Piñon-Juniper Woodlands of the Western United States: Are we on the Brink of Piñon Oblivion?

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I. ABSTRACT

Piñon-juniper woodlands are a broad classification of woodland situated in the semi-arid zone of northern Mexico and the western United States containing one or more species of piñon pine and/or juniper. The changing climate patterns are creating growing challenges for management of piñon-juniper. Higher temperatures and lower precipitation associated with climate change have led to reduced growth, reproductive problems, little or no seedling germination after disturbance, and tree death. Today, mature, seed producing piñons are most affected and are dying far faster than immature understory trees can replace them. The loss of piñon-juniper woodlands will have a negative impact on ecosystem services, food sources, and indigenous cultures. Piñon-juniper distribution has been changing for millennia, however, recent trends showing range expansion and contraction is occurring at an unprecedented rate and is of concern to many people. Piñon-juniper woodlands have been managed by indigenous peoples, but the appearance of Euro-American settlers and subsequent management had the most profound impact on woodlands of today. Current management focused toward restoration techniques involving restoring a natural fire and disturbance regimes, spatial patterns, and density. It is much debated because, the wide distribution and heterogeneity of piñon-juniper woodlands likely supported a wide range of stand structures and disturbance regimes, so not all management techniques may be appropriate everywhere. Many have created classifications to describe specific woodland types; future work will focus on fine turning classifications of piñon-juniper woodlands into smaller discrete classifications to develop more site specific management guidelines. Future management will likely be focused on adapting to changing climatic conditions by assisting in natural and supplemental regeneration, hazard mitigation through fuel reduction treatments, and assisting in artificial species migration.

II. INTRODUCTION: PIÑON-JUNIPER WOODLANDS

Piñon-juniper woodlands are a broad classification of woodland situated in the semi-arid zone of northern Mexico and the American states of Nevada, Arizona, Utah, Colorado, and New Mexico, with a minor presence in Texas, California, Oklahoma, and Wyoming (West et al. 1975), covering 17-27 million hectares across the range (Barger et al. 1972, West 1988). In Arizona, nearly 20% of the state is covered in woodland, with similar figures in New Mexico (29%) and Chihuahua, Mexico (20%) (Garrison and McDaniel 1982; Perez 1978; West et al. 1975). Piñon pines are a major component of piñon-juniper woodlands in over 14 million hectares of the southwestern United States, or roughly covering half of over 27 million hectares (Barger et al. 1972; Shaw et al. 2005). There is a similar classification of woodland called juniper woodlands or juniper savannas that do not have a piñon component, covering parts of northern California, eastern Oregon, and southwestern Idaho (Pieper 2008, Chambers et al. 1999). Adding the area covered by juniper woodlands pushes the figure for total land covered by piñon-juniper woodlands in North America to well over 30 million hectares, in addition to woodland areas in Mexico (Kuchler 1970). Figure 1 illustrates

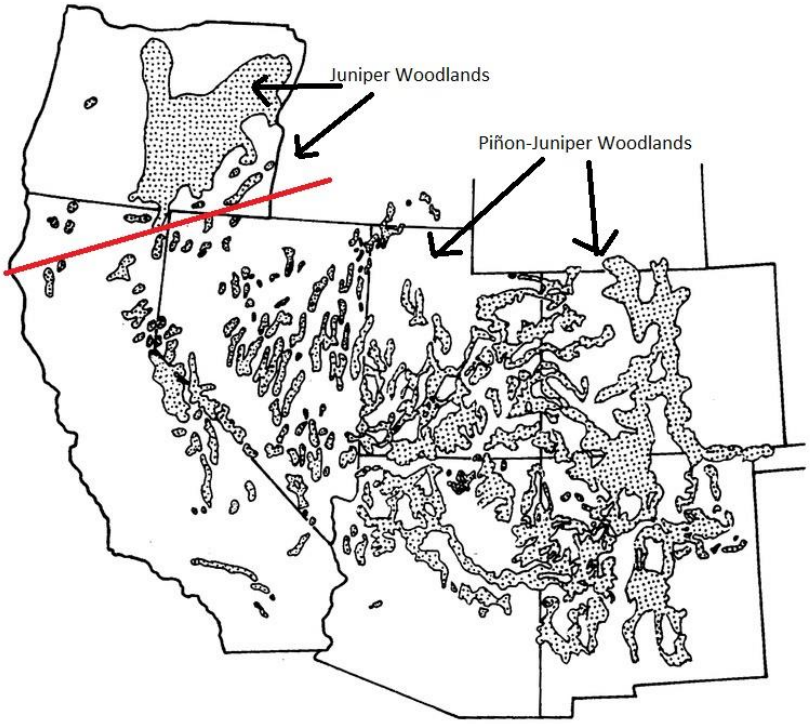


Figure 1 Distribution of piñon-juniper woodlands and juniper woodlands. Red line indicates division of juniper woodlands and piñon-juniper woodlands. Modified from Evans (1988).

the approximate distribution of piñon-juniper woodlands and juniper woodlands in the western United States.

Woodlands can be defined as consisting of short-statured overstory trees usually under 5 meters in height with relatively open canopies, and greater than 40% crown closure, however, woodlands with crown closures between 10-80% are also considered to fit the criteria of others who have attempted to categorize them (Gottfried et al. 1995; Heidorn 1994). Because of the broad definition of woodlands and heterogeneity of vegetation types which may fall under the description, many have developed categories for woodland types (West et al. 1975, Moir et al. 1987; West et al. 1998; West 1999; Thompson 1998; Jacobs et al. 1999; Romme et al. 2009). For the purpose of this paper, I am considering West's (1999) description of these areas as having at least one species each of either the drought-tolerant genera *Pinus* (subsection *Cembroides*) and *Juniperus* (section *Sabina*) and located in semi-arid habitats in the western United States. All stand compositions, including juniper woodlands and savannas, will be colloquially referred to as piñon-juniper woodlands, unless specifically noted.

Piñon-juniper woodlands host a wide array of ecosystem services and human uses (West 1999). A wide range of bird and mammal species have been identified that rely on piñon-juniper woodlands as important habitat (Miller & Wigand 1994). They have been also shown to be important to providing suitable environments for many grass and forb species, and important to watershed quality (Miller & Wigand 1994).

Piñon-juniper woodlands have constantly changed in distribution and range due to many factors including past management and fluctuations in climate patterns (Chambers et al. 1999). However, recent changes in species compositions, range, and distribution appears to be occurring at an unprecedented rate (Chambers et al. 1999). Climate change is expected to have a large and rapid impact on species compositions, woodland structure, and distribution of piñon-juniper woodland, perhaps causing some species to leave a system entirely. Landscape-scale die-offs of piñon pines related to climate change have already occurred in some areas (Floyd et al. 2009). Ramifications of such a striking change to the woodlands are largely unknown, but there will likely be negative impacts to ecosystem services and rapid alterations to ecosystem type, ecosystem properties, and land surface conditions (Breshears et al. 2005). The key to future management and persistence of piñon-juniper woodlands is understanding historic expansions, prehistoric migrations, and the effects of past human management. I will discuss geography and topography, overstory and understory structure, disturbance regimes and stand development, past and current distribution patterns, and management strategies, and conclude with a section discussing the future of piñon-juniper woodlands.

III. GEOGRAPHY & TOPOGRAPHY

Piñon-juniper woodlands are generally found on plateaus, foothills, rocky outcroppings, and lower mountain slopes occupying the warmest tree-dominated zone in the region. The woodlands are usually found ranging from 1350 meters to 2500 meters in elevation, but actual elevation may vary +/- 500 meters on extreme sites depending on topography, aspect, and geography (Gottfried et al. 1995). At higher elevations, piñon-juniper transitions into ecotypes containing ponderosa pine (*Pinus ponderosa*) or other higher elevation tree species, and transitioning to grasslands at lower elevations (Huffman et al. 2008).

Soils typical in piñon-juniper woodlands are diverse, having deep, clayey or sandy textured soils to relatively shallow and rocky soils (Romme et al. 2009; Clary 1987). They are derived from many parent substrates including granite, limestone, sandstone, volcanic, and alluvial deposits (Springfield 1976). The soils are generally poorly developed, but are very well-drained with varying compositions (Evans 1988). Typically, piñon-juniper woodlands can be found on soils belonging to the soil orders Entisol and Aridisol (Holochek 2011), however, woodlands may also be found on a variety of soils including Mollisol, Alfisol, Inceptisol, and Vertisol soils (Romme et al. 2009; Evan 1988). While most woodlands occur on poor soils, fertile soils may also support woodlands (Evans 1988). Piñon-juniper woodlands occur on a wide variety of slopes, aspects, and topographies (Gottfried et al 1995). Combined with soil types, physical location, and inappropriate management and land use (i.e. overgrazing, chaining, etc.), soils typical in younger piñon-juniper woodlands are highly erodible due to relative lack of understory cover (Davenport et al. 1998; Wilcox 1994; Hasting et al. 2003). However, the oldest stands are located on rocky and rough terrain with relatively absent understory vegetation, allowing trees to escape fire for centuries (Swetnam et al. 1992).

i. Climate

Piñon-juniper stands occupy the lowest elevation wooded zone in the region (Ronco 1990), but can cover a wide range of temperature and moisture regimes (Romme et al. 2009). Chambers et al. (2008) described woodlands as semi-arid, receiving between 18-55 centimeters of precipitation per year (Gottfried et al. 1995, Chambers et al. 2008; Romme et al. 2009). Precipitation amounts are largely influenced by shifts in sea temperatures (Enfield et al. 2001). These shifts in ocean temperatures can be short periods spanning several years in patterns such as

El Niño (Ropelewski 1987), or in some cases, trends may last decades such as shifts associated with the Pacific Multidecadal Oscillation or the Atlantic Multidecadal Oscillation (Enfield et al. 2001). These factors often contribute to long lasting and widespread drought throughout the range of piñon-juniper woodlands, but may also lead to widespread recruitment in favorable years.

