Fire History, Woodland Structure, and Mortality in a Piñon-Juniper Woodland in the Colorado National Monument

Authors: Kennard, Deborah K., and Moore, Amanda J.

Source: Natural Areas Journal, 33(3) : 296-306

Published By: Natural Areas Association

URL: https://doi.org/10.3375/043.033.0308

RESEARCH ARTICLE

•

Fire History, Woodland Structure, and Mortality in a Piñon–juniper Woodland in the Colorado National Monument

Deborah K. Kennard[1,3]

[1]Department of Physical and Environmental Sciences
Colorado Mesa University
Grand Junction, Colorado 81501

Amanda J. Moore[2]

[2]Bureau of Land Management
Gunnison, Colorado

•

[3] Corresponding author:
dkennard@coloradomesa.edu

Natural Areas Journal 33:296–306

ABSTRACT: The Colorado National Monument (COLM), on the northeastern edge of the Uncompahgre Plateau, supports a persistent piñon (*Pinus edulis* Engelm.) – juniper (*Juniperus osteosperma* (Torr.) Little) woodland, which has not been disturbed by large stand-replacing fires since modern fire records began. We examined the fire history of large (> 100 ha) stand-replacing fires, documented tree population structures, and characterized tree density, quadratic mean diameter (QMD), relative composition, and cumulative mortality using 431 – 0.1-ha plots distributed over 1600 ha of the Monument. We found no evidence of large stand-replacing fires (charred wood or truncated stand structures) in the study area. Stand ages inferred from size structures suggest that large stand-replacing fires have been absent for possibly a millennia. Tree population structures show a more stable stand structure for juniper; piñon pine population structures show a more recent and sustained regeneration pulse. Cumulative mortality of piñon pines was 18%, peaking at 47% in trees 20 – 24.5 cm diameter. Spatial patterns of juniper density, QMD, and mortality were more homogeneous than those of piñon pine. Results suggest temporal dynamics and spatial patterns of the COLM woodland are more influenced by drought and small fires (< 10 ha) than large fires (> 100 ha). This study provides important baseline data for changes that may be brought about by climate change in coming decades. It also stresses the importance of controlling cheatgrass (*Bromus tectorum*) and other invasive species to increase resistance of these persistent piñon–juniper woodlands to future fires.

Index terms: drought, fire history, *Juniperus osteosperma*, mortality, persistent piñon–juniper woodlands, *Pinus edulis*, spatial structure, Uncompahgre Plateau

INTRODUCTION

Piñon (*Pinus edulis* Engelm.) – juniper (*Juniperus osteosperma* (Torr.) Little) vegetation covers at least 40 million hectares in the western United States (Romme et al. 2009). There is a great variety of piñon–juniper community types included within this expanse, with each type differing in structure, composition, and disturbance regimes. Despite this diversity, however, only fairly recently have researchers begun to differentiate between the historical fire regimes of piñon–juniper savannas, which likely burned somewhat frequently and at low severities prior to 1900, and piñon–juniper communities that burned less frequently (Baker and Shinneman 2004). One such community is persistent piñon–juniper woodland, a type that is especially prevalent on upland sites in the Colorado Plateau. Persistent piñon–juniper woodlands may occur on rocky sites with less productive soils and, therefore, typically lack the surface fuels in the understory that would support frequent fires or as in Mesa Verde National Park on deep, productive loess-covered sandstones fragmented by geographic barriers to fire (Romme et al. 2009). Studies conducted in the past decade in persistent piñon–juniper woodlands across the Colorado Plateau have revealed centuries-long natural fire rotations ranging from ~300 – 600+ years (Eisenhart 2004; Floyd et al. 2004; Floyd et al. 2008; Huffman et al. 2008; Shinneman and Baker 2009).

These long fire rotations place persistent piñon–juniper woodlands in a unique category among southwestern forests and woodlands. Unlike many other forest communities in the western United States, including piñon–juniper savannas, multiple lines of evidence suggest that fire regimes of persistent woodlands across the Colorado Plateau have not been significantly altered by twentieth century fire suppression (Romme et al. 2003; Baker and Shinneman 2004). Therefore, rather than needing the reintroduction of fire or mechanical treatments to restore stand structure, these systems may represent rare examples of communities still within their historical range of variation. Furthermore, due to their long fire-free intervals, these persistent woodlands offer a rare signature of how long-term influences, such as climatic variability or disturbances other than fire, can influence woodland structure and development (Romme et al. 2009; Shinneman and Baker 2009).

Despite the growing body of research that points to infrequent fires in persistent piñon woodlands, researchers still call for local research to develop site specific ecologically sound resource management plans. In their extensive literature review, Baker and Shinneman (2004) concluded that many assessments of the current condition of piñon–juniper woodlands in the western United States are based on incorrect conclusions about the natural fire regime and that local research is essential to develop effective, scientifically-based restoration prescriptions. Local research is particularly valuable, as Floyd et al. (2008) point out, when preservation of natural ecological processes and conditions is a primary management goal, as in National Parks and wilderness.

Local studies can also provide a suitable historical context for interpreting the ecological significance of large fires and other disturbances, if and when they occur, as well as help managers predict what is likely to occur under a future scenario of climate change-induced drought. Researchers have documented disturbances that may alter the structure and composition of persistent piñon–juniper woodlands across the Colorado Plateau. The severe drought of 2002 – 2004 was shown to cause a large-scale die-off of piñon pines from *Ips confusus* infestation, resulting in a regional shift in species composition (Breshears et al. 2005) and the associated ecological dynamics. There have also been a series of large stand-replacing fires in piñon–juniper communities across the Colorado Plateau, such as those that occurred in Mesa Verde National Park in the last decade (Floyd et al. 2004). While these large fires may still fall within the historical range of variability of the disturbance regimes of persistent woodlands (Floyd et al. 2004), it is uncertain whether they may be the beginning of a trend in larger fires with changing climate.

The goal of this study was to characterize the fire history, stand structure, and mortality of a persistent piñon–juniper woodland in the Colorado National Monument in western Colorado. The Colorado National Monument (hereafter COLM) offers a rare opportunity to study persistent piñon–juniper woodlands that have been less affected by Euro-American settlement or recent fire than other sites on the Colorado Plateau. This area was designated a National Monument in 1911, only three decades after EuroAmerican settlement of the nearby valley. And, unlike other areas of protected persistent piñon–juniper woodlands on the Colorado Plateau, the COLM has had no major recorded fires in the past century. However, similar to other the sites across the region, piñon pine mortality from an *Ips confusus* infestation visibly increased in the COLM following the 2002 – 2004 drought. Specifically, our study goals were to: (1) estimate when the last large (> 100 ha) stand-replacing fire occurred, (2) characterize and compare the population structures of piñon pine and juniper, and (3) document levels of cumulative mortality.

METHODS

Study site

The Colorado National Monument lies along the northeastern flank of the Uncompahgre Plateau, a distinctive uplift within the larger Colorado Plateau in western Colorado, USA (Figure 1). The COLM includes 8296 ha of mesas and canyons, with the mesas rising approximately 700 m above the valley floor to an elevation of over 2000 m. The canyons that cut into the mesas expose a geographical stratigraphy ranging from the Precambrian Age to the later Morrison and Wingate formations. The area managed by the COLM is contiguous with a larger protected area of canyons, the 49,897 ha McInnis Canyons National Conservation Area managed by the BLM. More than half of this National Conservation Area (NCA) (30,553 ha) is managed as wilderness (Black Ridge Canyons Wilderness).

