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

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Photo: Dan Binkley

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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.

<u>Three General Kinds of Piñon-Juniper Vegetation</u>: 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.

<u>Potential Fire Behavior</u>: 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.

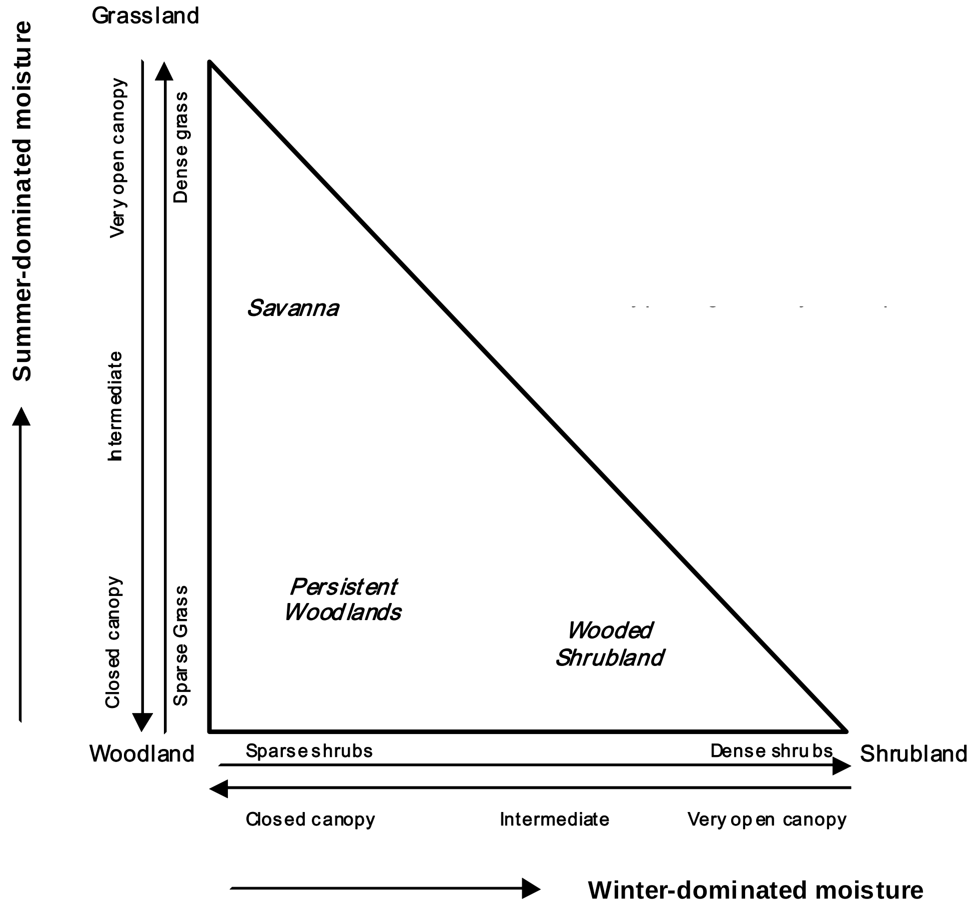

Figure 1. Generalized structure, i.e., relative proportions of trees, shrubs, and grass, and broad patterns of regional distribution in relation to gradients in seasonality of precipitation, in the three types of piñon and juniper vegetation discussed in this paper (Table 1). Note that local site conditions 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).*

