCUSSION

Pinyon pine recruitment in the 20th century in GSENM has resulted in significant increases in soil C and N storage. For carbon, soils under canopies have an average of ~3 kg of additional C than soils in interspace settings and for N the increase is ~0.1 kg. This accumulation happens largely over the course of the initial 100 years following tree establishment and translates into an overall soil accumulation rate of ~30 g $C \cdot m^{-2} \cdot yr^{-1}$ and 1 g $N \cdot m^{-2} \cdot yr^{-1}$, a number within the range of other studies of soil C accumulation following woody encroachment (Jackson et al. 2002). Despite this relatively large total rate of soil organic matter accumulation, the dynamics of the various organic matter pools are very different, with the highest rate of accumulation occurring in the fast-turnover POM pools, followed by far lower accumulation rates for the LF soil pools, and still slower accumulation into the physically protected MA pools.

### *Rates and forms of C accumulation*

The majority of the carbon and nitrogen changes associated with pinyon recruitment into sagebrush communities occurred under tree canopies. This pattern of organic matter accumulation under trees relative to plant interspaces follows the "island of fertility" patterns observed in other desert systems and suggests a net gain of soil organic matter following tree recruitment in these ecosystems (Schlesinger et al. 1996), assuming that interspace soils are not becoming highly eroded, a pattern that was not observed at these sites. This pattern is in contrast to observations from other dryland ecosystems undergoing vegetation change that suggest a net depletion of soil organic matter following woody encroachment (Jackson et al. 2002). Taken in combination, these studies illustrate that there may be different trajectories for organic matter changes in dryland ecosystems undergoing changes in the woody component of these systems. Indeed, global syntheses of woody encroachment highlight the sensitivity and sign of soil C

TABLE 2. Parameter values describing dynamics of soil C fractions based on Eqs. 3 and 4.

| Symbol | Description | Estimate | SE | Units |
|---|---|---|---|---|
| $k_{\mathrm{POM}}$ | POM turnover rate | 0.13 | 0.04 | $yr^{-1}$ |
| $C_{0(\mathrm{POM})}$ | initial >850 μm POM stock | 0.001 | 0.0005 | kg $C/m^2$ |
| $C_{\mathrm{S(POM)}}$ | steady-state >850 μm POM stock | 3.0 | 1.2 | kg $C/m^2$ |
| $1/k_{\mathrm{LF}}$ | LF residence time | 433 | 61 | yr |
| $C_{0(\mathrm{LF})}$ | intial LF C stock | 0.26 | 0.17 | kg $C/m^2$ |
| $I_{\mathrm{LF}}$ | input rate of C to LF | 10 | 1 | g $C \cdot m^{-2} \cdot yr^{-1}$ |
| $1/k_{\mathrm{MA}}$ | MA residence time | 567 | NA | yr |
| $C_{0(\mathrm{MA})}$ | initial MA C stock | 0.29 | NA | kg $C/m^2$ |
| $I_{\mathrm{MA}}$ | input rate of C to MA | 0.68 | 0.15 | g $C \cdot m^{-2} \cdot yr^{-1}$ |

*Note:* Abbreviations are: POM, particulate organic matter; LF, light fraction; MA, mineral-associated fraction.

TABLE 3. Carbon isotope ratios, C stocks, and calculated mean residence times (MRT) from selected soil fraction samples.

| Soil fraction sample, site, and age | $\delta^{13}C$ (‰) | SD | $\Delta^{14}C$ (‰) | SD | C stock at $t = 0$ (g C/m$^2$) | Measured C stock (g C/m$^2$) | MRT (yr) |
|---|---|---|---|---|---|---|---|
| Light fraction | | | | | | | |
| Deer Springs | | | | | | | |
| 131 yr | −24.6 | 0.15 | 89.7 | 1.6 | 0.26 | 1.29 | 613 |
| 50 yr | −25.3 | 0.15 | 21.1 | 1.5 | 0.50 | 0.57 | 406 |
| 325 yr | −23.1 | 0.15 | 30.9 | 1.7 | 0.26 | 3.73 | 470 |
| No Man's Mesa | | | | | | | |
| 300 yr | −23.2 | 0.15 | 28.4 | 1.9 | 0.26 | 0.31 | 237 |
| 51 yr | −24.4 | 0.15 | 72.8 | 1.6 | 0.80 | 1.09 | 299 |
| 123 yr | −23.8 | 0.15 | 59.2 | 1.6 | 0.80 | 0.93 | 575 |
| Mineral-associated fraction | | | | | | | |
| Deer Springs, 325 yr | −22.2 | 0.15 | −10.3 | 1.5 | 0.29 | 0.48 | 542 |
| No Man's Mesa, 300 yr | −22.4 | 0.15 | −13.3 | 1.7 | 0.29 | 0.48 | 591 |

*Notes:* The samples span a range of tree ages. The soil fraction years refer to tree age where the sample was taken. We assumed an initial C stock and used measured carbon stocks and tree age as the input parameters for estimating mean residence time (MRT), as described with Eq. 4. The following values are point estimates for soil fraction turnover times under each tree where radiocarbon values were obtained: the residence time (mean ± SD) for the light fraction = 433 ± 149 yr and for the mineral-associated fraction = 566 (not determined; $n = 1$).

responses and the role that climate, soil properties, and land use history play in mediating these responses (Asner and Martin 2004, Asner and Archer 2008).

Despite net gains in soil organic matter beneath tree canopies, this study suggests that increases in organic matter occur largely in surface organic matter pools that are not stabilized physically. In this study, >80% of the carbon differences between canopy and interspace sites was due to the accumulation of surface non-mineral-stabilized C and N fractions. Although there is limited formation of a litter layer in these ecosystems, these large-size-fraction (POM) surface pools (generally holding more than half the soil carbon in these systems) clearly consist of recognizable plant detritus, with relatively little decomposed and humified SOM. Our estimates of turnover times in this surface organic matter pool yield mean residence times that are considerably less than 20 yr. This pattern of relatively rapid surface SOM turnover leads to a short period of C accumulation following tree recruitment with near steady-state conditions reached between 50 and 100 yr (Fig. 3A), a result consistent with rapidly increasing NPP paired with relatively rapid surface SOM turnover. Although the size of the surface C pool and its relation to tree age is admittedly quite variable, these data suggest that C is unlikely to be stabilized in this pool for more than two to three decades. The tree populations on GSENM have a median age of ~100 yr, and so these systems are likely near steady state with respect to surface POM soil carbon accumulation; accordingly, little additional stabilization into this pool would be expected in these mature pinyon–juniper forests.

The rapid turnover and low degree of stabilization of the large-size-fraction organic matter under tree canopies in these ecosystems may be a consequence of high surface decomposition rates in desert ecosystems. A study of decomposition dynamics near Canyonlands National Park found very rapid rates of soil respiration when environmental (temperature and moisture) conditions were adequate, conditions that occurred on average approximately one out of five days of the year (Fernandez et al. 2006). More generally, desert ecosystems are prone to rapid rates of decomposition and carbon release with pulsed resource availability (Huxman et al. 2004), and surface C is particularly vulnerable to loss due to a high degree of UV photo-oxidation of organic matter (Austin and Vivanco 2006). In the longer term, the relatively short residence time and lack of physical stabilization of the surface POM fraction makes the pool quite vulnerable to loss if either C inputs cease or environmental conditions change in a manner that favors increased decomposition.

The very rapid turnover of the surface POM fraction in this study stands in contrast to the comparative persistence of both LF and MA carbon in the soil subsurface. All carbon in the LF and MA fractions turns over very slowly, and even non-mineral-associated C appears to be relatively stable if it is intermixed with mineral soil. These fractions range from a small proportion of the total SOM in the first 100 years of tree growth to as much as half the soil carbon under very old trees. More-rapid turnover estimates have been presented for light-fraction SOM in the Jornada Experimental Range in New Mexico, USA (Connin et al. 1997), and for the Rio Grande Plains of Texas (Liao et al. 2006*b*), results that may be partially due to the transient nature of woody encroachment and the use of steady-state approaches to estimating turnover times in systems that are undergoing net C accumulation. As a point of comparison, a steady-state turnover time estimate based on stocks and input rates for these sites for the LF and MA fractions in this study would yield a turnover time of ~90 yr for the LF and ~600 yr for the MA fraction. This estimate is closer to the more-rapid

LF or POM turnover estimate from other studies. However, the evidence for continuing changes in the LF and MA small C fractions (<850 μm) of soils 300 yr after tree establishment suggest that steady-state assumptions for these fractions are not appropriate. Additionally, the separation of POM and LF in this study suggests that low-density organic matter fractions may exhibit a range of turnover times with some relatively recalcitrant components within the overall low-density fraction.

### *The potentially ephemeral nature of dryland soil C storage*

The dominance of the light fraction, the non-mineral-stabilized SOM fraction, in carbon accumulation at GSENM has broad implications for carbon storage patterns in the western United States if the findings of this study are more broadly applicable. Work in Canyonlands National Park and in the Jornada desert (Connin et al. 1997, Fernandez et al. 2008) also suggests that most soil C is present in non-mineral or aggregate stabilized fractions, although studies in Texas show a greater proportion of stabilized woody biomass carbon in mineral fractions (Liao et al. 2006*a*). These variable patterns in the form of stabilized carbon are important to the long-term fate of soil fertility in these ecosystems and particularly to ecological response to management and disturbance. Environmental conditions or management activities such as thinning or rangeland improvement treatments that result in reduced NPP or the elimination of woody biomass could also trigger the loss of previously stabilized soil carbon and nitrogen if those fractions lack substantial physical protection. While soil carbon is often viewed as a potential multidecadal sink of carbon (Lal 2004), these results indicate that the majority of the SOM increase associated with 20th century tree recruitment in these sites is either in soil fractions with rapid turnover (e.g., the POM fraction) or in fractions that lack inherent physical protection mechanisms (e.g., the low-density fraction). Thus these pools could be prone to loss within two decades following disturbances that accelerate decomposition or oxidation. The extensive recent tree mortality in response to drought across the southwestern United States (Breshears et al. 2005) is one example of a broadly distributed change in vegetation dynamics that could precipitate large surface soil carbon losses in a relatively short time frame. Some pinyon–juniper woodlands have experienced up to 90% mortality in some stands. Such a dramatic alternation of forest structure over a few years may result in significant C release from these forests decades into the future. In addition to large-scale mortality, fires in pinyon–juniper-dominated ecosystems are another disturbance that may lead to relatively widespread loss of both aboveground and surface soil carbon. Soil carbon sequestration is a stated goal of the U.S. Healthy Forest Restoration Act of 2003, and so the ability to balance change in C stocks against other goals for ecosystem management will likely become an increasingly important issue for public land managers.

To provide long-term storage of soil carbon, non-mineral-associated carbon pools require some degree of physical or chemical protection. The accumulation of carbon in the LF and MA fractions is slow but illustrates some capacity of these soils to support additional carbon stabilization. The MA fraction, in particular, is a potential long-term storage pool, a point that contrasts with many agricultural ecosystems in which the mineral soil C fraction is relatively carbon saturated and unable to support additional C stabilization (Hassink 1997, Stewart et al. 2007). Once stabilized, the organic matter in the mineral fraction would be expected to have slow turnover times (Trumbore 2000), and this is the case for this study. Consequently, SOM age combined with the accumulation rates for both C and N shown in Fig. 3 suggest that this pool has the capacity to stabilize carbon on a multidecadal to century timescale but the accumulation rates ($\sim$1.4 g $C \cdot m^{-2} \cdot yr^{-1}$ and 0.1 g $N \cdot m^{-2} \cdot yr^{-1}$) are low and represent a very small fraction ($\sim$5%) of the published estimates for total soil C accumulation in dryland systems (Jackson et al. 2002) following woody encroachment.

In addition to carbon stabilized in mineral pools, this study and others suggest that subsurface soil pools may have some additional capacity for multidecadal SOM stabilization. In Texas shrublands, there is some degree of aggregate protection of soil carbon pools (Liao et al. 2006*a*) that could confer longer term stability to SOM pools (at least in the absence of soil disturbance). Recent work in the Rio Grande Plains of Texas also suggests that woody and root-derived SOM may have increased chemical recalcitrance that may be responsible for some slowing of decomposition rates for this material (Filley et al. 2008). Structural (biochemical) inhibition of decomposition could be one potential reason for the long apparent MRT of the LF fraction in GSENM; however, further work is need to determine whether it is SOM chemistry or environmental conditions that limit the potential for decomposition in desert ecosystems. While these mineral and subsurface pools may provide a longer term sink for carbon, they also appear to play a fairly minor role in the soil C balance in GSENM ecosystems.

### *Grazing impacts on soil C storage*

Grazing, like woody biomass changes, has a highly variable impact on soil carbon stocks. Our results indicate that grazing has had little to no influence on soil carbon storage in these sites, and this limited effect of grazing on soil carbon is in contrast to studies around Canyonlands National Park, where grazing causes large ($\sim$50%) decreases in soil carbon stocks (Neff et al. 2005, Fernandez et al. 2008). A number of the sites near Canyonlands and the GSENM sites examined here occur on Navajo Sandstones and have soils that are similar in texture, carbonate content, and soil nutrient status. Most of the study areas in and near Canyonlands National Park, however, have been in grasslands with relatively little woody cover and an overall grass cover of >30% in contrast to the GSENM study area, where grass cover is ~5%. These two types of settings are typically represented as distinctly different ecological sites and are likely to have had different historical grazing intensity; these differences in historical grazing are one possible cause of variable responses. Sites within Grand Staircase are upland pinyon–juniper ecological sites, whereas sites within Canyonlands are generally Desert Sandy Loam (Indian Ricegrass) sites. Although historical grazing intensity is difficult to establish, these lowland grass-dominated sites likely experienced higher intensity livestock use because the vegetation is more suitable to cattle grazing. An alternative to the historical grazing hypothesis for variable SOM responses is the difference in environmental conditions at these sites. The Canyonlands area sites that are particularly prone to carbon and nitrogen loss are lower elevation sites that fall into the Semi-Desert category in the NRCS ecological site descriptions that generally experience lower annual precipitation (and higher evaporative potential) than the Upland sites. The role of historical grazing intensity vs. climate as a control on ecological responses to land use is an important question given concerns about the sustainability of dryland livestock operations (Brunson and Huntsinger 2008) and ongoing forest management/rangeland restoration activities across the Colorado Plateau.

### *Managing for ecosystem C in the future*

The possibility of carbon storage in changing U.S. ecosystems has prompted increasing discussion of the possibilities of carbon management on public lands (Vine 2004, Meldahl and Kush 2006). Whether or not this is feasible depends to a large degree on the residence time and permanence of sequestered carbon pools. Additionally, any future management of ecosystems for carbon sequestration may come into conflict with existing federal lands policy that seeks to mitigate fire hazards (the healthy forests initiative) or, in the case of the western United States, policies that seek to reestablish rangelands following woody expansion. This study suggests that while there has likely been a significant multidecadal sequestration of C into soils following pinyon–juniper expansion in these ecosystems, the bulk of carbon stabilized in the process is present in short-residence-time pools. Thinning and overstory removal activities in pinyon–juniper ecosystems that seek to reduce hazardous fuel load should also be expected to cause relatively rapid declines in surface soil carbon and nitrogen storage. Similarly, the discussion about whether or not dryland ecosystems in the western United States are responsible for a substantial sink of carbon has centered largely on studies of the total soil carbon pool. Even if these ecosystems are storing substantial carbon in soils, the relatively large amount of carbon present in non-stabilized forms in soils at GSENM, Canyonlands, Rio Grande Plain, and the Jornada deserts (Connin et al. 1997, Neff et al. 2005, Fernandez et al. 2008, Filley et al. 2008) suggests that this carbon will be in forms not likely to be stabilized for more than a few decades.

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# Cheatgrass and Red Brome: History and Biology of Two Invaders

Chad R. Reid, Sherel Goodrich, James E. Bowns

**Abstract**—In recent history, there has not been a more ecologically important event than the introduction of cheatgrass (*Bromus tectorum*) and red brome (*Bromus rubens*) into the Intermountain West. These grasses are very similar in ecology and history and are separated mostly by function of elevation. Both species are from the Mediterranean region, and both arrived in the Western United States about the same time (1880). Cheatgrass and red brome have greatly affected fire frequency and intensity, which has been detrimental to native shrubs and other perennials in these systems. Red brome may have had an even greater impact, in that it has readily invaded non-disturbed areas, has had great impact on fire sensitive shrub species, and, to this point, we have not identified adapted species native or non-native for rehabilitating burned areas. Introduction of cheatgrass and red brome in the West has wreaked ecological havoc on the areas they have invaded and will continue to affect structure, function, and management of these areas well into the future. This paper will detail the history and ecology of these two highly invasive species.

## Introduction

In recent history, there has not been a more ecologically important event than the introduction of cheatgrass (*Bromus tectorum*) and red brome (*Bromus rubens*) into the Intermountain West. This paper details the history and biology of these two highly invasive species.

The reality of these annual grasses is well summed up by Peters and Bunting (1994) with the suggestion that the introduction of exotic annual grasses, including cheatgrass, into the Snake River Plain may have been the most important event in the natural history of that region since the last glacial period. Catastrophic ecosystem change for the western Great Basin has been suggested as a function of cheatgrass by Billings (1994).

These grasses are very similar in biology, ecology, and history and are separated mostly by function of elevation; we will review cheatgrass first and then follow with red brome.

## Cheatgrass

Cheatgrass—also called June grass, bronco grass, downy chess, and downy brome—is a winter annual that was introduced from the Mediterranean region in packing material and first found near Denver, Colorado, (Whitson and others 1991) or perhaps in the eastern coastal states prior to its entry into the West (Monsen 1994).

### Colonization of the West by Cheatgrass

Railways, roads, and contaminated grain seed are indicated as the principal means of initial spread of cheatgrass after which it was advantageous with heavy grazing and other disturbance (Billings 1994). One of the earliest reported collections of cheatgrass in the West was made in Washington by Sandberg and Lieberg in 1883, and a year later it was collected near Provo, Utah, by M. E. Jones (Billings 1994). In the course of extensive field surveys of about 1900, it was not reported for northern Nevada, and the first report for Elko County was in 1906 (Young and others 1987).

By 1946 it occupied at least 10 million acres in eastern Oregon (Monsen 1994). By the 1980s, Kunzler and others (1981) found it among the most abundant species in Gamble oak communities of central and northern Utah, and by the 1990s it was common to dominant over hundreds of thousands of acres in the Great Basin of Utah and Nevada. Roberts (1991) reported 900,000 acres (about 30%) of the Salt Lake District of the Bureau of Land Management was producing cheatgrass. It is present in all counties of Utah, where it is an integral part of the flora (Arnow 1987).

Cheatgrass is now widespread in North and South America in many plant communities of plains, deserts, foothills, and lower montane areas and especially where snow depth and temperatures allow for fall germination and some winter development or at least persistence of seedlings through winter.

It is most abundant in areas of between 6–16 inches of annual precipitation (Monsen 1994). It does not do well on saline soils, but its ability for rapid growth might allow it to make considerable growth from shallow, less saline moisture early in the season (Rasmuson and Anderson 2002).

Cheatgrass is an explosive invader in the Great Basin, Snake River Plain, Columbia Plateau, and other areas of the

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West where it proliferates with fire and other disturbance including roads and associated traffic, off road vehicle use, construction of recreation facilities, and livestock grazing.

It is highly flammable when dry, and with relatively little moisture it produces enough biomass to create continuous fine fuel that leads to high frequency and increasing size of subsequent fire, which perpetuates this plant and excludes many others including sagebrush.

Although some native and introduced grasses compete well with cheatgrass when the grasses are mature, the seedlings of few species can compete with cheatgrass. Francis and Pyke (1996) found that cheatgrass seedlings were superior competitors compared with seedlings of two cultivars of crested wheatgrass. However, they found that increasing densities of Hycrest cultivar of crested wheatgrass reduced cheatgrass biomass and tiller production.

Near complete control of cheatgrass might be necessary before seedlings of some perennial grasses, including desert needlegrass (*Achnatherum speciosum*), can be established (Rafferty and Young 2002).

In number of seeds produced per plant per unit of area, cheatgrass has the capacity to overwhelm native perennials at the seedling level even if the starting density of cheatgrass seeds is low (Young and Allen 1997).

## Management Implications

With dynamic expansion of cheatgrass with disturbance, it is desirable to promptly rehabilitate burned sagebrush and pinyon-juniper communities before cheatgrass has a chance to establish dominance of the site (Evans and Young 1978). In some cases, cheatgrass has been found to make an initial rapid increase and then greatly decline (Barney and Frischknecht 1974; Erdman 1970; Davis and Harper 1990).

A fire return interval of 3–6 years fueled by cheatgrass tends to wear down perennials. Regardless of some perennial plants being able to compete with cheatgrass at one point in time, the ability of cheatgrass to drive ecosystem dynamics over time is a function of high fire frequency as well as its aggressive growth features.