The area of the COLM is within the lands originally included as part of the 1868 Colorado Ute Reservation Treaty. It was then ceded in 1880, when the Utes were forcibly removed to Utah and settlers arrived to claim agricultural land in the valley adjacent to the Monument. The area was formally designated a National Monument in 1911 and came under management of the National Park Service. Livestock grazing was mostly removed from the area after 1911 and fenced out of most areas by the 1930s, with a few exceptions made for livestock trail drives through historic routes until the 1980s (Shinneman et al. 2008). A bison herd was introduced in the 1920s and removed in the 1980s. Much of the grazing pressure was concentrated in the canyon bottoms or the southernmost part of the park (NPS 2004), but the remote mesa tops were only lightly or never grazed by livestock. NPS monitoring in the COLM also documented the lowest frequency of invasive species on the mesa tops, with the highest frequency in the canyon bottoms (Perkins 2010). In this study, we focused on the mesa tops since they were less disturbed by grazing than other areas of the COLM and, therefore, would better represent pre-settlement conditions.

The climate of this area is semi-desert. Annual precipitation, measured at the COLM visitor's center (same elevation as the study area) since 1948, averages 29 cm. While this precipitation is fairly evenly spread throughout the year, spring and early fall tend to be wetter. Snowfall averages approximately 79 cm annually. The average summer high temperature is 32 °C and the average winter low temperature is -6 °C (NPS, www.nps.gov/colm/naturescience/weather.htm).

As mentioned previously, we focused on the mesa tops of the COLM. The mesa tops are relatively flat areas dissected by deep canyons. The elevation of the mesa tops ranges from 1760 – 2080 m asl. They are found on sandstone formations that have resisted erosion (stream-deposited Kayenta sandstone and wind deposited Entrada sandstone). Soils within these areas are made up primarily of eolian (wind blown) material and are classified as very shallow (< 25 cm deep). The texture of the soils ranges from loamy sands (coarse) to sandy clay loams (fine) (NPS 2005). Piñon–juniper woodlands with a sparse understory (less than 5% total cover by shrubs and herbaceous species) are the most widespread vegetation type on the upper mesas of the COLM (Von Loh et al. 2007). Tree canopy cover ranges up to 45% and biological soil crusts are well developed in the spaces between plants (Von Loh et al. 2007). Wyoming big sage brush (*Artemisia tridentata ssp. wyomingensis*) dominates patches of the upper mesas where wind-deposited soils are deeper.

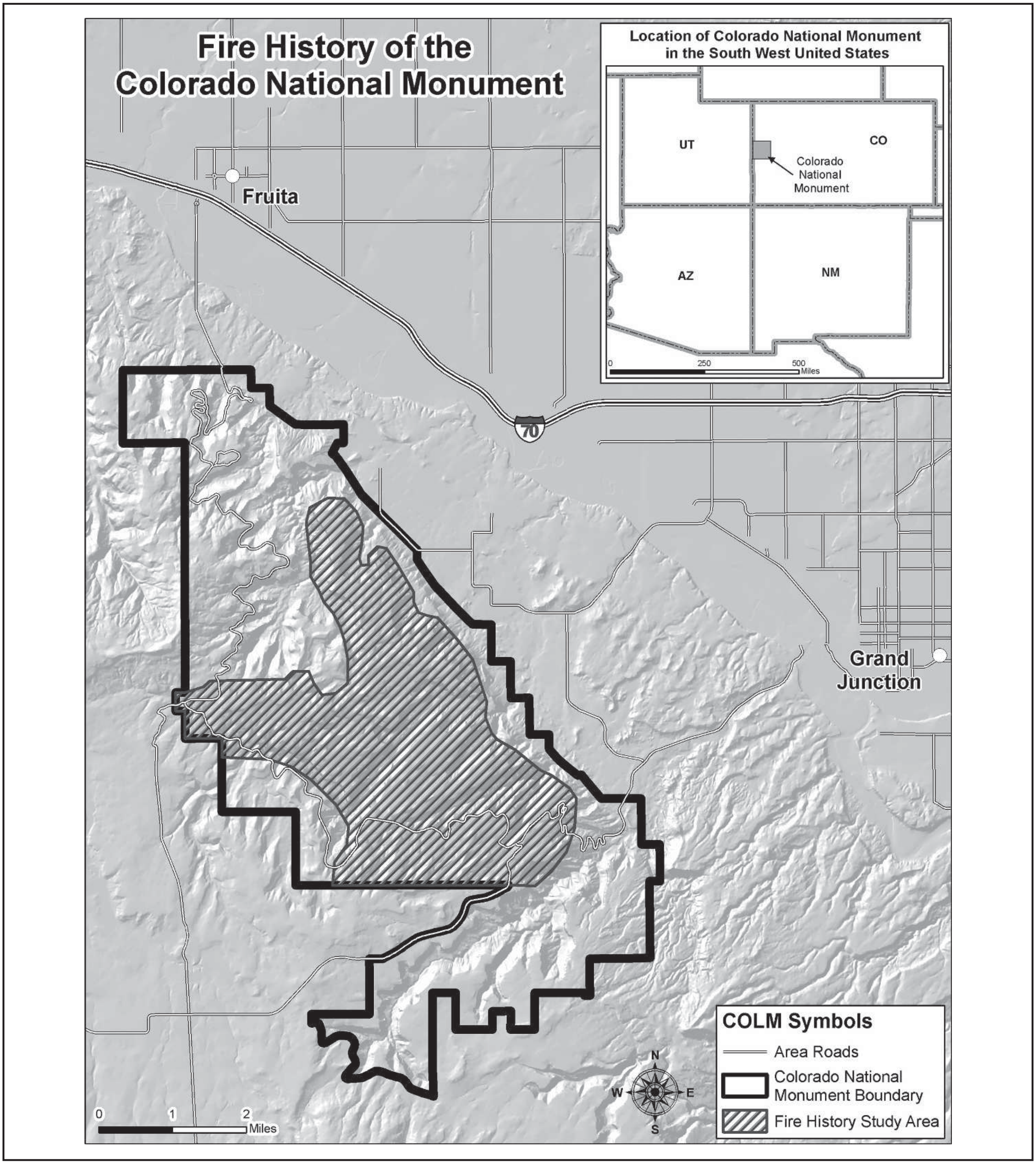

Figure 1. Location of study site in the Colorado National Monument in western Colorado.

Estimating fire history and stand ages

In this study, we focused on detecting evidence of large (> 100 ha) stand-replacing fires. Large, wind-driven fires in persistent piñon–juniper woodlands usually burn from crown to crown due to the lack of surface fuels, causing high mortality of piñon pines and junipers within the fire perimeter (Romme et al. 2009). This creates a landscape-scale fire scar that can be recognized in the decades following a fire since piñon pines and junipers require decades to regenerate (Miller and Tausch 2001). Centuries following these fires, their perimeters should still be recognizable as discrete patches with uniformly younger (smaller) trees than the surrounding area and charred wood on the ground. Within these patches, it can be assumed that the fire occurred prior to the dates of establishment of the oldest trees. This approach is similar to that used by Floyd et al. (2000, 2004, 2008), Huffman et al. (2008), and Shinneman and Baker (2009) in reconstructing the fire histories of piñon–juniper communities on the Colorado Plateau. The most commonly used technique for estimating fire histories, fire scar analysis, is less useful when working with piñon pines and junipers since both species are easily killed by fire (Barney and Frischknecht 1974; Koniak 1985). Although rare, researchers have detected fire-scarred individuals of both species (Brown et al. 2001; Huffman et al. 2008; Shinneman and Baker 2009), although many of these were found in ecotones of piñon–juniper and more flammable communities (ponderosa (*Pinus ponderosa*) or savanna). Fire scars are not useful for detecting large landscape-scale fires, which would typically be stand-replacing and consume fire-scarred trees (but see Baur and Weisberg 2009).