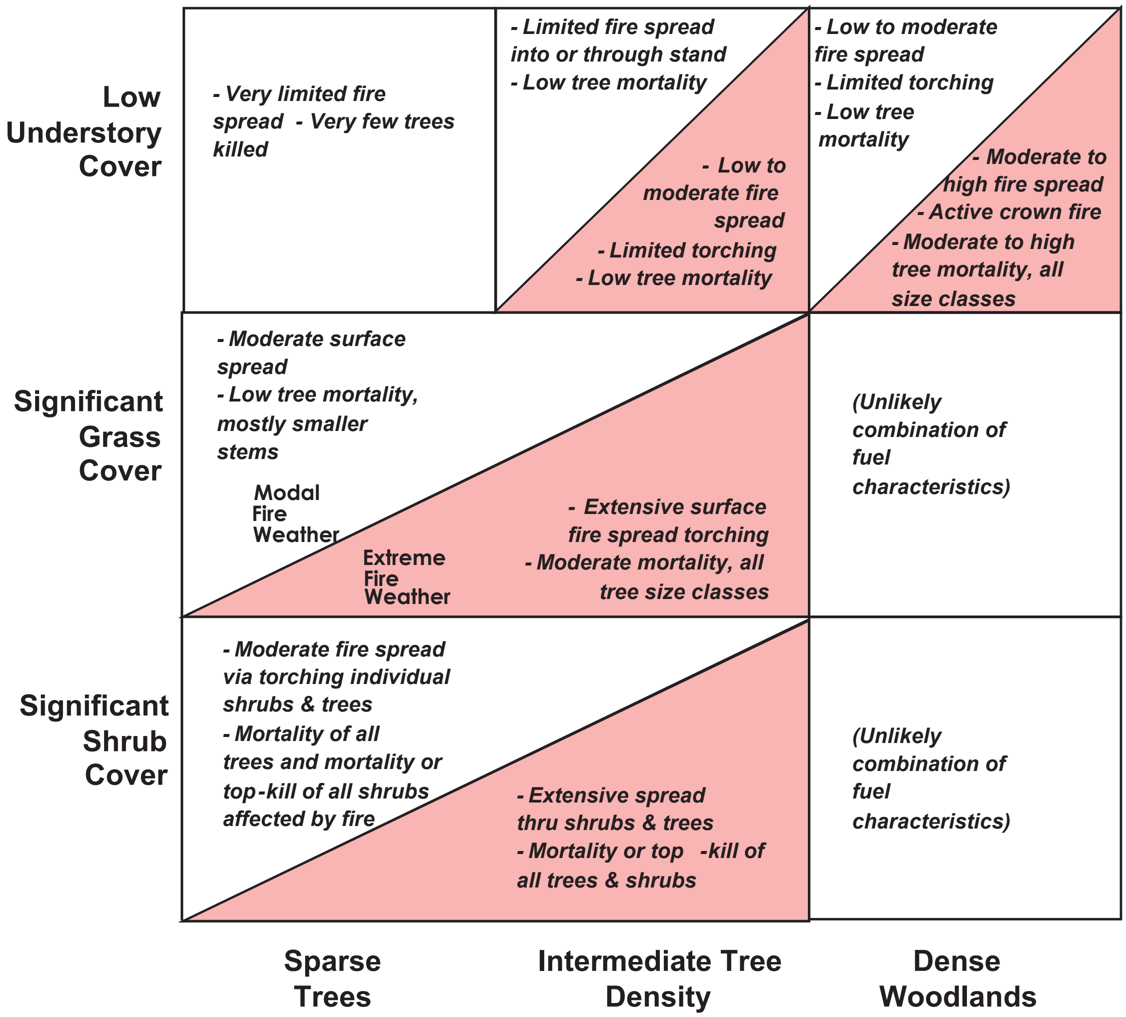

Figure 2. Probable fire behavior following a single ignition event in piñon and juniper vegetation with respect to variability in tree density (horizontal axis) and understory fuel characteristics (vertical axis). Split cells reflect variable fire behavior, spread dynamics, and tree mortality under "modal"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

<u>Kinds of Evidence</u>: *rigorous paleoecological studies, recent systematic observations of tree mortality*

<u>Explanation</u>: 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

<u>Kinds of Evidence</u>: *rigorous paleoecological studies, correlative studies, logical inference*

<u>Explanation</u>: 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.*

<u>Section IIb: What We Know About Piñon-Juniper Savannas</u>

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*.

<u>Explanation</u>: 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

<u>Kinds of Evidence</u>: *historic & recent photos, soils surveys*

<u>Explanation</u>: 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

<u>Kinds of Evidence</u>: *rigorous paleoecological studies, historic & recent photos*

<u>Explanation</u>: 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 tempera-tures, 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.*

<u>Section III: Evaluating the Mechanisms of Infill and Expansion</u>

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 vegeta-tion 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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