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

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Keywords: fire ecology, ecological restoration, wildland fire use

Abstract

The effects of 30 years (1974-2003) 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 Rincon Mountain Wilderness (RMW), Arizona. Tree density (trees per ha), diameter-class distributions, basal area and stand density index were compared among areas that burned 0, 1 and 2 or more times, and areas that burned mid-century (1940-1950) and again during the WFU era (1974-2003). In both the Rincon Mountain Wilderness and the Gila Wilderness, significantly fewer small-diameter (0-22.5 cm) trees occurred in areas that burned multiple times compared to areas that were unburned ($p < 0.05$). The density of large-diameter (45-90+ 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 Rincon

Mountain Wilderness, significantly more large-diameter trees occurred in areas that burned mid-century and again during WFU than in all other fire treatments. Mean 10-year basal area growth rates (1840-2000) were similar for both areas prior to a 1946 fire, but growth rates of mid-century burned trees increased significantly in the early 1900's ($p<0.05$). Thus, tree density and diameter growth reflect the thinning effect of that mid-century fire. Ponderosa pine forests in the Gila Wilderness and RMW are structurally diverse and resilient to fires burning during the natural fire season, suggesting that repeated WFU fires have restored forest resilience to fire.

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 in the late 19th century, fire suppression and other land uses have excluded all but the largest, most severe fires from ponderosa pine forests in the southwestern United States (Savage 1991, Swetnam and Baisan 1996). Disruption of this natural disturbance pattern has led to significant ecological changes (Covington and Moore 1994b, 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 1994b). Both the growth rates of large trees and nutrient cycling have declined (Covington et al. 1997). Habitat quality for key wildlife species has also declined (Long and Smith 2000).

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In 1968, the National Park Service began actively managing naturally ignited fires in Kings Canyon National Park, California. This represented one of the first formal acknowledgements of the negative impacts of fire exclusion (Parsons 2000). The USDA Forest Service followed suit in 1972, managing several lightning-ignited fires in the Selway-Bitterroot Wilderness area in Montana and Idaho. These actions 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. 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 disturbance process and mitigating hazardous fire conditions that might have resulted from fire exclusion (Rollins et al. 2001). 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).

The need for ecological restoration of ponderosa pine is now widely acknowledged by researchers and managers (Weaver 1951, Covington 2000, Allen et al. 2002). A major goal of ecological restoration of ponderosa pine ecosystems is restoring fire as a natural ecological process (Society for Ecological Restoration 1996). Fire is a keystone process in ponderosa pine forests (Morgan 1994) and is an essential component of restoration (Swetnam and Baisan 1996). However, changes resulting from fire exclusion include increased fuels and altered structural characteristics of ponderosa pine forests so that recent fires are more severe than those that occurred in the past (Covington and Moore 1994a). Current land use practices, including residential development along margins of fire-prone forests have increased wildfire threats to people and property. Reintroduction of fire after decades of exclusion, even in weather conditions that limit fire intensity, may be detrimental to forest health, for example, by damaging large, important trees or by burning more severely than was thought to occur historically (Covington et al. 1997, Allen et al. 2002).

Structural restoration, especially removal of small understory trees, is an important component of ponderosa pine forest restoration prescriptions (Covington et al. 1997). Fire alone may not kill small trees, and reducing the density of small trees by fire may not be possible without also damaging large overstory trees (Lynch 1959, Swezy and Agee 1991).

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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 will 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. A single fire may not sufficiently reduce surface fuels and stand density to increase resiliency to future fires, and fire hazard often remains high because tree injury and mortality after an initial fire add fuel to the understory (Harrington 1981, 1982, Sackett et al. 1996). 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). Small ponderosa pine trees are more susceptible to mortal fire injury than large trees (Harrington and Sackett 1990). Prescribed burning in ponderosa pine forests is often done in the early spring or fall when temperatures are cooler and fuel moistures are higher than in wildfires, and fire intensity is controlled in order to minimize extreme fire behavior. In contrast, wildland fires that escape initial attack typically occur under hot, dry and windy weather with dry fuels. Fire effects under these conditions, especially after years of fire exclusion, will likely be more severe than prescribed fires that are ignited under milder conditions. Observations from prescribed fire 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 large 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.