Using this approach, we estimated the ages of the largest piñon pine and juniper trees and using regression equations developed from tree ring analysis and mapped approximate stand ages across the mesa tops of the COLM. Sampling points were determined by overlaying a spatial grid at 200-m x 200-m spacing on the COLM area using ArcGIS. Grid points that fell across large contiguous areas on the mesa tops were selected for study. We focused on contiguous areas since these larger areas would allow us to detect the evidence of a large scale (> 100 ha) fire perimeter. Areas that were not contiguous with the main grid, or areas that were not accessible due to cliffs, were omitted. A total of 431 grid points were selected, encompassing an area of approximately 1600 ha (~40% of the mesa tops and ~20% of the entire COLM).

Grid points were located in the field using a Garmin etrex GPS unit over a three-year period, from September 2007 to June 2010. At each grid point, the largest piñon pine and the largest juniper within 10 m of the grid point were selected and their basal diameters measured. No junipers were found at nine of the grid points, so a total of 422 large junipers were measured. No piñon pines were found at 19 of the grid points, so a total of 412 large piñon pines were measured. While walking between grid points, visible evidence of previous fires, such as charred trees or charred downed wood, were also noted and GPS coordinates were recorded for the charred location.

Estimating tree ages

To develop tree age – diameter relationships, quadratic regression equations were developed from stem cross sections or increment cores of junipers (n = 23) and piñon pines (n = 23) that were aged by counting tree rings. The number of samples we were able to obtain was very small due to limitations on destructive sampling within the COLM. All of the juniper samples and 20 of the piñon pine samples were full trunk cross sections (cut as close to the ground as possible) obtained from fuel reduction treatments occurring in the COLM during 2007 – 2010. Three of the piñon pine samples were increment cores sampled in 2011 to extend the samples to larger size classes. From preliminary studies, we determined that the oldest junipers are several centuries older than the oldest piñon pines, and, therefore, the more useful focal species for this study. Due to the difficulty of obtaining reliable increment cores from large old junipers with anomalous stem growth and extremely dense wood, we did not core junipers, but limited the tree ring analysis to the full stem cross sections. All samples were sanded and ring counted fewer than 45 x magnifications. Half of the samples were cross checked by three researchers and the age of one sample was verified at the Laboratory for Tree Ring Research, Arizona. We attempted to cross date our piñon pine and juniper stem sections with two different master chronologies: one developed from piñon samples north of the COLM (J. Dean, unpubl. data and one developed from samples south of the COLM (W. Baker, unpubl. data). Our samples did not cross date with either of these chronologies closely, suggesting a site level chronology needs to be developed. However, due the difficulty of developing a master chronology with our small sample size, we based our age estimates on raw ring counts. Shinneman and Baker (2009) found that the average net error of raw ring counts of juniper collected from the Uncompahgre Plateau due to false and missing rings is +0.5 yr/100 yr; thus, we considered raw ring counts reasonable estimates of juniper age, particularly since we were only interested in maximum age.

Characterizing spatial patterns of stand structure and mortality

At the same 431 grid points where the largest trees were measured, 100-m^2 circular plots were established centered on each grid point. In each plot, the diameter of all piñon pines and junipers (both living and standing dead) rooted in the plot were measured. All trees were measured at the base of the trunk or stem. Measuring at the stem base made estimating tree ages more reliable as stem sections that were aged by counting tree rings were all cut as close to the ground as possible. Also, junipers at this site often branch low to the ground so the diameter below this branching, rather than 1.3-m height (dbh), is more indicative of the overall size class of the tree.

Recorded fire records

COLM has kept records of fire occurrence in the park since 1942. Recorded information includes the location and size of fire. We obtained these data to describe characteristics of more recent fires and to compare with field observations of charred snags or wood.

Analysis

Density, quadratic mean diameter, and percent mortality of both piñon pines and junipers, and the percentage of piñon pines of total tree density (relative composition) were calculated for each 100-m^2 plot. Quadratic mean diameter, which gives greater weight to larger trees than the arithmetic mean, was calculated as: $QMD = \sqrt{(\sum d_i^2)/n}$ where d_i is basal diameter and n is the total number of trees. To characterize the amount of variation between plots, coefficients of variation were calculated for all parameters.

RESULTS

Estimated stand ages

The oldest juniper stem cross section dated was approximately 920 years old (Figure 2). The regression equation using a quadratic model for tree age and basal diameter developed from the 23 juniper cross sections was: age = -0.2995(diameter)^2 + 35.453(diameter) – 209.98; r^2 = 0.73 (Figure 2a). We observed that most of the outliers in the linear regression were juniper stems where the pith was located close to, in some cases directly on, the edge of the stem. Plotting stem radius (the measured distance between the pith and the bark along the line where tree rings were counted) rather than diameter resulted in a better linear model (r^2 = 0.88, Figure 2b), showing that growth rates are not as anomalous as stem anatomy suggests. However, this equation was not used to predict ages of trees measured in the field due to the difficulty of determining the location of the pith without destructively sampling the tree.

There was not a single size class that dominated the largest junipers measured at each of the 422 grid points; instead, the histogram of these largest junipers was normally distributed. The average basal diameter of these largest junipers was 46 cm (Figure 3a), estimated to be approximately 863 years old based on the size x age relationship. Thirty-two percent of these largest junipers exceeded 54 cm, the size of the largest juniper cross section dated; therefore, we have low confidence in predicting the ages of these trees using the regression equation and instead group them into a general age class of > 900 years old (Figures 3a and b). This age class (> 900 yrs) was, in fact, the most abundant age class in the frequency distribution of estimated ages.