To beat cheatgrass in communities where sprouting perennial species have been depleted often requires prompt seeding of adapted perennials that are able to establish uniform stands with a single seeding. Few species are known to be able to establish stands with a single seeding in areas of less than 10–12 inches annual precipitation. Most of these, such as crested wheatgrass, have been introduced from Eurasia.

In recent years, great emphasis has been placed on seeding only natives after fire or other disturbance to maintain native plant communities. However, unless seedings are successful in keeping cheatgrass from dominating, the goal for natives is not achieved, and cheatgrass drives a departure from native ecosystems that exceeds the departure induced by crested wheatgrass.

Until native plant materials with the capability of competing with cheatgrass in low precipitation areas are available in large quantities, "pick your alien" (annual or perennial) will remain a dilemma for the native only concept. The option to not seed exotic perennials has and will likely continue to favor cheatgrass. Indeed, the concept of pure native communities has become not only problematic, but it is presented with catastrophic challenges by cheatgrass.

A growing body of evidence strongly suggests that cheatgrass presents a potential to turn the pure native concept into romantic fantasy. That body of evidence includes the presence, the abundance, and even dominance of cheatgrass in areas where native plant communities have been protected from disturbance.

Kindschy (1994) reported the presence and increase of cheatgrass in southeastern Oregon's Jordan Crater Research Natural Area that has been protected from human activities including livestock grazing.

On Anaho Island in Nevada, Tausch and others (1994) found cheatgrass has displaced native perennials despite a general absence of human-caused disturbance and fire. They attributed the increase to the competitive ability of cheatgrass.

In Red Canyon of the Green River, cheatgrass has been found as the most frequent species where livestock use and other post European related disturbance have been minimal (Goodrich and Gale 1999).

Young and Clements (1999) reported invasion of cheatgrass into ecologically high condition shadscale/greasewood communities in Nevada despite apparent lack of livestock grazing.

Young and Tipton (1990) cited two works from southeastern Washington that documented observations of cheatgrass successfully inserting itself into climax perennial grass/shrub communities that had been protected from fire and grazing for as long as 50 years. They proposed that the idea of cheatgrass spreading in a biological vacuum created by excessive grazing may be somewhat misleading or overstated.

Young and Allen (1997) have emphasized that site degradation is not necessary for cheatgrass invasion.

In western Utah, Harper and others (1996) found cheatgrass able to establish in ungrazed areas in desert shrub communities where, although native perennials were able to greatly suppress the size of cheatgrass plants, cheatgrass was able to maintain a presence by which it could expand upon disturbance including gopher mounds.

Austin and others (1986) found cheatgrass present in Red Butte Canyon of the Wasatch Mountains where livestock grazing was discontinued in 1905, which was essentially prior to cheatgrass reaching that area. Austin and others (1986) also found cheatgrass in Emigration Canyon of the Wasatch Mountains where livestock grazing was discontinued in 1957. They reported higher cover values for cheatgrass in 1983 than for 1935 in Red Butte Canyon and higher values in Red Butte Canyon than for Emigration Canyon in 1983. These values for cheatgrass are contrary to the concept of cheatgrass only increasing as a function of livestock grazing.

Knight (1994) reported that the cheatgrass problem is not restricted to land grazed by livestock, and he gave an example of an increase of cheatgrass following fire in Little Bighorn Battlefield National Monument in southern Montana. He suggested that managing vegetation of a National Monument so that it reflects presettlement conditions is a goal that may be impossible once certain introduced species become established.

Although some of these examples deal with areas that have been relatively little affected by human activities, nowhere is the climatic zone of cheatgrass in North America wholly isolated from the modern world. The rapidity, volume, and distance of transport of people and goods across the globe by air, water, and ground strongly suggest additional introductions of cheatgrass and other aggressive species from around the world will not only continue but will increase.

That livestock grazing has been a factor in the spread and abundance of cheatgrass is not disputed here. However, the suggestion by Young and Tipton (1990) that this factor has been overstated seems appropriate in that other factors of spread have perhaps received less attention. Disturbance of roadsides and water runoff from roads creates favorable habitat for several weedy species. Highway and off-road vehicles are highly efficient seed catching and dispersing agents. These features would have resulted in the spread of cheatgrass even in the absence of livestock grazing.

The reality of modern life and the aggressive nature of cheatgrass present challenges for managing wildland resources that will not be well addressed by clinging to concepts based on conditions that no longer exist. The world is not what it was prior to European settlement of the Americas. Air traffic, super highways, railways, and roads of high density were not part of the environment prior to 1492. Reality of today includes not only vehicles that travel hundreds of miles in a day with the potential to carry seeds not only across major drainages, but also across oceans.

A highly mechanized and highly mobile human population contributes to a high fire frequency that favors cheatgrass. The competitive nature of cheatgrass will not be reduced by the concept that native communities that are well managed or even untouched will keep it out. In some environments, it has demonstrated that it is a better competitor than native species. Within the ecological range of cheatgrass, basing potential natural plant communities of today on the environment prior to 1492 makes little sense.

Dealing with this force might require seeding some of the most aggressive and less fire prone perennials the world has to offer, regardless of origin. Although this concept is laced with the problem of the cure being worse than the malady where native communities are desired, such a desire is laced with a serious problem or dilemma of its own. The replacement of native ecosystems and their function by cheatgrass driven systems indicates a departure from "native" that exceeds that associated with seeding selected perennials.

Each fire on the Snake River Plain, valleys and foothills of the Great Basin, and other cheatgrass prone areas of the West tightens the grip cheatgrass has on these ecosystems. Opposition to seeding highly competitive perennials in cheatgrass prone areas is indicated to be a demonstration of values that are no longer a potential. Billings (1994) indicated that the potential has changed with the catastrophic ecosystem change induced by cheatgrass as indicated by Billings (1994).

The retort that cheatgrass is a function of past mismanagement of livestock on rangelands will do nothing to improve the condition, and it conveniently ignores the high likelihood that cheatgrass spread and dominance was inevitable with European settlement with or without livestock. Although the early advance of cheatgrass in the West was facilitated by livestock grazing, the ultimate spread of this species is a function of a number of factors, some of which could have advanced cheatgrass in the absence of livestock grazing. The dense network of roads, off-road vehicle use, and frequent use of these by an expanding population would have been the means of spread throughout the potential range of cheatgrass. Fire would have done the rest. It is quite likely that livestock simply set forward or accelerated the inevitable by a few decades.

The long-term trend in numerous crested seedings indicates that these seedings could be managed for the return and maintenance of at least native sagebrush (Huber and Goodrich 1999). In contrast, cheatgrass and its shortened fire cycles excludes sagebrush. Although crested wheatgrass is introduced, it presents an opportunity for greater diversity than does cheatgrass. Where stands of crested wheatgrass are managed for return of sagebrush, it can facilitate development of much greater structural diversity than does cheatgrass.

The use of natives at the present appears problematic; Britton and others (1999) evaluated performance of 24 taxa at a sagebrush site and 20 taxa at a greasewood site. The top performing 9 taxa at the sagebrush site and 10 taxa at the greasewood site were introduced.

Hull (1974) evaluated the performance of 90 plant taxa including many natives in rangelands of southern Idaho. Where annual precipitation was less than 25 cm (10 in), only 17 of the 90 taxa rated over 1 on a relative scale of 1–10. Of these 17 only 6 were natives and none of these natives rated over 2.1. Phases of crested wheatgrass rated from 7.7 to 9.5. Pubescent wheatgrass rated at 6.2 and intermediate wheatgrass rated at 5.1. No other taxon rated over 5.

In general, it seems that the expansion and dominance of cheatgrass has been more dramatic in the inherently grass-poor regions of the sagebrush ecosystem than in the inherently grass-rich regions. As indicated by Tausch and others (1994), the boundary between Wyoming big sagebrush and mountain big sagebrush in western Nevada represents a boundary below which moisture and other conditions favor annual grasses and above which perennial grasses are favored.

However, the Wyoming big sagebrush region of Wyoming compared to that of the Great Basin seems to be relatively rich in native grasses. In this grass-rich region, the invasion of cheatgrass has been comparatively mild. This contrast demonstrates that features other than livestock grazing are important in abundance of cheatgrass. Intensity of livestock grazing on sagebrush areas of Wyoming has probably been equal to, if not greater than, that in the Great Basin.

## Forage Value

Cheatgrass is nutritious when young and palatable to a wide range of ungulates and is highly preferred by mule deer during spring and fall (Austin and others 1994). Bighorn sheep have been observed using cheatgrass on the steep, southerly facing slopes of Red Canyon of the Uinta Mountains in winter when this winter annual is one of the few green herbaceous plants. It is of great economic importance to the domestic livestock industry in some places. Emmerich and others (1993) and DeFlon (1986) reported on range operations where cheatgrass is a major part of the winter and spring forage base. Kufeld (1973) rated it as valuable to elk in winter. This value is likely—at least in part—a function of the southerly exposures or other warm places that are often open in winter where this plant grows best.

On the Mojave Desert, Phillips and others (1996) found that domestic sheep showed high preference for this plant in spring. Cheatgrass seeds and new growth are valuable forage for chuckars (*Alectoris graeca*), Gamble quail (*Lophortyx gambelii*), and mourning doves (*Zenaidura macroura*) (Plummer and others 1968).

Disadvantages of cheatgrass include (1) high fire frequency, which greatly alters forage supplies and maintains rangelands in an annual condition frequently exposed to wind and water erosion; (2) a short green-feed period; (3) great variability of herbage production between moist and dry years; and (4) mechanical damage to mouth parts of grazing animals after drying and hardening of sharp seed-parts (Young and Allen 1997). Also cheatgrass has been associated with reduced nutritive quality of other species (Haferkamp and others 2001). DeFlon (1986) reported winter use of cheatgrass ranges was more favorable to achieving a stable forage base than was spring use. He explained that spring use lead to an increase in halogeton in his study area.

It appears that the consequences of cheatgrass remain little understood. Preservation of native plant communities based on hands-off management, seeding only locally collected native seed to avoid genetic contamination, and rangeland evaluation criteria that ignore cheatgrass in site potential are ideals difficult to implement where cheatgrass is well adapted. However, these and other preservation based ideals often prevail in planning, management, and legal maneuvering dealing with cheatgrass prone rangelands. Yet there seems to be little in the literature dealing with cheatgrass to support ideals of preservation on lands where cheatgrass is highly competitive.

The departure from native ecosystems inflicted by cheatgrass exceeds that of seeding selected, exotic perennial plants and other cultural practices that foster perennial plants that tend to reduce influence of cheatgrass. Ironically, advocates of preservation often support actions that favor cheatgrass over establishment of adapted perennial plants.

## Red Brome

As stated earlier, red brome is very similar to cheatgrass, thus most, if not all of the proceeding discussion on management and ecological implication also applies to red brome. The following is specific to the history and biology of red brome.

### Introduction and History

Red brome, often called an ecological equivalent of cheatgrass, is another Mediterranean winter annual that has invaded disturbed and undisturbed areas of western Northern America, especially the desert southwest. Red brome was brought to North America from the Mediterranean before 1880 (Watson 1880). Three possible scenarios have been proposed for introduction into this area: (1) California Gold Rush and Central Valley Wheat, (2) Southern California Shipping, and (3) Northern California Sheep.

The period of most rapid spread was from 1930 to 1942. The greatest spread into new regions during the past 50 years coincides with “warm” Pacific Decadal Oscillations regimes (El Nino) (Salo 2005). El Nino southern oscillations that result in consecutive years of above-average winter precipitation provides red brome 1st year germination and 2nd year high biomass. Increased $CO_2$ and N deposition also may be contributing to red brome’s success.

The early history of California includes the Mexican Period (1822–1846). It appears unlikely that red brome was introduced from Mexico because red brome was first reported in Mexico in 1931–1932 (Howell 1942), 50 years after its first collection in the United States.

In contrast to accidental introductions, red brome was seeded near the University of Arizona at Tucson from 1906 to 1908 for evaluation as a forage plant; this grass soon escaped and became established along the Santa Cruz River (Thornber 1909)

Red brome was also reported in northern Arizona in 1911 and collected near St. George, Utah, in 1926, where it increased in collections from this area for the next 35 years. In addition, it was also becoming common in waste places and cultivated areas around Las Vegas, Nevada, during this time (Maguire 1935).

Red brome was found throughout northeastern Nevada by the 1940s and continued to spread in designated natural areas of Arizona during this time. By the 1960s, red brome dominated even relatively undisturbed areas of Nevada, Utah, and Arizona and was also reported in New Mexico. Collections of red brome in south-central Utah increased dramatically after the construction of Glen Canyon Dam in 1963.

Red brome now occurs from British Columbia to northwestern Mexico and coastal California to western Texas; continued introductions may have provided new genotypes.

### Biology

The available literature suggests that red brome does not maintain a soil seed bank but exhibits early and uniform germination. In contrast, native annuals depend on soil seed banks, hedging in time; however, at least one researcher disputes this (J.A.Young, personal communication). Red brome has nearly uniform germination under cool, moist conditions typical of this region and can germinate with 0.5 inch of precipitation, Mojave native annuals appear to require twice that amount (Beatley 1966). One author contends this characteristic can lead to population crashes during drought, and winter droughts dramatically reducing red brome densities (Salo 2004). However, Beatley (1974) states that although its numbers vary from season to season where established, red brome has never been observed to miss a growing season.

### Fire Ecology

Red brome may be even more problematic than cheatgrass from a fire standpoint. Low humidity in its range leads to slower decomposition than cheatgrass, increasing fuel loading for a longer period of time. Red brome has been particularly troubling in this aspect because most of the systems it occurs in are not fire adapted, nor have we identified suitable native or non-native species for reseeding into these areas. These characteristics along with its ability to occupy non-disturbed blackbrush (*Coleogyne ramosissima*) sites, the shrub type most susceptible to fire in the region (Beatley 1966), is of great concern.

## Conclusion

The introduction of cheatgrass and red brome in the West has degraded invaded ecosystems and will continue to affect structure, function, and management of these areas well into the future. New land management paradigms will be required to manage these ecosystems.

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# Historical and Modern Disturbance Regimes, Stand Structures, and Landscape Dynamics in Piñon-Juniper Vegetation of the Western U.S.

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# Abstract

Piñon-juniper is one of the major vegetation types in western North America. It covers a huge area, provides many resources and ecosystem services, and is of great management concern. Management of piñon-juniper vegetation has been hindered, especially where ecological restoration is a goal, by inadequate understanding of the variability in historical and modern ecosystem structure and disturbance processes that exists among the many different environmental contexts and floristic combinations of piñon, juniper and associated species. This paper presents a synthesis of what we currently know, and don't know, about historical and modern stand and landscape structure and dynamics in three major and fundamentally different kinds of piñon-juniper vegetation in the western U.S.: persistent woodlands, savannas, and wooded shrublands. It is the product of a workshop that brought together fifteen experts from across the geographical range of piñon-juniper vegetation. The intent of this synthesis is to provide information for managers and policy-makers, and to stimulate researchers to address the most important unanswered questions.

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## Introduction

Piñon-juniper vegetation covers some 40 million ha (100 million acres) in the western U.S., where it provides economic products, ecosystem services, biodiversity, and aesthetic beauty in some of the most scenic landscapes of North America. There are concerns, however, that the ecological dynamics of piñon-juniper woodlands have changed since Euro-American settlement, that stands are growing unnaturally dense, and that woodlands are encroaching into former grasslands and shrublands. Yet surprisingly little research has been conducted on historical conditions and ecological processes in piñon-juniper vegetation, and the research that does exist demonstrates that piñon-juniper structure, composition, and disturbance regimes were very diverse historically as well as today.

Uncertainties about historical stand structures and disturbance regimes in piñon-juniper vegetation create a serious conundrum for land managers and policy-makers who are charged with overseeing the semi-arid landscapes of the West. Vegetation treatments often are justified in part by asserting that a particular treatment (e.g., tree thinning or prescribed burning) will contribute to restoration of historical conditions, i.e., those that prevailed before the changes wrought by Euro-American settlers. However, in the absence of site-specific information about historical disturbance regimes and landscape dynamics, there is danger that well-meaning "restoration" efforts actually may move piñon-juniper ecosystems farther from their historical condition. Some kinds of vegetation treatments may even reorganize ecosystems in such a way that restoration of historical patterns and processes becomes more difficult. Of course, ecological restoration is not the only appropriate goal in land management; but even where the actual goal is wildfire mitigation or forage enhancement, treatments are more likely to be effective if designed with an understanding of the historical ecological dynamics of the system being manipulated (e.g., Swetnam et al. 1999).

The purpose of this paper is to summarize our current understanding of historical stand structures, disturbance regimes, and landscape dynamics in piñon-juniper vegetation throughout the western U.S, and to highlight areas in which significant gaps in our knowledge exist. A separate but similar synthesis is in preparation for New Mexico and Arizona by D. Gori and J. Bate (personal communication). The authors of the geographically more extensive treatment presented in this paper gathered for a workshop in Boulder, Colorado, on August 22-24, 2006, to develop the information presented here. All have conducted research in piñon-juniper vegetation, and together they have experience with a wide diversity of piñon-juniper ecosystems, from New Mexico and Colorado to Nevada and Oregon.

The paper is organized in five parts. In Section I we present a brief overview of the variability in dominant species, climate, stand structure, and potential fire behavior of piñon-juniper vegetation across the West, to emphasize one of our key points---that this is a diverse vegetation type, for which a single model of historical structure and dynamics is inadequate, especially considering the magnitude of past and current management interventions. In Section IIa - IIc we summarize what we know about past and present conditions in piñon-juniper ecosystems in the form of a series of concise statements followed by more detailed explanations of each statement. The explanations include the level of confidence that we have in the statement, the kind(s) of evidence that support the statement, and the generality of the statement, i.e., whether it applies to all piñon-juniper ecosystems or only to a subset of these ecosystems (see next paragraph). By "past conditions" we mean the three to four centuries prior to the sweeping changes introduced by Euro-American settlers in the mid to late 1800s. In Section III we evaluate the possible mechanisms driving one of the most conspicuous features of piñon-juniper vegetation in many areas--the increase in tree density that has been observed during the past

100-150 years. We distinguish two somewhat different processes leading to higher tree density: (i) "*infill*" or increasing tree density within existing woodlands that were previously of lower density; and (ii) "*expansion*," i.e., establishment of trees in places that were formerly non-woodland (e.g., grassland or shrubland). In Section IV we suggest some general management implications that may follow from our understanding of piñon-juniper disturbance ecology, and in Section V we identify some key research needs.

Statements of HIGH CONFIDENCE generally are supported by some combination of (i) *rigorous paleoecological studies* that include adequate sampling and appropriate analysis of, e.g., cross-dated fire-scars, tree age structures, and macrofossils; (ii) *experimental tests of mechanisms* that incorporate adequate replication and appropriate scope of inference; or (iii) *systematic observations of recent wildfires, prescribed fires, or other disturbances (e.g., insect outbreaks)*, either planned before the event and documented by experienced, objective observers, or based on rigorous post-disturbance analyses using adequate and spatially explicit data. Statements of MODERATE CONFIDENCE generally are supported by (i) *correlative studies* that identify statistically significant associations between two variables but do not prove a cause-effect relationship; (ii) *anecdotal observations of recent fires,* i.e., opportunistic observations of wildfires or prescribed fires by experienced, objective observers, but not conducted in a systematic manner; or (iii) *logical inference,* i.e., deductive inferences from related empirical or experimental studies that are logical but not yet tested empirically. Depending on the details, other kinds of evidence may support either HIGH or MODERATE confidence: (i) *comparison of historic and recent photos of the same scene,* which documents changes in pattern or structure, but says little about the mechanism(s) causing the changes; or (ii) *written historical documentation* in the form of reports, articles, letters, and other accounts by reliable observers.

We intentionally refrain from making specific policy or management recommendations in this paper. Instead we provide the consensus among researchers of what we know (and don't know) about the science, and then highlight some of the broad conceptual implications of the science for framing policy and management decisions. We recommend that land managers, practitioners, and policy-makers rely primarily on the statements of broad applicability and high confidence in formulating management plans and priorities, and that researchers conduct new studies to critically test the statements of moderate confidence and generality. We also emphasize the importance of locally evaluating the kind(s) of piñon-juniper woodland being dealt with in any specific management situation, as well as incorporating social, economic, and political dimensions of management.

