Effects of Multiple Wildland Fires on Ponderosa Pine Stand Structure in Two Southwestern Wilderness Areas, USA

Zachary A. Holden[1,*], Penelope Morgan[1], Matthew G. Rollins[2], and Kathleen Kavanagh[1]

[1]Department of Forest Resources, University of Idaho,
Moscow, Idaho 83844-1133

[2]Fire Sciences Laboratory, Rocky Mountain Research Station,
USDA Forest Service, Missoula, MT 59807

[*]Corresponding author: Tel.: (208) 885-1202; e-mail: zholden@vandals.uidaho.edu

ABSTRACT

The effects of 30 years (1972-2003) of Wildland Fire Use for Resource Benefit (WFU) fires on ponderosa pine forest stand structure were evaluated in the Gila Wilderness, New Mexico, and the Saguaro Wilderness, Arizona. Tree density, diameter-class distributions, basal area, and stand density index were compared among areas that burned with different frequencies since 1972 and areas that burned mid-century (1940-1950) and again during the WFU era (1972-2003). In both the Saguaro Wilderness and the Gila Wilderness, significantly fewer small-diameter (5 cm to 22.5 cm) trees occurred in areas that burned multiple times since 1972 compared to areas that were unburned ($p < 0.05$) during this time. The density of large-diameter (>45 cm diameter breast height) trees in the Gila Wilderness was highly variable and did not differ significantly among fire treatments ($p > 0.32$). In the Saguaro Wilderness, significantly more large-diameter trees (>45 cm dbh) occurred in areas that burned mid-century and again during WFU than in all other fire treatments. Mean 10-year basal area increment growth rates (1840 to the present) of trees in the Gila Wilderness that experienced mid-century fires suggest that those fires may have had a thinning effect. Ponderosa pine forests in the Gila Wilderness and Saguaro Wilderness are structurally diverse and resistant to fires burning during the natural fire season, suggesting that repeated WFU fires have restored forest resilience to fire.

Keywords: ecological restoration, fire ecology, wildland fire use

Citation: Holden, Z.A., P. Morgan, M.G. Rollins, and K. Kavanagh. 2007. Effects of multiple fires on ponderosa pine on stand structure in two southwestern wilderness areas. Fire Ecology 3(2): 18-33.

INTRODUCTION

Southwestern ponderosa pine (*Pinus ponderosa)* ecosystems evolved with frequent, low-intensity fires that limited fuel accumulation and reduced densities of small trees (Cooper 1960, Swetnam and Dieterich 1985, Swetnam and Baisan 1996, Touchan *et al.* 1996). Historically, many southwestern ponderosa pine forests appeared much more open, with widely spaced trees and a lush, grassy understory. Intensive livestock grazing, fire suppression and other land uses have excluded all but the largest, most severe fires from ponderosa pine forests in the southwestern United States (Swetnam and Baisan 1996). Disruption of this natural burning pattern has led to significant ecological changes (Covington and Moore 1994*a*, Arno *et al.* 1995). Many stands now have higher densities of small and mid-sized trees, increased canopy closure, greater vertical fuel continuity and more surface fuels (Covington and Moore 1994*b*). One result of such changes is that recent fires are more severe than those that occurred in the past (Covington and Moore 1994*a*). In some places, both the growth rates of large trees and rates of nutrient cycling have declined (Covington *et al.* 1997). Habitat quality for key wildlife species has also declined (Long and Smith 2000).

Actions by the National Park Service in 1968 and by the Forest Service in 1972 began a program that has become known as the "wildland fire use for resource benefit" program (WFU), whereby naturally ignited fires are managed but allowed to burn under prescribed conditions (van Wagtendonk 2007). WFU fires must meet specific fuel and weather conditions such that fire is not likely to reach private property, threaten human life, or spread beyond a pre-designated maximum manageable area (Zimmerman and Bunnell 1998). Goals of the WFU program include restoring fire as a natural process and mitigating hazardous fire conditions that might have resulted from fire exclusion. These conditions have restricted the use of WFU programs mainly to large wilderness areas (Parsons 2000), although policy allows WFU programs wherever an approved fire management plan with specific provisions for WFU is in place (Zimmerman and Bunnell 1998). Alternatively, management-ignited prescribed fires are increasingly used as a means of reducing surface fuel loading and alter stand structure characteristics in ponderosa pine forests. Structural restoration, especially removal of small understory trees, is an important component of ponderosa pine forest restoration prescriptions (Covington *et al.* 1997). Using prescribed fire or WFU fires to restore ponderosa pine forest structure is a challenge because these fires must be hot enough to kill small trees without damaging large overstory trees (Lynch 1959, Swezy and Agee 1991). Currently being debated is the extent to which fire alone can be used to restore ponderosa pine forests and the level of mechanical removal of small understory trees that might be necessary before fire can be safely and effectively reintroduced. This issue is further complicated in the 17 million hectares of roadless and wilderness areas in the western United States, where limited access and legal restrictions make mechanical thinning treatments impractical.

Where fire use is appropriate, multiple fires may be necessary to structurally restore ponderosa pine forests. Although small ponderosa pine trees are more susceptible to mortal fire injury than large trees (Harrington and Sackett 1990), a single fire may not sufficiently reduce surface fuels and stand density to increase resiliency to future fires. Fire hazard often remains high because tree injury and mortality after an initial fire add fuel to the understory (Sackett *et al.* 1996; Harrington 1981, 1982). Observations from prescribed fires in fire-excluded forests suggest that when the accumulated fuel around the base of large trees burns, the resulting long-duration heating can result in mortality of trees that would otherwise be resistant to fire. Heavy fuel accumulation can also injure or kill large trees by damaging fine roots near the soil surface (Swezy and Agee 1991). Damage to overstory and understory trees by an initial entry fire may result in heavy post-fire fuel loading, increasing the intensity of a subsequent fire. Recent studies have evaluated the effects of multiple fires on stand structure in eastern hardwood forests (Signell *et al.* 2005) and ponderosa pine-Douglas-fir forests in Montana (Keeling *et al.* 2006). However, the effects of multiple fires on tree mortality in southwestern ponderosa pine forests are not well studied (but see Sackett *et al.* 1996).

The Gila Wilderness (GW) in western New Mexico and the Saguaro Wilderness area (SW) in southern Arizona contain large areas of ponderosa pine forests managed under wildland fire use programs (Webb and Henderson 1985, Gunzel 1974). Digital fire atlases within geographic information systems document the spatial extent and year of all known fires. In both areas, fires have burned repeatedly in many places during the 20th century (Rollins *et al.* 2001).

