Piñon-Juniper Is Not Just One Vegetation Type: Key Variation in Composition, Structure, and Disturbance Dynamics, With Implications for Wildlife and Ecosystem Management

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Piñon-juniper vegetation covers 100 million acres across the western United States, in areas of diverse topography, climate, and soils. Consequently, piñon-juniper vegetation varies greatly in structure and disturbance processes at both the stand and landscape scales. Until recently, however, this variation was not widely recognized, and piñon-juniper vegetation was interpreted and managed in much the same way throughout its range (Jacobs 2008a). Early studies were conducted primarily in the Great Basin and Mexican borderlands, where juniper expansion into grasslands and shrublands was a major ecological process and management concern. Results of these studies were then applied uncritically to piñon-juniper vegetation elsewhere, notably on the Colorado Plateau, where more recent research has revealed very different stand and landscape dynamics.

This later research resulted in a workshop in August, 2006, that brought together 15 scientists with piñon-juniper expertise from across the range of piñon and juniper in the western USA. These researchers summarized available data and understanding of these varied ecosystems and published their synthesis in three venues: an academically oriented, invited review paper in the journal *Rangeland Ecology and Management* (Romme et al. 2009), and two shorter articles geared toward managers and policy-makers (Romme et al. 2007, 2008). Key findings from these three articles and recent related literature are summarized below.

Variation in Tree Species Composition

Two piñon species and five juniper species are common components of piñon-juniper woodlands across the western USA (names from USDA Plants database, https://plants.usda.gov/java/). Local composition varies with geographic location, elevation, and synoptic climate (Jacobs 2008a). Utah juniper (*Juniperus osteosperma*) and two-needle pinyon, also called Colorado piñon (*Pinus edulis*), dominate woodlands in regions where precipitation is winter-dominated or bimodal, e.g., on the northern Colorado Plateau. Oneseed juniper (*J. monosperma*), alligator juniper (*J. deppeana*), and two-needle piñon dominate summer monsoonal regions, e.g., in central and southern Arizona and New Mexico. Rocky Mountain juniper *(J. scopulorum*) is common at higher elevations throughout most of the Rocky Mountain cordillera. Singleleaf pinyon (*P. monophylla*) and western juniper (*J. occidentalis*)

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dominate the summer-dry country of the Great Basin and eastern Sierra Nevada. Two additional piñons and four additional junipers are found along the southern and eastern boundaries of the American Southwest (Jacobs 2008a).

Variation in Stand Structure and Dynamics

The scientists at the 2006 workshop identified three fundamentally different kinds of piñon-juniper vegetation. This classification is based not on species composition, but on functional characteristics including typical stand structures, historical disturbance regimes, and temporal dynamics.

Piñon-juniper savannas are composed of trees within a grassland matrix, and are found in areas of summer-dominated precipitation, e.g., in southern and eastern Arizona and New Mexico, and southeastern Colorado. Tree densities probably were limited historically by periodic fires, but in the last century junipers have expanded extensively into former grasslands, especially on concave or depositional landforms, converting those grasslands to young savannas and woodlands in many places (e.g., Fuchs 2002, Jacobs 2008b, Poulos et al. 2009, Parker 2009, Margolis 2014).

Wooded shrublands are composed of trees within a shrub matrix, e.g., sagebrush (*Artemisia* spp.), Gambel or wavyleaf oak (*Quercus gambelii* or *Q. undulata*), mountain-mahogany (*Cercocarpus montanus*), and Utah serviceberry (*Amelanchier utahensis*). Tree density naturally waxes and wanes in response to decadal-scale variations in climate, fire, and insect activity. These woodlands are probably most prevalent in areas of winter-dominated or bimodal precipitation, e.g., in eastern Utah, western Colorado, and northwestern New Mexico.

Persistent piñon-juniper woodlands are found throughout the distribution of piñon-juniper woodlands in places that burn very infrequently, such as dry rocky upland sites, but especially on the northern Colorado Plateau where precipitation is winter-dominated or bimodal. Notably, these are not young woodlands created by recently expanding junipers and piñons, but are often ancient stands with centuries-old trees, snags, and fallen boles, and little or no evidence of fire. The paucity of fire in persistent piñon-juniper woodlands apparently is due to a generally discontinuous fuel structure (Romme et al. 2009). Heavy grazing has reduced fine fuels in many piñon-juniper woodlands, but fuels tend to be discontinuous and fires infrequent in persistent woodlands even where grazing has been absent or minimal for nearly a century, e.g., Mesa Verde National Park (Floyd 2003).

This classification, recognizing three broad kinds of piñon-juniper woodlands, is obviously a huge simplification of an extensive and varied vegetation type. Local floristic composition can vary greatly within each of these three functional piñon-juniper types. For example, Romme and Jacobs (2015; table 4) identified three floristic plant associations within the persistent piñon-juniper type and one within the piñon-juniper savanna type at El Malpais National Monument, New Mexico. Nevertheless, a functional classification of this kind highlights some of the important differences in composition, structure, and ecological dynamics to be found within piñon-juniper woodlands across the West and provides a first step in characterizing any specific woodland stand or landscape.