## Section I. Piñon-Juniper: A Diverse and Variable Vegetation Type

Woodlands dominated by various combinations of piñon and juniper species represent some of the most extensive and diverse vegetation types in western North America. For example, the Southwestern Regional GAP land cover maps (http://earth.gis.usu.edu/swgap/) show ca. 15% of the land area in New Mexico, Arizona, Colorado, Utah, and Nevada covered by vegetation of this kind. NatureServe, an international database of species and communities (http://www.natureserve.org/explorer/servlet/NatureServe?init=Ecol) lists 77 plant associations in the west in which a piñon is the dominant species (with or without junipers), and 71 associations in which junipers dominate (typically without piñon, or with piñon as a minor component). Piñon and juniper associations are found in almost every western state of the U.S., from California, Oregon, and Washington to North and South Dakota, Nebraska, Oklahoma, and Texas. Piñon and juniper associations also are widespread in Mexico, and juniper species extend north into

Canada and east to Virginia. Although the catch-all term "piñon-juniper" is typically applied to all of this diverse vegetation, it is important to note that one finds pure stands of juniper (very commonly) and of piñon (less commonly) as well as mixed stands.

This paper focuses primarily on piñon and juniper vegetation in the Intermountain West, the Southwest, the Southern Rocky Mountains, and the western edge of the Great Plains, including primarily the states of New Mexico, Arizona, Colorado, Utah, Nevada, and Oregon. Throughout this extensive region, woodlands of piñon and/or juniper are found on almost all landforms, including ridges, hill and mountain slopes, terraces, tablelands, alluvial fans, broad basins, and valley floors. Soils are similarly variable, ranging from relatively deep soils often high in clay or sand content, to shallow rocky soils, to rock outcrops where no soil is present but the trees are rooted in deep cracks of the bedrock. Woodlands of piñon and/or juniper occupy a broad zone of intermediate moisture and temperature conditions between the hot arid deserts of lower elevations and the cool mesic forests of higher elevations. Accordingly, soil temperature regimes range from mesic to frigid (e.g., Driscoll 1964, Miller et al. 2005).

There is a striking northwest-to-southeast gradient in the seasonality of precipitation: winter-spring precipitation predominates in the northwest, notably in the Great Basin, gradually shifting to a monsoonal summer pattern in the southeastern portion of the region including southern Arizona and New Mexico (Mitchell 1976, Jacobs in press). Total precipitation across most of the range of *Juniperus occidentalis* in the northwestern Great Basin varies between 25 and 40cm annually, falling mostly during winter storms, although this tree species can grow in areas receiving as little as 18cm (usually on sandy soils) or exceeding 50cm (Gedney et al. 1999). Annual precipitation amounts are similar where *J. monosperma* grows in south-central New Mexico, but in this latter region 60% or more falls between April and September, particularly during the late summer "monsoon." The Colorado Plateau (especially the southern portion), lying near the midpoint of this gradient, receives small peaks of precipitation in both winter and summer (http://www.cpluhna.nau.edu/Change/modern_climatic_conditions.htm).

Species composition and vegetation structure vary along the same northwest-to-southeast gradient. *Juniperus occidentalis* is the major woodland tree species in extreme northwestern Nevada, northeastern California, and eastern Oregon; *Pinus monophylla* and *Juniperus osteosperma* dominate woodlands elsewhere in the Great Basin; *Pinus edulis* and *Juniperus osteosperma* are the dominant woodland species across most of the Colorado Plateau and southern Rocky Mountains west of the Continental Divide; and *Pinus edulis* and *Juniperus monosperma* characterize the summer monsoon regions of New Mexico, east-central Arizona, and the southern Rockies east of the Continental Divide. Two other junipers also are common at higher elevations--*J. scopulorum* in much of the Colorado Plateau and southern Rockies, and *J. deppeana* in southern New Mexico and Arizona. In the western and northern regions, where precipitation is winter-dominated, the trees are typically associated with a major shrub component, notably big sagebrush (*Artemisia tridentata*) and other *Artemisia* spp., *Purshia tridentata*, *Chrysothamnus* spp., *Ericameria* spp., and *Cercocarpus* spp. Cool and warm season perennial tussock grasses also may be common associates, e.g., *Festuca idahensis, Pseudorogneria spicata, Achnatherum spp., Poa secunda,* and *P. fendleriana.* In eastern and southern regions, where the precipitation pattern is summer-dominated, piñon and/or juniper woodlands often support an understory of warm-season grasses, e.g., *Bouteloua gracilis*, *B. curtipendula*, *B. hirsuta B. eriopoda, Muhlenbergia pauciflora*, and *M. setifolia*, and woodlands may occur as patches within a grassland matrix. A diverse and highly variable mix of montane shrubs and chaparral species (e.g., *Quercus gambelii, Q. pauciloba,* and other *Quercus* spp*., Cercocarpus montanus, Amelanchier utahensis,* and *Purshia tridentata*) is an important component of piñon-juniper vegetation at higher elevations, notably in the Southern Rockies and Colorado Plateau.

Three General Kinds of Piñon-Juniper Vegetation: For the purposes of this paper, we identify three fundamentally different kinds of piñon-juniper vegetation, based primarily on canopy structure, understory characteristics, and historical disturbance regimes. The three kinds--persistent piñon-juniper woodlands, piñon-juniper savannas, and wooded shrublands--are summarized in Table 1, and their general structure and distribution in relation to seasonality of precipitation is depicted in Figure 1. There is great diversity within each of these general types, but this classification represents much of the variability in piñon-juniper vegetation across the western U.S. Research is underway to link these vegetation types to specific environmental characteristics that would allow for reliable prediction and mapping across large landscapes and regions, but at present we can identify only some very general environmental correlates. Because historical stand structures, disturbance regimes, and landscape dynamics were significantly different among these three basic types of piñon-juniper vegetation, we address each type separately in the summaries below.

Potential Fire Behavior: In all three kinds of piñon-juniper vegetation (Table 1), there are important interactions among canopy fuel structure, understory fuel structure, and fire weather conditions. **Continuity of canopy aerial fuels** is key in determining crown fire behavior, especially in woodlands where understory shrubs are relatively sparse, and is influenced most directly by total tree stem density, crown width, and crown fullness (often related to age). Understory vegetation also provides continuity among tree stems and ladder fuels, especially where tall shrubs are present. In wooded shrublands (Table 1), notably where *Artemisia tridentata* is the dominant shrub species, the shrub stratum may be more important than the trees in carrying fire, especially if the trees are widely spaced. Also fundamental to fire behavior is **total surface fuel loading**, influenced most directly by total biomass of the trees, shrubs, and other understory vegetation. A dense tree canopy may suppress the cover and biomass of shrubs and herbaceous plants, though some productive sites support both dense canopy and understory. Piñon and juniper also are able to become established and persist in very dry sites, with widely spaced trees and very little understory. These often-complex arrangements of overstory and understory factors form a matrix of likely fire behavior during a single fire event under modal (e.g., 80th percentile) and extreme (e.g., 95th percentile) fire weather conditions across the three basic piñon-juniper types, as summarized in Figure 2.

Actual fire weather is critical in most combinations of tree, shrub, and understory cover types; weather conditions determine the amount of tree mortality and the dynamics of fire spread both within a stand and across a landscape (Figure 2). However, stands with scattered trees among sparse understories of low shrubs and herbs almost always exhibit limited fire activity, given the lack of fuel, and the trees growing in such a stand are relatively protected from fire. Conversely, dense woodland conditions become highly flammable with time (i.e., fuel accumulation over decades or centuries) regardless of fine fuel conditions; the probability of ignition and duration of the fire season define the actual fire return intervals for these ecosystems in which fire is typically stand-replacing. It is also critical to recognize a difference between passive crown fires (torching of individual trees) versus active crown fires (running through the crowns of the trees) in piñon-juniper systems, which ties in both the overstory and understory fuel arrangements as well as extreme versus modal fire weather. If overstory and understory densities are relatively low, as in many very dry or rocky sites, even under the most extreme

Table 1. . Classification of piñon and juniper vegetation as treated in this paper. See Figure 3 for photos of each type.

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***(1) Persistent Piñon-Juniper*** *Woodlands* are found where site conditions (soils and climate) and disturbance regimes are inherently favorable for piñon and/or juniper, and where trees are a major component of the vegetation unless recently disturbed by fire, clearing, or other severe disturbance. Canopy structure varies considerably, from sparse stands of scattered small trees growing on poor substrates to relatively dense stands of large trees on relatively productive sites. Either piñon or juniper may dominate the canopy, or the two may co-dominate. The understory may be dominated by shrubs or forbs or less commonly by graminoids; a consistent feature of the understory is low total plant cover with frequent patches of bare soil or rock. Notably, these woodlands do *not* represent twentieth century conversion of formerly non-woodland vegetation types to woodland, but are places where trees have been an important stand component for at least the past several hundred years.

Persistent woodlands are commonly found on rugged upland sites with shallow, coarse-textured soils that support relatively sparse herbaceous cover even in the absence of heavy livestock grazing. However, they also occur in a variety of other settings, and their precise spatial distribution and bio-climatic context have not been characterized. Nevertheless, this type of piñon-juniper vegetation is found throughout the West. It appears to be especially prevalent on portions of the Colorado Plateau, where precipitation is bimodal with small peaks in winter and summer. Indeed, large expanses of the Colorado Plateau are characterized by ancient, persistent woodlands within spectacular canyon and plateau landscapes.

***(2) Piñon-Juniper Savannas*** are characterized by a low to moderate density and cover of trees within a matrix of a well-developed and nearly continuous grass or graminoid cover; shrubs may be present but usually are relatively unimportant. Either piñon or juniper may dominate the canopy, or the two may co-dominate. In places the density of trees may be enough to represent an open woodland rather than a savanna per se; nevertheless, the key feature of the piñon-juniper savanna is the relatively continuous grass cover in the understory.

Piñon-juniper savannas typically are found on moderately deep, coarse to fine-textured soils on gentle upland and transitional valley locations in regions where a large proportion of annual precipitation comes during the growing season. Soils and climate readily support a variety of plant growth forms including grasses and trees. This type of piñon-juniper vegetation appears to be especially prevalent in the basins and foothills of central and southern New Mexico and Arizona, where the precipitation pattern is dominated by the summer monsoon. Piñon-juniper savannas are relatively rare in the Southern Rocky Mountains, northern Colorado Plateau, and Great Basin, where precipitation has a stronger winter component.

**(3)** ***Wooded Shrublands*** are characterized by a dominant shrub stratum with a variable tree component that may range from very sparse to relatively dense. The tree component may be either piñon or juniper or both. Herbaceous cover also varies greatly, depending on local site conditions and history. The shrubs constitute the fundamental biotic community in these ecosystems; tree density naturally waxes and wanes over time in response to climatic fluctuation and disturbance (notably by fire and insects). Thus, these are areas of potential expansion and contraction of woodland within a shrub-dominated matrix (Romme et al. 2007).

Wooded shrublands are associated with a wide variety of substrates and topographic settings, from shallow rocky soils on mountain slopes to deep soils of inter-montane valleys. Wooded shrublands are often located in proximity to a persistent tree seed source on sites where competition from grasses and shrubs, drought, and periodic disturbance by fire, insects, and disease limit the development of mature trees or stands over time. Wooded shrublands appear to be especially prevalent in the Great Basin, where the precipitation pattern is winter-dominated, although they are found throughout the West.

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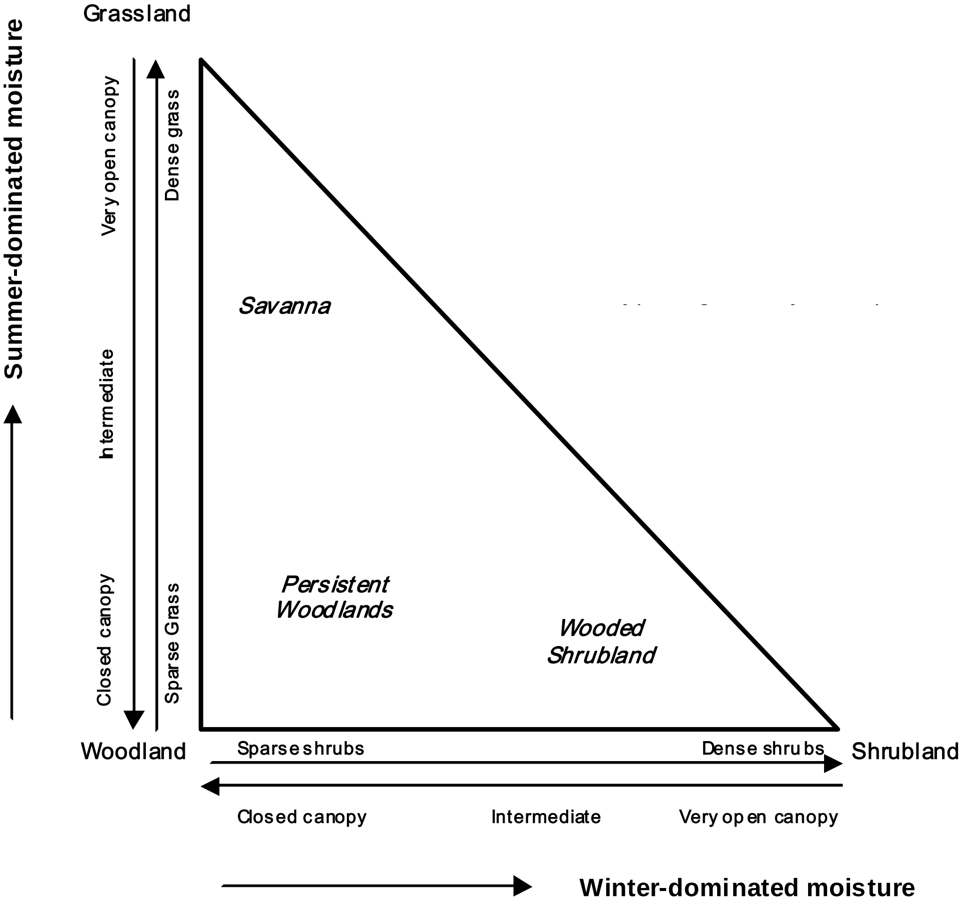

Esteban Muldavin/Craig Allen

Figure 1. Generalized structure, i.e., relative proportions of trees, shrubs, and grass, and broad patterns of regional distribution in relation to gradients in seasonality of precipitation, in the three types of piñon and juniper vegetation discussed in this paper (Table 1). Note that local site conditions may support any of the three types even in regions where one type is generally more prevalent.

weather conditions there simply may not be enough fuel for either active or passive crown fires to occur; the fire may simply go out before traveling through a stand (Figure 2).

## Section IIa: What We Know About Persistent Piñon-Juniper Woodlands

We define "persistent woodlands" as those found where site conditions (soils and climate) and disturbance regimes are inherently favorable for piñon and juniper (Table 1). Our group agreed on eight key ideas about persistent woodlands.

1. *Some persistent woodlands are stable for hundreds of years without fire, other than isolated lightning ignitions that burn only single trees or small patches and produce no significant change in stand structure. Many woodlands today show no evidence of past widespread fire, though they may have burned extensively in the very remote past (many hundreds or thousands of years ago).*

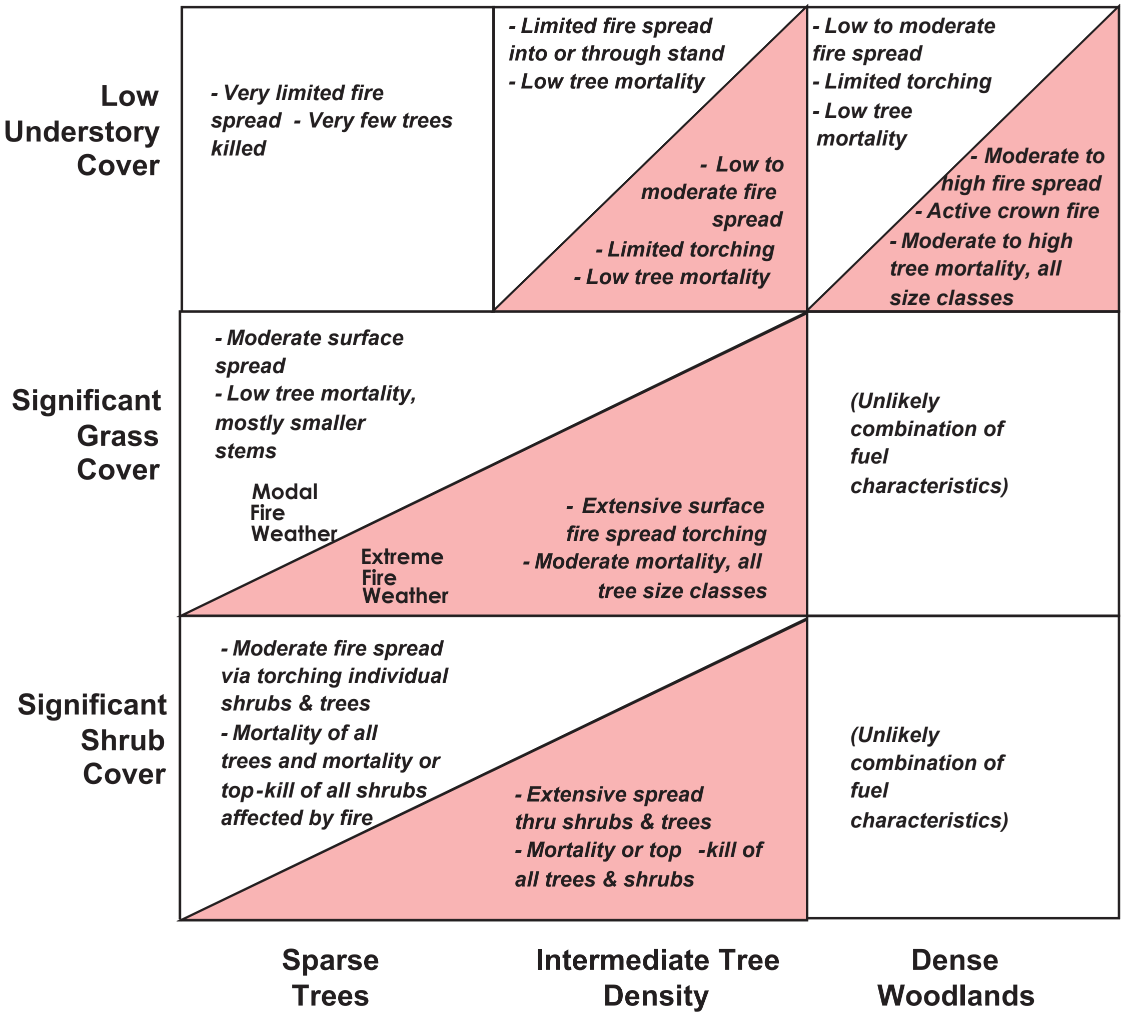

Figure 2. Probable fire behavior following a single ignition event in piñon and juniper vegetation with respect to variability in tree density (horizontal axis) and understory fuel characteristics (vertical axis). Split cells reflect variable fire behavior, spread dynamics, and tree mortality under "modal" (80th percentile) versus "extreme" (95th percentile) fire weather conditions.

* HIGH CONFIDENCE … BUT PRECISE GEOGRAPHIC APPLICABILITY NOT ADEQUATELY KNOWN

Kinds of Evidence: *rigorous paleoecological studies, presence of old trees and snags but no evidence of past extensive fire such as charred tree stems or extensive charcoal in soils*

Explanation: Many piñon and juniper woodlands exhibit little to no evidence that they ever sustained widespread fires during the period that trees have been alive in the stand. Living trees in these stands are often very old (300 to 1000 years) and exhibit multi-aged structure, with tree establishment often clumped but episodic within stands (e.g., Waichler et al. 2001; Eisenhart 2004; Floyd et al. 2004, 2008; Shinneman 2006). It is difficult to accurately gauge the time since the last major disturbance in such stands from living trees alone, because they typically contain even older logs or snags that overlap time spans of the living trees (i.e., they were not killed in a past stand-opening event). Charred snags and logs are either absent or extremely sparse. There may be individual charred boles or small patches of charred boles which apparently represent lightning ignitions in the past that failed to spread, but no extensive or continuous evidence of past fire.

Such woodlands are often located on rocky or unproductive sites with widely scattered trees, where understories are mainly bare ground with sparse vegetative cover (Figure 2). However, they also include some higher-density woodlands growing on more productive sites, and they may cover extremely large portions of some areas, such as the mesas, plateaus, and bajadas in southern Utah, western Colorado, northern Arizona, and northwestern New Mexico. Examples of locations where tree-ring data document old trees and a lack of widespread fire include pumice-sandy soils in central Oregon (Waichler et al. 2001); near the northeastern edge of the Uinta Range in Utah (Gray et al. 2006); the Tavaputs Plateau and several of the bajada communities on the fringes of southern Utah mountain ranges (E. K. Heyerdahl, P. M. Brown, and S. T. Kitchen, unpublished data); the Kaiparowits Plateau in Utah (Floyd et al. 2008); Mesa Verde, the Uncompahgre Plateau, and Black Canyon of the Gunnison in western Colorado (Eisenhart 2004, Floyd et al. 2004, Shinneman 2006); and the margins of the Chihuahuan Desert in central and southern New Mexico (Swetnam and Betancourt 1998 and unpublished data; Muldavin et al. 2003 and unpublished data). Persistent woodlands of this kind are especially prevalent in portions of the Colorado Plateau and Great Basin. They also probably occur throughout the range of piñon and juniper vegetation, although they may be less common in regions having monsoon-dominated precipitation patterns such as southern New Mexico (Fuchs 2002 and personal communication).