The rich history of well-documented fires and extensive spatial data available on the size and timing of fires provide a natural experiment with which to evaluate the effects of single and multiple fires on stand structure characteristics of ponderosa pine forests. In both the GW and SW, occurrence of mid-century fires prior to implementation of the WFU program allows us also to evaluate the effects of prior fire exposure on subsequent fire effects and resulting stand structure.

We evaluated the effects of 30 years of repeated wildland fires on the tree density in the GW and SW areas. In this comparative case study, we contrast two very different and geographically isolated wilderness areas, where the effects of wildland fires can be evaluated without the confounding effects of logging. As described by Fulé *et al.* (2003), we sampled across large areas in order to draw inference about the effects of a few fires. This broad-scale approach and the unique history of these sites necessarily limit the scope of inference to our study areas. However, similarities and differences observed at each site inform us about the extent to which these patterns can be generalized to other areas. A variety of data sources, including high-resolution historical aerial photographs and post-fire Landsat TM imagery, were available for the GW but not for the SW. Thus, our analysis for the GW was done in greater depth.

METHODS

Study Areas

Gila Wilderness Area. The 230,800-ha Gila Wilderness lies 70 km north of Silver City in west-central New Mexico (Figure 1), and ranges in elevation from 1,300 m to 3,300 m. The wilderness encompasses the Gila River and its headwaters, the Mogollon Mountains, and the Black Range. Volcanic events in the late Cretaceous period formed the parent material of the GW. Broad, relatively flat mesa tops characterize the northern portion of the complex, while the southern region is rugged. Climate patterns in the southwestern U.S. vary from annual to decadal time scales, with annual precipitation varying significantly by elevation (Beschta 1976, Sheppard *et al.* 2002). Precipitation is primarily bimodal, falling as snow in the winter or as rain in the summer during monsoon storms characteristic of the region. Thunderstorms are common in the summer months, resulting from rapid lifting of moist air from the Gulf of Mexico. These storms are localized, and often produce lightning that can ignite fires. The fire season in the GW typically begins in April, peaking in July at the height of the monsoon storm season. The mean fire interval for ponderosa pine forests at this site is 4 to 8 years, with a range of 1 to 26 years (Swetnam 1983). Mixed-severity fires occurred less frequently at higher elevations (Abolt 1996). Pinõn-oak-juniper woodlands occur at lower elevations of the wilderness and cover 23 % of the study area (2001). At middle elevations, extensive stands of ponderosa pine (approximately 21 % of the study area) cover mesa tops. Upper elevations support mixed Douglas-fir (*Pseudotsuga menzesii*), southwestern white pine (*Pinus strobiformis*), Engelmann spruce (*Picea engelmannii*), subalpine fir (*Abies lasiocarpa*), and aspen (*Populus tremuloides*) forests. The GW was grazed extensively through the 1950s but it has never been logged.

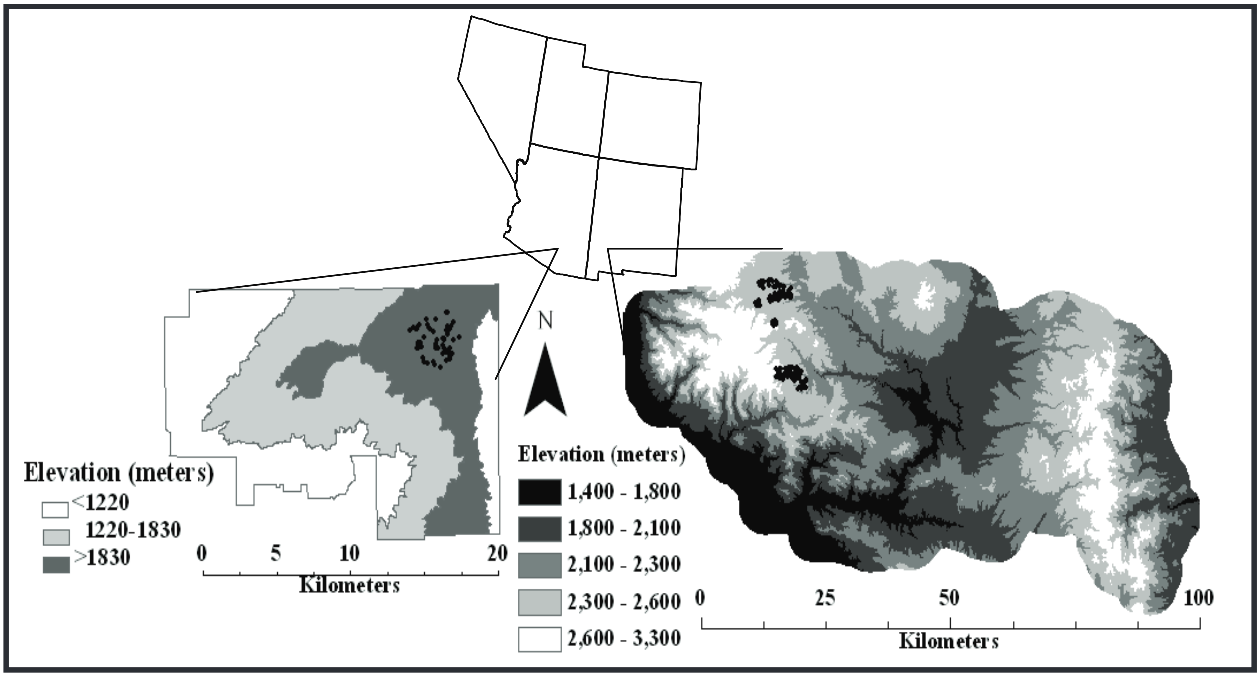

Figure 1. Study areas in the 29,500-ha Saguaro Wilderness, Arizona (left), and the 230,800-ha Aldo Leopold Wilderness Complex, New Mexico (right). Sample plot locations are indicated by black dots.