Romme et al. (2009) describes the high seasonality of precipitation as a northwest-to-southeast gradient. Areas of southern Arizona and New Mexico in the southeastern portions of the range have strong summer monsoonal moisture; a bimodal summer-winter pattern on the Colorado Plateau; and a winter-spring pattern in the Northwest and Great Basin (Mitchell 1976; Jacobs 2008). Precipitation can further be affected locally by geography, aspect, and slope (Gottfried 1995).

ii. Historical and Future Ranges

Piñon-juniper woodlands have changed drastically in range and density since the late Pleistocene and into the Holocene period (Miller & Tausch 2000; Miller & Wigand 1994). Piñon-juniper woodlands are largely affected by the long-term climate trends of temperature and precipitation (Thompson 1990; Miller & Tausch 2000). Studies of plant fossils, fossilized pollen, packrat middens, and dendrochronology (Miller & Wigund 1994; Ernst & Pieper 1996) have provided clues to the understanding of the current distribution and structure of piñon-juniper woodlands (Betancourt 1987).

Beginning about 10,000 years ago, the warming climate allowed juniper and piñon species to begin occupying higher elevations and moving northward in latitude (Betancourt 1987). Piñon-juniper woodlands replaced higher elevation mixed conifer forest types that had occupied the areas in the preceding 100,000 years of glaciation (Miller & Tausch 1994). During this time, piñon species migrated onto the Colorado Plateau from their refugia in the Chihuahuan and Sonoran deserts (Betancourt 1987). Junipers were also on the move during this time, spreading from the southern regions into the north and central Great Basin (Wigand et al. 1995).

The period of around 8,000-4,000 years ago (mid-Holocene) was a warm and dry period where the piñon-juniper woodlands occupied sites over 500m higher than currently found (Wigand et al. 1995; Jennings and Elliot-Fisk 1993). It was during this time when western juniper migrated into southeastern Oregon and northern California, forming the beginnings of the present-day juniper woodlands (Wigand 1987; Mehringer & Wigand 1990). During the mid-Holocene period, piñons also reached their furthest north distribution, extending beyond their current distribution (Betancourt 1987). Woodland distribution and abundance fluctuated continuously due to changes in climate. Toward the end of this period around 5,000-4,000 years ago, precipitation increased, allowing the western juniper to reach the northern-most areas of the Great Basin (Davis 1982, Mehringer 1986, Wigand 1987). The ideal climate spurred mass expansion of piñon and juniper woodlands across the Colorado Plateau and the Great Basin (Miller & Wigand 1994). Consequently, the period of favorable climate triggered rapid increases in grasses and forbs in the understory, preventing development of dense closed stands. The structure of these woodlands was fairly open with a dense herbaceous understory (Miller & Wigand 1994).

The mid-Holocene transitioned into the Neoglacial period around 3,000-2,000 years ago where wetter and cooler conditions prevailed (Davis 1981, Wigand 1987, Wigand et al. 1995). Western juniper woodlands and piñons expanded into much of their current range during the neoglacial period (Wigand et al. 1995). Distribution and density increased in the southern areas where the woodlands had previously occurred and are believed to resemble stands of today (Davis 1981, Wigand 1987).

The late Holocene began around 2,500 years ago and continued roughly until the appearance of European settlers around 140 years ago. The time span began with major droughts sparking large fires (Davis 1982, Wigand 1987; Chambers et al. 1998; Miller et al. 2001). Piñon-juniper woodland distribution and abundance was largely impacted by changing climate and subsequent frequent large fires, as shown by the presence of charcoal layers in sediment and pollen cores (Wigand et al. 1995; Miller et al. 2001). Shifts in climate and disturbance lead to widespread decline of junipers and perennial grasses (Wigand et al 1995). The Great Basin experienced widespread erosion with sediment deposits accumulating in drainage channels and alluvial fans during the beginning of the late Holocene (Miller et al. 2001). The resulting topography and soil structure continue to influence plant distribution and communities (Chambers et al. 1998). The Medieval Climate Anomaly (1,500-1,100 years ago) caused increases in summer precipitation and grass cover (Trouet et al. 2009). Woodlands increased in total area covered, both down in elevation and north in latitude, with piñon gaining considerable ground in this time (Wigand et al. 1995). A drying period followed, reducing woodland distribution again until the Little Ice Age began around 700 years ago. The Little Ice Age was cooler and wetter than the rest of the Holocene with upper tree lines in the California Sierra Nevada Mountains lower than even the Neoglacial period (Woolfenden 1996). Cooler and wetter conditions caused an increase of herbaceous species during this time, likely supporting higher fire frequencies, limiting overall woodland abundance and distribution (Wigand et al. 1995; Miller & Rose 1999; Wigand 1987; Miller & Wigand 1994).

The end of the Little Ice Age coincided with the appearance of the European settlers in the West and a general warming trend followed (Ghil & Vautgard 1991; Woolfenden 1996). The 20th Century had similar climatic conditions to what followed the Neoglacial period (Miller & Tausch 2002). However fire suppression since the 1800s has virtually excluded fires from the system, contrasting with the large increase in fire experienced following the Neoglacial period (Wigand et al. 1995; Miller et al. 2001). This has led to a rate expansion of piñon and juniper woodlands that has not occurred since the end of the Ice Age about 10,000 years ago (Miller & Wigand 1994; Tausch 1999). Expansion of piñon-juniper woodlands has been studied through field observations and repeat photography (Johnsen 1962; Blackburn & Tueller 1970). This phenomenon has been mostly attributed to overgrazing, fire exclusion, and climate change (Johnsen 1962; Blackburn & Tueller 1970; Yorks et al. 1994; Barger et al. 2009; Romme et al. 2009).

Climate change is expected to dramatically change species composition, structure, and distribution of piñon-juniper woodlands. Most areas in the West are expected to be heavily affected by climate change. The areas piñon-juniper woodlands occupy will experience higher temperatures, and may receive less precipitation, resulting in large shifts of piñon-juniper ecotones (Seager et al. 2007; Williams et al. 2012). These rapid shifts have already been observed in a piñon-juniper—ponderosa pine ecotone in northern New Mexico during a recent severe drought (Allen & Breshears 1998). Conversely, periods of drought may cause range expansion of juniper species into adjacent grasslands as demonstrated in areas of the Columbia Plateau (Miller & Rose 1995). Colorado piñon pine distribution is predicted to be severely affected by climate change, experiencing a large reduction in distribution by the end of the century, even disappearing altogether from many areas where it occurs now (Thompson et al. 1998; Cole et al. 2008). While there has been some research conducted on future mortality under climate change, there is limited information on how microsite, land use history, and past disturbance will affect future piñon-juniper stands under climate change.

iii. Responses to environmental gradients

Seasonality of precipitation and available soil moisture are important factors in determining woodland composition, stand conditions, density, and distribution (Gottfried et al. 1995). Elevation is an important factor in determining species composition and stand structure in piñon-juniper woodlands (West 1999). As elevation increases, percentage of vegetation cover increases (Pieper & Lymbery 1987). Lower elevations are warmer and drier, favoring an increase in juniper abundance and decrease in abundance of the less drought-tolerant piñon (Koepke and Kolb 2013). Tausch et al. (1981) explains that competitive superiority of juniper over piñon allows junipers to persist in lower and drier areas, but also higher and colder sites. This leaves piñon confined to the middle elevations of much woodland's range (Tausch et al. 1981; West 1999), however, depending on species present and site, piñon pines may dominate the upper elevations (Gottfried et al. 1995). Seasonal precipitation gradients are a main driver of species composition (Jacobs 2008), with higher elevation piñon-juniper creating an ecotone with ponderosa pine (*Pinus ponderosa*) and even mixed-conifer species; at lower sites, the woodland transitions to semi-arid shrubland or grassland (Jacobs 2008).

IV. WOODLAND STRUCTURE

Piñon-juniper woodlands have a highly varied understory and overstory composition. Piñon-juniper woodlands are characterized by the presence of at least one or more species of drought-resistant piñon pine (*Pinus spp.* subsection *Cembroides*) or juniper (*Juniperus spp.* section *Sabina*), however, absolute composition varies greatly across the range of the woodland (West 1999). Both junipers and piñons are extremely slow growing, routinely taking nearly 500 years to reach full size (Gottfried 2004). Sporadic regeneration cause piñon-juniper woodland stand ages to range widely from recently established even-aged stands to uneven-aged stands, with individual specimens over 1000 years old reported (Romme et al. 2009; Romme et al. 2002; Floyd et al. 2000).

Even though it may take a few centuries for piñons and junipers to reach full maturity, stands with individuals over 500 years of age are rare. The vertical stand structure is varied and does not follow conventional models for age and size (Tress and Klopatek 1987). Piñon-juniper woodlands have episodic regeneration due to past disturbance and climate patterns, leading to highly patchy stand structures (Evans 1988; Clifford et al. 2011; Tress and Klopatek 1987).

Romme et al. (2009) describes piñon-juniper woodlands as falling into three generalized categories that are related to gradients of soil moisture available to trees in relation to seasonality of the precipitation. Savannas are typically found in drier locations, receiving much of their precipitation in the summer (Romme et al. 2009). These areas have relatively open stand structure, usually with dense grass and forb understory. Wooded shrublands are also found in drier locations, but receive much of their moisture in winter, resulting in a denser stand structure with grasses and forbs, but genrally smaller statured trees than persistent woodlands (Romme et al. 2009).