The 230,800-ha Gila Wilderness and the 29,500-ha Rincon Mountain Wilderness area (RMW) in Arizona contain large areas of ponderosa pine forests managed under wildland fire use programs (Webb and Henderson 1985). 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 and more than 10,000 ha of ponderosa pine forests in the Gila Wilderness have burned 2 or more times in 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 provides a natural experiment with which to evaluate the effects of a single and multiple fires on stand structure characteristics of ponderosa pine forests. In both the Gila Wilderness and Rincon Mountain Wilderness, occurrence of mid-century fires prior to implementation of the WFU program allows us to evaluate the effects of prior fire exposure on subsequent fire effects and resulting stand structure.

The goal of this research project is to evaluate the effects of 30 years of repeated wildland fires on the tree density and stand structure of upper-elevation ponderosa pine forests in the Gila Wilderness and Rincon Mountain Wilderness areas. As described by Fulé et al. (2003), we are sampling 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 limits 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. In this comparative

case study, we contrast two very different and geographically isolated wilderness areas. A variety of data sources, including high-resolution historical aerial photographs, and post-fire Landsat TM imagery were available for the Gila Wilderness but not for the RMW. Thus, our analysis for the Gila Wilderness was done in greater depth.

Methods

Study Areas

Gila Wilderness

The 230,800-ha Gila Wilderness lies 70 km north of Silver City, in west-central New Mexico. Together with the Aldo Leopold Wilderness Area, it forms the Gila-Aldo Leopold Wilderness Complex (GALWC) (Figure 1). Encompassing the Gila River and its headwaters, the Mogollon Mountains and the Black range, elevations in the Gila range from 1300 to 3300 m. Volcanic events in the late Cretaceous period formed the parent material of the Gila Wilderness. Broad, relatively flat mesa tops characterize the northern portion of the complex, while the southern region is rugged. Climate patterns in the southwestern US 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 and as rain in the summer 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. Historically, low-severity (low tree mortality) surface fires occurred every 3-10 years in ponderosa pine forests of the Gila Wilderness (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 (Rollins et al. 2001). At middle elevations, extensive stands of ponderosa pine (approximately 21% of the study area) cover mesa tops above the Gila River. 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 Gila Wilderness was grazed extensively through the 1950's but it has never been logged.

Rincon Mountain Wilderness Area

The 29,700-ha Rincon Mountain Wilderness area of Saguaro National Park (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 Rincon Mountains and several adjacent ranges were uplifted during the mid-tertiary period and are now classified as metamorphic core complexes. Elevations of the Rincon Mountains range from 870 to 2600 m at the peak of Mica Mountain. Climate and weather patterns are generally similar to those described for the Gila Wilderness, 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 Rincon Mountains typically begins in April, peaking in July at the height of the monsoon storm season. The mean fire interval at this site is 6.1 years, with a range of 1-13 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 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 1520 m elevation. We focus this study on the upper elevations of the Rincon Mountains dominated by stands of ponderosa pine on south-facing slopes and mixed stands of ponderosa pine and Douglas-fir on north-facing slopes. The RMW was grazed but never extensively logged (Baisan 1983).

Field Methods

Sampling Design

Field data on forest stand characteristics in the RMW and the Gila Wilderness 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 multiple times from 1974-2003 (2x), and burned in the mid-century (1946) and again during WFU (2x+preWFU). In the Gila wilderness, we hereafter refer to this area as Johnson Mesa (See figure 4 for example photographs). These were stratified based upon fire perimeter data from digitized fire atlas records maintained by local land managers in both study areas. (Rollins et al. 2001, Morgan et al. 2001). In both wilderness areas, fire treatments (once, twice, and thrice burned areas) were selected from multiple combinations of different fires to avoid problems associated with pseudoreplication (Hurlbert 1984, Oksanen 2001). However, because so little area is unburned within this period (1974-2003) in the Gila Wilderness, 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 Gila Wilderness area was restricted to flat (0-9% slope) mesa tops in order to minimize the variability in stand structure associated with topographic variation, 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 RMW due to the area's rugged and highly variable topography.