Saguaro Wilderness Area. The 29,500-ha Saguaro Wilderness area of Saguaro National Park in the Rincon Mountains (Figure 1) is an uplifted dome of granitic gneiss that rises abruptly from the floor of the Sonoran desert (Baisan 1983). Sometimes called "sky islands," the Rincons and several adjacent ranges were uplifted during the mid-tertiary period and are now classified as metamorphic core complexes. Elevations range from 870 m to 2,600 m at the peak of Mica Mountain. Climate and weather patterns are generally similar to those described for the GW, with a bimodal precipitation pattern that varies significantly by elevation (Sheppard *et al.* 2002). Orographic lifting of moist air results in frequent summer thunderstorms. These storms bring lightning, and with it, fire. The fire season in the Rincons typically begins in April, peaking in July at the height of the monsoon storm season. The mean fire return interval at this site is 6.1 years, with a range of 1 to 13 years for large fires (Baisan and Swetnam 1990). The area's steep elevation gradient and diverse conditions give rise to a remarkably rich flora. Giant saguaro cacti *(Carnegiea gigantea)* and many species of desert trees and shrubs characterize the Sonoran desert vegetation at the base of the mountain. These convert to oak woodlands and then pinyon-juniper-oak woodlands above 1,520 m elevation. We focus this study on the upper elevations of the Rincons dominated by stands of ponderosa pine on south-facing slopes and mixed stands of ponderosa pine and Douglas-fir on north-facing slopes. The SW was grazed but never extensively logged (Baisan 1983).

FIELD METHODS

Sampling Design

Field data on forest stand characteristics in the SW and the GW were gathered in 2003 and 2004. Sample plots were selected based on a stratified random sample design, with each study area stratified into four different fire treatments: unburned since at least 1909, once burned (1x), burned two or more times from 1972-2003 (2x+), and burned in the mid-century (1946) and again during WFU (2x+preWFU) (Table 1). In the GW, we hereafter refer to this 2x+preWFU treatment area as Johnson Mesa (see Figure 2 for example photographs). The strata were based upon fire perimeter data from digitized fire atlas records maintained by local land managers in both study areas (Morgan *et al.* 2001, Rollins *et al.* 2001). In both wilderness areas, fire treatments were selected from multiple combinations of different fires to minimize problems associated with pseudoreplication (Hurlbert 1984, Oksanen 2001). However, because so little area is unburned within this period (1972-2003) in the GW, our control sampling was limited to one mesa and is thus unreplicated. All plots were randomly selected within each treatment and located a minimum of 50 m from trails. Sampling in the GW area was restricted to flat (0 % to 9 % slope) mesa tops in order to minimize the variability in stand structure associated with topography, wind, and slope effects on fire behavior and resulting post-fire effects, and to maximize the treatment effect being measured. Plot selection was random with respect to slope and aspect in the SW due to the area's rugged topography.

Table 1. Unique fire years and combinations of fires used to stratify plot selection in the Gila Wildnerness and Saguaro Wilderness areas.

Fire treatment	Fire year	# Plots
Gila Wilderness		
Unburned	NA	12
1x	1979	14
1x	1985	11
1x	1993	6
1x	1996	24
2x+	1979, 1985	11
2x+	1985, 1993	21
2x+	1979, 1985, 1993	26
2x+preWFU	1946, 1992, 1993	29
Saguaro Wilderness		
Unburned	NA	14
1x	1988	6
1x	1994	9
2x+	1988, 1997	4
2x+	1972, 1993	6
2x+preWFU	1943, 1954, 1986, 1988, 1993	11
2x+preWFU	1943, 1950, 1954, 1988, 1993	4
2x+preWFU	1943, 1954, 1988, 1993	10

Fire Atlas Accuracy Assessment

Fire atlases are rich but imperfect records of fire extent. In the Iron Creek Mesa area of the GW, some sample plots were located near the boundary of two intersecting fire perimeters. To check the accuracy of these two fire atlas perimeters we used to stratify our sampling, we `used Landsat TM images to map burned area perimeters of two fires (1985, 1993) sampled in the GW (Holden *et al.* 2005). Correspondence between the atlas-derived and imagery-derived fire perimeters was high, especially at the eastern and western edges that defined boundaries between fire treatments (1x, and 2x+ burned areas), confirming our sampling stratification for overlapping areas of the 1985 and 1993 fire perimeters. The relatively large number of smaller fires in the SW made a similar assessment of fire perimeter accuracy for that area impractical. However, recent work in the Saguaro Wilderness using randomly sampled fire-scarred trees to estimate 20th-century fire perimeters confirms that fires in that area, including older fires, were mapped with a high degree of accuracy in the digital fire atlas (Farris *et al.*, in press).

Figure 2. Unburned stands (upper right) and areas burned multiple times (upper left) in the Gila Wilderness. Unburned stands (lower right) and areas burned multiple (lower left) times in the Saguaro Wilderness.

Stand Structure Sampling

Stand structure data were collected within 11.3 m (400 m^2) fixed-radius plots following FIREMON fire effects monitoring protocol (Lutes *et al.* 2006). Tree species, height in m, and diameter at breast height in cm of each tree inside the plot was recorded. All trees larger than 5 cm dbh were measured within each plot. A total of 152 plots were sampled in the GW, and 40 plots were measured in the SW. Data for the SW were supplemented with an additional 24 fire effects monitoring plots using procedures specified by the National Park Service (http://www.nps.gov/fire/fire/fir_eco_mon_fmh.cfm) (NPS 2003). On these 24 plots, measurements included the dbh and height of all trees >15 cm dbh within 1,000 m^2 plots and all trees <15cm dbh within a 250 m^2 subplot.

Tree Ring Analysis

Tree age data for the GW and SW were collected within a randomly selected subset of sample plots in order to qualitatively assess general age structure patterns. General age patterns are described for both the SW and GW, but basal area increment (BAI) growth analysis was done only on tree cores from the GW. Two trees of each species present in a plot were systematically selected from each of four size classes: 5 cm to 20 cm, 20 cm to 40 cm, 40 cm to 60 cm, and 60+ cm. One increment core was taken from each tree at 30 cm above the ground. Increment cores were air dried, glued and mounted on wooden mounts and belt-sanded with successively finer grit paper (120 to 600), until the cellular structure of each core was visible. All cores were examined and aged using a 50x binocular microscope. Where the increment bore missed the pith, a pith locator was used to estimate the number of rings to tree center. Although we did not cross-date (Stokes and Smiley 1968), we acknowledge that cross-dating would improve identification of individual rings. However, our primary interest was in forest structure, thus the age data we report here are general and qualitative. In the GW 10 yr BAI growth data upon which we do perform quantitative analysis, only two rings (1951, 2002) were occasionally absent in our ring width series. False rings (where late wood begins to form in the spring and early summer due to moisture stress and then transition back to early wood) are easily identified by incomplete trachied closure. Because we were using 10 yr BAI averages and comparing relative differences between groups, missing rings would have had only a small influence on the results. The similarities in patterns of growth for the two groups of trees up until the 1920s give us confidence that our measurements were sufficiently accurate for our purposes.