It is possible that some of these stands could burn with larger patches of passive or active crown fire during extreme weather conditions, especially if understory density increased following prior wet years (Figure 2). However, in most such stands, other disturbances appear to be more important than fire in determining long-term structure and dynamics (see statement # 2 below).

2. *In some persistent woodlands, stand dynamics are driven more by climatic fluctuation, insects, and disease than by fire. For example, a widespread piñon mortality event occurred recently in the Four Corners region as a result of drought, high temperatures, and bark beetle outbreaks*.

* HIGH CONFIDENCE … BUT PRECISE GEOGRAPHIC APPLICABILITY NOT ADEQUATELY KNOWN

Kinds of Evidence: *rigorous paleoecological studies, recent systematic observations of tree mortality*

Explanation: Scientists and managers traditionally have placed greater emphasis on wildfire as a shaper of piñon-juniper woodland ecosystems than other types of natural disturbance. Increasingly, however, there is awareness that dynamics in many piñon-juniper woodlands are driven more by drought stress and its accompanying suite of diseases, insects, and parasites than by fire. Stand dynamics in persistent woodlands may be punctuated by episodic mortality or recruitment events that occur in response to extreme weather patterns (Betancourt et al. 1993, Swetnam and Betancourt 1998, Breshears et al. 2005). Indeed, studies of old woodlands often reveal an accumulation of coarse wood in the understory from trees that were killed by agents other than fire and have persisted due to the absence of fire (Betancourt et al. 1993, Waichler et al. 2001, Floyd et al. 2003, Eisenhart 2004).

Observations clearly indicate that drought stress is capable of altering woodland structures from landscape to regional scales. An example of episodic mortality related to extreme weather would be the recent impacts to southwestern woodlands caused by drought and warm temperatures (Breshears et al. 2005, Shaw et al. 2005, Mueller et al. 2005). Extensive mortality of *Pinus edulis* in the Four Corners region since 2000 has shifted canopy

Figure 3a. **Persistent woodland**, growing on a moderately productive site with a high percent canopy cover and sparse herbaceous understory. The canopy is composed of *Pinus edulis* and *Juniperus osteosperma*; the major understory shrub is *Artemisia tridentata*. Trees are of all ages, including many individuals >300 years old, and the stand contains no surface evidence of past fire. Navajo Point, Glen Canyon National Recreation Area, Utah, elevation ca. 2,100 m. Photo by W.H. Romme, 2005.

dominance of some stands from piñon to juniper (Mueller et al. 2005). Additionally, mortality data suggest that trees of cone-bearing age were more likely to die (Mueller et al. 2005; Selby 2005; M.L. Floyd et al., unpublished data; C.D. Allen, unpublished data) which likely will influence the trajectory of recovery for decades (note, however, that abundant piñon reproduction is now present in at least some affected stands; B. Jacobs, unpublished data).

Climate reconstructed from tree-rings throughout the Southwest suggests that the current drought is not unprecedented, and that droughts of a similar or greater magnitude have occurred many times in the past (Betancourt et al. 1993, Ni et al. 2002, Gray et al. 2003). For example, widespread tree mortality during a very severe “mega-drought” in the late 1500s may explain the rarity of piñon older than 400 years in the Southwest (Swetnam and Brown 1992, Betancourt et al. 1993, Swetnam and Betancourt 1998). Studies in the Southwest also demonstrate that recovery from drought may occur as a pulse of tree establishment and recruitment during the first wet period that follows the drought (Swetnam et al. 1999, Shinneman 2006). In some areas, in fact, recovery since the late 1500s “mega-drought” may be responsible for recent and ongoing increases in tree density (see Section III below).

Figure 3b. **Piñon-juniper savanna**, growing in relatively deep soils on gentle terrain, in a region where the precipitation pattern is summer-dominated. Trees are predominantly *Juniperus monosperma* with occasional *Pinus edulis*. Most trees are <150 years old, but there are at least three older trees per hectare. Blue grama (*Bouteloua gracilis*) is the dominant grass; cholla cactus (*Opuntia imbricata*) is also present. With a well-developed herbaceous stratum within a relatively productive environment, low tree density at this site may have been maintained historically by periodic fire. However, fire history studies have not been conducted in this area to confirm or reject this hypothesis. Near Mountainair, New Mexico, elevation ca. 1,950 m. Photo by W.H. Romme, 2006.

3. *Spreading, low-intensity surface fires had a very limited role in molding stand structure and dynamics of persistent piñon-juniper woodlands in the historical landscape. Instead, the dominant fire effect was to kill most or all trees and to top-kill most or all shrubs within the burned area. This statement also is true of most ecologically significant fires today.*

* HIGH CONFIDENCE … APPLIES TO PERSISTENT WOODLANDS THROUGHOUT THE WEST

Kinds of Evidence: *rigorous paleoecological studies, systematic observations of recent fires*

Explanation: Spreading, low-intensity surface fires (as opposed to stand-replacing fires) have been observed only rarely in piñon-juniper vegetation during the recent period since Euro-American settlement (Baker and Shinneman 2004). Apparently, such fire behavior also was rare in persistent woodlands prior to Euro-American settlement. Definitive fire-history evidence of a spreading low-intensity surface fire would include cross-dated fire scars at two or more locations along with intervening age-structure evidence that trees generally survived the fire (Baker and Shinneman 2004). However, few places provide such evidence. On the contrary, fire scars are conspicuously absent or rare in the great majority of piñon-juniper stands.

One possible example of historical low-severity spreading fire in piñon-juniper comes from northern New Mexico, at the upper ecotone between piñon-juniper and ponderosa pine forest, where two studies with cross-dated scars documented 10-13 spreading fires over a ca. 250-year period (Allen 1989, Morino et al. 1998). Evidence about tree survival between the fire-scarred trees was not collected, however, so it is not clear whether the fire actually burned the entire area, or spread primarily through ponderosa pine stringers and around the islands of piñon-juniper that may have lacked sufficient fine fuels to support low-intensity surface fires. Fire scars also were found at the ecotone between an open ponderosa pine forest and a piñon-juniper woodland in southern New Mexico (Muldavin et a. 2003); again, however, tree age data were not sufficient to confidently reconstruct the spatial patterns of fire spread within the piñon-juniper woodland.

A major problem in assessing the historical role (or lack of a role) of low-severity surface fire in piñon and juniper woodlands is that we do not know how often the trees scar when surface fire burns in their vicinity; this issue is addressed more fully below in Section V on research priorities. Nevertheless, available evidence indicates that low-severity fires generally were absent in persistent piñon-juniper woodlands, and if they did occur, they were likely patchy and of small extent (Baker and Shinneman 2004).

In contrast to the above, there is abundant evidence that fires in persistent woodlands since Euro-American settlement have been predominantly high severity, commonly killing all the trees and top-killing the shrubs and herbs within a fire perimeter, but often leaving some unburned islands of woodland (Baker and Shinneman 2004). Fire-history studies and historical evidence also document high-severity fires in multiple locations around the West during the pre-EuroAmerican era (Eisenhart 2004; Floyd et al. 2004, 2008; Bauer 2006; Shinneman 2006). Limited evidence suggests that fires occasionally could have been variable in severity, resulting in some low-severity areas on the margins of large high-severity fires or in small islands not burned at high severity (Baker and Shinneman 2004). Nevertheless, high-severity fire was likely the dominant type of fire in these woodlands in both historical and modern eras. However, fire extent and spatial patterns (especially patch size distributions of high severity fire) in pre-modern landscapes are not well known.

4. *Historical fires in persistent piñon-juniper woodlands generally did not "thin from below," i.e., they did not kill predominantly small trees. Instead, they tended to kill all or most of the trees within the places that burned regardless of tree size. This statement also is true of most fires today.*

* HIGH CONFIDENCE … APPLIES TO PERSISTENT WOODLANDS THROUGHOUT THE WEST

Kinds of Evidence: *rigorous paleoecological studies, systematic observations of recent fires*

Explanation: Almost all piñons and junipers are relatively fire-intolerant, because often they have thin bark and typically have low crowns. Unlike ponderosa pine, which self-prunes lower branches and develops thick bark with age, piñons and most juniper species are usually killed by fire even when mature. (We note, however, that older piñons can have bark >2 cm in thickness, and it is unknown how these trees may have responded to historical surface fires if they occurred. Mature *Juniperus deppeana* trees also can survive fire, and they commonly re-sprout if top-killed by fire.) The extent and spatial pattern of fire varies in time and space, from very small (<0.1 ha) and fine-grained to very large and coarse-grained (hundreds to thousands of ha), as a function of fuel structure and fire weather (Figure 2). Nevertheless, the dominant effect observed in recent fires in piñon-juniper vegetation has been complete or nearly complete tree mortality throughout the

Figure 3c. **Wooded shrubland,** composed of western juniper trees (*Juniperus occidentalis*) growing in a low sagebrush (*Artemisia arbuscula*) - Sandberg bluegrass (*Poa sandergii* ) community. Soils are shallow (15-30 cm) clay to clay loams overlying fractured basalt, which allows the tree roots to penetrate below the soil surface. The majority of trees sampled on this site exceeded 200 years, some approaching 800 years. Modoc Plateau in northeastern California, elevation 1,550 m. Photo by R.F. Miller, 1998.

area burned, and the effect was likely similar in historical fires.

5. *Historical fire rotations (i.e., the time required for the cumulative area burned to equal the size of the entire area of interest), and fire intervals at the stand level, varied from place to place in persistent piñon-juniper woodlands, but generally were very long (usually measured in centuries).*

* HIGH CONFIDENCE … APPLIES TO PERSISTENT WOODLANDS THROUGHOUT THE WEST

Kinds of Evidence: *rigorous paleoecological studies*

Explanation: We have few estimates of historical fire rotation for piñon-juniper woodlands based on adequate empirical data, but available studies report very long rotations. Examples include 410 or 427 years (depending on method of calculation) in Barrett Canyon in central Nevada (Bauer 2006), 480 years in southern California (Wangler and Minnich 1996), 400 - 600 years on the Uncompahgre Plateau in western Colorado (Shinneman 2006), and 400+ years on Mesa Verde in southwestern Colorado and on the Kaiparowits Plateau in southern Utah (Floyd et al. 2004, 2008). Note that “fire rotation” is a different concept and metric than “mean composite fire interval.” Because the latter metric may be influenced strongly by sampling intensity and scale (Hardy 2005, Reed 2006), we emphasize here the fire rotation concept, which is roughly equivalent to the average fire interval at a small point on the ground. We do not emphasize the absolute values that have been estimated for persistent piñon-juniper woodlands; rather we point out that historical fire rotations and point-intervals were much longer than is often assumed for piñon or juniper vegetation in general (e.g., Schmidt et al. 2002). We also note that modern fire intervals may be getting shorter, as explained in #6 below.

6. *Recent large, severe (stand-replacing) fires in persistent piñon-juniper woodlands are normal kinds of fires, for the most part, because similar fires occurred historically. However, the frequency and size of severe fires appears to have increased throughout much of the West since the mid-1980s, in piñon-juniper and also in other vegetation types. The causes of this recent increase in large piñon-juniper fires are uncertain, and it is unclear whether the very large sizes of some recent fires are exceptional or represent infrequent but nevertheless natural events.*

* MODERATE CONFIDENCE … APPLIES TO MOST PERSISTENT WOODLANDS THROUGHOUT THE WEST

Kinds of Evidence: *rigorous paleoecological studies, correlative studies, logical inference*

Explanation: Ages of live trees and charred juniper snags in piñon-juniper woodlands document the occurrence of large fires (at least hundreds of hectares in extent) in the 1700s on Mesa Verde in western Colorado and in the 1700s or 1800s on the Kaiparowits Plateau of southern Utah (Floyd et al. 2004, 2008). In central New Mexico, an extensive shrubland patch embedded within piñon-juniper woodlands of the Oscura Mountains is suggestive of a high-severity fire in the 1800s, though the tree-ring studies needed to confirm this hypothesis have not yet been conducted (Muldavin et al. 2003). Thus, we know that large severe fires occurred in piñon-juniper woodlands in the past, though we have little information on extents or spatial patterns of those fires.

An upsurge of large fires (>400 ha) in forested landscapes began in the mid-1980s in the western U.S.(Westerling et al. 2006). Increasing trends in large fire frequency and total area burned are particularly noticeable in some regions having extensive piñon-juniper woodlands (e.g., the Southwest and northern Great Basin). For example, a greater proportion of the piñon-juniper woodland on Mesa Verde has burned in the past decade than burned throughout the previous 200 years (Floyd et al. 2004).

Changes in fuel structure probably have contributed to the recent increase in large fires in some parts of the West. For example, fire exclusion in some ponderosa pine and dry mixed conifer forests has allowed fuel mass and vertical continuity to increase (Allen et al. 2002, Hessburg and Agee 2003), although recovery from nineteenth-century fires, logging, and livestock grazing, rather than fire exclusion, are likely the principal mechanisms of this change in other ponderosa pine forests (Baker et al. 2007). Invasion by highly flammable annual grasses (e.g., cheatgrass, *Bromus tectorum*) has increased horizontal fuel continuity and likelihood of extensive fire spread in many semi-arid vegetation types, including piñon-juniper woodlands and shrublands of the Great Basin and Colorado Plateau (Whisenant 1990).

However, large fire frequency also has increased in other forest types where changes in fuel conditions are probably far less important, e.g., in high-elevation forests of the northern Rocky Mountains (Schoennagel et al. 2004), leading Westerling et al. (2006) to suggest that an equal or more important mechanism may involve the warmer temperatures, longer fire seasons, and high amplitude of wet/dry years in recent decades. A similar increase in the frequency of large fires also has been documented in portions of Canada where changes in forest conditions due to land use are minimal, again suggesting a primary climatic mechanism (Gillet et al. 2004, Girardin et al. 2007). It should be noted that although increases in numbers of large fires and area burned are striking in some regions and in broad composite data from the western US and Canada, some sub-regions show little or no clear evidence of major changes in fire activity in recent decades (Westerling et al. 2006).

Given the very long fire rotations that naturally characterize persistent piñon-juniper woodlands (see statement #5), we cannot yet determine whether the recent increase in frequency of large fires occurring in this vegetation type represents genuine directional change related to changing climate or fuel conditions, or is simply a temporary episode of increased fire activity, comparable to similar episodes in the past. In any event, the suite of current and upcoming broad-scale environmental changes--warming temperatures, increasing tree densities (see statement #7), and expansion of fire-promoting species such as cheatgrass—all may all interact to dramatically increase the amount of burning in piñon-juniper and other vegetation types over the next century. See Section IV below on management implications for more on this idea.

7. *Tree density and canopy coverage have increased substantially during the twentieth century in some persistent woodlands, but not in all*.

* HIGH CONFIDENCE … BUT PRECISE MAGNITUDE OF INCREASE, CAUSES, AND GEOGRAPHIC APPLICABILITY NOT ADEQUATELY KNOWN

Kinds of Evidence: *rigorous paleoecological studies, historic & recent photos*
Explanation: From the late nineteenth through the twentieth century, tree abundance and/or size increased in many, though not all, persistent woodlands, as evidenced by repeat aerial photography (e.g., Manier et al. 2005) or tree-ring reconstructions of age structure (e.g., Eisenhart 2004, Floyd et al. 2004, Landis and Bailey 2005, Shinneman 2006, Miller et al. 2008). It should be noted that visual and re-photographic sources have limited ability to distinguish among changes in tree density, tree size, and canopy cover. For instance, re-sampling of permanent plots showed that a visually apparent increase in tree cover did not represent a substantial density increase, but primarily reflected enlarging of tree canopies as trees age (Ffolliott and Gottfried 2002). Nevertheless, it is clear that genuine increases in tree density have occurred over the last 100–150 years in many places throughout the West.

Infill of persistent woodlands has been well documented in many parts of the Great Basin. Tree age structures in old-growth woodlands of central Nevada show dramatic increases in establishment of new trees beginning ca. 1880 (Bauer 2006). On tablelands of southwest Oregon and southwest Idaho, where low sagebrush (*Artemisia arbuscula*) is the predominant woody layer but scattered *Juniperus occidentalis* also are present, sampling of live and dead trees reveals a gradual increase in tree densities since the late 1800s in many areas (Johnson and Miller 2006). In some places, however, the magnitude of infill has been relatively small. For example, in the Mazama Ecological Province, over 67% of the trees >1m in height became established prior to 1870, and most individuals <1m were growing slowly with very narrow rings--demonstrating that small trees actually may be relatively old, especially on sites with poor growing conditions (Waichler et al. 2001). In a dense old-growth woodland occupying several thousand acres in southeast Oregon, infill is occurring in the outer edges of the stand, but little infill has occurred in much of the main core where understory trees 0.5 and 1.0 m in height are 100 - 250 years old (R.F. Miller unpublished data).

Moving to the Colorado Plateau, age reconstructions in northern Arizona document infill on three common soil types (Landis and Bailey 2005). Infill also is occurring in portions of the Uncompahgre Plateau and Mesa Verde in western Colorado (Eisenhart 2004, Floyd et al. 2004, Shinneman 2006). Most of the infill on the Uncompahgre Plateau is by piñon rather than juniper (Shinneman 2006). However, the net increase in tree density in woodlands of the Colorado Plateau actually may be relatively small when viewed over a longer time frame, as periods of increasing tree density are balanced by periods of extensive mortality. Consistent with this idea, millions of piñon trees throughout the Four Corners region died in a recent severe mortality event (Breshears et al. 2005). Moreover, photographs of Mesa Verde from the late 1800s (e.g., Chapin 1892) show relatively dense woodlands not dissimilar in appearance from those of today. Further evidence of relatively little net change on the Uncompahgre Plateau comes from Manier et al. (2005), who compared aerial photographs from 1937, 1965-67, and 1994, and saw minimal net change in density or extent of piñon-juniper woodlands.

8. *The observed increase in tree density and canopy cover during the twentieth century in persistent piñon-juniper woodlands is likely not due to fire exclusion. However, the mechanisms driving tree infill and expansion are generally not well understood for any of the three piñon-juniper types (Table 1). Possible mechanisms are evaluated in Section III below.*

## Section IIb: What We Know About Piñon-Juniper Savannas

We define "savannas" as stands having a well-developed grass understory plus a low to moderate density of trees (Table 1). Stands having low tree density but an understory dominated by life forms other than graminoids are not treated here, but are included in the sections on “persistent woodlands” (above) and “wooded shrublands” (below). Our group reached consensus on three key ideas about piñon-juniper savannas.

9. *Pre-1900 disturbance regimes in piñon-juniper savannas are not well understood*.

Explanation: Fire, insects, and climatic variation all probably influenced the structure and dynamics of this vegetation type, but the precise role and relative importance of each of these processes, and their interactions, are poorly documented. Some of the key hypotheses about historical fire regimes in piñon-juniper savannas are presented and evaluated in Section III below. Rigorous testing of these hypotheses is a high-priority research topic, as explained in Section V below.

10. *In some regions, notably parts of southern New Mexico and Arizona, savannas were more extensive historically than they are today. During the late nineteenth and twentieth centuries, many savannas in these regions have been converted to piñon-juniper woodlands of moderate to high canopy coverage, and many former grasslands have been converted to savanna or woodland.*

* HIGH CONFIDENCE … BUT PRECISE GEOGRAPHIC APPLICABILITY NOT ADEQUATELY KNOWN

Kinds of Evidence: *historic & recent photos, soils surveys*

Explanation: Savannas are most common in regions where reliable precipitation during the growing season favors growth of grasses, and where total annual precipitation is sufficient to also support at least some trees. Such a region is in southern Arizona and New Mexico, where a major portion of annual precipitation comes in the summer monsoon. Extensive infill of former savannas, and conversion of former grasslands to savanna or woodland through tree expansion, are well documented in written and oral accounts (A. Leopold 1924, L. Leopold 1951), and in aerial and ground-based repeat photography (e.g., Sallach 1986, Miller 1999, Fuchs 2002) from this region. For example, a comparison of aerial photos of a southwestern New Mexico study area revealed that former grasslands and juniper savannas had been largely replaced by relatively dense stands of *Juniperus deppeana*, such that forests and woodlands having more than 40% tree canopy cover comprised <50% of the landscape in 1935, but had risen to >80% by 1991 (M. Miller 1999). However, infill of former savannas and expansion of trees into former grasslands is not uniform throughout the region: Sallach (1986) documented increasing tree densities in many locations as well as declines in the abundance of piñon and juniper in other places. Furthermore, although the pattern of infill and expansion is clear in many places from photographic evidence, the mechanisms of conversion from savanna to woodland or from grassland to savanna are often uncertain (see Section III below).