Fire Atlas Accuracy Assessment

Fire atlases are rich but imperfect records of fire extent. In the Iron Creek Mesa area of the Gila Wilderness, sample plots were sometimes located near the boundary of two intersecting fire perimeters. To check the accuracy of fire atlas perimeters we used to stratify our sampling, we used Landsat TM images to map burned area perimeters of two fires sampled in the Gila Wilderness (Holden et al. 2005b). 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 (1, 2 and 3 times burned areas), generally confirming our sampling stratification for overlapping areas of the 1985 and 1993 fire perimeters. The relatively large number of smaller fires in the RMW made a similar assessment of fire perimeter accuracy for that area impractical. However, recent work by Farris et al. (unpublished data) in the Rincon Mountain 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.

Stand Structure Sampling

Stand structure data were collected within 11.3-m (400 m^2) fixed-radius plots following FIREMON fire effects monitoring protocol (http://frames.nbii.gov). Tree species, height in m, diameter at breast height in cm, crown ratio, and height in m to live crown 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 Gila Wilderness and 40 plots were measured in the RMW. Data for the RMW were supplemented with 24 National Park Service fire effects monitoring plots (http://www.nps.gov/fire/fire/fir_eco_mon_fmh.cfm) for 64 plots total in RMW (NPS 2003). On these plots, measurements include the DBH and height of all trees >15 cm DBH within 1000-m^2 plots and all trees <15cm DBH within a 250-m^2 subplot.

Forest Age Structure Sampling

Tree age data were collected within a randomly selected subset of sample plots in order to determine forest age structure across treatments. Two trees of each species present in a plot were systematically selected from each of four size classes: 5-20 cm, 20-40 cm, 40-60 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 (120-600), 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. 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 Gila tree ring series upon which we do perform quantitative analysis, only two rings were occasionally absent in our ring width series (1951, 2002) and false rings are easily identified under a microscope. 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 (Figure 3) up until the 1920's

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 (data not shown). To determine whether these relative growth patterns were the result of the 1946 fire, periodic basal area increment (BAI) growth rates were measured from tree cores collected from the areas within the Gila Wilderness that burned in 1946 and areas that did not. For each treatment (burned twice, and burned twice plus pre-WFU), 18 cores were randomly selected from a subset of trees greater than 20 inches 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.

Data Analysis

For both the RMW and Gila Wilderness, parameters associated with tree size class distributions, including mean stem DBH, mean density (trees/ha), basal area (m^2/ha) and stand density index, were compared between fire treatments. We compared differences in mean density of trees 0-22.5 cm DBH as trees of this size are likely to be affected by surface fire (Harrington 1993, Sackett et al. 1996) and trees 47.5 cm or larger (large diameter trees are important for wildlife species and of interest to managers) between 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.

Due to concerns that plot-level averaging of stand structure metrics might lead to a loss of information, we employed a second method of data analysis. For both the Gila Wilderness and RMW, shape and scale parameters extracted from two-parameter Weibull functions were fit to tree density by DBH class data for each plot. Parameters for each function were analyzed as a 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. 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). 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 significant correlation between parameters of the Weibull function.

Results

Tree Density and Stand Structure

Stand structure differed significantly between fire treatments. Mean tree size (DBH) was higher and tree density was lower in areas subject to multiple fires in the both the Gila Wilderness (Table 1, Figure 2) and the RMW (Table 2, Figure 2). 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 in 1946) fire treatments ($p < 0.01$, Tables 1 and 2). Density of small-diameter trees (0-

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 RMW, where more large diameter trees were present in areas burned mid-century and again during the WFU era (table 2). Stand density index and basal area did not differ significantly (p values >0.05) among treatments in either the Gila Wilderness or the RMW. In both the Gila Wilderness and the RMW, shape and scale parameters of the Weibull distribution did not differ significantly between unburned and once burned areas ($p = 0.06$), but were significantly different for all other fire treatments ($p<0.01$, Table 1). These metrics, which describe the distribution of tree by size class across treatments shift toward larger tree size classes with increasingly frequent fires.

Trees in stands in all treatments varied in age from 40 to 320 yr, except on the unburned site where the oldest tree sampled was 240 yr old. Many of the trees aged were 140-160 yr old or 60-80 yr old, suggesting the presence of two major cohorts. On the mid-century burned Johnson Mesa, most of the aged trees were 150-200 yr old with a few much older trees (300+ yr old). Comparison of mean 10-yr basal area increment of trees from areas that did and did not burn mid-century before burning during the WFU era suggest that differences in stand structure between these areas reflects the thinning effects of that earlier fire. Growth rates of trees from both areas 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 3). Small fires in 1911 and 1913 occurred within 7 km of where these cores were collected. It is possible that these fires may have in fact burned over this area, which would explain the apparent increase in growth rates after 1920.