Preliminary evaluation of age-to-diameter relationships suggested that trees from areas that burned mid-century grew more rapidly than trees from other areas. To determine whether these relative growth patterns were the result of the 1946 fire, periodic (BAI) growth rates were measured from ponderosa pine tree cores collected from the areas within the GW that burned in 1946 and areas that did not burn. For each treatment (burned 2x+ and burned 2x+ preWFU), 18 cores were randomly selected from a subset of 128 ponderosa pine trees >50 cm in diameter. Radial increment growth of each tree was measured to the nearest 0.01 mm on a measuring stand using the Measure J2X software program (Robinson and Evans 2004). Mean 10 yr BAI growth patterns were compared from 1840 (pre-fire exclusion) to the present. Analysis of variance was used to test for statistically significant differences among years between treatments with a significance level set at alpha = 0.05.

Data Analysis

For both the SW and GW, parameters associated with tree size class distributions, including mean stem dbh, mean density (trees ha^{-1}), basal area (m^2 ha^{-1}) and stand density index, were compared among fire treatments. We were particularly interested in comparing differences in mean density of trees from 5.0 cm to 22.5 cm dbh as trees of this size are likely to be affected by surface fire (Harrington 1993, Sackett *et al.* 1996) and trees 45.0 cm or larger (large diameter trees are important for wildlife species and of interest to managers) among fire treatments. Data were analyzed using two-way analysis of variance procedures and Tukey's test for honest significant differences. Diagnostic plots of the residuals against fitted values for tree density suggested a lack of homoskedasticity. Data were transformed on a logarithmic scale for testing.

For both the GW and SW, shape and scale parameters extracted from two-parameter Weibull functions were fit to tree density by dbh class data for each plot. The Weibull function is highly flexible and commonly used to model forest stand structure (Bailey 1973, Avery 1994). Two parameters, shape and scale, describe the spread and overall shape of the function (Avery 1994). These metrics describe the distribution of tree by size class. Parameters for each function were analyzed as bivariate response variable using multivariate analysis procedures with fire treatment as a predictor variable. The Hotelling-Lawley test was used to test for statistically significant differences in Weibull parameters between individual fire treatment levels. This method of analysis is appropriate for these data because it incorporates tree-level information while retaining the appropriate n-value, and thus the correct degrees of freedom in the statistical model. Multivariate procedures were appropriate because of the information provided by both parameters of the Weibull function.

RESULTS

Stand structure differed significantly among fire treatments. Mean tree size (dbh) was higher and tree density was lower in areas subject to multiple fires in the GW (Table 2, Figure 3). In the SW, mean tree size (dbh) did not differ among treatments, while tree density was significantly lower in the 2x+ and 2x+preWFU treatments (Table 3, Figure 3). In the GW, mean dbh was significantly higher and tree density was significantly lower in the 2x+preWFU (areas burned twice in WFU and also mid-20th century) than in 2x (areas burned twice in WFU but not burned mid-century) fire treatments ($p < 0.01$, Table 2). Density of small-diameter trees (0.0 cm to 22.5 cm) differed significantly between fire treatments ($p < 0.05$) but the density of large trees (>45 cm dbh) was not different except in the SW, where more large diameter trees were present in areas burned mid-century and again during the WFU era (Table 3). Stand density index and basal area did not differ significantly ($p > 0.05$) among treatments in either the GW or the SW. In both the GW and the SW, shape and scale parameters of the Weibull distribution did not differ significantly between unburned and once

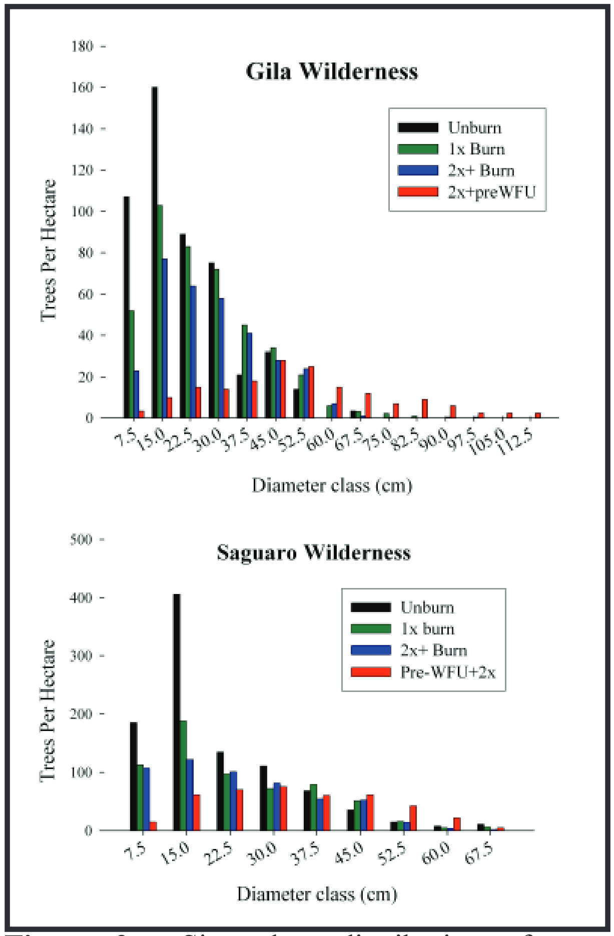

Figure 3. Size class distribution of trees compared for different burn treatments in the Gila and Saguaro Wilderness areas. Diamter at breast height class represents midpoint of size class.

Table 2. Gila Wilderness stand structure characteristics by fire treatment. Numbers in parentheses are 1 standard deviation. Small trees are 5 cm to 2.5 cm dbh and large trees are >45 cm dbh. Data followed by different letters within a row differ at $\alpha < 0.1$. Data followed by a different letter and a * indicate significant differences at $\alpha < 0.05$.

	Fire Treatment			
Structural characteristic	**Unburned n = 12**	**Burn 1x n = 55**	**Burn 2x+ n = 58**	**2x+preWFU n = 29**
Tree density (ha^{-1})	593 (377)a	478 (268)b	376 (191)b*	170 (75)c*
Mean dbh (cm)	23.2 (9)a	29.6 (10)b	31.1 (8.6)b	51.0 (16)c*
Basal area (m^2/ha)	27 (17)a	34 (11)a	29 (13)a	30 (25)a
Stand density index	490 (296)a	589 (51)a	495 (205)a	559 (328)a
Small tree density (ha^{-1})	308(198)a	185 (192)a	139 (165)b*	65 (62)b*
Large tree density (ha^{-1})	71 (69)a	113 (66)a	101 (70)a	114 (65)a
Weibull shape	5.2 (2.3)	7.7 (3.8)	7.4 (3.7)	15.0 (12.5)
Weibull scale	24.4 (9)	33.0 (11)	33.6 (9)	54.6 (16)
Shape/scale combined	a	b	b*	c*

Table 3. Saguaro Wilderness Trees/ha, trees/ha by size classes, SDI and BA. Small trees are 5 cm to 2.5 cm dbh and large trees are >45 cm dbh. Data followed by different letters differ at $p < 0.1$. Different letters followed by a * show significant differences at $p < 0.05$.