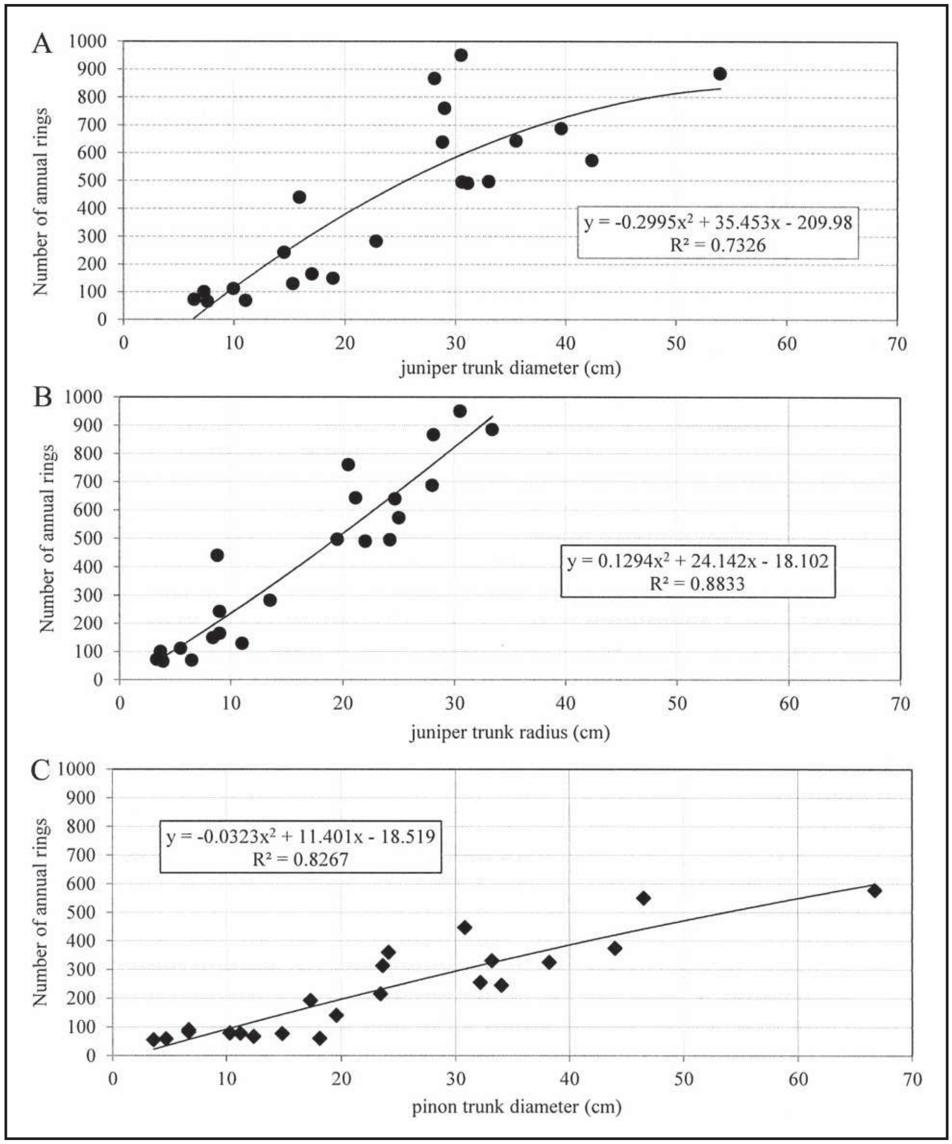

Figure 2. Age-size relationships for (A) juniper using trunk diameter, and (B) juniper using the same stem samples as in B, but using the radius of the stem section from the pith to the outermost ring, and (C) piñon pine using trunk diameter. Ages were determined using full stem cross sections for juniper (n = 23) and a combination of stem cross sections and increment cores (n = 20 and 3, respectively) for piñon pine.

The oldest piñon pine sample dated was approximately 577 years old (Figure 2c). The linear regression between tree age and basal diameter developed from the 23 piñon pine stem samples was: age = -0.0323(diameter)^2 + 11.40 (diameter) – 18.519; r^2 = 0.83. The 577 year old sample was from a 66 cm diameter tree, which is close to the upper end of sizes sampled in the study area (the largest piñon pine measured in the field was 76 cm diameter); therefore, we are more confident using the linear equation across the range of piñon pines measured in the field. The average basal diameter of

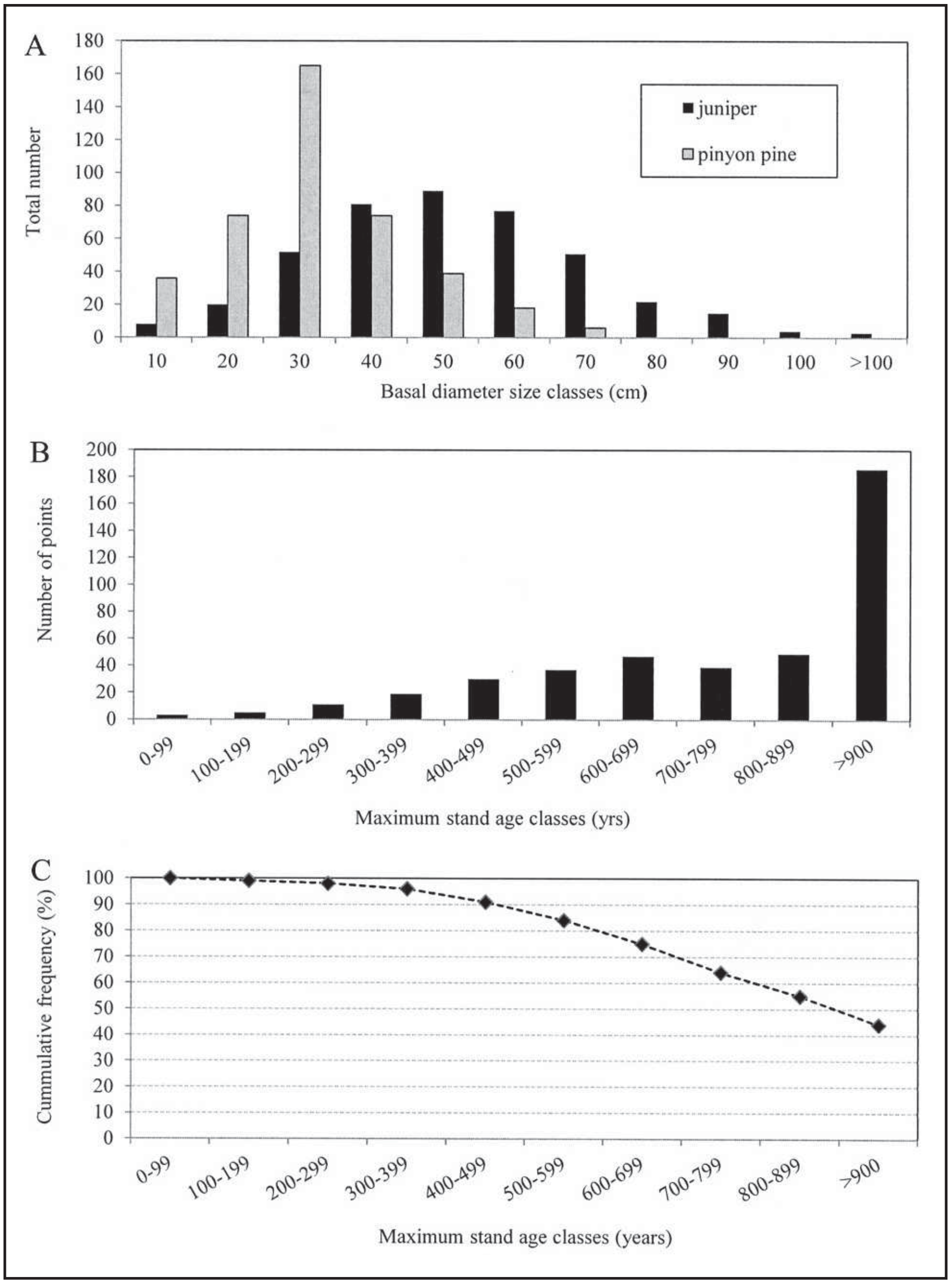

Figure 3. (A) Frequency distribution of the largest junipers and largest piñon pine individuals, as measured by basal diameter, at each of 426 sampling points in the study area. (B) Frequency distribution of stand ages estimated using the largest trees sampled at each point and regression equations. (C) Cumulative stand age distributions based on the data in B.

the 412 large piñon pines located near grid points was 27 cm (Figure 3a), estimated to be approximately 238 years old.

Comparing both species, junipers were the oldest (as estimated by regression equations) at 94% of the grid points. Considering the oldest trees at each grid point (juniper or piñon), more than half (56%) were trees older than approximately 800 years. A full 85% of grid points had trees older than approximately 500 years (Figure 3). Assuming these stand ages resulted from fires, this would result in a fire rotation ranging from 588 – 1428 years.

Evidence of past fires

There was no physical evidence of a large fire (> 100 ha) in the study area. No extensive areas of charred trees or charred wood on the ground were observed at sampling points or walking between sampling points. Two small areas of charred trees were found in the sampling area; both of these were found to be twentieth century fires that had been recorded in the COLM fire records and were both less than 1 ha. Evidence of charred trees was found at an additional 6 points, but was all limited to 1 – 3 trees each. The COLM fire records also show that small lightning-ignited fires are not uncommon in this park. Over the time period from 1942 – 2009, there have been numerous small fires (91 fires < 1 acre (0.4 ha)); 9 fires 1 – 4 acres (0.4 – 1.6 ha), 3 medium sized fires (6.9, 8.1, and 39.7 ha), but no fires larger than 39.7 ha.

