

Comments on piñon / juniper ecology in relation to Grand Staircase–Escalante National Monument.

Patrick J. Alexander | Ecologist | BLM National Operations Center | 1 Aug 2022

Context—My personal field experience with plant ecology is primarily in New Mexico, with some experience in Arizona & Colorado but relatively little in Utah. My understanding of plant ecology in GSENM at this point is based on published research, remote sensing products, aerial imagery, and AIM plot data, not field experience in GSENM. This document should be interpreted accordingly, as a provisional understanding from someone who is reasonably well-equipped to understand the ecological situation *within the limits* imposed by working at a distance, but is absolutely not able to ground-truth his understanding of the landscape from personal experience. Neither you nor I should be at all surprised to find my understanding inadequate!

Margolis (2014) provides a description of the relationship between fire, grazing, and piñon / juniper ecosystems for Glorieta Mesa, in north-central New Mexico. In short, the plant community was historically a savanna with relatively sparse piñon and juniper and a well-developed herbaceous layer dominated by perennial grasses. The historic fire regime was characterized by frequent (± 8 years, at Glorieta Mesa), low-intensity surface fires that caused mortality on young piñon & juniper, while rarely causing appreciable mortality on more mature trees. The introduction of grazing led to a reduction of fine surface fuels to below a level that could carry low-intensity surface fire. Fire became rare around 1900 and fire frequency was "0" from 1920 to 2000 in the Glorieta Mesa data. Without fire, piñon & juniper seedlings had low mortality. Tree density increased to ± 6 times that of the pre-grazing plant community. In southern New Mexico, juniper establishment appears to be particularly associated with years of very high winter / spring precipitation and good monsoonal precipitation (my unpublished assessment of precipitation records & aerial imagery). Prior research suggests juniper seedlings are susceptible to drought and that competition with perennial grasses (in addition to their role in fueling low-intensity surface fires) may reduce their ability to establish (Johnsen 1962; Smith et al. 1974). I believe this basic picture holds for most piñon / juniper and juniper savanna ecosystems in New Mexico, where *Juniperus monosperma* is usually the dominant juniper, often with *Juniperus deppeana*, *Juniperus scopulorum*, *Juniperus arizonica*, or *Juniperus pinchotii*. It may or may not hold for the northwestern corner of the state where *Juniperus osteosperma* is found. At the higher and lower elevation margins of piñon / juniper and juniper savanna ecosystems in New Mexico the dynamics probably become somewhat different. In any case, in approaching GSENM, my initial expectation was that this basic picture is the norm in piñon / juniper ecosystems, and that GSENM is likely to be similar. That didn't turn out to be a good expectation.

Remote sensing data for GSENM—The remote sensing data from the Rangeland Analysis Platform uses data from monitoring plots, including AIM plots, to develop ground-truthed estimates of fractional foliar cover from Landsat imagery. RAP appears to do a very good job of capturing variation in tree cover in GSENM, based on comparison of the RAP tree cover products to what I can see on aerial imagery going back to 1993. Where I see trees on aerial imagery, RAP also sees trees; where I see shrublands, bare ground, or vegetation treatments that removed trees on aerial imagery, RAP agrees that trees are sparse or absent. Using this kind of subjective assessment, I can't evaluate the relationship between the % tree cover reported by RAP and the true % tree cover, but I can verify that RAP is capturing the same relative densities of tree cover in different areas that I can see visually. However, I noticed some variation between years that seemed more likely to reflect differences in the quality of Landsat imagery, so decided to focus on a pair of rasters averaging the oldest (1986–1995) and newest (2012–2021) decades of available RAP data.

The RAP data shows relatively small increases (fractional tree foliar cover ± 2 –8% higher) from 1986–1995 to 2012–2021 across most of the piñon / juniper ecosystems of GSENM. So far as I can tell from aerial imagery, these result primarily from increases in the size of trees that were apparently mature or nearly mature in the earliest aerial imagery (± 1993). These increases in tree cover are not clearly evident when looking at the aerial imagery. Some areas show larger increases (± 10 –20%) from 1986–1995 to 2012–2021, and these appear to be associated either with trees recolonizing older vegetation treatments (pre-1986; for many of these the dates are not known) or growth of trees that apparently colonized pockets of shrubland within the broader matrix of piñon / juniper woodlands at some point in the early to middle 20th century. These larger changes are visually apparent in aerial imagery. The RAP data also shows widespread but small (± 1 –4%) declines in shrub cover in the piñon / juniper ecosystems of GSENM.