	Fire Treatment			
Structural characteristic	**Unburned n = 14**	**Burn 1x n = 15**	**Burn 2x+ n = 10**	**2x+preWFU n = 25**
Tree density (ha^{-1})	980 (377)a	631 (268)b*	540 (191)b*	420 (75)b*
Mean dbh (cm)	25.3 (8.8)a	26.5 (9.1)a	27.2 (5.5)a	26.6 (9.1)a
Basal area (m^2/ha)	38 (26)a	31 (16)a	26 (11)a	44 (25)a
Stand Density index	690 (296)a	574 (277)a	481 (190)a	713 (243)a
Small tree density (ha^{-1})	726 (530)a	399 (330)b*	331(127)b*	146 (211)b*
Large tree density (ha^{-1})	42 (51)a	29 (28)a	24 (16)a	82 (60)b*
Weibull shape	2.0 (0.4)	2.3 (0.7)	2.05 (2.1)	3.1 (3.1)
Weibull scale	26.7 (9.0)	28.7(10.2)	28.1 (3.4)	37.9 (8.9)
Shape/scale combined	a	a	b*	c*

burned areas ($p = 0.06$), but were significantly different for all other fire treatments ($p < 0.01$, Table 2 and Table 3).

Trees we sampled in the SW and GW varied in age from 40 yr to 320 yr, except on the unburned site where the oldest tree sampled was 240 yr old. Nearly 50 % of the trees aged were 140 yr to 160 yr old or 60 yr to 80 yr old, suggesting the presence of two major cohorts (established circa 1840 and circa 1920). In the GW, mean 10 yr BAI growth rates of trees from areas that did and did not burn mid-century are similar up until the early 19th century. While growth rates continue to follow the same relative pattern, they begin to diverge after 1920 and become significantly different after 1940 (Figure 4).

DISCUSSION

Ecological and Management Implications

Our results suggest that repeated WFU-era fires have reduced the density of small-diameter trees without significantly affecting the density of the largest trees. These results are consistent with mortality patterns observed in wildfire (McHugh 2003) and prescribed fire studies in ponderosa pine forests elsewhere in the southwestern US (Sackett *et al.* 1996). The WFU fires we studied burned during summer under weather and fuel moisture conditions that would likely have promoted variable and sometimes more intense fire behavior than experienced in most prescribed burns, but it appears that the largest trees have sufficient resistance to survive.

The Weibull parameters, which describe the distribution of trees by size class across treatments, reflect a significant shift toward larger tree size classes with increasingly frequent fires (Table 2, Table 3). However, tree-size class distributions in many stands in both the SW and GW are still skewed toward small to medium diameter (15 cm to 30 cm dbh) trees (Figure 3), even in sites burned three or more times in the 20th century. Such distributions likely reflect pulses of successful tree establishment during favorable environmental conditions (Savage *et al.* 1996) and variable thinning by subsequent fires. Even more highly skewed distributions than these have been observed in other ponderosa pine forests from which fires have largely been

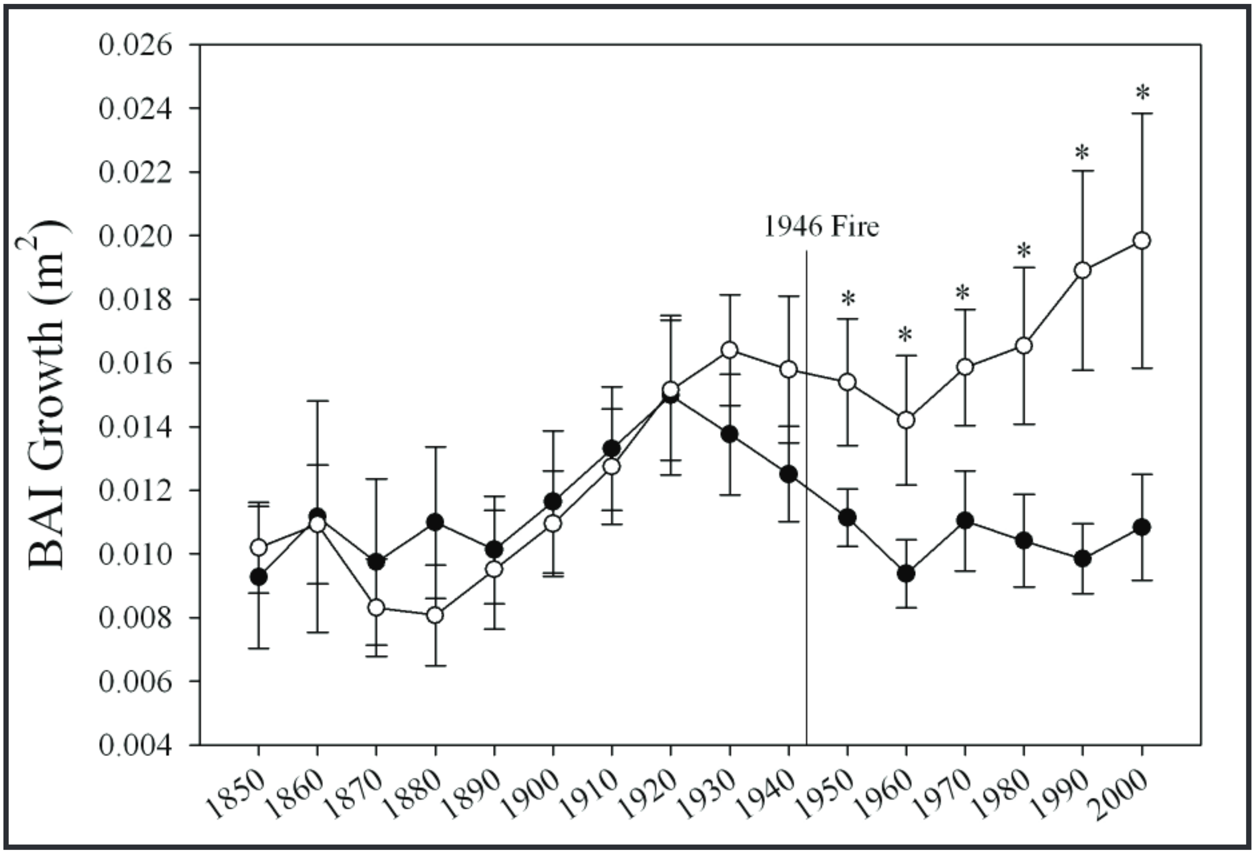

Figure 4. Basal area growth of 18 large (>50 cm dbh) ponderosa pine trees for stands in the Gila Wilderness burned in 1946 and then again during WFU (open symbols) contrasted with stands that burned only during the WFU era (filled symbols). Years represent midpoint of 10-year intervals. Error bars represent ±1 standard deviation. * indicates significant differences at $\alpha = 0.05$.

excluded, especially if large trees have been cut (White 1985, Oliver 2001, Moore *et al.* 2004).