The spatial distributions of the largest junipers revealed few distinct patches of uniformly smaller trees that could suggest a previous stand-replacing fire. The only potential area was a patch of 12 contiguous grid points with junipers less than 26 cm diameter (~450 years old) in the upper portion of the lower study area. This area differed from surrounding areas in that it consisted of piñon–juniper intermixed with sage shrublands. While this could represent a seral area regenerating from a past fire, we found no indication that a fire had occurred (charred wood, stumps, or snags).

Stand structure and composition

Overall stand density was 827.4 trees/ha across the study area (all size classes, live and dead; Table 1). Piñon pines were more abundant with an overall density of 487.5 trees/ha compared to 339.9 trees/ha for juniper; however this was mainly due to a higher abundance of smaller trees (< 5 cm diameter). Including that size class, piñons comprised 59% of trees overall; but excluding trees < 5 cm diameter, piñons comprised 50%. Piñon pine density was more variable between plots than juniper density as reflected by its higher coefficient of variation (Table 1).

The average quadratic mean diameter of juniper was more than double that of piñon pines, reflecting the overall larger size of junipers (Table 1). QMD of piñons was also skewed to the lowest class (Figure 4), while the frequency of juniper QMDs

Table 1. Summary statistics on general woodland stand structure in Colorado National Monument calculated from 431 100 m^2 plots.

Stand traits and species	Mean	SD	Median	Range	CV
Diameter of largest trees w/in 10 m of gridpoints (cm)					
Juniper	46.6	19.0	45.8	3-120	41%
Pinon	27.4	12.8	26.2	1-76.5	47%
Total density -all trees live and dead (#/ha)					
Juniper	339.9	267.3	300	0-1500	79%
Pinon	487.5	436.1	400	0-2900	89%
Both	827.4	588.3	700	0-3300	71%
Density- live trees only (#/ha)					
Juniper	291.0	229.9	300	0-1200	79%
Pinon	399.8	389.1	300	0-2900	97%
Both	690.0	501.2	600	0-3300	73%
Pinyon:Juniper ratio	59%	28	57%	0-100%	47%
Quadratic Mean Diameter					
Juniper	23.7	14.9	24.3	0-82.5	63%
Pinon	10.5	8.3	9.5	0-48.5	80%
Both	19.1	10.2	18.4	0-53.0	53%
Percent dead (%)					
Juniper	14	21	0	0-100	150%
Pinon	18	29	0	0-100	161%

was more normally distributed. Similar to density, the QMD of piñon pine was more variable between plots than that of juniper as reflected by its higher coefficient of variation (Table 1).

The size class structure of juniper showed an abundance of trees in the < 5 cm size class, but more evenly distributed numbers of trees in the 5 – 39.9 cm size classes (Figure 5). Notably, there were slightly more junipers in the 20 – 24.9 cm size class than the neighboring size classes. The size class structure of piñon pines had a distinctive reverse J-shaped curve, with an abundance of small trees < 5 cm diameter (Figure 6).

Mortality

Overall mortality of piñon pines was 18%; however, mortality ranged from 30% – 47% in trees 10 – 40 cm diameter, peaking in the 20 – 24.5 cm size class (Figure 6). Juniper mortality overall was lower (14%), and ranged below 30% for all size classes except for 60 – 65 cm diameter trees (Figure 5). In general, juniper mortality was distributed more evenly across the study area than piñon mortality, as indicated by its lower coefficient of variation (Table 1).

DISCUSSION

Fire history and stand age

We found no physical evidence of large (> 100 ha) stand-replacing fire in our study site but ample evidence of small fires (< 10 ha); therefore, the range of possible fire rotations calculated from stand ages (588 – 1428 yrs) may reflect the influence of these many small fires in addition to the large fires that were the focus of our study. However, multiple lines of evidence support the idea that large stand-replacing fires have been absent for time periods approaching a millennia. For example, junipers estimated to be > 900 years old were found at 44% of sampling points throughout the study area. Since this evidence suggests that fire intervals at this site may approach or exceed the average life span of junipers, non-living indicators such as charcoal may provide a more accurate estimate of fire intervals. Scott et al. (2001) dated 11 charcoal layers sampled from the major canyon draining the easternmost boundary of the study area using ^{14}C techniques and found the most recent date was 1180 YBP (Scott et al. 2001). While it is likely this

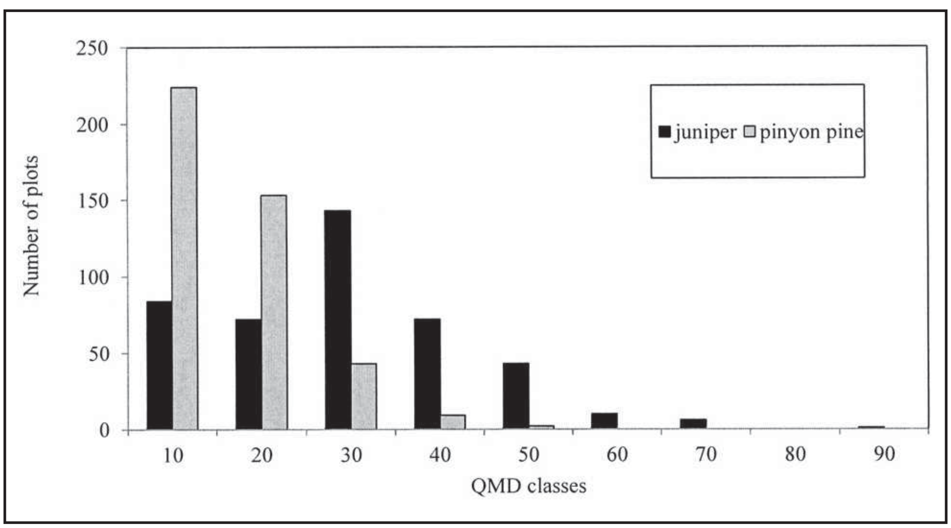

Figure 4. Frequency distribution of the quadratic mean diameters calculated separately for juniper and piñon pine trees measured in 431–100 m^2 plots in the Colorado National Monument. All size classes were included in the calculation, from seedlings and larger.