Discussion

Our results suggest that repeated WFU-era fires have reduced the density of small-diameter trees without significantly affecting the density of larger trees. These results are consistent with mortality patterns observed in wildfire (McHugh 2003) and prescribed fire mortality studies in ponderosa pine forests elsewhere in the southwestern US (Sackett et al. 1996), even though the WFU fires burned during summer and with variable and sometimes more intense fire behavior than experienced in most prescribed burns. WFU fires were often much larger than typical prescribed burns, often burning thousands of hectares over weeks and months resulting in highly variable fire behavior and effects.

Tree size-class distributions in many stands in both the RMW and Gila Wilderness are skewed toward small-to-medium diameter (15-30 cm DBH) trees (Figure 2), 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 subsequent variable thinning by fires. Even more highly skewed distributions than these have been observed in other ponderosa pine forests from which fires have largely been excluded, especially if large trees have been cut (White 1985, Oliver 2001, Moore et al. 2004).

Tree size-class distributions from areas in the Gila Wilderness and RMW that burned mid-century differ greatly from other areas that did not burn during that time period. In the Gila Wilderness, large diameter (82.5-112.5 cm DBH) trees occurred more frequently in areas burned mid-century and largely absent from areas that did not burn in mid-century. Biophysical characteristics at these sites, mid-century climate, or long-term history, such as 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 to the mid-century (1946) fires that occurred there. Indeed, growth patterns of trees measured on this site compared to other areas suggest this is the case (Figure 3). Mid-century fires probably killed some small-diameter trees that in subsequent fire-free years would have grown sufficiently large to be resistant to subsequent fires. Residual trees grew faster on burned sites relative to sites without mid-century fires. 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). This study retrospectively evaluates the effects of early (mid-century) fires on resulting stand structure and growth patterns in a southwestern ponderosa pine forest.

Stand age at the time of burning may have been critical in shaping the present structure of forests in the Gila Wilderness. 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 one hundred years reveal divergent patterns in subsequent stand development in the Gila Wilderness. The 1946 fire in Johnson Mesa may have occurred early enough to kill many of the small diameter trees likely from the 1920's cohort, aiding in the development of open stands dominated by large-diameter trees. In contrast, Iron Creek Mesa, which burned in 1979 and again in 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 1). It appears that even after three wildland fires, the successional trajectories of stands on Iron Creek Mesa are set. Most of the remaining trees are now relatively large and fire-resistant. Surface fuels have been reduced by previous fires, likely decreasing future fire intensity and the likelihood of extreme fire behavior. Thus, continued burning, even under

summer wildfire conditions may thin recently recruited small understory trees but is less likely to significantly affect mid-sized and larger trees that established during a period of fire exclusion. As such, achieving open stand structure characteristics like those of reference sites (Stephens and Fule 2005) and those observed on Johnson mesa sites may take many years.

Limitations

This study has several important limitations. First, 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. Second, our primary unburned (control) area in the Gila Wilderness 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 RMW. The lack

of unburned control areas is a testament to the effectiveness of the WFU program in restoring fire to the landscape. However, it 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 Gila Wilderness. Third, 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.

Ecological and Management Implications

Many upper elevation ponderosa pine stands in the RMW and Gila Wilderness 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. 2005a) and large logs were present across the study sites (Rollins, unpublished data). Fire-caused canopy openings ranging in size from 0.25-20 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 (Holden et al., 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.

Conclusions

Forest structure in the RMW and Gila Wilderness reflects a combination of recent and historical fires. Stand structure in the areas we sampled were remarkably diverse, even in areas that burned multiple times, with small-diameter trees, snags and logs present. Recent fires have contributed to the resilience of many stands to signicant fire impacts. In some areas, early to mid-century fires, when stands were relatively young, likely had significant impact resulting in open stands of larger-diameter trees. Comparisons between these and areas that burned repeatedly during the 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.