This study retrospectively evaluates the effects of early (mid-century) fires on resulting stand structure and growth patterns in a southwestern ponderosa pine forest. Tree-size class distributions from areas in the GW and SW that burned mid-century differ greatly from other areas that did not burn during that time period. Furthermore, in the GW, very large diameter (>82.5 cm dbh) trees occurred more frequently in areas burned mid-century and were largely absent from areas that did not burn in mid-century. Biophysical characteristics at these sites and fires unrecorded in the fire atlas may have contributed to the difference in stand structure we observed. However, we attribute the structural condition of forests in this area of the GW to the effects of the mid-century (1946) fire that occurred there. A mid-century fire would have had two important effects on the structure of the forests in which we sampled. First, this fire would have killed some small-diameter trees that in subsequent fire-free years would have grown sufficiently large to be resistant to subsequent fires. Second, surviving trees grew faster on burned sites relative to sites without mid-century fires, possibly due to the reduction of competition for resources. It is not clear why growth rates of trees on sites with and without mid-century fires begin to diverge starting around 1920. Atlas data indicate that small fires in 1911 and 1913 occurred within seven km of where these cores were collected. It is possible that fires in these years may have in fact burned over this area, which could contribute to the apparent increase in growth rates after 1920. Indeed, growth patterns of trees measured on this site within the GW suggest this is the case (Figure 4). Increased basal area growth following mechanical thinning has been documented in ponderosa pine forests in Montana (Sala *et al.* 2005) and Arizona (Skov *et al.* 2005).

Stand age at the time of burning may have been critical in shaping the present structure of forests in the GW. Comparison of mean tree diameters and density on Johnson Mesa and Iron Creek Mesa, two areas that burned more than once but at different times in the last 100 years reveal divergent patterns in subsequent stand development in the GW. The 1946 fire on Johnson Mesa may have killed many of the small diameter trees likely from the 1920s cohort, aiding in the development of open stands dominated by large-diameter trees. In contrast, Iron Creek Mesa, which did not burn in 1946 but burned in 1979, 1985, and 1993, while relatively open compared to other forests in the southwestern US, has more than twice the average tree density than on Johnson Mesa (Table 2). It appears that even after three late-century wildland fires, the successional trajectories of stands on Iron Creek Mesa are set. Most of the surviving trees are now relatively large and fire-resistant. Surface fuels have been reduced by the three WFU fires, likely decreasing future fire intensity and the likelihood of extreme fire behavior (Table 2). Thus, continued burning, even under summer wildfire conditions, will serve to thin recently recruited small understory trees but is less likely to significantly affect the mid-sized and larger trees that established during the long fire-free interval before 1979. As such, achieving open stand structure characteristics like those of reference sites (Stephens and Fulé 2005) and those observed on Johnson mesa sites may take many years.

Many upper elevation ponderosa pine stands in the SW and GW are structurally diverse and resilient to burning under most fire weather conditions. Repeated fires continue to reduce density of small trees while not significantly reducing the density of large trees. Snags in these study areas persist after repeated surface fires (Holden *et al.* 2005) and large logs were present across the study sites (M.G. Rollins, Forest Service, unpublished data). Fire-caused canopy openings ranging in size from 0.25 ha to 20 ha have been observed across both study areas. Our preliminary comparisons of historical aerial photographs and more recent satellite imagery and fire atlas data suggest these openings occur when fires burned after a long period of fire exclusion and where high pre-fire tree densities were observed in the aerial photographs (Z.A. Holden, U. Idaho unpublished data). While crown fires are still possible in dense stands, subsequent fires resulted in fewer changes in the forest. Thus, most of the risks, in terms of mortality to medium- and large-diameter trees are associated with the first fire after long periods of fire exclusion. Subsequent fires, even when they occurred only twice in 30 years, did not significantly further change the size distribution of trees in these wilderness areas. The unique management history of these wilderness areas limits the extent to which we can draw inferences to other study areas. However, our results do suggest that the length of fire-free intervals may be important in determining the magnitude of ecological effects of fires. However, it is unclear to what extent this might be a structural rather than time interval threshold.

Limitations

We stratified our data according to fire frequency derived from fire atlas data; however, the accuracy of fire atlases are imperfect, especially in areas characterized by low-severity fires, because it is difficult to map fire perimeters when few of the larger trees are killed (Morgan *et al.* 2001). It is possible that fire perimeters, especially older fires, weren't accurately mapped, that some areas within a fire perimeter may not have burned, or that small fires occurred but were not detected or recorded (Morgan *et al.* 2001). These are simply the realities associated with using historical fire perimeter data as a tool for stratification and sampling. While we attempted to confirm our sampling using historical Landsat imagery, there is no "truth" to confirm that either data source is correct. It is also possible that because of small unrecorded fires or small unburned islands within historical fire perimeters, sample plots were located in areas that burned with a different fire history than that recorded in the fire atlas. It is also worth noting that fire behavior would have varied among sites and across years.

Our primary unburned (control) area in the GW burned in 2002, months before we planned to sample there, leaving us with only one area that had not burned in the 20th century. We acknowledge that the relatively small number of plots (n = 12) sampled across this unreplicated mesa may not represent the variety of stand conditions one might expect to see across a broader range of unburned conditions. Similarly, large unburned areas were lacking in the SW. The lack of unburned control areas is a testament to the effectiveness of the WFU program in restoring fire to the landscape. However, this lack limits the extent to which we can draw inferences about the level of tree mortality that occurs after initial wildland fire use fires in the GW.

Pre-fire conditions are lacking for both study areas. Thus, we can't rule out the possibility that stand structure differences exist because of long-term site differences (historical fire frequency, site productivity, etc.) or other factors such as fire behavior and burn severity differences leading to different patterns of tree establishment and mortality, rather than from the effects of the fires within which we measured.