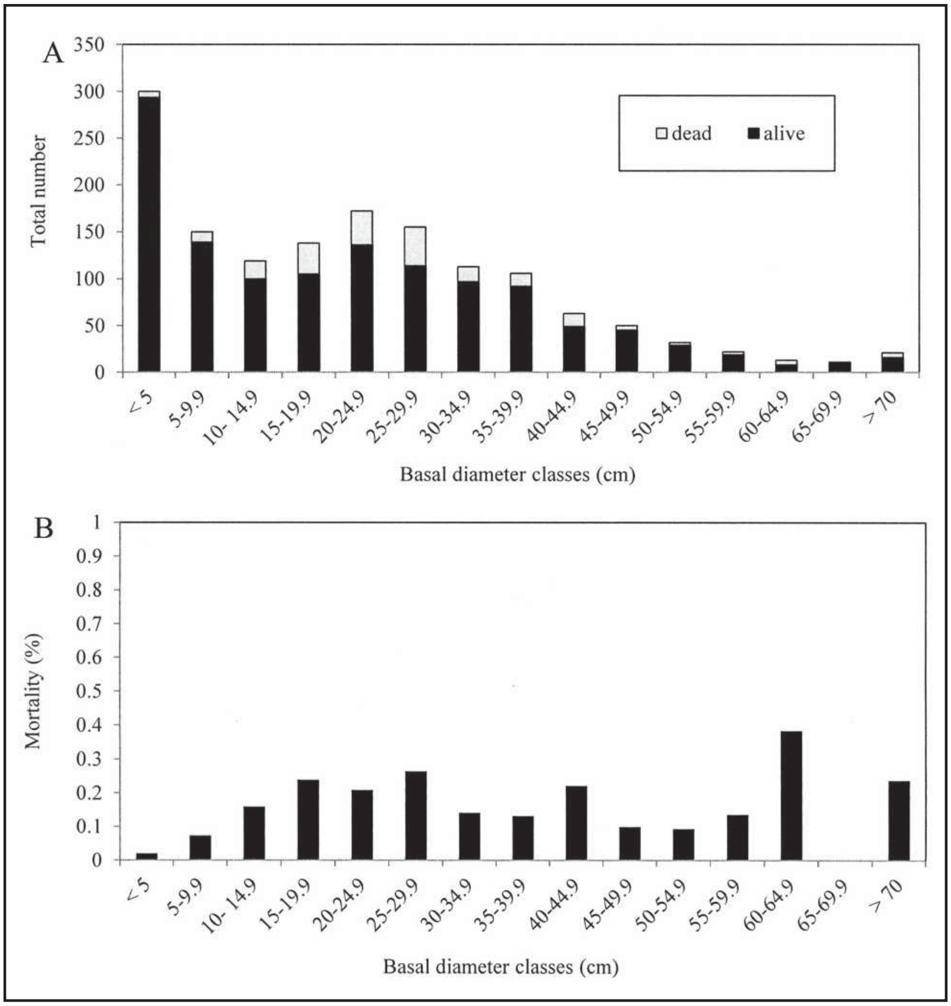

Figure 5. (A) Juniper size class distribution (live and standing dead) and (B) juniper percent mortality by size class of all juniper trees in measured in 431 100 m^2 plots (4.31 ha) in the Colorado National Monument. Size classes are based on the diameter of stems measured at the stem base.

date exceeds the date of the last wildfire (due to the fact that the charcoal may have come from piñon pines or junipers that were already several centuries old at the time of the fire), the average interval between the 11 charcoal dates was 790 yrs, which also points to very long fire-free intervals. A third source of evidence is the lack of charred wood in the study area. Research from this region suggests that dead piñon pine or juniper wood on the ground persists for as long as 1100 years and 600-year-old dead wood is relatively common (Baker et al. 2008). Charred wood is expected to persist even longer due to the fact that char is resistant to decay. Indicative of this, direct evidence of old fires (300 – 400+ yrs old) in the form of charred wood has been found in other studies of persistent piñon–juniper woodlands (Eisenhart 2004; Floyd et al. 2008; Bauer and Weisberg 2009; Shinneman and Baker 2009).

A significant finding of this study is that the fire rotation of the COLM may be longer and the woodlands older than other persistent piñon–juniper woodlands documented in the Colorado Plateau. For example, Shinneman and Baker (2009) found fire rotations of > 400 – 600 yrs at sites throughout the Uncompahgre Plateau; Floyd et al. (2008) found similar fire rotations at Navajo Point, Utah; Huffman estimated rotations of 340 and 290 years at sites in Arizona and New Mexico, respectively. The oldest juniper tree dated from the COLM, 922 years old, was at least a century older than other old trees (piñon pine or juniper) documented from other sites in the region. For example, Baker et al. (2008) dated an 800 year-old piñon pine in the Douglas Creek Arch area in northwestern Colorado; Shinneman and Baker (2009) dated a 699 year-old juniper from a site further south on the Uncompahgre Plateau in western Colorado; Floyd et al. (2008) dated a 578 year-old piñon pine in Navajo Point, Arizona; and Floyd et al. (2000) aged a ~500 year old juniper in Mesa Verde National Park.

The rarity of large landscape-scale fires in the COLM is not likely due to a lack of ignitions. In the 67 years since the COLM started keeping records, there have been on average 1.5 small fires recorded each year. What is striking is that over such a long time period, none of these ignitions has resulted in a large fire.. It is well known that conditions conducive to large fires have occurred many times recently in Mesa Verde National Park, where over nine major fires (> 100 ha) have burned through piñon–juniper woodlands, petran chaparral, and Douglas fir (*Pseudotsuga*) in the past century (NPS 2008). Of significant concern is a recent spate of large fires in piñon–juniper woodlands within 40 km of the COLM, where 15 large fires (> 100 ha) have occurred since 1989. The total area burned by these 15 fires is over 10,000 ha, more than the total area of the COLM.

Differences in weather, topography, and fuels may explain why the COLM has a longer fire rotation than nearby piñon–juniper woodlands. Being located on the northeastern flank of the Uncompahgre Plateau, the COLM is often on the leeward side of major weather systems that generally move from the southwest/west to the northeast/east. All of the large fires that occurred within 40 km of the COLM since 1989 were located on the western, or windward, side (13 fires) or top (2 fires) of the plateau.

Topographical fuel breaks may also play a role. Most of the study area is isolated by sheer cliffs (100+ m tall) and represents essentially islands or peninsulas of fuel. This decreases the chances of large fires spreading into these areas from ignitions outside the perimeters, and it decreases the chances of an ignition within the perimeter growing into a large fire before being

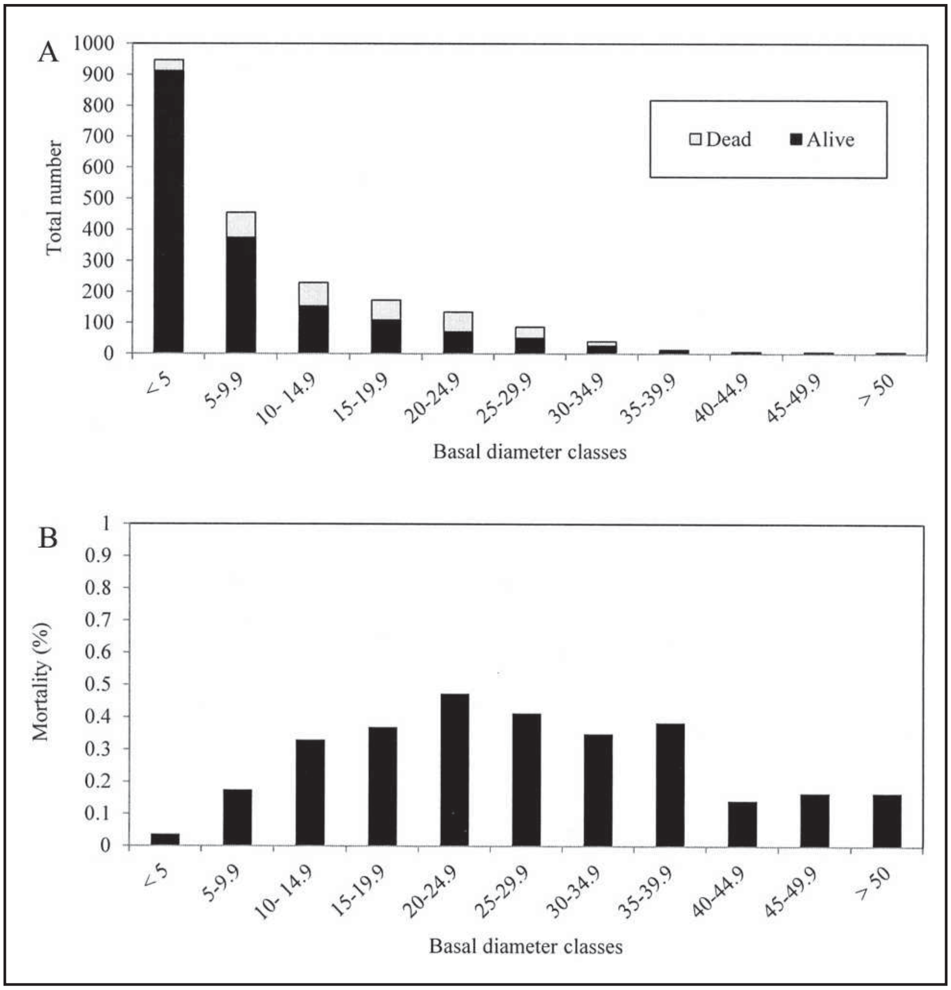

Figure 6. (A) Piñon pine size class distribution (live and standing dead) and (B) piñon pine percent mortality by size class of all juniper trees in measured in 431 100 m^2 plots (4.31 ha) in the Colorado National Monument. Size classes are based on the diameter of stems measured at the stem base.

stopped by a cliff. There are numerous large patches of exposed sandstone within the study area that also serve as barriers to fire spread.