While resilience to burning (i.e. restoration of surface fire regimes) is an important component of ponderosa pine forest restoration, stand structure in the Gila Wilderness and the RMW reflect a legacy of fire exclusion. 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 last thirty years. These data suggest that repeated fires, even under traditional wildfire conditions may not mortally injury medium and larger trees that have become fire resistant as a result of years of fire exclusion. It

remains unclear how stand structure in the RMW and Gila Wilderness 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.

Acknowledgments

This research was supported in part by funds provided by the Rocky Mountain Research Station, Forest Service, US 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 Amie-June Brumble, Tyler Morrison, Alisa Keyser, Cassie Koerner, Ayn Shlisky, Ceci McNicoll, Reese Lolley. The Gila National Forest staff and Wendel Hann provided invaluable logistical support. We appreciate the comments of the two anonymous reviewers.

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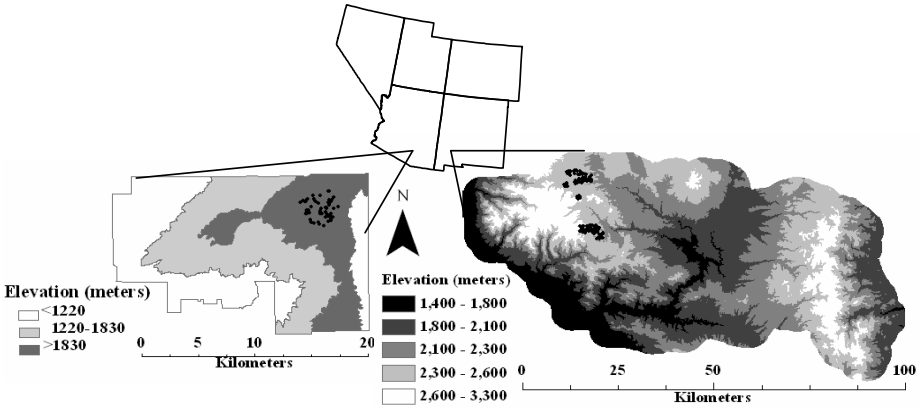
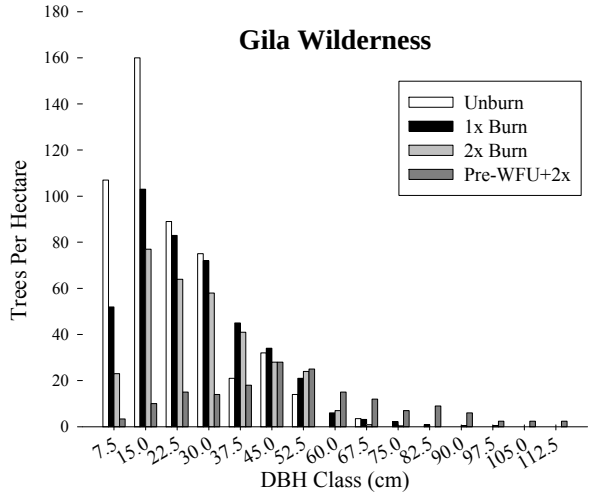


Figure 1. Study areas in the 29,500-ha Rincon Mountain Wilderness, AZ (left) and the 230,800-ha Aldo Leopold Wilderness Complex, NM (right). Sample plot locations are indicated by black dots.



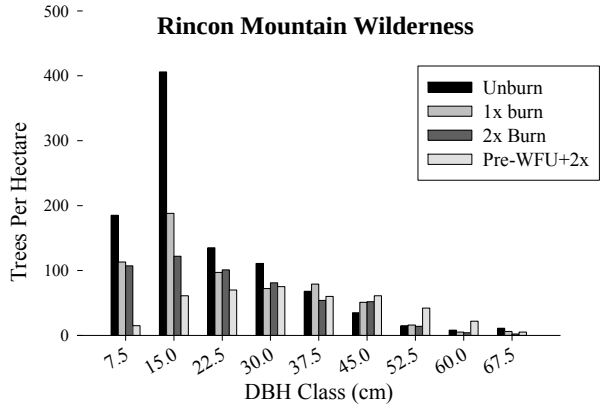


Figure 2. Size class distribution of trees compared for different burn treatments in the Gila and Rincon Mountain Wilderness areas.