Photographic evidence of tree infill and expansion is often impressive, but we lack historic photo coverage for much of the West. Consequently, other methods are frequently needed to determine whether any particular woodland today represents a persistent woodland of long duration or a former savanna or grassland in which tree infill or expansion during the past century has transformed the area into a woodland. A long-term view of vegetation change over centuries or millennia can be obtained from packrat middens, if available (see Section III below); however, packrats tend to collect vegetation in the rocky areas around their nests, such that middens may not reflect changes occurring in areas far away from the rocks where some of the most dramatic recent tree expansion appears in photographic comparisons (Swetnam et al. 1999). An age structure composed entirely of young trees, coupled with an absence of large dead boles, stumps, or other evidence of past disturbance by fire or wood harvest, indicates that a site was not wooded for at least a few centuries prior to the establishment of the extant trees (Jacobs et al. in press). Probably the strongest evidence that an area was persistently occupied by savanna, grassland, or shrub-grassland in the past is the presence of a mollic epipedon, which typically develops where grasses are a dominant vegetation component over long time periods. However, in some areas the upper soil horizons have been entirely lost through previous grazing and erosion, thus complicating accurate soils interpretations (see Section IV below on management implications for more on this problem).

11. *The principal mechanisms driving tree infill and expansion during the twentieth century are not well understood for piñon-juniper savannas or any of the three piñon-juniper types (Table 1) and probably vary from place to place. Possible mechanisms are evaluated in Section III.*

## <u>Section IIc: What We Know About Wooded Shrublands</u>

We define "wooded shrublands" as places where shrubs are dominant, but site conditions also can support trees during favorable climatic conditions or during long periods without disturbance (Table 1). Substantial tree mortality occurs during unfavorable climatic periods or following disturbance; hence these are places of potential expansion and contraction of the tree component (Romme et al. 2007). Our group reached consensus on four key ideas about wooded shrublands.

12. *Spreading, low-intensity surface fires had a very limited role in molding stand structure and dynamics of wooded shrublands in the historical landscape. Instead, the dominant fire effect was to kill most or all trees and to top-kill most or all shrubs within the burned area. This statement also is true of most ecologically significant fires today.*

* HIGH CONFIDENCE … APPLIES TO WOODED SHRUBLANDS THROUGHOUT THE WEST

<u>Kind(s) of Evidence</u>: *rigorous paleoecological studies, systematic observations of recent fires*

<u>Explanation</u>: The fuel structure in wooded shrublands typically is not conducive to a spreading, low-severity fire that would consume fine fuels without killing the dominant trees or shrubs, because the fine fuels are usually discontinuous (Figure 2). The major fuel components are the crowns of live shrubs and/or trees, which, if ignited, tend to burn completely with considerable heat release and death of the plant (Baker 2006, R. Tausch personal observations). Thus, as in persistent

woodlands, fires in wooded shrublands typically kill all of the trees and top-kill all of the shrubs and herbs within the areas that burn; usually the only surviving plants are those in patches that do not burn (see statements #3 and 4 for more on this idea).

13. *Increasing density of piñon and/or juniper within previously shrub-dominated areas, via infilling and expansion, is occurring extensively in some regions, notably the Great Basin, but is of relatively limited extent in other areas, notably western Colorado.*

* HIGH CONFIDENCE … BUT PRECISE GEOGRAPHIC APPLICABILITY NOT ADEQUATELY KNOWN

Kinds of Evidence: *rigorous paleoecological studies, historic & recent photos*

Explanation: Increasing density of piñon and/or juniper within sagebrush and other shrubland types has been widely documented in the western United States. Evidence includes aerial and ground-based repeat photography, and stand reconstruction using dendroecological methods (Cottam and Stewart 1940, Blackburn and Tueller 1970, Tausch et al. 1981, Rogers 1982, Miller and Wigand 1994, Soulé and Knapp 1999, Soulé et al. 2004, Johnson 2005, Bauer 2006, Johnson and Miller 2006, Weisberg et al. 2007). Increases in woodland area are occurring both through infilling of pre-existing sparse woodlands and from expansion of trees into formerly treeless shrublands.

Some of the most impressive infill and expansion have occurred in portions of the Great Basin, where woodland area may have increased by an order of magnitude since the mid-nineteenth century (Miller and Tausch 2001). For example, in stand reconstructions across an extensive area in northwest Utah, central Nevada, southwest Idaho, and southeast Oregon, extant and dead trees dating to the period prior to 1860 were found in only 16 - 67% of current woodland stands, suggesting the current area occupied by trees has increased 150 - 625% since 1860 (Miller et al. 2008). In this study, old trees (>140 years) usually were scattered in low densities across the landscape with no evidence that pre-1860 stands were as dense as many stands today. In another study, old trees (>140 years) accounted for less than 10% (usually <2%) of the individuals >30 cm in height (Johnson and Miller 2007). Similarly, Gedney et al. (1999) compared U.S. Forest Service surveys conducted in 1938 and 1988 across eastern Oregon and reported a 600% increase in area occupied by *Juniperus occidentalis*. Rates of increase in tree cover are very fast in some areas, e.g., ca. 10% per decade (Weisberg et al. 2007) or even a doubling every 30 years (Soulé et al. 2004). Bauer (2006) observed a sharp increase in the rate of tree establishment beginning ca. 1880, when the stem density doubling interval decreased from 85 to 45 years. However, there is geographic variability in the rate of density increase: for example, across six woodland stands in the northern portion of the Great Basin, tree age structures revealed a gradual shift from substantial increases in piñon and junipers to relatively limited establishment during the past 140 years (Miller et al. 2008).

In contrast to the extensive changes documented in woodlands of the Great Basin, studies on the Uncompahgre Plateau in western Colorado indicate that tree expansion into shrublands has been far more limited, and that the total area of piñon-juniper woodland has not increased substantially either in the twentieth century (Manier et al. 2005) or over recent centuries (Eisenhart 2004, Shinneman 2006). Although infill of pre-existing woodlands has occurred in this region in recent decades, the net increase in tree density over longer time periods may be minimal due to episodic mortality events (see statement #14).

Shrub-dominated soils typically do not develop a mollic epipedon that can be used as in savannas or grasslands to distinguish areas where trees expanded into former shrublands from persistent woodlands recovering from previous disturbance. However, other kinds of evidence, as described in statement #10 (e.g., the presence/absence of large old trees, living and dead), can aid in reconstructing local site history. An intriguing potential indicator of former sagebrush communities is the presence of sage-grouse leks. Some areas of current woodland are documented to have supported sage-grouse populations in the late 1800s and early 1900s. Sage-grouse hens re-nest in the same general sagebrush-dominated areas year after year, and their mature offspring do the same; colonization of new areas is slow (Dunn and Braun 1985, USDI BLM 1994, Connely et al. 2004, Schroeder and Robb 2004). Thus, documented past utilization by sage-grouse in a woodland today is evidence that the woodland has developed within a former sagebrush community.

14. *In addition to increases in piñon and juniper density in some areas, loss of piñon and juniper (especially from marginal sites) also has occurred recently and in the past.*
* HIGH CONFIDENCE … BUT PRECISE GEOGRAPHIC APPLICABILITY NOT ADEQUATELY KNOWN

Kinds of Evidence: *rigorous paleoecological studies, historic & recent photos*

Explanation: Although recent woodland expansion has received much attention, contraction of woodlands also has been documented, both recently and in the past. As noted in statement #2 on persistent woodlands, a "mega-drought" in the late 1500s probably killed many southwestern piñon trees, and a very recent and extensive die-back occurred between 2002 and 2004 in the Four Corners region as a result of drought, high temperatures, and bark beetle outbreaks. Substantial piñon mortality also occurred in parts of New Mexico during the severe drought of the 1950s (Swetnam et al. 1999). Some twentieth century expansions of woodland trees into sagebrush on the Uncompahgre Plateau in western Colorado appear now to be undergoing reversals as young trees are dying in recent droughts (K. Eisenhart, unpublished data). Thus, for thousands of years, tree expansion and contraction may have been a normal part of climatically driven fluctuations in woodland densities, perhaps especially at the ecotones with sagebrush, grasslands, and other non-woodland vegetation. It follows that the recently documented woodland expansion may be reversed by future contractions of woodland in at least some areas.

15. *The principal mechanisms driving tree infill and expansion during the twentieth century are not well understood for wooded shrublands or any of the three piñon-juniper types (Table 1) and probably vary from place to place. Possible mechanisms are evaluated in Section III.*

## Section III: Evaluating the Mechanisms of Infill and Expansion

A *pattern* of increasing tree density in many persistent woodlands, savannas, and wooded shrublands, and of tree expansion into former grasslands and shrublands, is well documented (see statements #7, 10, and 13). However, the *mechanism(s)* driving these changes is unclear. This is an important issue, because infill and expansion often are attributed primarily to effects of fire exclusion; consequently vegetation treatments designed to reduce or eliminate piñons and/or junipers often are justified in part by the assumption that past and present land uses have produced "unnatural" increases in tree density. Although this assumption is probably correct in some situations, clearly it is not correct in all. For example, exclusion of low-severity surface fires during the twentieth century cannot be the primary reason for infill of persistent woodlands, because low-severity fire was never frequent in these ecosystems even before Euro-American settlement (see statements #1, 2, and 3); furthermore, in many places we can explain increasing tree density as recovery from severe fire or anthropogenic clearing in the past, or as natural range expansion near the biogeo-graphical limits of a tree species. Therefore, we begin this section by reviewing these two relatively well understood mechanisms for increases in local tree density or extent (i.e., recovery from past severe disturbance and natural range expansion) in Sections IIIa and IIIb below.

But what is driving infill of persistent woodlands, savannas, and wooded shrublands, and expansion of piñon and juniper into former grasslands and shrublands, in the many places across the West where there is no evidence of earlier severe fire or clearing, and where infill and expansion are occurring near the center of the species' biogeographical distribution? In Sections IIIc - IIIe we evaluate the three most cogent explanations that have been offered: (i) direct and indirect effects of livestock grazing, (ii) fire exclusion, and (iii) climatic effects. Surprisingly little empirical or experimental evidence is available to support or refute any of these hypotheses; most interpretations are based on logical inference. Consequently, we cannot now come to any firm conclusions about the mechanisms driving infill and expansion of piñon and juniper in many locations. Nevertheless, we review existing evidence and data gaps for each of these three hypotheses, and we highlight this question as a high-priority research topic in Section V of this paper.

***Section IIIa. Recovery from Past Severe Disturbance***: Although fires are very infrequent in persistent woodlands, large severe fires do occur under some weather conditions (Figure 2), and recovery of the former woodland structure requires many decades to centuries (e.g., Erdman 1970; Floyd et al. 2000, 2004). Evidence of a stand-replacing fire also will remain conspicuous for many decades or centuries, in the form of charred snags and downed wood. Thus, a stand of young piñons and/or junipers growing amidst charred juniper snags and other forms of partially burned wood is *not* testimony to abnormal effects of fire exclusion, but simply represents recovery from a past high-severity fire.

Similarly, many areas that were chained in the 1950s and 1960s now support dense stands of young piñons and/or junipers that may give the appearance of expansion into grasslands or shrublands (e.g., Paulson and Baker 2006;143-146); however, closer inspection often reveals windrows of large, dead tree boles that were piled up during the chaining operation, along with stumps and seeded non-native grasses. Such a stand of young trees does not represent abnormal expansion of trees into non-woodland habitats, but is another example of natural recovery from severe disturbance. Widespread harvest also occurred during the Euro-American settlement era to provide materials for fence posts, firewood, construction materials, and charcoal to support the mining industry, e.g., in the Nevada Great Basin, (Young and Budy 1979) and in territorial New Mexico (Scurlock 1998;128-129). Sallach (1986) interpreted twentieth century increases in tree density in many places in New Mexico as recovery of pre-existing woodlands following severe human disturbance (wood-cutting and clearing for pasture improvement) rather than infill or invasion of previously sparse woodlands and grasslands. In some portions of the Southwest, woodlands still may be recovering from centuries of deforestation and other land uses by prehistoric and historic Puebloan peoples (Wyckoff 1977, Samuels and Betancourt 1982, Kohler and Matthews 1988, Allen et al. 1998, Allen 2004:64-66, Briggs et al. 2007).

Unfortunately, the extent, intensity, and specific locations of historic and prehistoric fire, harvest, and clearing generally are not well known. Nevertheless, particularly if a burned or cleared stand was a persistent woodland (Table 1), then local site conditions are inherently favorable for trees, and we should expect trees to be re-establishing naturally on the disturbed site.

***Section IIIb. Natural Range Expansion***: The presence of young piñon and juniper trees near the species' current geographical range limits may represent natural, long-term change in biogeographical extent rather than unnatural expansion into non-woodland habitats. Studies of sub-fossil pollen deposits and packrat (*Neotoma* spp.) middens reveal that many low-elevation conifer species, including junipers, piñons, and ponderosa pine, have been expanding their ranges throughout the Holocene (the past ~12,000 years) from glacial refugia sites in the Southwest and northern Mexico. In response to increasing temperatures and perhaps aided by moist periods, piñons expanded rapidly into the central and northern parts of the western United States at the end of the Pleistocene (Betancourt 1987, Nowak et al. 1994, Swetnam et al. 1999, Wigand and Rhode 2002), while junipers may have expanded with increasing temperatures, but during drier periods (Lyford et al. 2003).

This natural range expansion continues today. For example, the northernmost *Pinus edulis* population in Colorado, near Fort Collins, has been present for only about 400-500 years, and piñon continues to increase and expand into adjacent shrub and grassland communities (Betancourt et al. 1991). Similarly, the northernmost outlier of piñon in northeastern Utah at Dutch John Mountain colonized as recently as the 1200s (Gray et al. 2006). *Juniperus osteosperma* also has been expanding its range in Wyoming and adjacent sites in Utah and Montana for the past several thousand years, both at a regional scale by moving into new mountain ranges and at local scales by expanding populations where it has already established. In fact, juniper populations in some parts of Wyoming may represent the first generation of trees in these areas (Lyford et al. 2003). In addition to latitudinal range expansions following the Pleistocene, piñons and junipers have moved to higher or lower elevations in response to the climate changes that have occurred during the Holocene; for example, woodlands in the Great Basin have alternately expanded across large areas of landscape during favorable climatic periods and retreated to smaller refuge areas during less favorable periods (Miller and Wigand 1994). Thus, some expansions (and contractions) of piñons and junipers represent species' responses to natural processes such as climate change, rather than a consequence of land use or other human activities.

Unfortunately, not all of the specific locations where natural biogeographic range expansion is occurring have been mapped. Therefore, this mechanism should be considered in local site evaluations, especially where a site is located near the margins of the species' range.

***Section IIIc. Direct Effects of Livestock grazing***: Extensive livestock grazing began in the late 1800s in many parts of the western U.S. (Wooton 1908, Oliphant 1968, Dahms and Geils 1997, Scurlock 1998, Allen et al. 2002, Hessburg and Agee 2003)--and extensive infill and expansion of piñon and juniper began at the same time in many areas (e.g., Miller and Rose 1999, Fuchs 2002, Landis and Bailey 2005; C. D. Allen unpublished data). The coincidence in time between the onset of grazing and of increasing tree density suggests a direct cause-effect relationship, the mechanism presumably being that heavy grazing reduced herbaceous competition with tree seedlings and thereby enhanced seedling survival. Support for this mechanism comes from Johnsen's (1962) report of markedly better growth of juvenile *Juniperus monosperma* in places where grass had been removed.

However, empirical evidence for or against the grazing mechanism is sparse and mixed. Shinneman (2006) found greater densities of young trees in grazed areas on the Uncompahgre Plateau in western Colorado than in nearby ungrazed areas. In contrast, Harris et al. (2003) reported comparable twentieth-century increases in tree density in both grazed and un-grazed areas in a southern Utah study site; and lightly grazed areas often appear to contain as many young trees as heavily grazed areas in the northern Great Basin (R.F. Miller, personal observation) and in south-central New Mexico (E.H. Fuchs, personal communication). It is well known that grazing effects can be extremely variable across different soil types within the same climatic zone. For example McAuliffe (2003) notes that grazed soil types with shallow argillic horizons are much more resistant to woody plant encroachment than are sites that promote deeper infiltration. Moreover, the mechanistic relationship between herbaceous competition and tree seedling establishment has received little experimental testing beyond Johnsen's early study.

Thus, we simply lack adequate empirical or experimental information with which to confidently evaluate the importance (or lack of importance) of the *direct* effects of livestock grazing as a key mechanism driving tree infill and expansion during the past 150 years. However, the *indirect* effect of livestock grazing also may have been important because sustained heavy grazing reduces grasses and other herbaceous fuels which foster fire spread under both modal and extreme fire weather conditions. In some western ponderosa pine and dry mixed conifer forests, exclusion of low-severity fires has been a principal mechanism driving tree density increases during the twentieth century (e.g., Allen et al. 2002, Hessburg and Agee 2003), although in other western ponderosa pine forests the principal mechanisms were nineteenth-century fires, logging and livestock grazing, rather than fire exclusion (Baker et al. 2007). Thus, the importance of this indirect effect of grazing hinges on the importance of fire exclusion in driving infill and expansion of piñons and junipers (IIId below).

***Section IIId. Fire exclusion***: A logical argument can be made that fire exclusion since the mid-1800s is a primary cause of piñon and juniper infill in savannas and wooded shrublands, and of tree expansion into former grasslands and shrublands. Southwestern savannas and grasslands in particular often produce continuous fine fuels conducive to frequent and wide-spreading fires, and they occur in regions where wet/dry climatic cycles are common. Thus, it is logical to suppose that historical fires in these ecosystems were frequent enough to kill most of the fire-intolerant piñons and junipers that continually became established among the fire-tolerant grasses. Fire behavior and effects in wooded shrublands (especially those with tall shrubs) differ from fire in savannas in that the shrub fuels typically support higher flame lengths, greater heat release, and greater likelihood of extensive tree mortality (Figure 2); and post-fire recovery of the shrubs is often slower than recovery of burned grasses. Nevertheless, in both kinds of ecosystems recurrent fires may have maintained tree densities well below what could potentially be supported by local climate and soils; higher-density stands may have persisted only in relatively fire-safe sites, such as on rocky outcrops or in rocky draws, where fire spread or high-severity fire was inhibited. Support for the fire exclusion hypothesis comes from the fact that extensive infill and expansion began to occur in many places in the late nineteenth century, coincident with the onset of livestock grazing and the resulting reduction in the frequency of extensive surface fires. Grazing intensity was greatly reduced in most of the West after 1930, but effective governmental fire suppression began to be more effective at about that time (Pyne 1982), and additional land use changes-- notably those resulting in fragmentation of landscapes, including roads, buildings, and cleared fields-- have generally precluded the extensive fires that may have burned prior to the late nineteenth century in many areas.

Although this interpretation is logical, it has a major empirical shortcoming-- namely that the assumption of frequent historical fire is unproven (even untested) in many areas. In the relatively few fire history studies that have been conducted in piñon and juniper vegetation, fire-scarred trees (perhaps the most conclusive direct evidence of previous fires) are typically rare or absent (Baker and Shinneman 2004). There are questions about how to interpret the paucity of fire-scarred piñons and junipers (see the discussion of methodological issues in Section V on research priorities), but a general lack of fire scars is consistent with the idea that fires actually were infrequent in all or most kinds of piñon and juniper vegetation in the past. If fire was in fact infrequent in piñon and juniper vegetation prior to the late 1800s, then fire exclusion cannot be the major driver of tree infill and expansion during the last century.

Thus we see that two logical, but contradictory, interpretations can be made about the historical role (or lack of a role) of fire in limiting piñon and juniper infill and expansion. To critically evaluate both interpretations, we need more spatially extensive empirical data on piñon-juniper fire history, especially in piñon and juniper savannas.

In the absence of adequate empirical data, interpretations of fire history often are based instead on anecdotal observations and logical inference. There is also a tendency to import observations from areas of very different biophysical conditions and treat them as generalities when data are sparse. For example, late nineteenth century fires in some desert grasslands of southeastern Arizona are documented from newspaper accounts (Bahre 1991;138-141), and it is also inferred that fires must have been relatively frequent to prevent shrub encroachment of some desert grasslands (McPherson 1995). It might be assumed from this evidence that fire played a similar role in desert grasslands, piñon and juniper savannas, and open woodlands that have grassy understories, throughout much of the Southwest. However, desert grasslands in Arizona differ in composition and climate from those in New Mexico, and grasslands at the edge of the Great Plains in eastern New Mexico differ yet again. Moreover, Wright (1980;16) states that the pre-1900 role of fire in grasslands of southern Arizona and New Mexico is simply unknown, and that fire was possibly unimportant ecologically in at least some kinds of desert grassland (e.g., black grama (*Bouteloua eriopoda*) communities). We have a similarly inadequate understanding of the (probably complex) ecological role of fire in piñon and juniper savannas of Arizona and New Mexico.