Persistent woodlands are the densest stands found on the wettest sites, with the high tree density resulting in minimal understory of grasses and forbs (Romme et al. 2009).

i. Species composition and distribution

Piñon-juniper woodland species composition follows a similar northwest-to-southeast gradient as Romme et al. 2009 describes as being related to occurrence of monsoonal moisture. Figure 2 shows the distribution of select overstory piñon and juniper species in relation to seasonal precipitation patterns. Composition of the piñon and juniper species is a heterogeneous mix across the woodlands. Subtle physiological differences amongst populations and species have a profound ecological significance across regional and elevational gradients (Nowak et al. 1999; Moore et al. 1999).

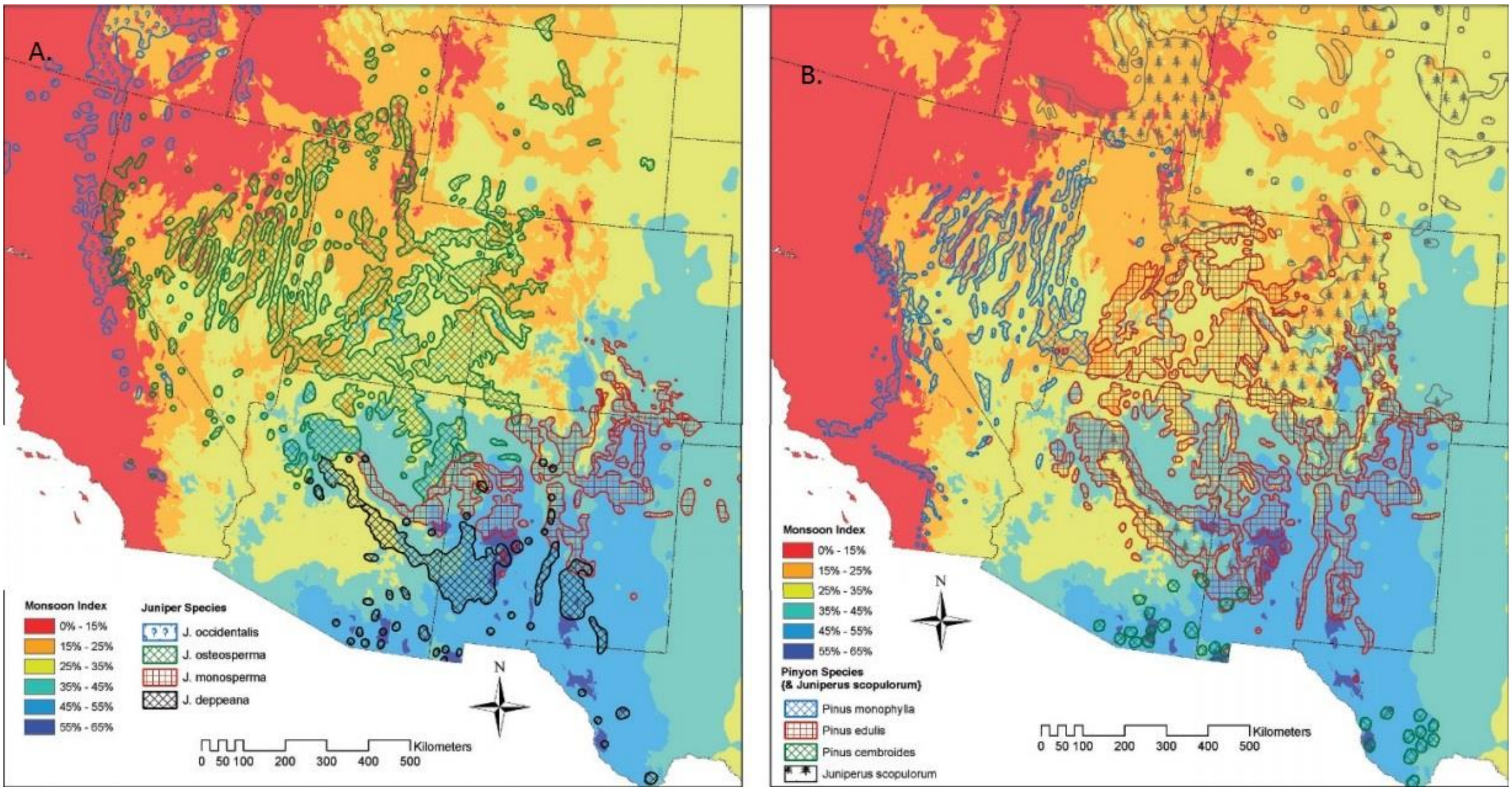


Figure 2: A.) Describes distribution of select juniper species in relation to seasonal precitation patterns. B.) Describes select piñon species in relation to seasonal precipitation patterns. Monsoon index refers to percentage of yearly precipitation received through monsoonal moisture. Adapted from Romme et al. (2009).

All of these species do not occur within every area, but can be described in overlapping ranges with distinct species compositions. These distinct compositions fall into five general ecological provinces (Fig. 3) as described by McLaughlin (1986)—the Colorado Plateau, Rocky Mountains, Great Basin, Colombia Plateau, and Mogollon Rim provinces. The table below (Table 1) shows the generalized distribution of piñon and juniper species across the five ecological provinces.

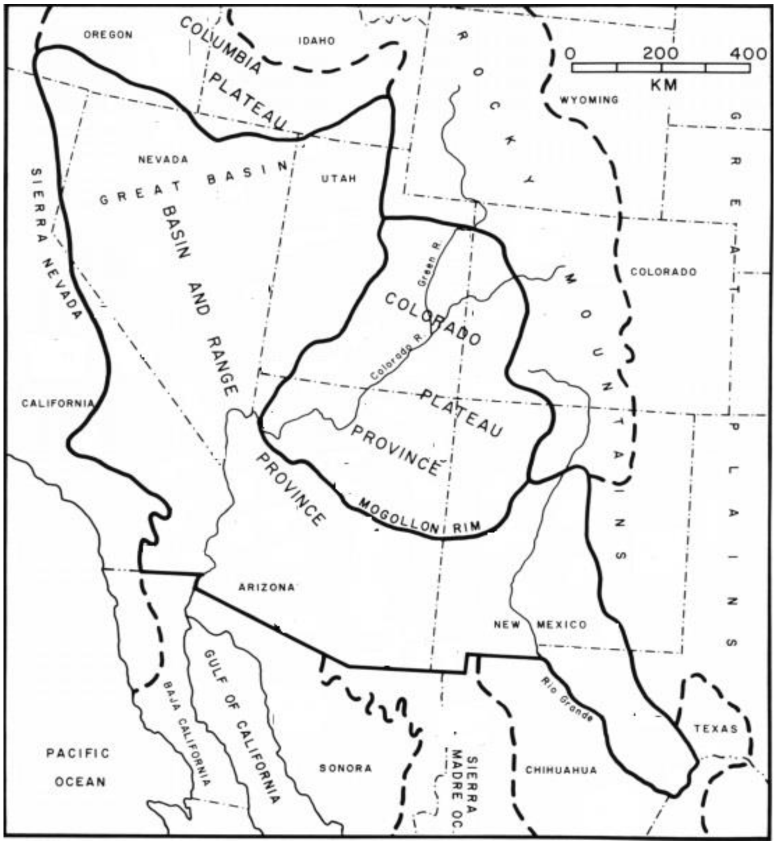


Figure 3 Generalized ecological provinces in the West. Adapted from McLaughlin (1986)

Distribution of piñons and junipers across 5 ecological provinces described by West (1999) & Malusa (1992)			
Ecological Province	Pines	Junipers	Others
Great Basin/Range	*Pinus monophylla* *P. californiarum* subsp. *californiarum*	*J. osteosperma* *J. occidentalis* var. *australis*	
Mogollon Rim	*P. californiarum* subsp. *fallax* *P. edulis* *P. discolor* *P. californiarum*	*J. monosperma* *J. deppeana* *J. scopulorum* *J. osteosperma*	*Quercus* spp. *Cerocarpus* sp. *Arctostaphylos* spp *Cupressus arizonica*
Columbia Plateau		*J. occidentallis* var. *occidentallis*	*Artemesia* spp.
Rocky Mountains	*P. edulis*	*J. monosperma*	
Colorado Plateau	*P. edulis*	*J. osteosperma* *J. monosperma*	

Table 1 Distribution of piñons and juniper across 5 ecological provinces described by West (1999) and Malusa (1992)

ii. Piñon Pine Species

Piñon pines are a drought tolerant group of pines in the family Pinaceae belonging to the genus *Pinus* subsection *Cembroides.* Piñons are relatively small trees reaching a height of 3-16 meters and a diameter at breast height of 15-75 centimeters at full maturity (Howell 1940). Piñons are extremely slow growing trees, typically growing 10-15 centimeters a year in height; at this rate it takes 75 to 200 years to fully mature reproductively (Floyd et al. 2009). Piñons may begin producing cones around 35 years old, however, reliable seed production normally does not occur for 100 years. Once producing seed, trees may produce viable seed for centuries (Chambers et. al 1999). Seeds are mainly disseminated by several species of bird including Steller's, piñon, and scrub jays, and Clark's nutcracker (Balda 1987). The birds cache large amounts of seeds underground in ideal germination conditions (Balda 1987). Piñons are very long lived trees routinely reaching 500 years in age; specimens of *Pinus edulis* over 1,000 years old have been reported in parts of Dinosaur National Park in Colorado (personal communication, M. Lisa Floyd-Hanna, February 2015).