Some Applications to Wildlife and Ecosystem Management

Management opportunities and constraints are greatly influenced by the kind of piñon-juniper (PJ) ecosystem being considered. For example, the greatest departure from historical conditions is likely to be seen in PJ savannas, especially where oneseed juniper has converted former grasslands into new savannas and woodlands during the past century. Persistent PJ woodlands, on the other hand, generally have changed the least from historical conditions. Persistent PJ woodlands are also most likely to support a unique suite of wildlife species that are closely linked with and dependent upon this particular kind of habitat, e.g., Pinyon Jay (*Gymnorhinus cyanocephalus*) and Juniper Titmouse (*Parus inornatus*). Thinning or burning these woodlands is likely to reduce habitat quality for these PJ specialists.

Reducing tree density to improve forage production for livestock, or habitat for grassland-associated wildlife species, will often be most successful in PJ savannas, where soils and climate are inherently favorable for grasses, and least successful in persistent PJ woodlands where rocky soils favor trees and shrubs over grasses.

Prescribed fire can be applied effectively in many PJ savannas for a number of objectives, but prescribed fires in persistent PJ woodlands notoriously fail to spread through surface fuels, or spread uncontrollably into the canopy. In wooded shrublands dominated by shrub species that resprout after fire, e.g., Gambel oak (*Quercus gambelii*) or mountain-mahogany (*Cercocarpus montanus*), prescribed fire can be an effective way to improve forage for deer (*Odocoileus hemionus*) and elk (*Cervus elaphus*). However, if the dominant shrubs tend to be killed by fire without resprouting, e.g., big sagebrush (*Artemisia tridentata*), then prescribed burning may produce little improvement in forage but promote an unwanted increase in nonnative plants such as cheatgrass (*Bromus tectorum*).

Piñon-juniper vegetation is managed for a wide range of objectives, and in some cases the same kind of treatment can achieve several different objectives. But this is not always true. For example, the term *restoration* is sometimes used loosely to bundle multiple objectives, but restoration has a precise definition as "… an intentional activity that initiates or accelerates the recovery of an ecosystem with respect to its health, integrity and sustainability … attempts to return an ecosystem to its historic trajectory," (Society for Ecological Restoration, http://www.ser.org/). Genuine restoration often does not achieve some of the other objectives we have for PJ vegetation, and accomplishing some of those other objectives may actually move an ecosystem farther from its historical condition.

Floyd and Romme (2012) explore some of the conceptual issues in restoration of PJ ecosystems. Restoration may focus on structural parameters, e.g., canopy density and understory composition, or on *functional aspects*, e.g., fire frequency and effects. A structural restoration can return a site to its condition at a particular time in the past, but that may be only one point within the natural range of variability on that site, and it may not persist without repeated treatment. Therefore, a functional or process-oriented restoration may be more sustainable, by allowing natural dynamics to continually reshape the ecosystem within historical bounds. Restoration may be passive or active. *Passive* restoration merely removes any chronic stressors from an ecosystem that is generally in good condition, e.g., minimizing impacts of grazing or recreation, without directly altering structure or process by actions like thinning or planting. *Active* restoration directly alters the ecosystem to restore parameters that have been lost and cannot be recovered without intervention.

The specific type of restoration needed depends on the current condition and history of a PJ ecosystem. Floyd and Romme (2012) suggest some very general guidelines and review specific examples of restoration experience in different kinds of PJ. If a PJ ecosystem remains within or very close to its historical range of variability (Weins et al. 2012), then passive restoration may be all that is needed to maintain that condition; active treatments like canopy thinning could actually degrade stands by disturbing the soil and encouraging non-native plant establishment. An example requiring only passive restoration is the old-growth PJ woodlands on Mesa Verde (Floyd 2003). In other cases the overstory may be in good shape—within the historical range of tree density, composition, and cover—but the understory has been degraded by excessive grazing or recreational impacts.

Piñon-juniper woodlands on the Kaiparowitts Plateau fit this description: native graminoids and herbs were greatly reduced by more than a century of summer grazing, and nonnative cheatgrass has proliferated in place of the natives, even though the overstory remains intact (Floyd et al. 2008). Restoration of this ecosystem requires both passive and active approaches: passive to reduce grazing intensity and active to reduce the cheatgrass and possibly to also plant seed of species native to the site, but the overstory needs no treatment.

In other places both the overstory and understory have been degraded by past land use, and active treatment is needed in both components of the ecosystem. Such a stand in Bandelier National Monument was restored by mechanically thinning the overly dense overstory and scattering the slash over the bare ground between retained tree canopies. Within 3-5 years, understory plant cover increased several-fold and run-off and sediment production were reduced by an order of magnitude (Jacobs 2015). The treated stand also was more resilient in the face of a severe drought, a wildfire, and a bark beetle outbreak that occurred a few years after the treatment.

A final class of PJ ecosystems that may be in special need of restoration is found in areas that have recently burned in high-severity wildfires. A large portion of the old-growth woodland on Mesa Verde burned between 1989 and 2008 (Floyd et al. 2004). Severe fires in this kind of ecosystem are natural, although infrequent, but postfire vegetation dynamics have been anything but natural. Nonnative annual and biennial plant species such as cheatgrass, muskthistle (*Carduus nutans*), and Canada thistle (*Cirsium arvense*) have expanded rapidly into the burned areas and are the dominant species in many places. Notably, the stands having understories of resprouting shrubs, e.g., Gambel oak or Utah serviceberry, have been more resistant to invasion by nonnative plants than have the stands with few sprouting shrubs. This may be because the sprouting shrubs compete strongly with the nonnatives. Active restoration by means of planting native grass seeds in the burned areas has substantially reduced the abundance of the nonnatives, although it has not eliminated them altogether (Floyd et al. 2006).

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