A more discontinuous canopy may also contribute to the longer fire-free intervals at the COLM. Overstory stand density found in this study was 30% lower than at other piñon–juniper sites across the Uncompahgre Plateau as reported by Shinneman and Baker 2009 (410 vs. 589 live trees/ha, > 5 cm diameter). Working in the same area, Eisenhart (2004) found that higher elevations (> 2000 m) coincide with conditions more conducive to fire, such as greater canopy continuity and increased lightning. Most of the mesa tops at COLM are below 2000 m, consistent with the low fire frequencies we found. While overstory stand density was lower at the COLM relative to sites surveyed across the Uncompahgre Plateau by Shinneman and Baker, this was due to fewer piñon pines at this site (188 vs. 366 live pinions/ha > 5 cm diameter). Notably, the juniper component was identical (223 live junipers/ha > 5 cm diameter). Shinneman and Baker did note that in their survey the oldest woodlands were juniper-dominated and found at lower elevations.

Lack of fine surface fuels, especially continuous flammable components, are recognized as the limiting factor in the spread of fires in some persistent woodlands (Romme et al. 2009); and, therefore, differences in surface fuels are also likely to play a part in the contrasting fire frequencies of areas within and outside the COLM. The burned areas outside the COLM have experienced significantly more livestock grazing than our study site. It would be fruitful to survey these areas to determine if increased grazing pressure resulted in a higher cover of invasive species such as cheatgrass (*Bromus tectorum*), providing a contiguous layer of fine surface fuels more conducive to fire spread. Invasive plant monitoring in the COLM revealed that the mesa tops had the lowest frequency of invasive plant species while the canyon bottoms had the highest (Perkins 2010). This would be consistent with the grazing history of the park, since, historically, grazing was mostly limited to the canyon bottoms. Cheatgrass is not currently on the COLM high priority list for weed control; however, it will be important to continue to monitor and control this and other invasive species to prevent the spread of future fires through these non-native fuel components.

Mortality

Piñon mortality found in this study was lower than in other studies documenting mortality following the 2002 – 2004 drought. This is consistent with remote sensing data Breshears et al. (2005) used to detect changes in piñon–juniper woodlands following this drought. While piñon pine mortality was fairly widespread in the southwestern U.S., the areas of highest mortality were further south than the COLM and reached peaks as high as 90% in New Mexico (Breshears et al. 2005). Since we only documented standing dead trees, our mortality figures may have underestimated actual level, since many piñon pines may have fallen since the drought and *Ips* outbreak and not have been included in sampling. Conversely, junipers are more deeply rooted and less susceptible to root disease and, therefore, may remain upright as snags for decades. Therefore, we believe that the juniper mortality documented (14%) is not as much of a result of the recent drought, as in the piñon pines, but represents mortality accumulated over a much longer time period.

The higher piñon mortality will change the relative composition of woodland stands at the COLM, but only slightly – from 59% to 58% piñon pine (live and dead vs. live trees only). However, the higher mortality rates in larger size classes will result in a more notable shift in overstory composition (trees > 5 cm diameter) from 50% to 46% (live and dead vs. live trees only). Since mortality was higher in trees of cone-bearing age, there may be further changes in woodland composition for several more cohorts (Mueller et al. 2005).

Woodland structure

Stand structures at COLM support the general observation by Romme et al. (2009) that living trees in persistent piñon–juniper woodlands are typically very old (300 – 1000 yr) and exhibit a multiaged structure that reflects episodic tree recruitment events. The population structure of piñon pine at COLM follows a reverse-J distribution and is structurally similar to those documented by Shinneman and Baker (2009) who used age classes to show a pulse of piñon regeneration that started in the 1800s. The population structure of junipers at the COLM reveals at least one major episode of regeneration several centuries ago, although it is difficult to pinpoint exactly the time period using size classes. In their detailed study, Shinneman and Baker found six episodes of juniper regeneration on the Uncompahgre starting in the 1550s and concluded that the age structures of both species were strongly influenced by broad-scale multidecadal climate patterns. While junipers increased slightly during drought periods, they generally maintained a relatively steady density over the past 500-year period. In contrast, an extended dry period from 1620s – 1820s preceded the dramatic increase in piñon pines that occurred around 1800. It is notable that the COLM has an abundance of junipers older than 500 years. There is a general lack of both piñon pines and junipers in the southwest older than 400 – 500 years believed to be associated with a megadrought in the 1500s (Swetnam and Betancourt 1998); however, this megadrought is not believed to have been important as far north as the COLM.

Taken together, the results of this study support the conclusion by Romme et al. (2009) that temporal stand dynamics in piñon–juniper are driven more by climatic fluctuation (and the resulting insects and disease) than by fire. These temporal patterns include both chronic, low-level tree mortality and episodic mortality and recruitment events. On a spatial scale, the lack of large fires, but consistent occurrence of small fires, would result in a very fine-grained forest structure. Evidence of this fine-grained pattern is reflected particularly by a relatively consistent juniper density and QMD across the study area. The importance of smaller patch-scale events in creating a fine-grained pattern of forest stand structure in persistent piñon–juniper woodlands has been noted by other researchers (Floyd et al. 2008; Huffman et al. 2008). Notably, piñon density, mortality, and QMD show more spatial variability than juniper. This difference in spatial patterning between piñon and juniper may be related to their drought-tolerances and differential ability to survive across a wide range of microhabitats. The higher drought and insect tolerance of juniper may explain why it has maintained a more stable stand structure for centuries and a population that is more evenly structured across space.

CONCLUSION

This study supports the findings of other recent studies in persistent piñon–juniper woodlands on the Colorado Plateau, which have found long fire-free intervals (Floyd et al. 2004, 2008; Shinneman and Baker 2009; Huffman et al. 2008). As these authors also concluded, the implications of these long fire rotations are that these woodlands have not been substantially changed by fire exclusion in the past century and, therefore, are not outside of their historic range of variation in stand structure, fire frequency, and fire behavior. As such, prescribed underburns or mechanical thinning of these forests do not represent ecological restoration and, in fact, can do long-term damage by removing old-growth trees and opening up sites for invasion by introduced species.

While consistent with these earlier studies, the present study significantly expands the known range of fire-free intervals for piñon–juniper woodlands. With an abundance of trees approaching 1000 years, no evidence of previous large fires, and charcoal layers that date to 1180 BP, this area has likely not experienced a large-scale fire for time scales approaching a millennia. The current spatial pattern of stand structure at the COLM represents what old-growth woodlands can attain in the absence of large, intense disturbances. These structures reflect a relatively uniformly distributed juniper population, but a more spatially variable distribution of piñon that may reflect the patchy nature of more frequent (relative to juniper) recruitment and mortality episodes.