A similar paucity of empirical fire history data plagues our efforts to understand what is driving tree infill and expansion in sagebrush-dominated communities and associated wooded shrublands of the Great Basin and Colorado Plateau. For example, historical fire rotations (time required for cumulative area burned to equal the size of the entire area of interest) in Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) and low sagebrush, which formed the matrix within which many of the woodland communities existed, are estimated to have been 100-450 years (Baker 2006). With such long rotations, it would seem that fire was not frequent enough historically to prevent extensive tree establishment, and that the long fire intervals observed during the twentieth century are not far outside the historical range of fire intervals--all implying that fire exclusion cannot be the major driver of piñon and juniper infill and expansion. However, the "sagebrush community" is very heterogeneous, and a single broad characterization of historical fire rotations cannot adequately convey the complex historical role of fire in these ecosystems. For example, historical fire rotations were substantially shorter in the more mesic mountain big sagebrush (*A. tridentata* ssp. *vaseyana*) communities than in the more xeric Wyoming big sagebrush and low sagebrush communities (Baker 2006), and conversion of shrubland to woodland also can occur more rapidly (within only several decades) in the more mesic habitats (R. Miller, personal observations)--suggesting that the late-nineteenth and twentieth-century reduction in fire frequency was in fact a major cause of tree expansion in at least some shrublands.

In sum, we simply do not have adequate empirical data on historical fire regimes to determine how important (or unimportant) fire exclusion has been in allowing infill and expansion of piñon and juniper trees in savannas and wooded shrublands across the West. Obtaining additional fire history data is therefore a high research priority (see Section V).

***Section IIIe. Climatic effects***: The onset of extensive infill and expansion of piñon and juniper in the late nineteenth century in many areas coincided not only with the beginning of fire exclusion, but also with the end of the "Little Ice Age" and the beginning of a general warming trend and changes in precipitation patterns that continued through the twentieth century. Occupying as they do the transition zone between mesic forests at higher elevations and environments too dry for trees at lower elevations, piñon-juniper communities may be especially sensitive to even subtle changes in temperature and precipitation.

It is possible that some or even much of the infill and expansion of piñon and juniper that has occurred during the past 150 years is a more-or-less natural response to short-term and long-term climatic fluctuation. Two long-term twentieth century data-sets from desert and semi-desert areas of southern New Mexico and Arizona reveal that relatively high winter precipitation generally favors woody plants over herbaceous species, and that specific periods of extensive shrub establishment coincided with periods of wet winters (Neilson 1986, Brown et al. 1997). These two studies focused primarily on expansion of shrubs, not trees, but other studies demonstrate that recovery of woodlands from drought may occur as a pulse of piñon recruitment during the first wet period that follows the drought (Swetnam et al. 1999, Shinneman 2006). For example, in two study areas on the Uncompahgre Plateau in western Colorado, piñon abundance began increasing in the late 1700s, during a wet period that followed a long dry period (Eisenhart 2004, Shinneman 2006). Note that the apparently climate-driven increase in tree density in this area occurred more than half a century before arrival of Euro-American settlers and associated effects of livestock grazing and fire exclusion. In the twentieth century there were two very wet periods in the Southwest--during the first two decades of the century and the period from the mid-1970s to the mid-1990s (Swetnam and Betancourt 1998). These are both periods when many piñon trees became established in the region (e.g., Floyd et al. 2004).

Additional support for the climate hypothesis comes from observations of recent contraction as well as expansion of piñon and juniper woodlands. Although much of the twentieth century was apparently favorable for tree establishment and survival, extensive piñon mortality occurred in the Southwest during severe droughts of the 1950s and from the mid-1990s through the early 2000s (see statements #2 and #14).

The major shortcoming of the climate hypothesis is that the evidence is mostly correlative, with limited experimental data with which to evaluate the specific mechanisms by which piñon and juniper respond to specific climatic changes. We also have a poor understanding of how climatic variability influences growth and abundance of the herbaceous component of piñon and juniper vegetation, notably grasses, which in turn influences fuel structure and potential fire behavior.

***Summary of potential mechanisms driving tree infill and expansion***: It is widely assumed that infill and expansion of piñon and juniper represents primarily an "unnatural" consequence of human land use, in particular the effects of fire exclusion. It is important to stress, however, that human land use in the western U.S. since the mid to late 1800s has occurred against a backdrop of climatic and natural disturbance-driven fluctuations in tree establishment and survival. All of these processes interact, and the relative importance of each (direct grazing impacts versus fire exclusion versus climatic fluctuation) probably varies spatially and temporally across the vast expanse of heterogeneous environments occupied by piñon and juniper vegetation. Thus, simplistic and over-generalized explanations of the processes driving infill and expansion should be avoided, and new research should be conducted to disentangle the relative effects of the various mechanisms underlying piñon and juniper infill and expansion in different eco-regions within the western U.S. (see Section V on research priorities).

## Section IV: Some Management Implications

It was not the task of our group to make specific recommendations or guidelines for management of piñon-juniper vegetation. However, in this section we caution against overly simplistic management assumptions, and discuss how the information presented in the fifteen statements above can be used to help inform specific management policies or management plans that involve this vegetation type.

A. *Different management goals--e.g., wildfire mitigation, forage improvement, wildlife habitat improvement, and ecological restoration--cannot necessarily all be achieved by the same kinds of treatment in piñon and juniper vegetation*.

Four widespread management actions being conducted in piñon and juniper vegetation are (i) wildfire mitigation, which usually focuses on fuel reduction through selective cutting, mastication, or low-severity prescribed fire; (ii) forage improvement for livestock by reducing tree competition with palatable grasses and forbs, sometimes supplemented by planting grasses; (iii) habitat restoration for wildlife, e.g., sage grouse and other shrub-dependant species; and (iv) ecological restoration, in which the goal is to re-establish structural and functional characteristics similar to what existed prior to Euro-American settlement. It is important to emphasize that any given vegetation treatment is unlikely to achieve all of these goals simultaneously. For example, extensive low-severity prescribed burning would not represent restoration in persistent woodlands that rarely burned historically, but rather a novel kind of disturbance. Similarly, reduction of tree density in savannas or wooded shrublands where substantial infill has occurred primarily due to the climatic conditions of the twentieth century might represent restoration of pre-1900 stand *structure* but not of the ecological *processes* that formerly maintained a lower-density structure. Chaining piñons and junipers, followed by planting grass, could improve forage production, but would not necessarily represent ecological restoration, especially in persistent woodlands where trees were always relatively dense.

Where the multiple goals of wildfire mitigation, forage improvement, wildlife habitat improvement, and ecological restoration might potentially be most congruent is in former savannas, grasslands, and shrublands where substantial infill or expansion of trees has been adequately documented to have occurred primarily because of EuroAmerican land uses (e.g., fire exclusion), as opposed to natural processes (e.g., climatic fluctuations). In these situations, reduction of tree density to former levels can reduce crown fire potential while promoting recovery of understory components, if soils and native flora are still in good condition (see below); these changes also may improve habitat conditions for selected wildlife species of interest. Note that this initial treatment to restore canopy structure would not be adequate in itself to fully restore pre-EuroAmerican conditions: it also would be necessary to restore and maintain the key natural processes that formerly maintained the savanna, grassland, or shrubland structure, and to reform the land uses that led to the changes in woodland structure to avoid a repeated cycle of restoration and degradation.

In many or most situations, however, it probably is not possible to achieve all of these goals with one kind of vegetation treatment; thus, managers should be very clear about the specific goals and objectives of piñon-juniper treatments.

B. *Ecological restoration is neither straight-forward nor easy*.

Restoration of pre-EuroAmerican conditions is not the only potentially appropriate management goal for any given piece of ground. But where ecological restoration is the primary objective, appropriate treatments will differ among the different kinds of piñon-juniper vegetation (Table 1). A field key to the three kinds of piñon-juniper vegetation that we recognize in this paper is provided in Romme et al. (2007); we regard this key as tentative pending additional clarification of the distribution and characteristics of the three types (see Section V on research priorities).

In many woodlands, the most urgent restoration needs are related not to the canopy but to the understory, soils, and watershed function. Invasive species, local extirpation of key native species, erosion, and other changes in structure and composition at the ground level may be more significant than increased tree density. This may be especially true of many persistent woodlands, where current tree densities may not be abnormal, but the understory is in poor condition. If such sites burn in the future, they may become dominated by aggressive and persistent non-native species instead of progressing through a normal post-fire successional trajectory (e.g., Floyd et al. 2006), potentially resulting in a permanent type conversion.

In any situation, it is important that restoration treatments reflect an understanding of historical conditions and mechanisms of twentieth-century change. A landscape perspective also is needed in evaluating current conditions and designing treatments to improve current conditions: the historical landscape in most places was a heterogeneous mix of stand structures, reflecting underlying heterogeneity of soils, topography, and disturbance history. One practical and conservative approach is to use existing stand age-structure and underlying topographic and edaphic cues as a template for the appropriate spatial pattern and intensity of treatment. In many Great Basin landscapes, for example, tree expansion into former shrublands is most likely to have occurred where soils are deeper and herbaceous productivity is greater. If the objective is to restore pre-1850 structure and if tree expansion is adequately documented to have resulted primarily from EuroAmerican land uses (e.g., fire exclusion) as opposed to natural processes (e.g., climatic fluctuations), then such sites likely would be the appropriate locations for removal of all or most trees. This patchiness issue is very important but often neglected: areas of persistent woodland and areas of tree expansion commonly are finely intermingled, but management prescriptions that ignore this spatial pattern may homogenize a formerly heterogeneous landscape. As with all ecological restoration efforts, the land uses that led to the changes in woodland structure also will need to be reformed to avoid a repeated cycle of restoration and degradation.

Mechanical thinning and prescribed burning have been applied widely in piñon and juniper woodlands to restore historical savanna or open woodland structure, but these treatments have had mixed results. Grass cover may increase in a two-to-three year window after cabling and slash treatments (e.g., Brockway et al. 2002, Ansley et al. 2006) but some treatments (often associated with grazing) have shown few long-term benefits with regard to grass increase or tree reduction (Rippel et al. 2003, Schott and Pieper 1987). Some of these failed cases may involve misguided attempts to convert persistent woodlands (e.g., on shallow, rocky soils) to savannas that were never supported in those locations, while in other instances loss of soil resources may preclude successful reestablishment of herbaceous understory (Davenport et al. 1998, Hastings et al. 2003).

In any treatment, whether fire mitigation or restoration, care is needed to avoid unnecessary damage to understory plants and soil crusts. Fires (including prescribed ones) and mechanical treatments may expose soils and disturb soil crusts, accelerate erosion rates, and inadvertently create barriers to savanna restoration (Roundy et al, 1978). Severe fires may result in nearly complete mortality of understory vegetation and loss of soil protection from raindrop impact. Shrub mortality from fire is of particular concern where big sagebrush is an important component of the plant community, because the shrubs sometimes require decades to recover, to the detriment of sage-grouse and other shrub-dependent fauna (Wisdom et al. 2005, Baker 2006). Both prescribed fire and mechanical treatments may facilitate invasion by cheatgrass and other undesirable non-native species. Cheatgrass in particular is a major threat to the long-term ecological integrity of western woodlands and shrublands, in part because it may allow fire to recur in a stand far more frequently than historically (D'Antonio and Chambers 2006). In general, then, treatments may either increase or decrease soil erosion, integrity of the plant community, and likelihood of achieving restoration, all depending on the details of implementation (Blackburn 1983).

In places where it has been adequately documented that a formerly open savanna or wooded shrubland was maintained in the past by frequent fire, and that twentieth century fire exclusion is the primary mechanism causing conversion to closed woodland, this anthropogenic change in canopy structure may have been accompanied by substantive changes in soils, fuel structure, and local fire regimes; under these circumstances restoration of pre-1900 conditions may be extremely difficult if not impossible. The combination of increasing tree cover and heavy grazing may have suppressed herbaceous growth, thereby reducing the horizontal continuity of fine fuels and the potential for subsequent surface fires (Allen 1989, Hastings et al. 2003). Loss of herbaceous cover plus hoof action also may have caused severe erosion of surface soil horizons, thereby precluding ready re-establishment of herbaceous cover even if grazing pressure is diminished and tree cover is reduced (i.e., hysteresis). Such changes in soil, vegetation, and fuel structure could persist for a very long time and result in a new fire regime more similar to that of persistent woodlands than to the former savanna fire regime (i.e., a state change may have occurred: Holling 1973, Suding et al. 2004, Briske et al. 2005). This is a logical hypothesis if we assume that the historical savanna or woodland supported greater herbaceous cover and frequent fires, and if true it would seriously constrain management options. Note, however, that in many locations the mechanism(s) driving tree infill and expansion have not been adequately documented by field studies that test alternative explanations of change (hypotheses IIIa to IIIe).

Given the global climatic changes that have already occurred and are projected to occur over the twenty-first century, coupled with the effects of past land use and invasion by non-native species such as cheatgrass, it may be impossible in many places to precisely restore the kinds of piñon-juniper ecosystems that existed 150 - 400 years ago (Millar et al. 2007). It remains important to understand the ecological conditions that prevailed during that earlier period in order to understand the patterns and mechanisms of change during the past 150 years--but that particular reference period (which encompasses The "Little Ice Age" in many areas) is likely an unachievable target for restoring many ecosystems in the twenty-first century. The uncertainties and potential magnitude of ecological change that we will likely face in the next century argue for caution and humility as we design our management goals and specific vegetation treatments in piñon-juniper and other vegetation. The uncertainties also underscore the importance of continued research, monitoring, and evaluation of treatment effectiveness in the spirit of adaptive management.

C. *Regardless of treatment goals, it is important to regard all treatments as experiments to be monitored as part of an adaptive management strategy*. *Collaborative implementation and evaluation of current and upcoming vegetation treatments by managers and researchers can be an especially effective way to improve our understanding of piñon-juniper ecosystems and improve the efficacy of future treatments.*

Piñon and juniper treatments designed primarily to improve livestock forage have been conducted for several decades and in numerous geographic locations across the western U.S. However, we have much less experience with treatments aimed at wildfire mitigation or ecological restoration in piñon-juniper vegetation, and much less information on their effectiveness or possible undesirable side-effects. As noted in the section just above, results of restoration efforts in piñon-juniper vegetation have been mixed. Nevertheless, intensive treatments designed for wildfire mitigation and ecological restoration in piñon-juniper vegetation are underway or planned in many parts of the West today. Many or most of the current federally sponsored initiatives do not require any systematic, structured, or experimental monitoring of effects (e.g., BACI design). A stronger experimental design likely will be included in upcoming recommendations by the Conservation Effects Assessment Project (http://www.nrcs.usda.gov/technical/nri/ceap/), but funding for effective monitoring of treatments may remain inadequate. Given the fundamental uncertainties about the ecology of piñon-juniper vegetation, it is important that managers and policy-makers acknowledge this uncertainty when presenting management plans to the public; and despite the lack of rigorous monitoring requirements and associated funding at the federal level, we hope that managers will monitor treatment effects to the greatest extent possible (recognizing constraints of budget and personnel), both because there is potential for unexpected or undesirable results of treatments and because we may learn a great deal from what in effect are broad-scale ecological experiments.

In particular, we emphasize the outstanding opportunities for collaboration between scientists and managers in current and upcoming piñon-juniper treatments. By adding a well-designed research and monitoring component to a practical management-oriented project, not only is it possible to evaluate the efficacy of a given project, but also it can improve our understanding of the more general ecological processes at work in piñon-juniper vegetation. For example, we can infer much about historical fire regimes by carefully documenting fire behavior and fire effects on soils and vegetation under varying conditions of fuel structure and ambient weather (see Section V on research priorities).

## **Section V: Some Research Priorities**

Our analysis of what we know and do not know about piñon-juniper vegetation suggests five high-priority research topics related to historical structure and dynamics of these ecosystems as well as the patterns and mechanisms of change since the mid-to-late 1800s. Outlined below in no particular order are the information needs that we think are most urgent for informing management of piñon-juniper vegetation across the western U.S., particularly where ecological restoration is a primary goal.

### *1. Improve our methodology for reconstructing fire history and tree population dynamics in piñon-juniper vegetation.*

It is not surprising that rigorous empirical fire history studies are relatively few for piñon-juniper vegetation, because these systems present significant methodological challenges. Interpretations of fire history and tree demography should be based on multiple converging lines of evidence, including (if available) fire scar analysis, tree age structure (including both dead and living trees), soil characteristics, charcoal abundance and spatial distribution, packrat midden analysis, comparisons of historic and modern photos, and written documentation.

Perhaps the key methodological issue is how to interpret the general scarcity of fire scars on piñon and juniper trees (Baker and Shinneman 2004). Fire-scars on live piñon are especially rare in study sites on the Colorado Plateau (Eisenhart 2004; Floyd et al. 2004, 2008; Shinneman 2006), but they seem to be somewhat more common in regions to the south and west. Fire-scarred piñon have been found in Chihuahuan Desert Borderlands (Camp et al. 2006); in the Oscura, San Andres (Muldavin et al. 2003) and Sacramento (Wilkinson 1997, Brown et al. 2001) Mountains of southern New Mexico; and in the Jemez Mountains of northern New Mexico (C. Allen, unpublished data). Large old *J. deppeana* in southern New Mexico frequently contain basal scars, but it is not certain that all of these scars originated from fire and most junipers cannot be cross-dated due to frequent false rings and asymmetrical growth patterns. Fire-scarred piñon have been found in the Great Basin, with dates that correspond to known fires of the past 30 years (Bauer 2006, Py et al. 2006); these latter scarred trees were concentrated around the margins of stand-replacing fires and in unburned patches within the fire perimeters. Even where fire-scarred piñons and junipers are found, however, they are almost always few in number.

Lack of fire scars could indicate that either (i) no fires occurred within the life spans of the extant trees, or (ii) fires occurred but the fires simply were not recorded on those trees. Piñons and junipers are generally fire-intolerant, and when fire occurs beneath them typically they are killed. However, fires being carried by grasses or shrubs occasionally have been observed to burn around piñon and juniper trees, scorching the outer crowns but not igniting the trees, perhaps due high fuel moisture levels in the trees foliage and a lack of suitable fuel directly beneath and adjacent to the trees (R. Tausch, personal communication). This kind of fire behavior could explain a lack of fire scars, despite fire occurring in the stand, but the frequency and overall significance of such a burning pattern are unknown at this time. Baker and Shinneman (2004) examined a site on the Uncompahgre Plateau in western Colorado where an early written report described a fire in piñon-juniper woodlands, but they found no fire-scarred trees or other evidence of past fire. This observation could support the idea that low-severity fires can burn and leave no evidence of their passing; but it also may simply reflect a fire so small that its ecological effects were negligible or a fire whose location was not accurately reported in the early report.

Uncertainty about how to interpret the fire-scar record (or lack of a record) is a major impediment to better understanding of historical fire regimes in piñon-juniper vegetation throughout the West. One particularly effective approach for improving our understanding of fire-scar formation in piñon would be for researchers to collaborate closely with managers conducting prescribed burns. It would be instructive to examine piñon trees within the perimeters of recent fires, both immediately post-fire and over subsequent years. However, the most rigorous and direct way to study fire-scarring in live piñon would be to experimentally burn around individual trees, under varying conditions of ambient weather and tree characteristics, and subsequently record the rates of tree survival and scar formation.

2. *Conduct rigorous empirical studies of fire history and tree demography in places where extensive infill and expansion have occurred but local history is unknown.*

As emphasized in Section III, the *pattern* of increasing tree density in many former savannas, grasslands, and sparsely wooded shrublands is well documented, but the *mechanism* is unclear. If fire was formerly frequent in a given area, then fire exclusion is likely the major cause of increasing tree density during the twentieth century. But if fire was not frequent historically, then another mechanism must be more important. The role of fire in shaping historical stand and landscape structure cannot be assessed until we have a better understanding of historical fire regimes among the various kinds of piñon-juniper vegetation.

3. *Evaluate the history and status of what appear to be persistent woodlands within regions where the precipitation pattern is summer-dominated.*

Most of what we know about persistent piñon-juniper woodlands comes from research conducted in the Great Basin and Colorado Plateau, where the precipitation pattern is either winter-dominated or more or less equally divided between winter and summer. Less research has focused on these kinds of woodlands in monsoon-dominated regions such as southern New Mexico (but see Wilkinson

1997, Brown et al. 2001, Muldavin et al. 2003, and Camp et al. 2006). We list this as a research priority because a number of managers and practitioners in New Mexico and Arizona--field people who have a great deal of on-the-ground experience--think that persistent woodlands were never an important component of the historical vegetation in this region (e.g., H. Fuchs, personal communication). Interpretations of ecological history in these woodlands needs to be based on convergence of multiple sources of information, e.g., fire scars (where available), tree age structures, soils characteristics, and comparisons of historic and recent photographs. Investigations also need to be conducted at multiple spatial scales, from individual stands to landscapes.