Finally, we have tried to characterize stand structure across a broad region with a relatively small sample size, which also contributes to variability in these data. Therefore, we interpret these data with some caution.

CONCLUSIONS

Forest structure in the SW and GW reflects a combination of recent and historical fires, site conditions, and other factors. Stand structure in the areas we sampled were remarkably diverse. Even in areas that burned multiple times, small-diameter trees, snags, and logs were present. Recent fires have contributed to the resilience of many stands to significant fire impacts. In some areas, early- and mid-century fires that occurred when stands were relatively young likely had significant impact and resulted in open stands of larger-diameter trees. Comparisons between these areas and others that burned repeatedly during the late-century WFU era suggest that the timing of these fires relative to stand age or tree size may have been critical in determining successional trajectories of forest stand structure.

Stand structure in the GW and the SW also reflect a legacy of fire management and suppression. Repeated fires have apparently reduced densities of small diameter trees, but stands of dense sapling and pole-sized trees are still common even in areas burned multiple times during the WFU era. These data suggest that repeated fires, even those that occur during dry conditions, may not mortally injury medium and larger trees that have become fire resistant as a result of long fire-free intervals. It remains unclear how stand structure in the SW and GW will continue to develop as WFU natural fires continue to burn both areas. Important questions remain about the effects of continued, repeated burning on stand structure and of the physiological responses of individual trees.

ACKNOWLEDGEMENTS

This research was supported in part by funds provided by the Rocky Mountain Research Station, Forest Service, U.S. Department of Agriculture (Research Joint Venture Agreement 02-JV-111222048-252), the USDA/USDI Joint Fire Science Program (JFSP 01-1-1-06), the University of Idaho, and The Wilderness Society through the Gloria Barron Scholarship awarded to Z. Holden. We appreciate field assistance from A. Brumble, T. Morrison, A. Keyser, C. Koerner, A. Shlisky, C. McNicoll, and R. Lolley. The Gila National Forest staff and W. Hann provided invaluable logistical support. We appreciate the comments of C. Miller and two anonymous reviewers.

LITERATURE CITED

Abolt, R.A. 1996. Surface and crown fire histories of upper elevation forests via fire scar and stand age structure analyses. Thesis, University of Arizona, Tucson, USA.

Arno, S.F., M.G. Harrington, C.E. Fiedler, and C.E. Carlson. 1995. Restoring fire-dependent ponderosa pine forests in western Montana. Restoration and Management Notes 13(1): 32-36.

Avery, T.E., and H.E. Burkhart. 1994. Forest measurements. 4th edition. McGraw-Hill, New York, New York, USA.

Bailey, R.L., and T.R. Dell. 1973. Quantifying diameter distributions with the Weibull function. Forest Science 19: 97-104.

Baisan, C.H. 1983. Fire history of the Saguaro Wilderness, Saguaro National Monument. Senior thesis, University of Arizona, Tucson, USA.

Baisan, C.H., and T.W Swetnam. 1990. Fire history on a desert mountain range: Saguaro Wilderness, Arizona, USA. Canadian Journal of Forest Research 20: 1559-1569.

Beschta, R.L. 1976. Climatology of the ponderosa pine type in central Arizona. Arizona Agricultural Experiment Station Technical Bulletin, 228.

Cooper, C.F. 1960. Changes in vegetation, structure, and growth of southwestern pine forests since white settlement. Ecological Monographs 30(2): 129-164.

Covington, W.W., P.Z. Fulé, M.M. Moore, S.C. Hart, T.E. Kolb, J.N. Mast, S.S. Sackett, and M.R. Wagner. 1997. Restoring ecosystem health in ponderosa pine forests of the southwest. Journal of Forestry 95: 23-29.

Covington, W.W., and M.M. Moore. 1994*a*. Postsettlement changes in natural fire regimes and forest structure: ecological restoration of old-growth ponderosa pine forests. Pages 153-182 in: R.N. Sampson and D.L. Adam, editors. Assessing forest ecosystem health in the inland west. Hawthorn Press, Stroud, Gloucester, United Kingdom.

Covington, W.W., and M.M. Moore. 1994*b*. Southwestern ponderosa forest structure: changes since Euro-American settlement. Journal of Forestry 92 (1): 39-47.

Farris, C.A., C.H. Baisan, and T.W. Swetnam. In press. Spatial and temporal corroboration of fire-scar based fire history reconstructions in a frequently burned wilderness area in southern Arizona. Canadian Journal of Forest Research.

Fulé, P.Z., T.A. Heinlein, W.W. Covington, and M.M. Moore. 2003. Assessing fire regimes on Grand Canyon landscapes with fire scar and fire record data. International Journal of Wildland Fire 12: 29-145.

Gunzel, L. 1974. National policy change: natural prescribed fire. Fire Management 35(3): 4-8.

Harrington, M.G. 1981. Preliminary burning prescriptions for ponderosa pine fuel reductions in southeastern Arizona. U.S. Department of Agriculture Forest Service Research Paper RM-402.

Harrington, M.G. 1982. Stand, fuel, and potential fire behavior characteristics in an irregular southeastern Arizona ponderosa pine stand. U.S. Department of Agriculture Forest Service Research Note RM-418.

Harrington, M.G. 1993. Predicting *pinus ponderosa* mortality from dormant season and growing season fire injury. International Journal of Wildland Fire 3(2): 65-72.

Harrington, M.G., and S.S. Sackett. 1990. Using fire as a management tool in southwestern ponderosa pine. Pages 126-131 in: J.S. Krammes, editor. Proceedings of the symposium on effects of fire in management of southwestern natural resources. U.S. Department of Agriculture Forest Service General Technical Report RM-191.

Holden, Z.A., P. Morgan, M.G. Rollins, and G.R. Wright. 2005. Ponderosa pine snag densities following multiple fires in the Gila Wilderness, New Mexico. Forest Ecology and Management 221: 140-146.

Holden, Z.A., A.M.S. Smith, P. Morgan, M. Rollins, and P. Gessler. 2005. Evaluation of novel thermally enhanced spectral indices for mapping fire perimeters and comparison with fire atlas data. International Journal of Remote Sensing 26(21): 4801-4808.

Hurlbert, S.H. 1984. Pseudoreplication and the design of ecological field experiments. Ecological Monographs 54(2): 187-211.

Keeling, E.G., A. Sala, and T.H. DeLuca. 2006. Effects of fire exclusion on forest structure and composition in unlogged ponderosa pine/Douglas-fir forests. Forest Ecology and Management 237: 418-428.