This study also provides important baseline data for changes that may be brought about by climate change in coming decades. As many ecologists have noted, we may be witnessing a unique period where vegetation patterns are being shifted over large areas in a short period of time. With the expected increase in drought conditions in the Southwest, we may see a sharp decline in the piñon component of many piñon–juniper woodlands. If climate change brings weather conditions more conducive to wind-driven fire, or if cheatgrass and other invasive species that create contiguous fine surface fuels continue to spread, we may lose large areas of piñon pine and juniper and the communities they support. The one manageable component of this system that could increase the resilience of the piñon–juniper woodlands to fire is controlling cheatgrass.

ACKNOWLEDGMENTS

The National Park Service supported this study through the Colorado Plateau Cooperative Ecosystem Studies Unit (Cooperative Agreement Number H1200-09-0005). We thank the following students in the Department of Physical and Environmental Sciences at Colorado Mesa University for contributing to data collection: Seth Wilsey, Kate Burden, Zach Walker, Sara Fleishman, Sam Morrison, Josh Jahner, Julie Robinson, Becca Frenier, Julianna Aycock, Shad Johnson, Ben Potter, Andy Cook, and Jake Pollert.

Deborah K. Kennard is an associate professor in the Department of Physical and Environmental Sciences at Colorado Mesa University in Grand Junction, Colorado.

Amanda J. Moore graduated from Colorado Mesa University with a B.S. in Environmental Science and is currently GIS Coordinator for the Bureau of Land Management Gunnison Field Office and is pursuing a M.S. in Environmental Protection and Natural Resource Management with a focus in GIS through the University of Denver.

LITERATURE CITED

Baker, S.G., J.S. Dean, and R.H. Towner. 2008. Final report for the old wood calibration project, Western Colorado. Centuries Research, Inc. Montrose Community Foundation and State Historical Fund, Project No. 06-M1-018, Montrose, Colo.

Baker, W.L., and D.J. Shinneman. 2004. Fire and restoration of piñon–juniper woodlands in the western United States: a review. Forest Ecology and Management 189:1-21.

Barney, M.A., and N.C. Frischknecht. 1974. Vegetation changes following fire in the piñon–juniper type of western central Utah. Journal of Range Management 27:91-96.

Bauer, J., and P.J. Weisberg 2009. Fire history of a central Nevada pinyon–juniper woodland. Canadian Journal of Forest Research 39:1589-1599.

Breshears, D.D., N.S. Cobb, P.M. Rich, K.P. Price, C.D. Allen, R.G. Balice, W.H. Romme, J.H. Kastens, M.L. Floyd, J. Belnap, J.J. Anderson, O.B. Myers, and C.W. Meyer. 2005. Regional vegetation die-off in response to global change-type drought. Proceedings of the National Academy of Sciences 102:15144-15148.

Brown, P.M., M.W. Kaye, L.S. Huckaby, and C.H. Baisan. 2001. Fire history along environmental gradients in the Sacramento Mountains, New Mexico: influences of local patterns and regional processes. Ecoscience 8:115-126.

Eisenhart, K.S. 2004. Historic range of variability and stand development in piñon–juniper woodlands of western Colorado. Ph.D. diss., University of Colorado, Boulder.

Floyd, M.L., D.D. Hanna, and W.H. Romme. 2004. Historical and recent fire regimes in Piñon–Juniper woodlands on Mesa Verde, Colorado, USA. Forest Ecology and Management 198:269-289.

Floyd, M.L., W.H. Romme, and D.D. Hanna. 2000. Fire history and vegetation pattern in Mesa Verde National Park, Colorado, USA. Ecological Applications 10:1666-1680.

Floyd, M.L., W.H. Romme, D.D. Hanna, M. Winterrowd, D. Hanna, and J. Spence. 2008. Fire history of piñon–juniper woodlands on Navajo Point, Glen Canyon National Recreation Area. Natural Areas Journal 28:26-36.

Huffman, D.W., P.Z. Fulé, K.M. Pearson, and J.E. Crouse. 2008. Fire history of piñon–juniper woodlands at upper ecotones with ponderosa pine forests in Arizona and New Mexico. Canadian Journal of Forest Research 38:2097-2108.

Koniak, S. 1985. Succession in piñon–juniper woodlands following wildfire in the Great Basin. Great Basin Naturalist 45:556-566.

Miller, R.F., and R.J. Tausch. 2001. The role of fire in juniper and pinyon woodlands: a descriptive analysis. Pp. 15–30 *in* K.E.M. Galley and T.P. Wilson, eds., Proceedings of the Invasive Species Workshop: the Role of Fire in the Control and Spread of Invasive Species. Fire Conference 2000: The First National Congress on Fire Ecology, Prevention, and Management. Miscellaneous publication no. 11, Tall Timbers Research Station, Tallahassee, Fla.

Mueller, R.C., C.M. Scudder, M.E. Porter, R.T. Trotter, III, C.A. Gehring, and T.G. Whitham. 2005. Differential tree mortality in response to severe drought: evidence for long-term vegetation shifts. Journal of Ecology 93:1365-2745.

[NPS] National Park Service. 2005. Final general management plan – environmental impact statement. Colorado National Monument, Grand Junction.

[NPS] National Park Service. 2004. Colorado National Monument: NCPN Phase III Report Appendix E. Available online <http://science/matire/m[s/gpv/im/units/ncpn/NCPN.cfm>.

[NPS] National Park Service. 2008. Fire history: 1933 – 2008 Map. Available online <www.nps.gov/meve/parkmgmt/upload/fire_map_2008.pdf>.

Perkins, D.W. 2010. Invasive exotic plant monitoring at Colorado National Monument, 2009. Natural Resource Technical Report NPS/NCPN/NRTR-2010/354, National Park Service, Fort Collins, Colo.

Romme, W.H., M.L. Floyd-Hanna, and D.D. Hanna. 2003. Ancient piñon–juniper forests of Mesa Verde and the West: a cautionary note for forest restoration programs. Pp. 335-350 *in* Fire, Fuel Treatments, and Ecological Restoration: Conference Proceedings, 16–18 April 2002, Fort Collins, Colo. Proceedings RMRS-P-29, U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, Colo.

Romme, W., C. Allen, J. Bailey, W. Baker, B. Bestelmeyer, P. Brown, K. Eisenhart, L. Floyd, D. Huffman, B. Jacobs, R. Miller, E. Muldavin, T. Swetnam, R. Tausch, and P. Weisberg. 2009. Historical and modern disturbance regimes of piñon–juniper vegetation in the western US. Journal of Range Management 62:203-222.

Scott, R.B., A.E. Harding, W.C. Hood, R.D. Cole, R.F. Livaccari, J.B. Johnsons, R.R. Shroba, and R.P. Dickerson. 2001. Geologic map of Colorado National Monument and adjacent areas, Mesa County, Colorado. Geological Investigations Series I-2740, U.S. Geological Survey, Denver, Colo.

Shinneman, D.J., and W.L. Baker. 2008. Ecological restoration needs derived from reference conditions for a semi-arid landscape in Western Colorado, USA. Journal of Arid Environments 72:207-227.

Shinneman, D.J., and W.L. Baker. 2009. Historical fire and multidecadal drought as context for piñon–juniper woodland restoration in western Colorado. Ecological Applications 19:1231-1245.

Swetnam, T.W., and J.L. Betancourt. 1998. Mesoscale disturbance and ecological response to decadal climatic variability in the American southwest. Journal of Climate 11:3128-3147.

Von Loh, J., K. Landgraf, A. Evenden, T. Owens, S. Blauer, and M. Reid. 2007. Vegetation Classification and Mapping Project Report, Colorado National Monument. Natural Resource Technical Report NPS/NCPN/NRTR-2007/061, National Park Service, Fort Collins, Colo.