Table 2 Major piñon species found in the west. Derived from West (1999) and Malusa (1992). Taxonomy verified with USDA PLANTS Database (http://plants.usda.gov/)

List of Piñon Species described by West (1999) & Malusa (1992)	
Common Name	**Scientific Name (syn.)**
border piñon	*Pinus discolor*
singleleaf piñon	*P. monophylla*
California piñon	*P. californiarum* subsp. *californiarum* *syn. (P. monophylla* subsp. *californiarum)*
Arizona single leaf piñon	*P. californiarum* subsp. *fallax* *syn. (P. monophylla* subsp. *Fallax)* *syn. (P. edulis* var. *fallax)*
Colorado piñon	*P. edulis*
Mexican piñon	*P. cembroides*
Parry piñon	*P. quadrifolia*

There are seven species of piñon pine that occur in allopatric distribution in the western and southwestern United States (Aldon and Springfield 1973). This means these species arose from a common ancestor after vicariance. There are also at least seven species of piñon pines in Mexico (Bailey & Hawksworth 1987, Zavarin 1987), but these will not be discussed here. Research done on piñon pines has mostly been focused on describing new species and work in detailing the magnitude of divergence between the currently accepted taxon (Zavarin et al. 1985, Zavarin 1987, Flores-Rentería et al. 2013). Although many of the species of piñon pines have distinct ranges and characteristics, their distributions may overlap, creating hybrids (Christensen et al. 1995; Chambers et al. 1999). This has resulted in great debate over correct classification, ranges, and subspecies in the taxonomic world (Malusa 1992; Chistensen et al. 1995; Lanner & Van Devender 1998; Romme et al. 2009). Figure 5 shows distribution areas of piñon pines species across the West.

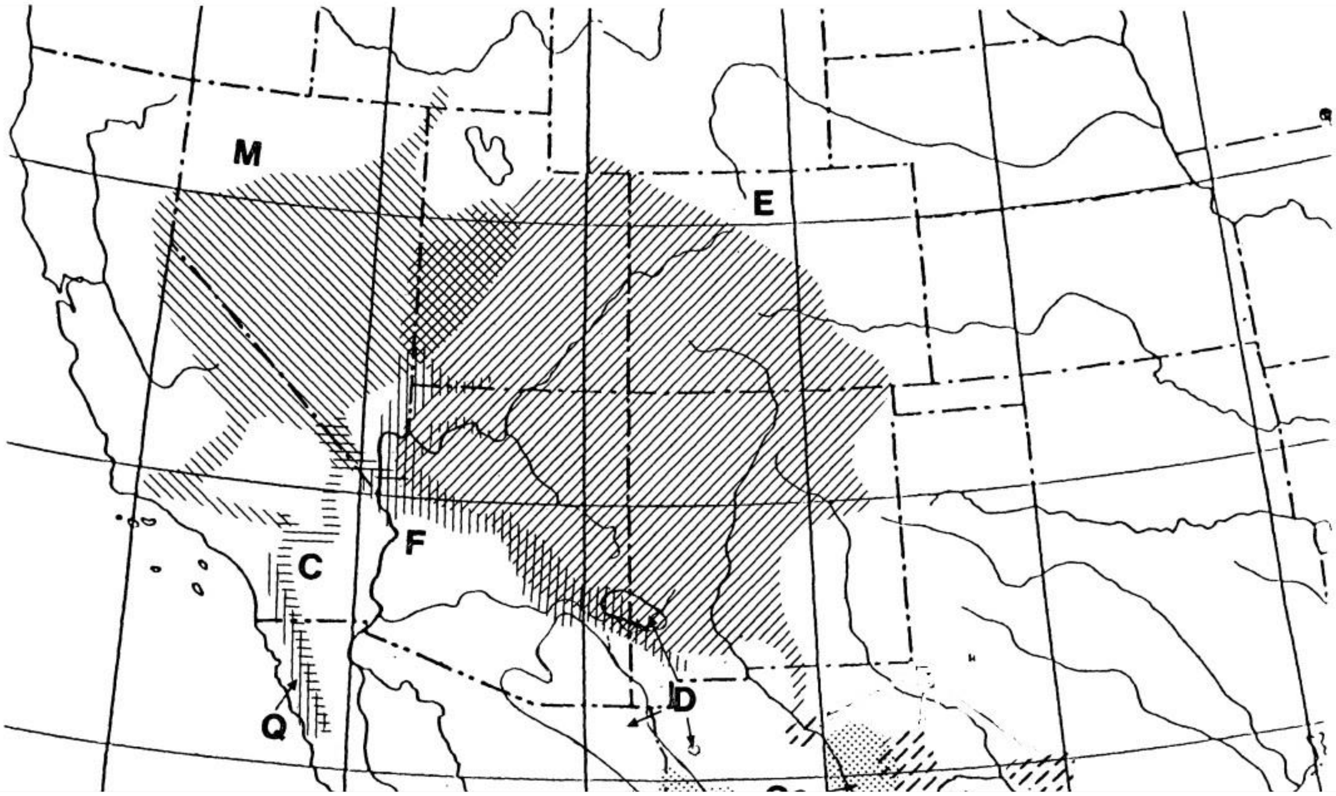


Figure 4 Distribution of select piñon pines; M=P. monophylla, E=P. edulis, Q=P. quadrifolia, C=P. californiarum, F=P. californiarum var. fallax, D=P. discolor. Source: Malusa (1992). Taxonomy verified with USDA PLANTS Database (http://plants.usda.gov/)

iii. Juniper Species

Junipers are a diverse group of woody coniferous shrubs and short statured trees belonging to the family Cupressaceae in the genus *Juniperus.* Junipers are found throughout the northern hemisphere, with the genus containing upwards of 70 species. Although as many as 17 species and varieties of junipers may be found in North America (West 1999), there are six main species of junipers in piñon-juniper woodlands of the western and southwestern United States (Miller & Wigand 1994; Pieper 2008, USDA PLANTS Database). These New World species are part of the section *Sabina* (USDA PLANTS Database).

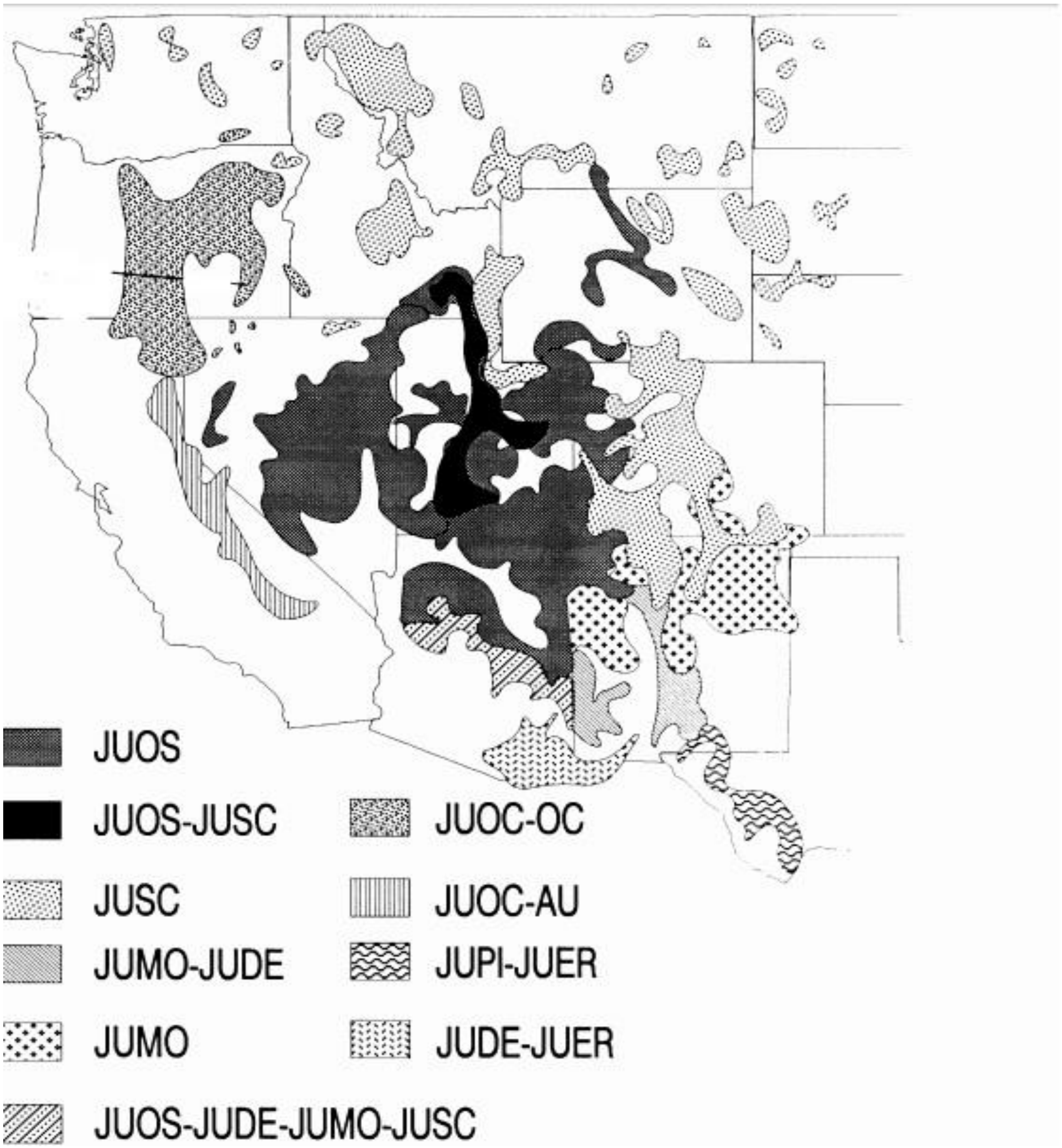
Junipers are usually at least a minor component of piñon-juniper woodlands, but are also the defining characteristic of the juniper woodlands of the northern Great Basin and the Northwest (Franklin & Dyress 1988). Juniper woodlands lack a piñon component and cover over a million hectares across eastern Oregon, southern Idaho, and northern California (Dealy et al. 1978, Cronquist et al. 1972). The species represented is western juniper (*Juniperus occidentalis* var. *occidentalis*), favored over piñons in the cooler summers and wet winter/dry summer climate (Mitchell 1976). Juniper woodlands generally contain numerous understory sagebrush (*Artemesia* spp.) and grass species (Miller & Rose 1995).