An important component of this research would be to determine whether the stands that now appear to be persistent woodlands in this region are actually former fire-maintained savannas or open woodlands that have been degraded by livestock grazing and fire exclusion (see Section IV on management implications for more on this idea). The distinction between genuine "persistent woodlands" that burned infrequently in the past and formerly fire-maintained savannas or woodlands is important, especially in identifying targets and methods for ecological restoration. For example, "persistent woodlands" may still be within their historical range of variability, whereas degraded woodlands would be strong candidates for restoration to pre-1900 conditions.

4. *Conduct experimental studies to evaluate the specific effects and interactions of fire exclusion, climatic fluctuation, and livestock grazing in driving expansion and infill of piñons and junipers during the past 100 – 150 years.*

Although we have a number of correlative studies that suggest hypotheses about the environmental drivers of infill and expansion (see Section III), we now need experimental studies to critically test those hypotheses. Because the relative importance of fire exclusion, climate, and grazing probably varies from place to place, it would be important to conduct such studies in many different geographical areas.

5. *Develop maps showing where woodlands and savannas existed prior to Euro-American settlement versus woodlands that have expanded into formerly non-woodland vegetation types.*

Managers often have inadequate information about specific locations in the landscape where the vegetation has been greatly altered by anthropogenic activities (e.g., places where woodlands are expanding due fire exclusion) and where anthropogenic influences have been relatively minimal (e.g., persistent woodlands). Such information is essential, however, for planning appropriate treatment placements if the goal is to explicitly restore historical vegetation structure. In addition, an understanding of the historical context in which the present biota developed can help inform managers about the feasibility and methods of achieving other objectives such as forage improvement. To meet this need, we must first develop consistent criteria that can be used in the field for distinguishing pre-existing woodlands or savannas from former grasslands or shrublands that have been invaded by trees. Relevant criteria might include soil characteristics as well as size, apparent age, and density of living and dead trees. Once these distinctions can be made reliably in the field, woodlands can be sampled over a large heterogeneous region, correlations can be identified between woodland type and environmental variables (elevation, substrate, topography, etc.), and maps can be generated in a GIS environment for use in planning and implementing restoration and other vegetation treatments. This kind of research is underway in a few places (e.g., Jacobs et al. in press, B. Bestelmeyer and S. Yanoff, unpublished data; both studies were conducted in New Mexico), but it is needed throughout the range of piñon and juniper vegetation.

A similar combination of field and GIS techniques can be used to predict where in the landscape the greatest and most ecologically significant changes are likely to occur in the future. In Mesa Verde National Park, for example, Floyd et al. (2006) mapped the locations of piñon-juniper woodlands having the greatest probability of post-fire invasion by non-native species, and Turner et al. (2008) mapped the areas most likely to be affected by shortened fire intervals and associated erosion events as a consequence of cheatgrass invasion.

## Conclusions

Piñon-juniper is one of the major vegetation types in western North America, covering some 40 million ha across nearly a dozen states. Until recently, management tended to emphasize removal of piñon and juniper trees to improve livestock forage, wild ungulate habitat, and water yield. However, the biodiversity, aesthetics, and ecosystem services provided by intact piñon-juniper vegetation are increasingly recognized and appreciated, and management objectives have expanded beyond simple tree removal. These ongoing efforts at ecosystem management are hindered by inadequate information about several basic elements of the ecology of piñon-juniper vegetation, including (i) prehistoric and historic disturbance regimes, (ii) mechanisms driving twentieth century changes in vegetation structure and composition, and (iii) the variability in ecosystem structure and process that exists among the diverse combinations of piñons, junipers, and associated shrubs, herbs, and soil organisms throughout the 40 million hectares of western North America that are covered by this vegetation type. This paper presents a summary of what we currently know and don't know about historical and modern stand and landscape structure and dynamics in piñon-juniper vegetation. Its intent is to provide a source of information for managers and policy-makers, and to stimulate researchers to address the most important unanswered questions. The process of producing this paper—assembling experts from throughout the range of piñon and juniper to synthesize our current knowledge—was effective. Such an approach, similar to that used by Allen et al. (2002), could be applied to other kinds of vegetation for which historical structure and dynamics are uncertain or controversial.

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# Influence of Livestock Grazing and Climate on Pinyon Pine (*Pinus edulis*) Dynamics

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## Abstract

Over the last century there has been marked expansion and infilling of pinyon (*Pinus* spp.)–juniper (*Juniperus* spp.) woodlands into grassland and shrubland ecosystems across the western United States. Although range expansions in pinyon-juniper populations have been documented with changing climate throughout the Holocene, over the last century, local scale impacts such as livestock grazing, changes in fire regimes, and increasing atmospheric $CO_2$ concentrations are thought to be more recent drivers of pinyon-juniper woodland distribution. Our objective was to examine the role of historical livestock grazing relative to past climate in regulating pinyon (*Pinus edulis* Engelm.) recruitment and growth over the last century in a persistent pinyon-juniper woodland. We compared pinyon dynamics on a remote mesa site, minimally grazed by domestic livestock, to a nearby historically grazed site in southeastern Utah. The presence of a significant number of old trees ($>200$ yr) at both sites suggests that these populations were well established prior to the 20th century. No differences in pinyon density or basal area were observed between grazed and ungrazed sites. Stand age structure data showed that pinyon recruitment across these sites was highly synchronous, with a large recruitment event occurring during the early 1900s; 17% of the pinyons on both sites dated to the 1920s. Climatic conditions during this decade were consistently cool and wet—conditions known to support enhanced recruitment and growth in pinyon pines. Pinyon growth was also strongly synchronous across sites ($r = 0.96$). Pinyon growth was significantly correlated with winter/spring precipitation and negatively correlated with June temperature. Taken together, our results suggest that past climate may be more important in structuring pinyon populations than historical land use in these persistent pinyon-juniper woodlands. Given future climate projections of increasing temperature and more extended drought periods, regeneration of pinyon populations following the recent regional-scale dieback may be slow. Moreover, prolonged drought combined with potentially slow regeneration times for pinyon under future climate scenarios could result in substantial declines in pinyon populations across the region, a result that land managers should consider when planning for future restoration treatments in persistent pinyon-juniper woodlands.

## Resumen

Durante el último siglo se ha presentado una marcada expansión y establecimiento de bosques de piñón (*Pinus* spp.)–junípero (*Juniperus* spp.) dentro de ecosistemas de pastizales y matorrales a través del oeste de Estados Unidos. Aunque el rango de expansión en las poblaciones de piñón-junípero se ha documentado con cambios en el clima a través de los últimos años; durante el último siglo, los impactos a escala local como pastoreo de ganado, cambios en regímenes de fuego y el incremento de las concentraciones de $CO_2$ en la atmósfera, se cree que son los más recientes responsables de la distribución de los bosques de piñón-junípero. Nuestro objetivo fue examinar la función histórica del pastoreo de ganado en relación con las condiciones climáticas presentadas en el pasado, sobre la regulación de piñón (*Pinus edulis* Engelm.) analizando su crecimiento durante el último siglo en un bosque persistente de piñón-junípero. Comparamos la dinámica del piñón en un sitio de una remota meseta, mínimamente pastoreada por ganado doméstico, con sitios cercanos pastoreados históricamente en el sureste de Utah. La presencia de un número significativo de arboles mayores de ($>200$ años) en ambos lugares sugiere que estas poblaciones fueron bien establecidas con anterioridad al siglo XX. No se encontró diferencias en la densidad de pinos o en el tamaño de su área basal analizada entre sitios pastoreados y no pastoreados. Los datos de la estructura de la edad de la población mostraron que los árboles evaluados en estos sitios fueron altamente sincronizados, con una gran cantidad de eventos que ocurrieron a principios de 1900; 17% de los piñones en ambos sitios datan de la década de 1920. Condiciones climáticas durante esta década fueron consistententemente frías y húmedas—estas condiciones sirvieron para apoyar el mayor número de incorporación y desarrollo de los pinos piñoneros. El desarrollo de los piñoneros estuvo también fuertemente sincronizado entre los diferentes sitios ($r = 0.96$). El crecimiento de los piñoneros se relacionó significativamente con la precipitación de invierno/primavera y negativamente correlacionado con la temperatura del mes de junio. Agrupando, nuestros resultados sugieren que las condiciones climáticas presentadas en el pasado podrían ser más importantes en la estructura de las poblaciones de los pinos piñoneros que el uso histórico de la tierra realizado en los bosques de juníperos. Dadas la futuras proyecciones climáticas sobre el incremento en la temperatura así como mayores y más prolongados periodos de sequia, la regeneración de las poblaciones de piñones

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siguiendo la reciente escala-regional la muerte regresiva puede ser lenta. Además, de las sequias prolongadas y la combinación con periodos de regeneración potencialmente lentos para los piñones, bajo un escenario cambios climáticos a futuro esto podrían resultar en una principal disminución en las poblaciones de piñonero en la región, como resultado los manejadores de estas áreas deberían considererarlo a la hora de planear futuros tratamientos de restauración en bosques persistentes de piñón-junípero.

**Key Words:** age structure, climate change, Grand Staircase–Escalante National Monument, land use change, pinyon, tree growth, tree ring, woodland expansion, woody encroachment

## INTRODUCTION

Over the last several decades it has become widely recognized that increasing dominance of woody plants over herbaceous plant species is occurring at rapid rates across a diverse range of ecosystems globally (e.g., Archer 1994; Miller and Wigand 1994; Van Auken 2000; Briggs et al. 2005; Fensham et al. 2005; Andersen and Baker 2006; Dalle et al. 2006). Of particular importance to the management of western United States public lands has been the expansion and infilling or "thickening" of pinyon pine (*Pinus* spp.) and juniper (*Juniperus* spp.) populations, a pattern observed over the last century across widely diverse climatic regimes, soil types, and understory vegetation communities (Blackburn and Tueller 1970; Miller and Wigand 1994; Miller and Rose 1999; Tausch and Nowak 1999; Wall et al. 2001; Weisberg et al. 2007).

Throughout the Holocene, pinyon and juniper species have fluctuated along elevational and latitudinal gradients—dynamics that have generally been linked to changing climate throughout this period (Van Devender and Spaulding 1979; Betancourt 1990; Nowak et al. 1994; Wigand et al. 1995; Jackson et al. 2005; Gray et al. 2006). Past pinyon-juniper range expansions generally occurred during cooler, wetter time periods with warm, dry periods resulting in woodland declines and replacement by grasslands and desert shrub communities (see review by Tausch 1999). During the last century a number of human caused changes, such as declines in the use of fire by Native Americans, the large-scale introduction of domestic livestock grazing by Euro-American settlers and subsequent increases in fire return intervals, and increasing atmospheric $CO_2$ concentrations all in combination with favorable climatic conditions are believed to have set up the conditions for rapid expansion of pinyon-juniper populations across western landscapes (Miller and Wigand 1994; Tausch 1999).

Of the changes that occurred across these landscapes during the late 19th into the early 20th century, the most extensive impacts on plant communities came from the large-scale introduction of domestic livestock grazing (Pieper 1994). The timing of the introduction of livestock in record numbers coincided with cooler, wetter conditions across the West in the early 20th century (Hereford et al. 2002; Gray et al. 2006), conditions that were favorable for tree establishment. Thus the question still remains as to whether widespread expansion and infilling across the range of pinyon-juniper woodland types was driven by past climate, changing land use with the large-scale introduction of domestic livestock, or the likely scenario a combination of these factors. Resolution of these questions has clear relevance to the management of a dominant vegetation type of the western United States.

Pinyon-juniper woodlands occur across a diverse array of climatic regimes, soil types, and understory plant communities: in southern Utah alone, there are over 20 distinct ecological site types where pinyon and juniper species are the dominant overstory component (http://www.ut.nrcs.usda.gov/technical/technology/range/ecosites.html). Thus efforts to more broadly understand the important drivers leading to the observed changes in pinyon-juniper woodland dynamics over the last century have often been confounded by the highly variable climatic regimes, understory vegetation structure, and historic disturbance regimes across these ecosystems.

Following this, our objective in this study was to compare the role of historical livestock grazing and past climate to better understand how these factors have influenced modern pinyon (*Pinus edulis* Engelm.) distributions in two upland sites (2 000–2 200 m in elevation) on the Colorado Plateau that differed in historical land use. Vegetation structure at these sites can best be described as upland "persistent" woodlands (Jacobs et al. 2008; Romme et al. 2008), which are characterized by low understory plant cover, variable tree cover ranging from small sparsely distributed trees occurring on low productivity sites to closed canopy dense stands occurring at more productive sites, and fire return intervals on century to multicentury time scales (Romme et al. 2008). As the term "persistent" suggests, these areas often maintain significant presence of pinyon and juniper species over several hundred years, and the more recent changes in the structure of these woodlands over the last century have been recruitment into interspaces between older trees or "infilling" as opposed to expansion into adjacent vegetation types. The persistent pinyon-juniper woodland type is found throughout the western United Sates and is particularly well represented across large expanses of the Colorado Plateau.

Our study was conducted at Grand Staircase-Escalante National Monument in southern Utah at a paired historically grazed and relatively ungrazed protected site. Historical photos from these sites showed dramatic tree cover increases over the last 70 years (Harris et al. 2003). Previous research at these sites showed that total tree cover (*Pinus edulis* Engelm. and *Juniperus osteosperma* [Torr.] Little) was similar across these sites (Harris et al. 2003; Guenther et al. 2004) but that pinyon cover was nearly twofold higher at the historically grazed site as compared to the minimally grazed protected site (Guenther et al. 2004). As a result, we focused our research efforts on changes in pinyon pine recruitment and growth dynamics over the last 150 years—the time period since the large-scale introduction of domestic livestock.

Here we addressed the question, is there evidence that the large-scale introduction of domestic livestock in the late 1800s resulted in increased recruitment and growth of pinyon pine in persistent woodlands? There are a number of proposed mechanisms by which trees may outcompete herbaceous plant communities under grazing, such as reduction of fine fuels and subsequent increases in fire return intervals, decreased competition for soil resources with removal of understory plants, and increasing shrub density, which serve as nurse plants for trees (Miller and Wigand 1994; Tausch 1999). If the introduction of livestock to these areas in the 1800s was an important driver of pinyon population dynamics at these sites, we would expect to observe enhanced recruitment and growth at historically grazed sites during the mid-1800s into the early 1900s as compared to the minimally grazed relict site.

Synchronous pinyon recruitment and growth dynamics across sites that differed in historical livestock grazing, however, would suggest that more regional scale factors such as climate may be the important driver of pinyon-juniper expansion over the last century. Following this, we addressed a second question: How has past climate influenced pinyon recruitment and growth dynamics? Climatic variables that influence pinyon recruitment differ from those that influence annual growth. Successful pinyon recruitment is dependent on a complex set of abiotic and biotic factors that regulate initiation of cone production through seed production and subsequent germination, a process that takes three years to complete. Cool and wet climatic conditions have been shown to be generally favorable for pinyon cone production and subsequent seed germination and establishment (Forcella 1981; Chambers et al. 1999). Thus we would expect that climate-driven pinyon recruitment events should generally occur during cool-wet time periods that persist over multiple years. In contrast to pinyon recruitment, pinyon growth has been shown to be mostly responsive to winter and early spring precipitation of the current growing season (Fritts et al. 1965; Adams and Kolb 2005).

## METHODS

### Study Site

This study was conducted in 2005 within Grand Staircase-Escalante National Monument (GSENM), which is located in southern Utah near the Arizona border. Sites were located within the Grand Staircase geographic section of the monument, which is characterized by a succession of geologic cliffs and benches that increase in elevation, crossing a range of substrates from south to north (Doelling et al. 2000). To examine the role of historical land use on pinyon pine dynamics over the last century, we located pinyon-juniper woodland sites within GSENM that differed in past land use but occurred in areas with similar substrates, topography, elevation, and climate. No Man's Mesa (NMM; lat 37°19′28″N, long 112°06′54″W) is a 715-ha wilderness study area within GSENM that has experienced very little domestic livestock grazing over the last century. The steep cliffs surrounding the mesa top make this area relatively inaccessible to domestic livestock. The only documented domestic grazing on NMM was a period of goat grazing during the late 1920s. A trail was built to the top of NMM in 1927, and 800 goats were relocated to the top of the mesa. The following spring 1 300–1 500 wether goats grazed the mesa for approximately six weeks (Mason et al. 1967). Based on the area of NMM, the stocking rate was approximately 1–2 goats $\cdot$ ha$^{-1}$ during that time period.

We paired this relatively ungrazed mesa site with Deer Spring Point (DSP; lat 37°17′56″N, long 112°10′12″W), a site located approximately 2.5 km southwest of NMM that has been grazed by domestic livestock since the mid-1800s (Guenther et al. 2004). Although exact numbers for livestock are not available for DSP, inferences may be drawn from the history of grazing in the region. Grazing was first introduced to southern Utah in the 1860s. By 1913, an estimated 113 000 sheep and 15 000 cattle were located within Kane County, a 10 641-km$^2$ area in southern Utah in which our sites are located (http://www.nps.gov/glca/naturescience/grazing.htm). These pinyon-juniper woodland sites are generally considered to be fair on the spectrum of poor to excellent in providing forage for domestic livestock.

**Table 1.** Topographic and physical soil characteristics on the minimally grazed No Man's Mesa as compared to historically grazed sites at Deer Spring Point. Values are means calculated across six transects at each site.

| | No Man's Mesa | Deer Spring Point |
|---|---|---|
| Elevation (m) | 2 139 | 2 169 |
| % slope | 5.7 | 5.7 |
| Aspect at transect stops | | |
| % north-facing | 68 | 67 |
| % south-facing | 4 | 5 |
| % east-facing | 14 | 14 |
| % west-facing | 14 | 14 |
| Soil texture | | |
| % sand | 91 | 89 |
| % silt | 6 | 7 |
| % clay | 3 | 4 |

Both NMM and DSP receive on average 360 mm of precipitation each year and are located at an elevation ranging from 2 000 m to 2 200 m (Table 1). Bedrock geology and soil classification maps in a Geographic Information System were used to establish transects on similar soil and geologic settings on NMM and DSP. Six transects were established at each of the sites. Each transect was 300 m in length with sampling points occurring at 25-m intervals. Soils are derived from eolian and alluvial Carmel-Page Formation sandstone and classified as Pinepoint soils, which are well-drained sandy soils with sand content ranging from 85% to 90% across all transects (Table 1). Pinyon pines are co-dominant with Utah juniper (*J. osteosperma*) at these sites.

### Pinyon Population Structure

Population structure of pinyons at our sites was examined by estimating density, basal area, and age across each of the transects. Absolute density of pinyon was estimated using the point-centered quarter method (PCQ). PCQ is a plotless method to estimate tree density, which is generated from "nearest neighbor" measurements and is generally less time consuming and requires fewer sampling points to adequately capture population densities compared to plot-based methods (Pollard 1971). At each transect stop, we divided the area into four quarters along a north-south and east-west line. The distance from the transect stop to the nearest tree (defined as > 4 cm diameter at the root collar) was measured within each of the quarters. Pinyon density was then calculated based on the density estimator:

$$D \approx \frac{3}{\pi\left(r_i^2\right)},$$

where $D$ is pinyon density and $r_i$ is the average distance in meters to the nearest tree across the four quarters (Pollard

1971). To avoid biased estimates of densities due to nonuniform tree density, we calculated a density estimator at each transect stop as opposed to the transect as a whole, which has been shown to be a less biased estimator of population density in nonuniform forests (Jost 1993). Tree diameter was measured at 10–15 cm height on each of the trees with a dbh tape, which converts measurements of tree circumference to diameter. These measurements were used to estimate basal area of pinyons at each site. Basal area for each tree was calculated using the equation

$$G=\pi_4^{d^2},$$

where $G$ is the basal area of the nearest pinyon in each quarter and $d$ is the diameter in centimeters. The four basal area estimates for each stop were then averaged and multiplied by the density value for that location to obtain an estimate of pinyon basal area along each transect.

Tree age was estimated using dendrochronology or tree ring dating techniques. An increment borer was used to collect cores from each of the pinyon trees within each sampling quarter ($n = 52$ at each transect). Cores were collected approximately 10–15 cm above the ground. Tree cores were analyzed at the Institute for Arctic and Alpine Research's Dendrochronology Lab, University of Colorado, Boulder. Increment cores were mounted (Stokes and Smiley 1968) and progressively sanded with FEPA (Federation of European Producers of Abrasives) 120-, 220-, 320-, and 400-grit (162, 68, 44.7–47.7, and 33.5–36.5 μm, respectively; Orvis and Grissino-Mayer 2002) sandpaper using a Bosch belt sander (Robert Bosch Corp., Farmington Hills, MI). Cores were also hand-surfaced using 400-grit and 1 200-grit sandpaper (33.5–36.5 μm). Cores were then visually cross-dated using a previously developed pinyon chronology for NMM (C. Woodhouse, unpublished data, 2003). For samples difficult to visually cross-date, undated inner sections were run against the previously developed chronology using COFECHA (Holmes 1983). COFECHA is a software program used to check the quality of the cross-dating. All matching dates found using the COFECHA program were then visually verified. We recorded the first year of secondary growth to calculate tree age. Although we made three to five attempts in the field to obtain cores that included pith, we were not always successful. We estimated dates for cores without pith by overlaying sets of concentric circles on the inner rings of the core (Applequist 1958). To correct for the time to coring height, we collected pinyon seedlings from a range of sites and soil types across the region. Average time to a coring height of 15 cm is approximately 11 yr. Once an inner ring date was obtained from the cores, 11 yr was subtracted from this date to correct for the time to coring height.