Unlike the southern New Mexico sites I'm familiar with, where large and visually conspicuous pulses of juniper establishment occurred around 20–30 years ago in areas where junipers had previously been sparse, I did not see any evidence of big establishment events while looking at aerial imagery in GSENM. Small junipers are harder to see, though, forcing me to rely more on zooming in to small areas scattered around the landscape than on more systematically assessing larger parts of the landscape. So, my understanding of any recent juniper establishment events is probably not that good, and I would be especially unlikely to spot establishment events in the last decade or so. Given the resolution of Landsat imagery (30 m pixels), RAP & similar Landsat-derived products are not able to distinguish young from mature junipers.

AIM data for GSENM—The time horizon of the AIM and LMF data doesn't support inference of changes over time in GSENM. The primary role of these data in the context of this document, then, is calibration of the RAP Landsat estimates.

Published literature in GSENM—Harris et al. (2003) used a combination of field methods and remote sensing to compare the vegetation of Deer Spring Point (grazed) to No Man's Mesa (very little livestock grazing in the early 20th century, none known since; native ungulates apparently also having little impact on the vegetation). They found that total foliar cover was significantly higher and bare soil significantly lower at the grazed Deer Spring Point compared to the relatively ungrazed No Man's Mesa. Shrubs had significantly higher fractional cover¹ at Deer Spring Point, cryptobiotic crusts, grasses², and cacti significantly higher fractional cover at No Man's Mesa³. Tree cover was not significantly different between the two sites.

Guenther et al. (2004) made a similar comparison between these two sites using field methods without remote sensing data. Guenther et al. report their results as absolute foliar cover values, which are difficult to directly compare with the fractional cover values of Harris et al. However, the two studies reach similar conclusions. Contrary to Harris et al., Guenther et al. find that bare ground is significantly lower on No Man's Mesa than at Deer Spring point. Consistent with Harris et al., Guenther et al. find that on the relatively ungrazed No Man's Mesa cryptobiotic crust cover is significantly higher, total foliar cover and shrub cover significantly lower than on the grazed Deer Spring Point. Guenther et al. find no significant difference in tree

¹ This is a somewhat unusual way of reporting the data, as the fraction of total foliar cover rather than as foliar cover. Given that the total foliar cover also differs between the two sites, it is possible for the two sites to have significantly different fractional cover values even if the absolute foliar cover values do not differ.

² For grasses, the significant difference apparently holds for fractional cover but not absolute foliar cover.

³ There's an error in Table 2 of Harris et al. that can cause some confusion here. I believe the first row should be read as "grasses", the second row—which is not labelled—as "forbs".

cover between the two sites. Grasses a minor component ($\pm 3\%$ foliar cover) of both sites, with no significant difference between them. Annual plants are sparse ($< 2\%$ cover) on both sites, but significantly higher at the grazed Deer Spring Point. Non-native plants were few at Deer Spring Point, absent from No Man's Mesa.

Barger et al. (2009) followed with research on tree demography and growth at these two sites. They found no significant differences in tree density, diameter, basal area, growth rate, or age of the oldest trees. Trees were significantly younger at the relatively ungrazed No Man's Mesa (average age 124 years) than at the grazed Deer Spring Point (average age 157 years). If grazing were causing an increase in tree establishment, we would expect the opposite, for trees to be younger at Deer Spring Point. Barger et al. (2009) highlighted the synchrony in establishment and growth rates between the two sites as evidence that climatic variation was the primary factor controlling these processes, not the difference in grazing history. The main limitation of these two studies is that they look at a single grazed / ungrazed comparison. The ecological dynamics may be different in other piñon / juniper ecosystems in GSENM.

Harris & Asner (2003) used a combination of field methods and remote sensing data to evaluate how vegetation changes with distance from livestock water sources. They distinguished photosynthetic vegetation, non-photosynthetic vegetation, and bare ground, but did not distinguish between plant functional groups (perennial grasses, trees, etc.). To address differences in livestock travel distance over different types of terrain, they categorized slopes into three categories ($1-5^\circ$, $5-15^\circ$, $>15^\circ$). They found significantly less vegetation (of both categories) and significantly more bare ground near livestock water sources. On the steepest terrain category, this effect extends ± 0.3 km from water sources, while on shallower slopes it extends $0.5-0.7$ km from water.

Witt & Shaw (2010) summarized plot data from piñon / juniper woodlands in GSENM, collected from 1981 to 2005 as part of the U.S. Forest Service's Forest Inventory & Analysis program. They focus on increased mortality, especially of piñon but also of juniper, during drought from 2003 to 2005.