Table 3 List of juniper species described by West (1999). Taxonomy verified with USDA PLANTS Database.(http://plants.usda.gov/)

List of *Juniperus* Species described by West (1999)	
Common Name	**Scientific Name**
Utah juniper	*Juniperus osteosperma*
western juniper	*J. occidentalis* var. *occidentalis*
Sierra juniper	*J. occidentalis* var. *australis*
alligator juniper	*J. deppeana*
oneseed juniper	*J. monosperma*
Rocky Mountain juniper	*J. scopulorum*

Junipers found in piñon-juniper woodlands are coniferous small trees or large multistemmed shrubs, rarely over 12 meters in height and 250 centimeters in diameter at root collar (Gottfried 2004; Adams 1975). Individual plants may be monoecious or dioecious and feature awl- or scalelike foliage (Adams 1975). Small female seed cones represent berry-like fruits; not true fruits, but a seed cone covered by a fleshy or semi-woody covering (Ciesla et al. 1998). Holthuijzen and others (1987) have described seed dispersal as primarily by birds.

Figure 5 Distribution of junipers in the West; JUOS = Utah juniper, JUSC = Rocky Mountain juniper, JUMO = one-seeded juniper, JUDE = alligator juniper, JUOC-OC = western juniper var. occidentalis, JUOC-AU = western juniper var. australis, JUPI = Pinchot's juniper, JUER = redberry juniper. Taken from Miller and Wigand (1994), they partially derived from Critchfield and Little (1966). Taxonomy verified with USDA PLANTS Database.



iv. Understory

Because of the wide distribution of piñon-juniper woodlands across the West, a diverse palette of understory shrubs, forbs, and grasses can be found. West (1999) describes precipitation patterns as a main driver in understory composition and structure. Therefore, understory forbs and grass composition and structure are similar to surrounding ecosystems and ecotones at lower and higher elevations and are much more complex than the overstory (West et al. 1998; West 1999). Generally, understory cover can be described as a function of total canopy cover (Pieper 1990). As total canopy cover increases, total understory biomass decreases (Pieper 1990; Jameson 1967; Short et al. 1977). Likely factors that contribute to the reduction in understory biomass are litter accumulation, reduced light resources with a denser canopy, and competition for water and other soil nutrients (Jameson 1967). Other forest types in the western United States have been observed to have similar patterns of understory growth (Everett et al. 1983). Plant communities in these areas are complex, usually not having every species in every area, creating a very complex matrix of patches within the woodlands. Generally, more understory is present in north facing aspects as opposed to south facing slopes (Everett et al. 1983).

The western juniper woodlands of the Columbia Plateau and piñon-juniper stands of the northwestern Great Basin are associated with a mixture of cool-season bunchgrasses and a major shrub component (West 1999; Romme et al. 2009). Understory plants here are derived from the Arcto-tertiary Geoflora, where trees were once dominant (Axelrod 1976). Winter moisture followed by dry summers favors shrub species able to tap deep water such as *Artemisia* spp. (Section *Tridentatae*), *Purshia tridentata*, *Chrysothamnus* spp., *Ericameria* spp., and *Cerocarpus* spp. Plants completing their life cycle before the dry season such as the cool-season bunch grasses include: *Festuca idahoensis*, *Pseudoegneria spicata*, *Achnatherum*, *Poa secunda*, and *Poa ferndleriana* (Romme et al. 2009). Dominant forb species may belong to *Lupinus*, *Penstemon*, *Castelleja*, *Balsamorihiza*, and *Allium* (Romme et al. 2009; Moir 1979; Pieper 1992).

Areas in the rest of the range including the Mogollon Rim, Colorado Plateau, and northern areas of Mexico receive a large proportion of annual precipitation during the summer monsoon rains. A large number of suffrutescents such as *Senecio longilobus*, *Guitierrezia* spp., *Brickellia* spp., *Halopappus* spp., *Artemisia* spp., and *Salvia*. spp may be found (West 1999; Romme et al. 2009). Succulent and similar plants belonging to the Cactaceae family such as *Opuntia* spp. and *Cylindropuntia* spp. may be found; plants belonging to Asperagaceae including *Agave* spp., *Nolina* spp., *Daslyrion* spp., and *Yucca* spp. may be found in this area also (West 1999). Warm season sod-forming grasses and bunch grasses pepper the understory from surrounding semiarid grasslands or southern short and mixedgrass prairies (West 1999). These warm season grasses may include species from the genera *Aristada, Digitaria, Eragrostis, Bouteloua*, *Hialria*, *Sporabolus, Muhlenbergia*, and *Lycurus* (West 1999; Romme et al. 2009).

Some forbs in this area are a heat-adapted groups of plants, remnants of the Madro-tertiary Geoflora (Axelrod 1958). Examples of these genera are *Croton, Euphorbia, Ipomoea, Solanum*, and *Polygala* (Pieper 1992; Manzanares et al. 1998). A large number of annual plant varieties exist, but numbers can vary widely year to year due to fluctuations in local climate or may nearly lack an entire understory due to overgrazing and degradation (Threshow & Allan 1979).

v. Overstory density & spatial patterns

Density and spatial patterns of the overstory can be explained by past disturbance, management, and climate. The wide distribution and varied terrain give rise to a wide variety of densities and spatial patterns (Romme et al. 2009). The advancement of junipers into surrounding grasslands in the last 150 years has been studied and documented (VanAuken 2009), but others have shown reduction in area covered or density in some areas (Shaw et al. 2005; Floyd et al. 2009).

In the juniper woodlands of the Northwest, area occupied by juniper has increased 150-600% since the mid-1800s, showing no evidence many juniper woodlands were ever as dense as they are today (Miller et al. 2008). Before the start of this invasion, junipers over 140 years old were scattered in low density with few young trees; today, only 16-67% of stands contain trees from before 1860 (Miller et al. 2008). The Great Basin has experienced similar range expansion of piñon-juniper woodlands, but studies have shown relatively low establishment in the last half of the 20th century (Betancourt et al. 1993; Mueller et al. 2005; Miller et al. 2008).

The Colorado Plateau and Mogollon Rim have shown a neutral net change in total area covered by piñon-juniper woodlands, but there has been extensive increase in density and area in some places, and contraction and die-offs in others (Romme et al. 2009). Studies have shown increases in densities of woodlands, with many areas infilled by Colorado piñon (Floyd et al. 2004; Shinneman & Baker 2009). Densities in some areas of northern New Mexico and Southern Colorado have similar densities compared to photographs taken in the late 1800s (Romme et al. 2009). However, climate change-driven mortality due to drought-induced stress has caused a reduction in density and total area covered (Floyd et al. 2009).

V. DISTURBANCES & STAND DEVELOPMENT IN PIÑON-JUNIPER WOODLANDS

Many factors contribute to current stand dynamics in our piñon-juniper woodlands (Romme et al. 2009); however, climate is thought to be a main driver (West & Van Pelt 1987). The region where piñon-juniper woodlands exists is characterized by periodic sustained drought (Miller & Wigand 1994), leading to widespread drought-induced mortality (Shaw et al. 2005). Increases in the greenhouse gases associated with climate change are projected to increase intensity and duration of droughts (Hoerling & Kumar 2004).

The roles of disturbances such as insects, climatic changes, diseases, and fire regimes are not totally understood or agreed upon (Romme et al. 2009; Floyd et al. 2009). However, heterogeneity of piñon-juniper woodlands are at least partly due to fire regimes and land use by humans (Gifford 1978; Miller & Tausch 2000; Hastings et al. 2003)

i. Drought and Insect Interactions

Most mortality in piñon-juniper woodlands is due to a web of interactions between drought, diseases, and insects (Allen & Breshears 1998; Shaw et al. 2005). While there was extensive mortality in the 1980s and 1990s, a majority of mortality has occurred since 2000 (Hicke et al. 2016). Landscape-scale mortality in piñon-juniper woodlands are expected to increase in the future due to climate change, causing dramatic shifts in ecotones (Breshears et al. 2005). A drought in the 1950s in the Southwest caused a rapid shift in ecotone due to *P. pondersosa* mortality (Allen & Breshears 1998).

Beginning in 1998, a prolonged drought caused widespread piñon pine mortality in the Four Corners states (McPhee et al. 2004; Floyd et al. 2009). Most mortality occurred from 2002-2004 in Colorado piñon (*P. edulis*) (Anhold & McMillin 2003). Ultimately, 1.2 million hectares were affected in central and northern Arizona, northern New Mexico, Utah, and southern Colorado, with some areas reporting up to 100% mortality, primarily in piñons (Shaw et al. 2005; Anhold & McMillin 2003; Floyd et al. 2009).