Long, J.N., and F.W. Smith. 2000. Restructuring the forest: goshawks and the restoration of southwestern ponderosa pine. Journal of Forestry 53: 67-72.

Lutes, D.C., R.E. Keane, J.F. Caratti, C.H. Key, N.C. Benson, S. Sutherland, and L. Gangi. 2006. FIREMON: Fire effects monitoring and inventory system. U.S. Department of Agriculture Forest Service General Technical Report RMRS-GTR-164CD.

Lynch, D.W. 1959. Effects of a wildfire on mortality and growth of young ponderosa pine trees. U.S. Department of Agriculture Forest Service Research Note INT-66.

McHugh, C.W., and T.E. Kolb. 2003. Ponderosa pine mortality following fire in northern Arizona. International Journal of Wildland Fire 12: 7-22.

Moore, M.M., D.W. Huffman, P.Z. Fulé, W.W. Covington, and J.E. Crouse. 2004. Comparison of historical and contemporary forest structure and composition on permanent plots in southwestern ponderosa pine forests. Forest Science 50(2): 162-176.

Morgan, P., C.C. Hardy, T.W Swetnam, M.G. Rollins, and D.G. Long. 2001. Mapping fire regimes across time and space: understanding coarse and fine-scale fire patterns. International Journal of Wildland Fire 10: 329-342.

National Park Service. 2003. Fire monitoring handbook. National Park Service National Interagency Fire Center, Boise, Idaho, USA.

Oksanen, L. 2001. Logic of experiments in ecology: is pseudoreplication a pseudoissue? OIKOS 94: 27-38.

Oliver, W.W. 2001. Can we create and sustain late successional attributes in interior ponderosa pine stands? Large-scale ecological research studies in northeastern California. Pages 99-103 in: R. Vance; C.B. Edminster; W.W. Covington, and J.A. Blake, compilers. Ponderosa pine ecosystems restoration and conservation: steps toward stewardship. U.S. Department of Agriculture Forest Service. Proceedings RMRS-P-22.

Parsons, D.J. 2000. Restoration of natural fire to United States wilderness areas. Personal, societal, and ecological values of wilderness. Pages 42-46 in: Sixth World Wilderness Congress proceedings on research, management, and allocation, volume II. U.S. Department of Agriculture Forest Service Proceedings RMRS-P-14.

Robinson, W.J., and R. Evans. 2004. A microcomputer-based tree-ring measuring system. Tree-Ring Bulletin 4059-64.

Rollins, M.G., T.W. Swetnam, and P. Morgan. 2001. Evaluating a century of fire patterns in two Rocky Mountain wilderness areas using digital fire atlases. Canadian Journal of Forest Research 31: 2107-2123.

Sackett, S.S., S.M. Haase, and M.G. Harrington. 1996. Lessons learned from fire use for restoring southwestern ponderosa pine ecosystems. In Conference on adaptive ecosystem restoration and management: restoration of cordilleran coniferous landscapes of North America. U.S. Department of Agriculture Forest Service General Technical Report RM-GTR-278.

Sala, A., G.D. Peters, L.R. McIntyre, and M.G. Harrington. 2005. Physiological responses of ponderosa pine in western Montana to thinning, prescribed fire and burning season. Tree Physiology 25: 339-348.

Savage, M., P.M. Brown, and J. Feddema. 1996. The role of climate in a pine forest regeneration pulse in the southwestern United States. Ecoscience 3(3): 310-318.

Sheppard, P.R., A.C. Comrie, G.D. Packin, K. Angersbach, and M.K. Hughes. 2002. The climate of the U.S. southwest. Climate Research 21: 219-238.

Signell, S.A., M.D. Abrams, J.C. Hovis, and S.W. Henry. 2005. Impact of multiple fires on stand structure and tree regeneration in central Appalachian oak forests. Forest Ecology and Management 218: 146-158.

Skov, K.R., T.E. Kolb, and K.F. Wallin. 2005. Differences in radial growth response to restoration thinning and burning treatments between young and old ponderosa pine in Arizona. Western Journal of Applied Forestry 20(1): 36-43.

Stephens, S.L., and P.Z. Fulé. 2005. Western pine forests with continuing frequent fire regimes: possible reference sites for management. Journal of Forestry 103(7): 357-362.

Stokes, M.A., and T.L. Smiley. 1968. An introduction to tree-ring dating. University of Chicago Press, Illinois, USA.

Swetnam, T.W. 1983. Fire history of the Gila wilderness. Thesis, University of Arizona, Tucson, USA.

Swetnam, T.W, and C.H. Baisan. 1996. Historical fire regime patterns in the southwestern United States since AD 1700. Pages 1-32 in: C.D. Allen, editor. Proceedings of the second La Mesa Fire symposium, Los Alamos, New Mexico. U.S. Department of Agriculture Forest Service General Technical Report RM-GTR-286.

Swetnam, T.W, and J.H. Dieterich. 1985. Fire history of ponderosa pine forests in the Gila Wilderness, New Mexico. Pages 390-397 in: J. Lotan, B.M. Kilgore, W.C. Fischer, and R.W Mutch, technical coordinators. Proceedings of a symposium and workshop on wilderness fire. U.S. Department of Agriculture Forest Service General Technical Report INT-182.

Swezy, M.D., and J.K. Agee. 1991. Prescribed-fire effects on fine-root and tree mortality in old-growth ponderosa pine. Canadian Journal of Forest Research 21: 626-634.

Touchan, R., C.D. Allen, and T.W Swetnam. 1996. Fire history and climatic patterns in ponderosa pine and mixed-conifer forests of the Jemez Mountains, northern New Mexico. U.S. Department of Agriculture Forest Service General Technical Report RM-GTR-278:33-46.

van Wagtendonk, J.W. 2007. The history and evolution of wildland fire use. Fire Ecology 3(2): 3-17.

Webb, D.R., and R.L. Henderson. 1985. Gila Wilderness prescribed natural fire program. Pages 413-414 in: J.E. Lotan, B.M. Kilgore, W.C. Fischer and R.W. Mutch, technical coordinators. Proceedings of a symposium and workshop on wilderness fire. U.S. Department of Agriculture Forest Service General Technical Report. INT-182.

White, Alan S. 1985. Presettlement regeneration patterns in a southwestern ponderosa pine stand. Ecology 66(2):589-594.

Zimmerman, G.T., and D.L. Bunnell. 1998. Wildland and prescribed fire management policy, implementation procedures reference guide. National Interagency Fire Center: Boise, Idaho, USA.