Published literature for other Colorado Plateau piñon / juniper ecosystems—Floyd et al. (2000, 2004, 2015) studied piñon / juniper ecosystems on Mesa Verde. They found that the fire-return interval in the natural fire regime was ± 400 years and that low-intensity surface fires did not play an appreciable role in this ecosystem in the time period covered by their data (± 1700 to 2004). The natural fire regime is apparently characterized by infrequent but high-intensity, stand-replacing fires. They also found that the outlines of fires that took place 200–300 years ago were still visible on aerial imagery, and that charred snags from fires of this age could still be found when the sites were visited in person. Where historical fires were more frequent on Mesa Verde, plant communities are dominated by resprouting shrubs (*Quercus gambelii*, *Amelanchier utahensis*, *Cercocarpus montanus*, &c.) rather than piñon / juniper woodlands. Recent changes in the piñon / juniper woodlands include a period of unusually large, high-intensity fires from 1996 to 2003, apparently the result of drought following two decades of unusually wet weather that led to increased plant biomass. Mortality of piñon (and juniper, to a lesser extent) due to the combined effects of hotter temperatures, drought, and engraver beetles (*Ips confusus*) has continued to 2015, having the greatest effect on mature piñons. Floyd et al. (2015) highlight the possibility that the effects of recent climatic shifts could lead to overall decline or loss of the current, old-growth or persistent, piñon / juniper woodlands at Mesa Verde.

Floyd et al. (2008) studied stand structure on Navajo Point, at the south end of Kaiparowits Plateau. They estimated the fire return interval at ≥ 600 years, and found that there had not been fundamental change in stand structure associated with human activity during the 20th century. Similar to work on Mesa Verde, they could identify the outlines of fires 100–300 years old on aerial imagery, and could find charred snags from fires of this age during field work. Kennard & Moore (2013) studied stand structure and fire history in persistent piñon / juniper woodlands of Colorado National Monument, west of Grand Junction. They found no evidence

of large (>100 ha) stand-replacing fires and infer that such fires have been absent from this ecosystem for a perhaps a millennium. Drought and small (< 10 ha) fires, rather than large fires, have probably been the primary causes of tree mortality. Shinneman & Baker (2009a) studied several dozen sites around the margins of the Uncompahgre Plateau, south of Grand Junction and west of Montrose. They found high-intensity, stand-replacing fires occurring at >400–600 year intervals and no evidence of low-severity surface fires. They identified long-term patterns of tree mortality and drought driven by climate. Using sites within Colorado National Monument as an ungrazed comparison, Shinneman & Baker (2009a) found significantly higher seedling /sapling densities at grazed sites for piñon but not for juniper. Both Floyd et al. (2008) and Kennard & Moore (2013) highlight the risk of cheatgrass (*Bromus tectorum*) reducing the resistance of persistent piñon / juniper woodlands to fire & resilience of woodlands following fire. Shinneman & Baker (2009b) also discuss the role of cheatgrass in this context. They did not find a significant difference in post-fire cheatgrass cover between seeded and unseeded sites. However, regression models found that higher native plant species richness and higher cover of biological soil crusts were associated with lower post-fire cheatgrass cover. Shinneman & Baker (2009b) suggest that grazing is likely to play a role in altering fire ecology due to the effect of grazing on ecosystem attributes related to post-fire resilience.

Vankat (2017) studied change from 1935 to 2011 in piñon / juniper woodlands on the south rim of the Grand Canyon, in Grand Canyon National Park. They divided piñon / juniper woodlands into three categories: persistent piñon / juniper woodland, seral piñon / juniper woodland (affected by stand-replacing fires prior to 1935), and transitional piñon / juniper woodland (at the higher elevations of piñon / juniper ecosystems, grading into ponderosa woodlands). Within persistent piñon / juniper woodland, they found that drought-induced mortality on more mature piñon and juniper was followed by establishment events, primarily of juniper. Density and basal area of tree increased substantially in seral piñon / juniper woodland, apparently representing post-fire recovery towards the tree cover typical of persistent piñon / juniper woodland. Transitional piñon / juniper woodland was stable. Vankat attributes change primarily to climatic variation over time. He echoes Floyd et al. (2000, 2004, 2008), Shinneman & Baker (2009), and Romme et al. (2009) in describing the natural fire regime as having infrequent (recurring on time scales of multiple centuries) but high-intensity, stand-replacing fires that play a smaller role within the context of climate-driven mortality and establishment cycles.