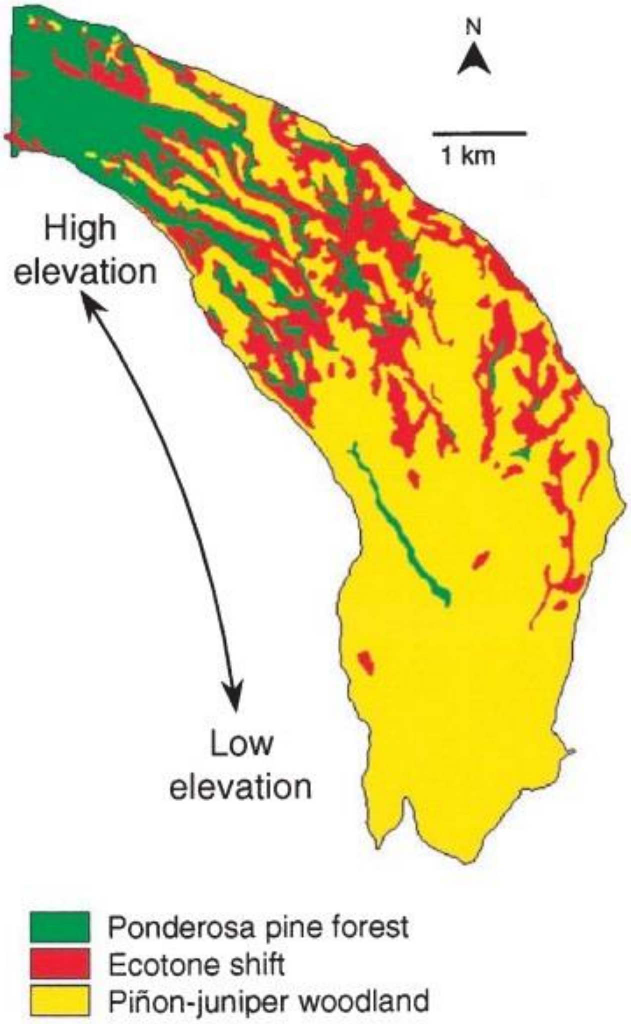


Figure 6. Changes in vegetation cover between 1954-1963 on a site in northern New Mexico. Green shows persistent ponderosa pine forest. Yellow shows persistent piñon–juniper woodland, and red shows the ecotone shift zone where forest changed to woodland. From: Allen and Breshears (1998).

Most mortality was attributed to piñon *Ips* (*Ips confusus*) infestations in stands severely affected by water stress (Manion 1991; Negron & Wilson 2003). This predisposition to massive *I. confusus* outbreaks during droughts was also observed in the droughts of the 1950s and early 1900s (Cole et al. 2004; McPhee et al. 2004). Most of the mortality in 2002-2004 occurred in stands between 1650-2400 meters in elevation where Colorado piñon is dominant (Klienman et al. 2012; Floyd et al. 2009). Characteristics such as density, tree diameter, or infestations of piñon dwarf mistletoe (*Arceuthobium divarcatum*)predisposing stands to massive piñon *Ips* mortality are still being researched and debated (Negron & Wilson 2003; Floyd et al. 2009; Klienman et al. 2012). Floyd et al. (2009) found there was no correlation between densities and *Ips* mortality across field sites in the Four Corners area, although they note the 2002-2004 drought was so severe, patterns denoting density-dependent relationships were not evident. However, it has also been argued stand density is a large factor in susceptibility to piñon *Ips* attack (Klienman et al. 2012). Larger diameter trees are known to be most susceptible to *Ips* infestation, and ultimately more predisposed to die during massive droughts (Wilson & Tkacz 1992; Negron & Wilson 2003; Floyd et al. 2009). Other common causes of stress and mortality are twig beetles (*Pityophthorus* sp.), pitch mass borer (*Dioctria ponderosae*), piñon needle scale (*Matsococcus acalyptus*), piñon blister rust (*Cronartium occidentale*), and black stain root disease (*Leptographium wagneri)* (Eager 1988; Floyd et al. 2009; Negron & Wilson 2003). These pathogens often do not kill a piñon outright, but predispose it to mortality from other agents (Eager 1988).

There has been recent work examining studying the complex interactions between drought, insects, and mycorrhizae (Smith & Read 2010; Gehring & Bennett 2009). Presence of mycorrhizae has been show to significantly reduce attacks by root herbivores (Gange 2001), but other studies have shown presence of some mycorrhizal relationships to have mixed results in providing benefits against aboveground herbivores (Gehring et al 1998; Gange et al. 2002; Gehring & Whitman 2002). Drought many have significant impacts on mycorrhizal relationships, leading to a decrease in plant fitness (Gehring et al. 1998). Decrease in fitness leaves the overstory (namely piñon pines) more vulnerable to pathogens such as piñon *Ips,* thus, increasing tree mortality (Gehring et al. 1998).

The resulting stand structure after drought shows a decrease in overall canopy cover (Clifford et al. 2005), large amounts of woody debris on the ground, and a change in dominant species (Shaw et al. 2005). These changes often will have lasting effects on stand characteristics. The severe drought of the early 2000s caused widespread mortality of Colorado piñon many magnitudes higher than the associated juniper species (Mueller et al. 2005; Floyd et al. 2009). The shift in species dominance in these areas will have a profound effect on stand dynamics into the future (Mueller et al. 2005). While drought can be a driver of community characteristics in piñon-juniper woodlands, favorable climate patterns coinciding with mast seed production can spur widespread regeneration of many of the woody species (Barger et al. 2009; Zlotin and Parmenter 2008). One notable instance of widespread regeneration of piñon pine occurred in the 1920s (Barger et al. 2009).

ii. Fire History

Many have debated historical fire regimes of piñon-juniper woodlands (Floyd et al. 2000; Romme et al. 2003; Brown et al. 2001). It is likely piñon-juniper woodlands have supported a wide range of fire regimes in the past including frequent low-severity fires, moderate-severity fires, and infrequent high-severity stand-replacing fires (Jacobs 2008). Huffman and others (2008) suggested fire was an integral part of higher elevation piñon-juniper woodlands, but were infrequent (~300+ years) and high-severity. Other studies have also shown evidence of long fire return intervals, suggesting a frequent low-intensity fire regime was uncommon (Baker & Shinneman 2004; Huffman et al. 2008). Romme et al. (2003) suggested a combination of topography and fuel patchiness makes it difficult for a fire to be carried through the system until large amounts of woody debris have accumulated.

The total proportion of yearly precipitation during the growing season is one of the many climatic factors that dictates variability across piñon-juniper woodlands by mediating types and abundance of understory cover (Jacobs 2008; Poulos et al 2009; Bauer & Wiesberg 2009; Kennard & Moore 2009; Margolis 2014). For example, piñon-juniper woodlands in the southeast of the region receive a majority of their annual precipitation during the growing season, allowing dense grass and forb cover to serve as fine fuels for frequent low-severity fires; but areas of the northeastern Colorado Plateau receive more winter precipitation, resulting in low understory cover unable to carry frequent fire (Jacobs 2008). Those systems are generally shaped by infrequent (>400 years) stand replacing fire regimes (Margolis 2014). Because these systems are mostly characterized by stand replacing fires, woodland areas that are dense and mature are likely to have high mortality following fire (Margolis 2014).

Interestingly, both junipers and piñons are capable of developing fire scars (Young & Evans 1981; Tausch & West 1988), but due to the nature of some infrequent fire regimes, no fire scars will develop since many of those fires are stand replacing. Most data on fire history is based upon studies of fire scars from species in adjacent ecotones, such as ponderosa pine forests, or a mix of piñon pine, ponderosa pine, and juniper fire scars (Miller & Rose 1999; Brown et al. 2001; Huffman et al. 2008). While juniper species are very difficult to use tree rings as a tool for dating individual trees, piñon pines have high potential to be used to reconstruct fire histories because they readily produce reliable growth rings (Floyd et al. 2009). Junipers are also difficult to use for researching fire history through fire scars because the thin bark does not generally allow the tree to survive fire. Some juniper species, however, can regenerate prolifically following fire through sprouting (i.e. alligator juniper), but this still leaves a lack of fire scarring (Bauer & Weisberg 2009). While piñon pines are highly likely to succumb to fires when they are young, mature trees with thick bark are able to survive more severe fires, thus, creating fire scars (Bauer & Weisberg 2009).

There are important interactions among different types on piñon-juniper woodlands such as understory fuel structure, canopy fuel structure, and fire weather conditions (Romme et al.

2009). Romme et al. (2009) explains the continuity and abundance of canopy fuels in persistent woodlands and wooded shrublands leads to high likelihood of a high-intensity stand replacing fire, partly due to a lack of understory vegetation unable to carry surface fire. The open and sparse canopies of savannas lead to abundant fine surface fuels leading to frequent low-intensity fires, thus maintaining an open savanna structure (Romme et al. 2009). However, piñon pines and junipers are able to establish under a wide range of conditions forming complex arrangements of understory and overstory structures; ranges in fire behavior and conditions can also vary widely from year to year affecting residual structure, composition, and function over time (Romme et al 2009).

iv. Management Strategies

Even before European settlers appeared in the West, Native Americans had a profound impact on piñon-juniper woodlands (Denevan 1992). The native peoples used piñon-juniper woodlands for hunting and gathering and cleared woodlands for fuelwood and agriculture (Ernst & Pieper 1996). Research has shown overuse of woodlands is not a recent phenomenon (Kohler & Matthews 1988). Evidence from studies in southwest Colorado suggests widespread deforestation of woodlands around Native American settlements coincides with population crashes and settlement abandonment (Kohler & Matthews 1988). Kohler and Matthews (1988) found evidence of woodland depletion and overuse in southwest Colorado by the Ancestral Puebloans was demonstrated by transition in type of firewood used and an overall decrease in abundance of piñon seeds. Various other studies have also shown an overall decrease in piñon-juniper woodlands resulting from Native Americans use (Vivian & Mathews 1973; Samuels & Betancourt 1982; Van Devender 1987). The Ancestral Pueblo farming techniques within piñon-juniper woodlands were generally slash-and-burn type agricultural systems due to the thin and nutrient poor characteristics of the soil (Kohler 1992). Stiger's (1979) studies of seed records show that presence of Colorado piñon and juniper seeds decrease as incidence of maize and beans increases at Mesa Verde, suggesting widespread deforestation during this time.