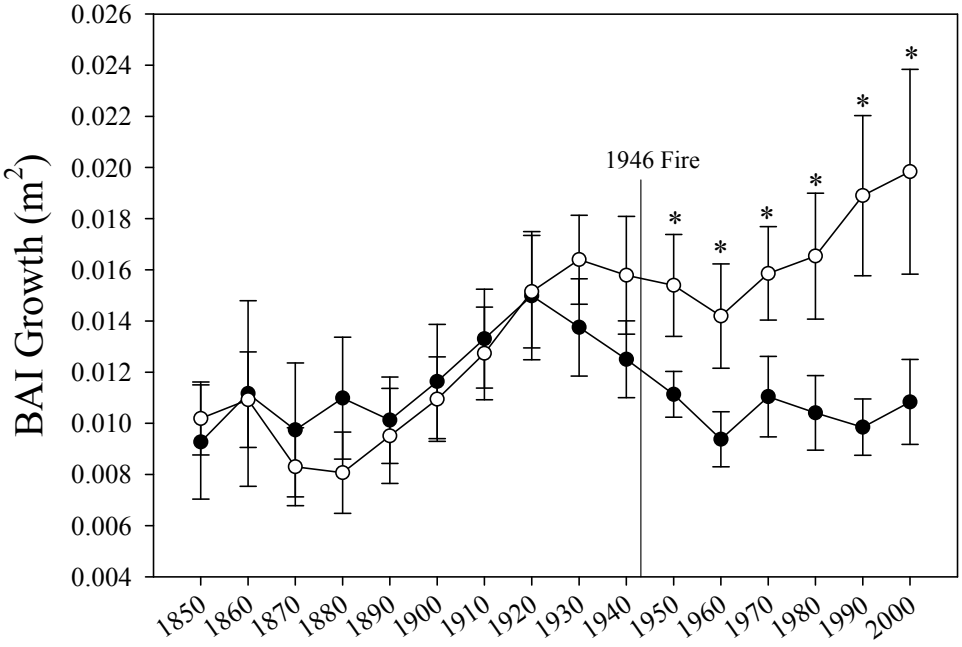


Figure 3. Basal area growth of trees for stands burned in 1946 and then again during WFU (open symbols) contrasted with stands that burned only during the WFU era (filled symbols). Years represent midpoint of 10-year intervals. Error bars represent ± 1 standard deviation. * indicates significant differences at α= 0.05.

Figure 4. 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 Rincon Mountain Wilderness.

Table 1. Gila Wilderness tree density and size (trees per hectare), mean stem diameter, basal area (BA) and stand density index (SDI) with TPH by size class, and Weibull shape and scale parameters for each fire treatment. Numbers in parentheses are 1 standard deviation. Data followed by different letters within a row differ at $\alpha < 0.1$ Different letters followed by a different letter and a * indicate significant differences at $\alpha < 0.05$.

Fire Occurrence	Unburned n=12	Burn 1x n=55	Burn 2+3x n=58	Pre-WFU +2x n=29
Trees/ha	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*
BA (m^2/ha)	27 (17)a	34 (11)a	29 (13)a	30 (25)a
SDI	490 (296)a	589 (51)a	495 (205)a	559 (328)a
TPH (5-22.5 cm)	308(198)a	185 (192)a	139 (165)b*	65 (62)b*
TPH (22.5-45 cm)	125 (93)a	180 (192)a	90 (66)a	125 (141)a
TPH (>45 cm)	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 2. Rincon Mountain Wilderness Trees/ha, trees/ha by size classes, SDI and BA. Data followed by different letters differ at $p<0.1$. Different letters followed by a * show significant differences at $p< 0.05$.

Fire Frequency	Unburned n = 14	Burn 1x n = 15	Burn 2+3x n = 10	Pre-WFU +2x n = 25
Trees/ha (SD)	980 (377)a	631 (268)b*	540 (191)b*	420 (75)b*
DBH (cm)	25.3 (8.8)a	26.5 (9.1)a	27.2 (5.5)a	26.6 (9.1)a
BA (SD)	38 (26)a	31 (16)a	26 (11)a	44 (25)a
SDI (SD)	690 (296)a	574 (277)a	481 (190)a	713 (243)a
5-22.5 cm TPH	726 (530)a	399 (330)b*	331(127)b*	146 (211)b*
>45 cm TPH	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*