Romme et al. (2009) provided a synopsis of research on piñon / juniper ecosystems and fire ecology. They divide piñon / juniper ecosystems into three general categories: persistent piñon / juniper woodlands; piñon / juniper savannas; wooded shrublands. The authors suggest that, although these broad categories are useful, there is substantial variation within each category as well. They identify persistent piñon / juniper woodlands as especially prevalent on the Colorado Plateau⁴, and as associated especially with shallower or rockier soils and climates with relatively bimodal (rather than primarily winter / spring or primarily monsoonal) precipitation. In a prior paper, Romme et al. (2007) provide a key allowing sites to be assigned to one of these three categories. They characterize persistent piñon / juniper woodlands as being primarily areas where total tree foliar cover is >10% and at least 10% of the trees are over 150 years old, or with lower tree foliar cover at particularly rocky, unproductive sites or following severe disturbance (fire, woodcutting, etc.). In both papers, Romme et al. characterize persistent piñon / juniper woodlands as burning infrequently, with fires typically small and stand-replacing; having relatively low and discontinuous surface fine fuel loads under most natural conditions; being stable over centuries in the absence of fire and not increasing in tree cover due to recent changes in fire regime; having patterns of tree mortality and establishment driven primarily by climate

⁴ So far, the papers I've found on piñon / juniper ecosystems of the Colorado Plateau either call them persistent woodlands or present data showing that they match the Romme et al. definition of persistent woodlands.

rather than fire; having recent periods of more numerous or larger fires driven primarily by climate, secondarily by invasion of cheatgrass. They also state that there have been 20th century increases in tree cover in many or most persistent piñon / juniper woodlands (Romme et al. 2007) but with little landscape-scale effect on total tree cover (Romme et al. 2009). A potential role of grazing in changing tree cover, tree demography, or fire regime in persistent piñon / juniper woodlands is discussed in both papers, but they state (in Romme et al. 2009) that the empirical evidence is "sparse and mixed".

Conclusions—My current understanding of the fire ecology and tree dynamics in piñon / juniper ecosystems of GSENM is that: These are primarily persistent piñon / juniper woodlands and are probably not significantly affected by grazing or fire suppression with regard to fire frequency or severity, or the abundance or age distribution of trees. The natural fire regime is probably characterized by relatively small (100–1000 ha²), infrequent (fire return interval of 300–600 years⁵), but high intensity, stand-replacing fires. Natural climatic variation on decadal time scales and, in recent decades, unnatural increases in temperature and changes in the timing or amount of precipitation are probably the major drivers of change in tree abundance / age composition. Climate change is probably the primary anthropogenic force affecting piñon / juniper dynamics in GSENM, and is not directly amenable to management at the scale of GSENM.

This basic picture seems to be inconsistent with the fire regime / vegetation condition classes in LANDFIRE products. I do not have a good enough understanding of how this LANDFIRE product is developed to have a good guess why this might be. I view remote-sensing products like this primarily as a way of extending our understanding over much larger parts of the landscape than we can directly sample. As a result, I approach them by first trying to if they are consistent with an understanding based other data sources. When this is not the case, I consider other data sources—especially direct field measurements—to have priority and the remote-sensing product to be in doubt⁵. Or, at least, while it may tell me something useful, I find myself uncertain what that something might be. I've found several times that a remote-sensing product is accurately reporting something about the landscape, but not what I expected it to report.⁶

There may well be multiple piñon / juniper types at GSENM, which differ in one or more of the characteristics described above. Other piñon / juniper types are more likely to be found on deeper and less rocky soils relative to the average of piñon / juniper ecosystems at GSENM.

A focus on a single site, No Man's Mesa, in studies of ungrazed ecosystems in GSENM is a major constraint on our ability to interpret the available research as characterizing piñon / juniper ecosystems in the monument as a whole.

The largest published effects of grazing on the ecology of piñon / juniper ecosystems in GSENM are decreases in cryptobiotic crust cover & increases in shrub cover in grazed areas, and decreases in total foliar cover near water sources / in areas more readily accessible to livestock. The few papers I've run across so far that directly address native & introduced plant diversity and annual plant cover suggest that grazing causes declines

⁵ A related observation on this point: I often use southern New Mexico as my test case, since this is where I can most easily & accurately check a remote-sensing product against what's happening on the landscape. There are reasons to suspect that southern New Mexico is not a very good test case for this particular LANDFIRE product. Nonetheless, I can't really match up the patterns of variation depicted in the LANDFIRE fire regime / vegetation condition class product with patterns of variation in plant ecology on the landscape.

⁶ In one case, I found that a product intended to depict areas more susceptible to juniper encroachment instead showed me where junipers were historically—presumably, naturally—more abundant. Certainly useful, but with management implications close to the opposite of what was intended.

in native plant diversity, higher introduced plant diversity, higher annual cover, and greater susceptibility to invasion by cheatgrass, especially after fire. Given the severity of cheatgrass invasion throughout much of the western U.S., a better understanding of this topic would be desirable. The papers I've come across so far are more suggestive than definitive. However, I haven't made any effort to review the literature on cheatgrass except insofar as it overlaps with studies of tree dynamics & fire ecology. I don't know how well other research fills this gap.

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