Since the arrival of Coronado in the mid-1500s in the Southwest, livestock have been a continual presence in piñon-juniper woodlands (Ernst & Pieper 1996). The Spanish colony implemented fire suppression around settlements for the first time in the late-1500s, likely causing the initial increase in density and distribution of piñon-juniper woodlands (Ernst & Pieper 1996). Evans (1988) outlines the reasons for expansion of piñon-juniper woodlands during this time is because of reduction of understory cover unable to carry frequent fire due to overgrazing, fire exclusion understory grass and forb degradation resulting in relatively no competition for establishing piñon and juniper. These three factors have led to an overall increase in density of woodlands, especially in areas where frequent fire probably maintained more open stand structures (Evans 1988).

Clearing and cutting of piñon-juniper woodlands was extensive in many areas when Anglo-American settlers arrived in the mid-1800s (Evans 1988). Piñon and juniper wood had many uses such as fuel, charcoal, fence posts, railroad ties, and beams for mines (Gottfried 2008; Evans 1988). Mining in the West lead to dramatic increase in use of wood converted to charcoal used for ore smelting (Evans 1988).

As mining and other needs for new infrastructure dwindled in the early 1900s, management of piñon-juniper woodlands was generally focused on increasing grass and forb production used for grazing through removal of the overstory until the 1970's (Tausch & Tueller 1977; Evans

1988). These techniques used were referred to as "range improvements" and included bulldozing, cabling, chaining, chopping or burning individual trees, and understory seeding of mainly non-native forage grasses (Tausch & Tueller 1977). It was erroneously thought managing for multi-resources by tree removal would increase water yield (Gottfried et al. 1995). However, studies in Arizona failed to prove any benefit in watershed health and water yield (Clary et al. 1974).

Chaining was the predominant range improvement technique used (O'Meara et al 1981). Two bulldozers would attach an anchor chain between them and would uproot trees as they moved across the woodland (O'Meara et al. 1981). This was nearly always followed by planting non-native forbs and grasses such as crested wheatgrass (*Agropyron cristatum*) for grazing of livestock (Redmond et al. 2013). The legacy of these misguided treatments has led to widespread overgrazing by livestock leading to degradation and loss of piñon-juniper woodlands (Jacobs & Gateway 1999). Evident to mangers early on, by 1936, it was estimated over a quarter of a billion hectares of rangelands in the West were degraded (Evans 1988). Range improvement techniques were curtailed in the 1970s, however, widespread grazing has continued, pushing the type and severity of disturbance past the historical natural range of variability (Evans 1988). The end of range improvement techniques saw new pressure by increased fuelwood demand exacerbated by the oil crisis of the 1970s (Gottfried 2008). Management shifted to a multi-resource management approach to sustain wood harvest into the future (Gottfried 2008).

It is unclear how past management has continued to affect current piñon-juniper woodlands (Aro 1971; Redmond et al. 2013; Tausch and Tueller 1970). Studies by Redmond et al. (2013) showed after 40 years, the resulting legacy structure from chaining to have an overall increase in non-native understory grasses planted during the initial range improvements, but an overall reduction in overstory trees. Utilizing piñon-juniper woodlands for grazing reduces understory cover and species present (Potter & Krenetsky 1967). Removing livestock from the system has been shown to positively benefit understory plants (Potter & Krenetsky 1967). Silvicultural techniques used on piñon-juniper woodlands may have unintended outcomes affecting site productivity, such as scattering slash after thinning and removal of juniper causing high rates of soil-surface erosion and runoff (Pierson et al. 2007; Stoddard et al. 2008).

Recent management has been aimed at restoration treatments reducing fuel loads in the wildland/urban interface and maintaining ecological integrity of soils, overstory trees, and understory grasses and forbs (Dellasala et al. 2004; Huffman et al. 2009), but some traditional range improvement techniques continue to be used (Brockway et al. 2002). These treatments are aimed at reclaiming areas believed to have been grasslands, improve degraded lands, and increasing forage production. There has been much debate on what historical reference the restoration treatments should be based upon (Romme et al. 2003; Baker & Shinneman 2004; Huffman et al. 2009; Romme et al. 2009; Huffman et al. 2013). Covington et al. (1994) argue fire exclusion since the late-1800s have caused the fires in piñon-juniper woodlands to change from frequent low-severity fires to high-severity stand-replacing fires. However, others believe fire exclusion has had little effect on the ecosystem (Floyd et al. 2004). Fuels reduction projects have great potential to push the woodland out of the historical range of variability by altering natural disturbance, stand development, and ecological processes; none the less, fuel reduction treatments are being carried out in areas of particular importance such as near residences or in recreation areas (Romme et al. 2004; Huffman et al. 2008). These mechanical treatments and prescribed burning are aimed at reducing risk of crown fire by increasing resilience by reducing surface fuel loads, removing ladder and canopy fuels, and retaining older fire-resistant trees (Huffman et al. 2008;

Fule et al. 2002). Prescribed fire may have mixed results because of the sometime sparse nature of surface fuel loads attributed to long fire return intervals, therefore, it is still unclear applying these techniques to the system would have the desired affects (Huffman et al. 2008; Erskine & Goodrich 1999).

VI. SYNTHESIS

Recent management has been aimed at restoration treatments, but there has been much debate on what historical reference the restoration treatments should be based upon. Much of the debate concerns the return of frequent fire as this type of fire regime was probably not part of many piñon-juniper woodlands. Many areas never have supported a short fire interval, but were more often stand-replacing with a return interval of over 500 years. Some believe introducing fire on these systems is just another poor management decision. Effective management is often hindered by a lack of information available on specific assemblages of piñon and junipers. Because of the heterogeneity of piñon-juniper woodlands, a variety of spatial patterns, densities, and disturbance regimes existed throughout the past. Further research still needs to be done to tease out the factors contributing to overall stand dynamics, so we can correctly manage piñon-juniper woodlands to persist and thrive into the future.

Large-scale die-offs of piñon pines related to climate change are expected to increase in the future, altering species compositions, woodland structure, and distribution of piñon-juniper woodlands, perhaps causing some species to leave a system entirely (Floyd et al. 2009). Climate envelope models are developed to predict potential future locations of species. First, conditions in which a specific species currently exists are identified and quantified; then, areas where those conditions exist in future climate models are identified, which can be interpreted as future suitable locations (Rehnfeldt et al. 2006). Models (Figure 7) predict the future distribution of *P. edulis* under climate change to be very bleak, virtually disappearing from many areas it currently occupies. This trend can be seen with not only piñon species, but junipers as well (not shown). This suggests there will be a dramatic change in ecosystem types in many areas where piñon-juniper woodlands now occupy. Ramifications of such a striking change to the woodlands are largely unknown, but there will likely be wide reaching impairments to ecosystem services and rapidly altered ecosystem type, ecosystem properties, and land surface conditions (Breshears et al 2005). Managing piñon-juniper woodlands in the future will be complex. Many believe the outlook for piñon-juniper woodlands is very bleak. Climate change is expected to occur at a rate faster than piñon-juniper woodlands can respond and adapt (Allen & Breshears 1998). Widespread

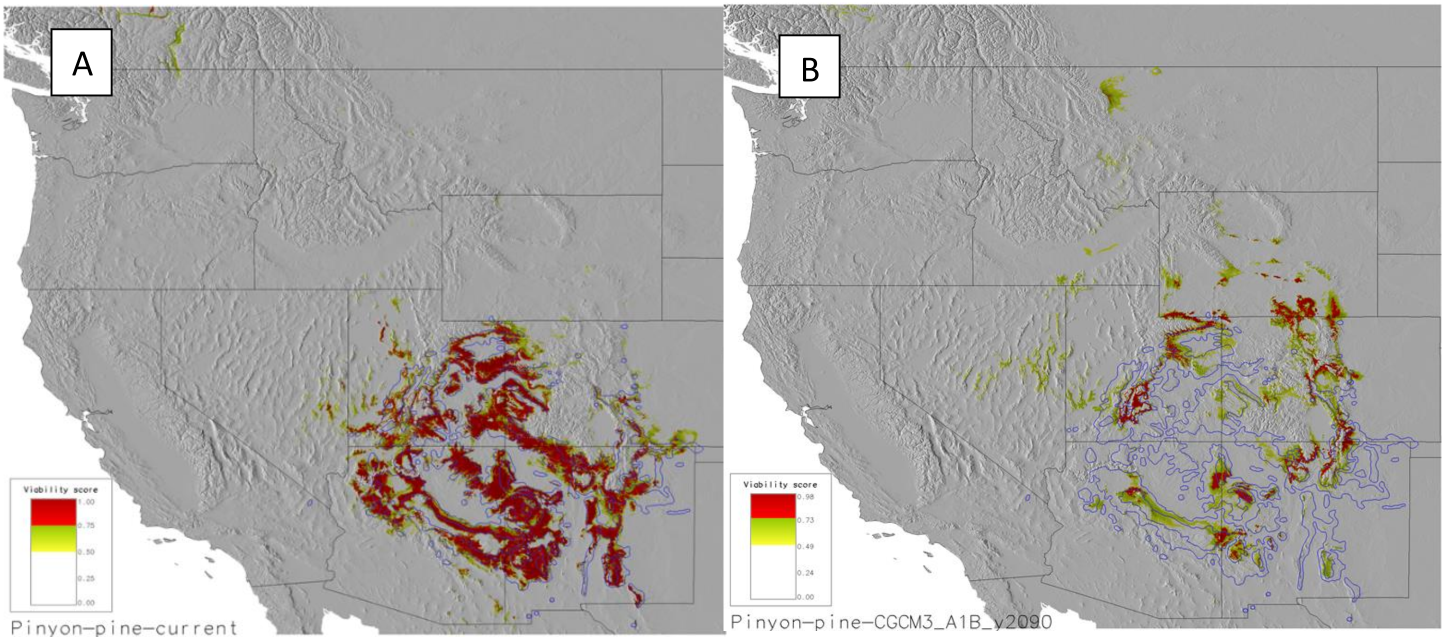


Figure 7 A.) Model displaying current range for P. edulis. B.) Future (yr. 2090) climate model displaying future range of P. edulis; Red indicates high likelihood of P. edulis occurring in that area. The green/yellow indicates conditions that may be suitable to support P. edulis, but are less likely. Developed by Nicholas Crookston, USFS, using climate output from model CGCM3 A1B (Third Generation Coupled Global Climate Model, High Emission scenario (A1B)). Accessed from: http://charcoal.cnre.vt.edu/climate/species/speciesDist/Pinyon-pine/; see also Rehfeldt et al. 2006 for further description of models