### Pinyon Growth

We examined the relationship between pinyon growth and past climate at our sites. We measured tree-ring width on a Velmex ring width measuring system at the Institute of Arctic and Alpine Research at the University of Colorado at Boulder. One pinyon tree core from each transect stop was chosen for ring width measurement. Of this subset, we chose only those trees that were older than 1950 to be in the chronology for a sample size of 62 and 52 for NMM and DSP, respectively. Raw ring widths were reported to 0.001 mm. To enhance the environmental growth signal from the pinyon populations, raw ring widths were standardized using the software program ARSTAN 6.05P (Cook 1985). Within ARSTAN, series data were first detrended with a negative exponential function, a type of growth curve observed across a range of organisms. If this growth function failed to adequately model changes in growth rates over time, then the series was modeled with a linear regression. Raw ring width was divided by the "expected" or modeled value to obtain detrended values in the series. Biweight means were then calculated for each year to create a site chronology. These values were then compared to average precipitation and temperature values for the previous October through August of the current growing season.

**Table 2.** Pinyon pine stand structure at historically ungrazed sites (No Man's Mesa) as compared to historically grazed sites (Deer Spring Point). Values are means with standard errors in parentheses. Differences in letters denote statistical differences in mean values evaluated at $P < 0.05$.

| | No Man's Mesa | Deer Spring Point |
|---|---|---|
| Density (pinyons $\cdot$ ha$^{-1}$) | 282 (113) | 295 (41) |
| Diameter at 10 cm (cm) | 19.9 (1.2) | 20.6 (1.1) |
| Basal area (m$^2 \cdot$ ha$^{-1}$) | 8.7 (3.9) | 13.2 (2.6) |
| Mean tree age (yr) | 124 (5)$^a$ | 157 (7)$^b$ |
| Average diameter growth rate (cm $\cdot$ yr$^{-1}$) | 0.20 (0.02) | 0.17 (0.02) |
| Establishment year of oldest trees | 1 589 (63) | 1 587 (59) |

### Statistical Analyses

We used a Student's $t$-test to examine the role of historical livestock grazing in structuring the pinyon populations by comparing pinyon density, basal area, and growth rates across historically grazed (DSP) and minimally grazed sites (NMM). Pearson's correlation coefficients were calculated to examine the strength of the relationship between pinyon recruitment and growth to monthly climatic variables. Divisional climate data for southeastern Utah (Division 7) for years 1896–2007 were obtained from the National Oceanic and Atmospheric Administration National Climate Data Center (http://www7.ncdc.noaa.gov/CDO/CDODivisionalSelect.jsp). SPSS 16.0 was used in all statistical analyses (SPSS 2007).

## RESULTS

### Pinyon Recruitment

The oldest trees at our sites dated to time periods before Euro-American settlement (Table 2), suggesting that pinyons have been present at these sites for more than 200 years (min/max oldest tree date across 6 transects, NMM = 1 360/1 780, DSP = 1 450/1 790). Pinyon stand structure was similar at NMM and DSP, with no observed differences in density or basal area between these sites (Table 2). Mean tree age, however, was significantly older at DSP relative to NMM (Table 2). Differences in mean tree age were primarily due to the greater number of trees at DSP that dated to time periods before 1700 as compared

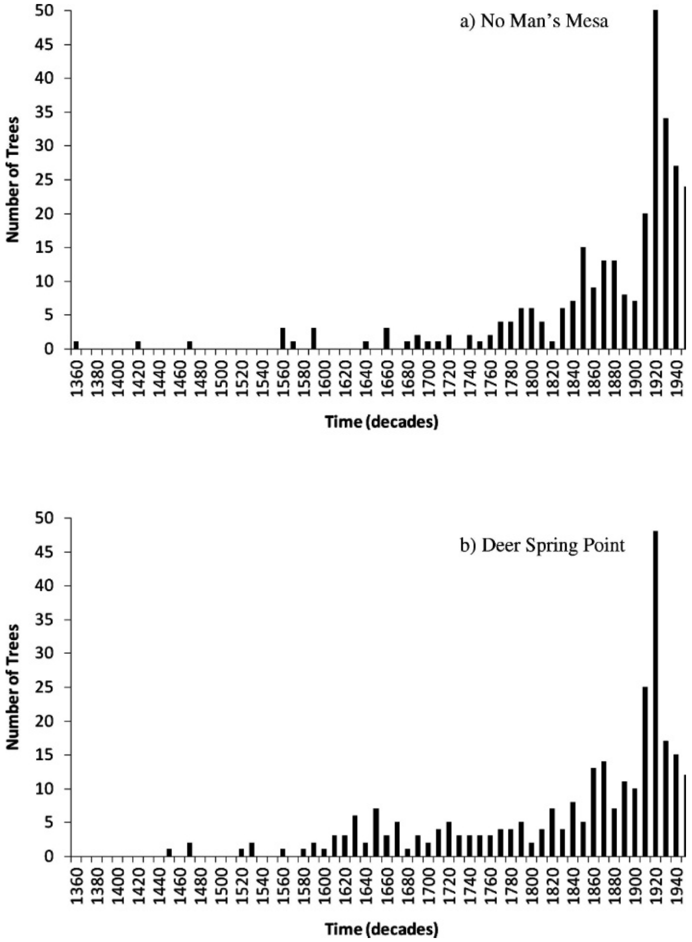

**Figure 1.** Pinyon recruitment at **a**) No Man's Mesa and **b**) Deer Spring Point. Only trees > 4 cm in diameter were aged; thus pinyon recruitment over the last several decades (1960–present) was not estimated.

to NMM (DSP = 41, NMM = 17). Although there were differences in mean tree age, the proportion of old trees dating to presettlement time periods was similar across the two sites: 36% and 38% of the trees dated to periods before the 1850s across NMM and DSP, respectively (Fig. 1).

Recruitment dynamics were highly correlated between the two sites (Fig. 2; $r = 0.88$, $P = 0.003$). Of particular interest, our data show a large pulse in recruitment during the 1920s; 17% of the trees sampled across both sites dated to the period of 1920–1929 (Fig. 1). Climatic conditions during this time period were consistently cool and wet (Fig. 3). During the period 1920–1929, average growing season temperature and annual precipitation were below and above normal in 7 and 8 of the 10 years, respectively.

## Pinyon Growth

Similar to pinyon recruitment, pinyon growth dynamics were similar at NMM and DSP. Average pinyon growth rates (tree diameter/age) between NMM and DSP were not significantly different, averaging 0.20 and 0.17 $\mathrm{cm \cdot yr^{-1}}$, respectively (Table 2). Furthermore, annual growth across these sites was highly correlated (Fig. 4; $r = 0.94$, $P < 0.001$). These data taken together suggest that the introduction of livestock did not significantly impact pinyon growth rates over the last 150 yr. Pinyon growth rates, however, were strongly influenced by winter through early summer precipitation (Fig. 5). At NMM and DSP, pinyon growth was positively correlated with precipitation levels in the previous October-to-June period and negatively correlated with June temperature (Fig. 5).

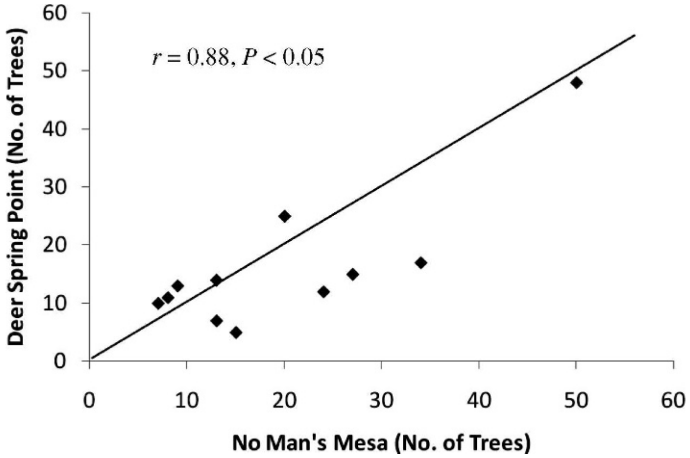

**Figure 2.** Correlation of pinyon recruitment dynamics (trees/decade) across No Man's Mesa and Deer Spring Point for the period 1850–1950, the period when heavy livestock grazing has been estimated to have occurred across the western United States. A 1:1 line is shown for comparison.

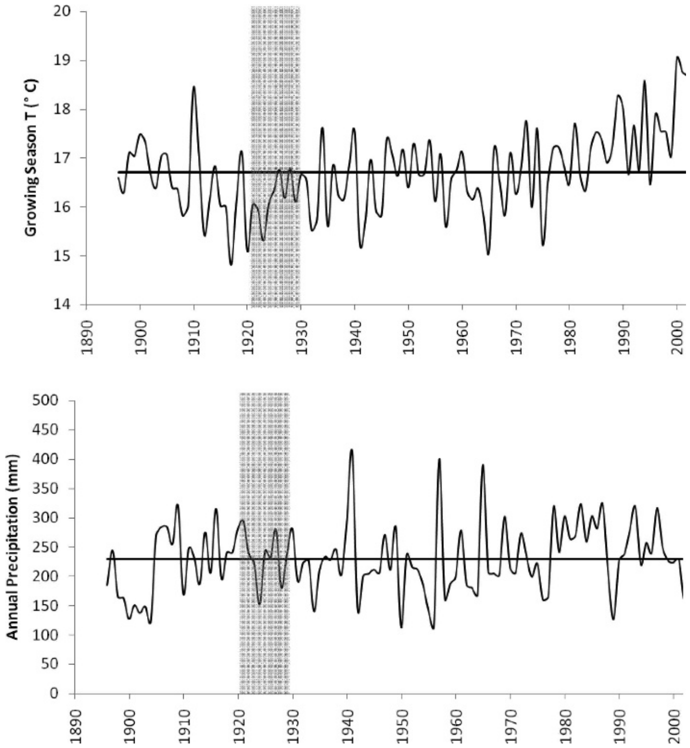

**Figure 3.** Total annual precipitation and average growing season temperature (March–August) record for southern Utah (Division 7). Solid horizontal lines represent the mean across the period of 1896–2007. The transparent gray bar indicates the time period of the large pinyon recruitment event at our sites.

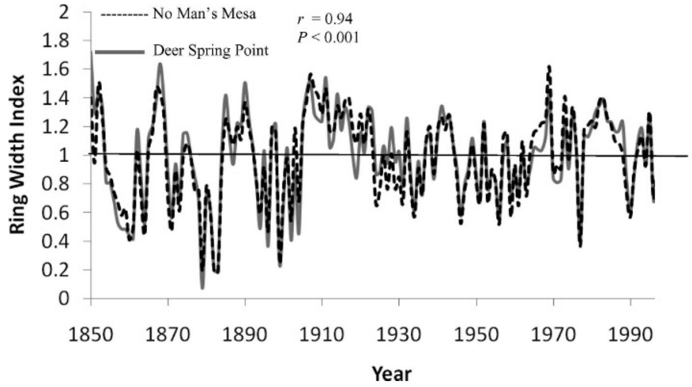

**Figure 4.** Pinyon growth from 1850 to 2005 at historically grazed (No Man's Mesa) and ungrazed (Deer Spring Point) sites. Tree ring values are standardized ring widths.

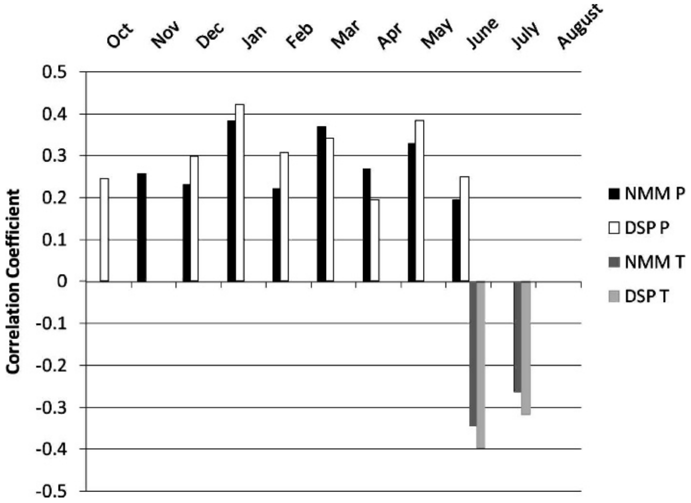

**Figure 5.** Correlation of pinyon growth to monthly climatic variables over the period 1896–2005 for the relatively ungrazed No Man's Mesa (NMM) site and the historically grazed Deer Spring Point (DSP) site. Pinyon standardized ring widths were correlated with average monthly temperature (T) and precipitation (P) during 11 months. Late fall and winter climatic variables (October–March) were previous to the current growing season. Late spring and summer climatic variables (April–August) were during the current growing season. Only correlation coefficients that were statistically significant at levels of $P < 0.05$ are shown.

## DISCUSSION

### Role of Historical Livestock Grazing in Pinyon Recruitment and Growth

Livestock grazing has clearly played a role in promoting woody expansion and infilling across a range of ecosystems globally (Cottam and Stewart 1940; Madany and West 1983; Soulé et al. 2003; Tefera et al. 2007; Schlecht et al. 2009). Historical livestock grazing, however, appears to have been less important than past climate in structuring the pinyon population within the persistent pinyon-juniper woodland sites in our study. Pinyon recruitment and growth dynamics were highly correlated across sites that differed in past land use, and evidence of a large-scale recruitment event during a cool, wet climatic period suggests that climate during the early-20th-century wet period may have played an important role in structuring the modern pinyon population. The enhanced early century pinyon recruitment dynamics in our study are similar to those reported for the Uncompahgre Plateau in western Colorado, which suggests that conditions were favorable for pinyon establishment across the region (Eisenhart 2004). Other studies have observed similar patterns in recruitment whereby recent expansion of woody plants was strongly regulated by climate and the role of historical livestock grazing was negligible (Silva et al. 2001; Fensham et al. 2005). It is important to note, however, that limited goat grazing did occur in the late 1920s on NMM (1927–1928), and this grazing event cannot be ruled out as having an influence on pinyon recruitment and growth dynamics on NMM during the 1920s.

Declines in grass and herbaceous understory under intensive livestock grazing, which may influence tree-grass competition and increase fire return intervals, have been hypothesized as important mechanisms by which trees outcompete grasses and herbaceous species (Miller and Wigand 1994). Given the low levels of herbaceous cover available for livestock grazing across persistent pinyon-juniper woodlands (Romme et al. 2008) and historic fire regimes that are characterized by stand-replacing fires on a rotation upwards of 400 yr at sites similar to ours in the region (Floyd et al. 2000, 2004), it is unlikely that the competitive environment and historic fire regime were significantly altered with the introduction of livestock in the late 1800s. Grass and herbaceous understory cover at these sites ranged from 5% to 6% in a recent study (Guenther et al. 2004), values that are in the range of what we would expect to observe in this Upland Shallow Loam (Pinyon-Juniper) ecological site (ftp://ftp-fc.sc.egov.usda.gov/UT/Range/D35XY/035XY315UT.pdf). Although fire history was not reconstructed for these sites, our stand age structure data suggest that no large stand-replacing fires have occurred across these sites for at least 200 yr. Oldest tree dates for all NMM and DSP transects were similar and ranged from 1362 to 1788 and 1437 to 1797, respectively. Results from a recent study by Floyd et al. (2008) conducted at similar pinyon-juniper sites in southern Utah showed no alteration of historic fire regimes after a century of livestock grazing and fire suppression policies.

### Importance of Climate in Regulating Pinyon Populations

In our study, pinyon recruitment was strongly synchronous across sites that differed in historical livestock grazing, suggesting that more regional scale factors such as climate were more important in structuring the modern pinyon population. Large recruitment events, such as the one that occurred in the pinyon population during the 1920s, may have a disproportionate effect on structuring pinyon-juniper woodlands. Nearly one out of every five pinyons sampled across our sites recruited in the 1920s. The period of cooler, wetter conditions began in the decade previous to the 1920s and continued into the early years of that decade. These conditions have been observed in the climate record of southern Utah (Fig. 3) and across much of the Southwest (Ni et al. 2002). This early-20th-century wet period was preceded by a multiyear drought at the turn of the century (1896–1904), which may have resulted in regional tree mortality, thus setting the stage for a regional recruitment event (Plummer 1904; Phillips 1909). Large-scale recruitment of other western forest species such as ponderosa pine (*Pinus ponderosa*) occurred across much of the

Southwest United States during this same time period (Savage et al. 1996; Brown and Wu 2005).

If multiyear cool, wet periods are needed for large-scale pinyon establishment events, what are the potential implications for future recruitment events among pinyon populations in this region given projections of warmer and drier conditions across the West? Our sites have experienced a 0.7°C average increase in growing season temperature in the last 50 years as compared to the previous half century (Fig. 3; 1900–1950 average = 16.4°C, 1958–2007 average = 17.1°C). Taken with the climate projections of increasing temperatures, more prolonged drought periods, and a general increase in aridity across the region (Christiansen et al. 2007; Seager et al. 2007), it is possible that large pinyon recruitment events, such as the one documented in the 1920s at our sites, will be less frequent in the future. Indeed, studies of *P. edulis* masting events across central New Mexico showed declines in seed production during the most recent multiyear drought (1997–2004; Zlotin and Parmenter 2008).

As we predicted, pinyon growth was positively correlated with winter through early summer precipitation. Most interestingly, pinyon growth was also negatively correlated with June temperature. This negative correlation is likely due to drought stress incurred during the driest month of the growing season before the onset of the summer monsoon, whereby higher temperatures impact available soil water through evaporative processes. In this region of the Colorado Plateau, the first xylem cells in pinyon are just being formed during the last week of May (Fritts 2001); thus environmental conditions during this period are critically important for pinyon growth. Recent regional pinyon tree ring analyses (N. N. Barger and C. Woodhouse, unpublished data [in prep], 2009) suggest that this effect of summer temperature is a trend that has been observed over the last several decades that was not apparent early in the century.

Although there has been a general trend toward expansion of pinyon-juniper woodlands over the last century, in the last several years (i.e., 2000–2004) a multiyear drought has caused large-scale dieback of pinyons throughout the Colorado Plateau (Breshears et al. 2005; Shaw 2006). Acknowledging the need for cool wet climatic conditions spread over several years to promote a large-scale pinyon recruitment, it is likely that with future climate projections of increasing temperature and more extended drought periods, future regeneration of pinyon populations following regional-scale dieback will be slow. Moreover, prolonged drought combined with potentially slow regeneration times for pinyon under future climate scenarios could result in substantial declines in pinyon populations, a result that land managers should consider when planning for future restoration treatments across these landscapes.

### Management Implications

Pinyon-juniper ecosystems are the third largest vegetation type in the United States and one of the largest vegetation types administered by federal land management agencies. To date, nearly 7% of pinyon-juniper woodlands on the Colorado Plateau, many of which occur in persistent woodland sites such as ours, have been treated over the last several decades using a variety of methods to remove trees in efforts to restore pinyon-juniper ecosystems to a more historical condition. Our data suggest that the large pinyon recruitment event in the 1920s, trees that now make up a significant overstory component in these sites, was likely a natural or climate-driven event. Additionally, nearly one-third of the pinyons dated in our study established in presettlement time periods (before 1850), which suggests that pinyons have long been established at this site. Although these persistent woodland sites have been historically grazed by livestock and there is evidence from previous studies that cover and composition of the understory vegetation has been altered, in particular increases in shrub and declines in biological soil crust cover (Harris et al. 2003; Guenther et al. 2004), a recent study of the rangeland health attributes of pinyon-juniper woodlands in GSENM suggests that pinyon-juniper sites have been relatively unaffected by historical land use (Miller 2008). If one of the overarching management goals in mechanically treating persistent pinyon-juniper woodlands is to restore these ecosystems to a more historical condition, results from our study in addition to several others conducted in this woodland type on the Colorado Plateau suggest that anthropogenic changes in grazing and fire regimes have not played a significant role in structuring pinyon-juniper populations over the last century (Floyd et al. 2000; Eisenhart 2004; Floyd et al. 2008; Romme et al. 2008). Thus the call to restore mature, persistent pinyon-juniper woodland to a more historic condition using mechanical methods and prescribed fire must be closely scrutinized.

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