drought-induced mortality has already been documented in the piñon-juniper/ponderosa pine ecotone, suggesting some areas in the ponderosa pine forest type may warm to the point they cannot support piñon-juniper woodlands (Mueller et al. 2005; Koepke et al. 2010).Should we accept that a shift in ecosystem type is inevitable and plan for managing grasslands instead? Piñon pine has high potential to disappear from many areas, but junipers will remain much longer (Miller & Rose 1995; Floyd et al. 2009). While loss of piñon pines will be devastating to local people and some wildlife species, managing for remaining junipers important to maintaining at least some of the ecosystem services piñon-juniper woodlands provide.

Further research should focus on further classifying piñon-juniper woodlands into more specific groups to tailor restoration treatments for specific sites (Romme et al. 2009). Managing natural resources without the ability to classify ecological communities through scientific justification is very difficult (Hironaka 1987). Few have attempted to classify woodlands into more discrete groupings (but see Falco 2014; Romme et al. 2007; Romme et al. 2009). By defining a variety of classes of piñon-juniper woodlands through statistically validated classifications, managers will be able to develop more specific guidelines available for their specific woodland type and site. Creating classifications for specific assemblages of piñon-juniper woodlands provides the framework for collection and utilization of information for management justification (West et al. 1998). Statistical data from certain sites may be compared to other sites similar in classification (West et al. 1998). By using classifications justified by actual data, correct management techniques may be applied to reduce the risk of implementing inappropriate actions (West et al. 1998).

We need to review our current management and objectively determine the validity of the reasoning. Chaining and other range improvements have decreased significantly since the 1970s, but we need to put an end to practices aimed at grassland restoration. Density reduction needs to be curtailed, because regeneration is extremely difficult and sporadic. Although piñons are shade-intolerant, most establishment occurs under nurse trees, shrubs, or downed woody debris. This favors juniper and piñon regeneration in the understory. When piñons and junipers begin to creep into adjacent areas, we need to embrace it, even if it means sacrificing grazing area. In fact, the recent woody encroachment into adjacent grasslands could actually be its re-establishment on sites previously occupied by woodlands (Samuels & Betancourt 1982; Dick-Peddie 1993; Brockway et al. 2002; Romme et al. 2009; Huffman et al. 2012). Allowing woody invasion into grasslands may negate losses of piñon-juniper woodlands in other areas, thus adding wildlife habitat. Conversely, mortality of piñon-juniper woodlands in some areas may lead to expansion of grasslands and habitat for wildlife (i.e. greater sage-grouse, pronghorn) lost through woody invasion into grasslands. These range contractions and expansions are a natural response to climate change, not a relic of past management and use (Romme et al. 2009), therefore, should be encouraged.

As I have shown, the ranges piñon-juniper woodlands have been expanding and contracting for millennia. Encouraging natural migration into previously unoccupied landscapes may be the only way we see piñon-juniper woodlands survive. Climate change may have a variety of effects on piñon-juniper woodlands, mostly concerned with transition of ecosystem type to another. Aside from a piñon-juniper woodland stand remaining similar to current structure, there are four generalized shifts from current structure which may occur in these areas. Thus, there are a different set of strategies that should be used for each type of ecosystem transition type. For detailed management suggestions and implications and considerations, see Table 4.

One shift may be woody encroachment of piñon and juniper species into adjacent grasslands. I recommend we allow this invasion to occur as the range expansion of piñon-juniper woodlands in this area may negate areas of woodland lost to climate-change induced mortality. By promoting diversity of species, a wide range of wildlife species may flourish, particularly if widespread mortality leads to habitat loss in other areas. However, allowing woody encroachment heavily alters ecosystem function of grasslands and may even lead to further degradation of soil conditions.

A shift in ecotone from pure stands of ponderosa pine to piñon-juniper/ponderosa pine encourages natural migration up in elevation of piñon and juniper species. Without allowing piñons and junipers to move into ponderosa pine ecosystems, piñon-juniper woodlands may disappear in many areas, since its distribution is being pushed up in elevation by warming climate. We should even implement artificial regeneration of piñon and juniper as part of an assisted migration strategy. However, all the efforts may be in vain since encouraging these shifts may result in increased coarse woody fuel loads, leading to catastrophic fire. Abandoning ponderosa pines could have a dramatic effect on wildlife species due to changes in ecosystem services.

Due to massive mortality of the more drought-susceptible piñon pines, the structure may shift to the more open juniper savanna. More open ground in the interspaces will result in more grass and forb production, which may be beneficial for grazing of both domestic livestock and wild ungulates. The fine fuels on the ground increase fire potential, so fire mitigation may need to be performed in the wildland/urban interface. More open growing space may allow for species migration from lower ecotones.

Table 3 Recommendations and implications for four recent climate-change induced ecosystem changes

Recommendations and Implications for Management Actions		
Area of interest	***Management action***	***Implications & Considerations***
Piñon-juniper encroachment into adjacent grasslands/ shrublands	• Hands-off approach • Allow and encourage range expansion; it • Introduce fire to maintain open canopy structure • Promote diversity of species • Monitor soil and understory conditions and adapt management	• Reduction in grazing capacity in areas of encroachment • Heavily alters ecosystem function of grasslands • Introducing fire may not be appropriate for all systems • Encroachment exacerbates soil erosion in some areas by preventing understory establishment • Allows natural species migration due to climate change • Allows alternative land uses and income (i.e. biofuel, etc.) • May negate effects of range contractions in other places
Shift from ponderosa pine forest type to piñon-juniper/ponderosa pine ecotone	• Reduce juniper density; leave snags and logs on site • Pruning and individual tree removal as needed for fire mitigation • Encourage natural regeneration of ponderosa, piñon, and juniper • Artificial regeneration of piñon and juniper • Monitor and adapt to encourage shift of ecotone	• Increased fuel load may lead to large fires • Artificial regeneration is expensive and may fail • Abandoning ponderosa pine will reduce future harvest revenues • Shifts wildlife composition due change in forest structure • Allows for natural species migration to higher elevations
Shift from piñon-juniper woodland to juniper savanna	• Reduce fuel load for fire mitigation • Introduce frequent low-severity fires to mimic natural fire regimes • Encourage and/or seed understory grasses and forbs • Manage for coarse woody debris and understory vegetation for frequent low-severity fires • Monitor and adapt if conditions show transition to grassland/shrubland or revert to piñon-juniper woodland.	• Managing for only junipers will lead to loss of biodiversity • Introducing fire may not be appropriate for all sites • Seeding is expensive and may introduce invasives • May allow for increased grazing capacity • Allows for species migration from lower ecotones • Maintains some ecosystem services on piñon-juniper woodlands
Shift from piñon-juniper woodland to grassland/semi-arid desert.	• Fire mitigation by removing woody debris to reduce threat to people and infrastructure • Encourage natural regeneration or seed native plants from a lower ecotone, especially where erosion is of concern • Monitor and adapt if conditions for degradation or further shift of ecosystem	• Removal of woody debris may lead to lack of nurse trees for adequate seedling establishment in short term • Could allow for increased grazing capacity • May add compensatory wildlife habitat to mitigate loss (i.e. sagebrush grouse, pronghorn, etc.) • Allows migration of plants from lower ecotones • Change in microclimate may make the site inhospitable to tree regeneration

Perhaps the most dramatic shift from piñon-juniper woodlands may result from massive mortality of both piñon and juniper. The resulting ecosystem would be dominated by grasses and forbs, many from lower ecotones. The stark change in microsite may leave the area inhospitable to tree regeneration for decades or centuries. Domestic ungulates would benefit from the added area for grazing and habitat (i.e. pronghorn, cattle). The contraction of piñon-juniper in these areas will compensate for areas where piñon-juniper encroachment has already occurred.

This shift may also allow for expansion of range loving wildlife habitat into previously wooded areas such as greater sage-grouse and pronghorn. The increased fire regime associated with grasslands could pose a risk to people and infrastructure in the wildland/urban interface, so fire mitigation may need to be performed.

Piñon-juniper woodlands have been dynamic throughout history and continue to change. The key to piñon-juniper woodland management is understanding the relationship between community type and site potential, and projecting future distribution of community types under different climate change scenarios. The ultimate future of management of piñon-juniper woodlands will be focused on adapting to changing climate and encouraging natural processes within the woodlands. However, history has shown piñon-juniper woodlands have never been static, and continually change over time. To manage piñon-juniper woodlands under a changing climate, landowners and managers should be encouraged to focus on promoting resilience (capacity of a system to respond to perturbation or disturbance and recover quickly) rather than maintenance of the status quo or static condition. Allowing ecological processes without manipulation ensures piñon-juniper woodlands will persist into the future, even if they are not in areas we traditionally would think to find them.

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