CHAPTER 1

EVALUATION OF NOVEL THERMALLY ENHANCED SPECTRAL INDICES FOR MAPPING FIRE PERIMETERS AND COMPARISONS WITH FIRE ATLAS DATA

Abstract

We evaluated the potential of two novel Landsat Thematic Mapper (TM)-derived spectral indices for both discriminating burned from unburned areas and for producing fire perimeter data (as a potential surrogate to digital fire atlas data) within two historical wildland fires (1985 and 1993) in ponderosa pine (*Pinus ponderosa*) forests of the Gila Wilderness, New Mexico, USA. Index-derived images and manually-selected burned area polygons derived from Landsat TM imagery were compared to fire perimeters recorded within a digitized fire atlas. For each fire, the highest spectral separability was achieved using the newly proposed Normalized Burn Ratio-Thermal ($NBRT_1$) index (M=1.18, 1.76, for the two fires respectively). Correspondence between fire atlas and manually-produced perimeters was high. However, automated imagery classification approaches using even highly separable indices performed poorly. Landsat imagery may be a useful supplement to existing historical fire perimeter mapping methods, but the timing of the post-fire image will influence the separability of burned and unburned areas.

Keywords: spectral separablity, M-statistic, VI6T, NBRT, NDVI, Landsat

Introduction

Wildfires are a critical disturbance agent in many parts of the world (Morgan et al. 2001). The occurrence of wildfires has important management and research implications spanning a wide range of scientific disciplines. Numerous studies have sought to determine the extent of burned area (Pereira 1999; Smith et al. 2002) and the long-term effects of fires on ecosystem health (Morgan et al. 2001). The broad extent and remote nature of many fires makes remotely sensed imagery an obvious tool for fire science and management (Morgan and others 2001). Satellite sensor imagery has been used to map areas burned in a diverse range of vegetation types, including shrublands (Pereira 1999), chaparral (Minnich 1983), boreal forests (Fraser et al. 2001), and savannas (Smith et al. 2002). Recent burned area mapping studies have used spectral indices that employ a two-band combination of near infrared (NIR) with short-wave infrared (SWIR) or thermal-infrared (TIR) bands. These 'two-dimensional' indices have been demonstrated in several environments to provide greater discrimination between burned and non-burned areas compared to two-dimensional indices that use only visible and NIR bands (Chuvieco and Congalton 1988; Trigg and Flasse 2001).

The cost of Landsat imagery has generally consigned use of this data type to validation (Eva and Lambin 1998, Pereira 1999), where researchers typically use only Landsat visible to SWIR bands in supervised classifications, and generally exclude the TIR bands. Landsat imagery is also used widely by natural resource managers in the United States to produce fire perimeters databases or 'fire atlases', which are frequently used to assess fire hazard and risk and departure of vegetation from historical conditions. Fire atlases are typically constructed weeks to years after fire events using personal accounts, maps of the area burned, aerial photographs and in recent years, satellite sensor imagery (Minnich 1983; Morgan et al. 2001). Fire atlases do not normally include information on the internal variations within the burned area but instead provide land managers with the location and overall extent (i.e. the overall perimeter) of the area burned (Morgan et al. 2001).

The objective of this letter is to provide an initial evaluation of two novel spectral indices, which incorporate changes in the NIR, SWIR and TIR bands, for discriminating between burned and unburned areas using spectral separability statistics. This letter assesses both a new variant of an existing two-dimensional index and evaluates several variations of a novel ‘three-dimensional’ index that uses all of the NIR, SWIR, and TIR Landsat bands. A secondary objective is to assess the potential of archived Landsat imagery to derive fire perimeters as surrogates for historical fire perimeter data for two surface fires in a ponderosa pine forest of the southwestern United States.

Methods

Study Area

The Iron Creek Mesa area is located within the 230,800 ha Gila Wilderness, and is part of the Gila-Aldo Leopold Wilderness Complex in west-central New Mexico (Figure 1). Elevations in the Gila range from 1300 to 3300 m, and this study focuses on the mid-elevation ponderosa pine forests that dominate Iron Creek Mesa and surrounding drainages. We selected two fires for analysis, using historical fire atlases of the region. The 28,000 ha Gilita Fire burned across Iron Creek Mesa from June 1993 to 10 September 1993, and the

18,000 ha Iron Fire burned across the northern part of the Gila Wilderness from July, 1985 until 15 August 1985. These large fires killed few of the dominant large ponderosa pine and thus represent a challenge for mapping perimeters from space.

Imagery Preparation

Landsat 5 TM scenes corresponding to the first available cloudless post-fire image dates were selected for both fires (Iron Fire TM image acquired 22 October 1985; Gilita Fire TM image acquired 26 September 1993). We converted the reflective and thermal bands of each Landsat TM scene into top-of-atmosphere reflectance and brightness temperature respectively. Using both the fire perimeter maps as a guide and visual interpretation of the Landsat imagery, we randomly selected 300 pixels We classified selected pixels as either burned or unburned and subsequently used them for separability analyses and image classification. Pixels from burned and unburned areas for each index-derived image were assessed for and generally met assumptions of normality (Figure 3). Following Pereira (1999), the *M*-Statistic was then used to assess the utility of each Landsat band and spectral index listed in Table 1 to discriminate between burned and unburned areas. The optimal index was identified as the index with the highest consistent *M*-statistic over both fires. Using this optimal index, the TM images were then classified as burned if they fell within the mean ± 2 standard deviations of the burned index values.

Following previous studies that incorporated the thermal bands of satellite sensors (e.g. Eva and Lambin 1998), we assessed the potential of incorporating the Landsat TIR band into the existing Normalized Burn Ratio (NBR) index. NBR and several other fire indices rely on the principle that burning an area results in a lowering of the NIR reflectance with a corresponding increase in both the mid-infrared reflectance (Chuvieco and Congalton 1988) and brightness temperature (Eva and Lambin 1998). Therefore, we sought to enhance the expected post-fire changes in the NBR by incorporating the Landsat thermal band. These indices are hereafter referred to as $NBRT_i$ (Table 1). Furthermore, a modified version of the VI3T spectral index (Barbosa et al. 1999), herein referred to as the VI6T index (Smith et al. submitted), was applied by replacing the Advanced Very High Resolution Radiometer (AVHRR) band 3 values with the Landsat thermal band (band 6).

Accuracy Assessment

Historical fire atlas data are generally coarse-scale data that show the spatial extent of a fire. As we were primarily interested in comparing Landsat-derived fire perimeters with a coincident fire atlas, we did not attempt to assess the unburned patches within the overall fire perimeter. Therefore, we calculated the accuracy of the index-derived technique by measuring the degree of omission and commission along the fire perimeter compared to that measured by both hand-digitizing the Landsat imagery and using fire atlas data.

Results

The M-statistic values for each Landsat band and spectral index for both fires are displayed in Table 2. For the 1985 fire, the $NBRT_1$ index was most spectrally separable. M-statistic values of all the indices for the 1993 fire were notably higher than for the 1985 fire (Table 2).

The correspondence between the fire atlas and manually-digitized burned area perimeters were high for both the 1985 and 1993 fires (84% and 89% overlap, respectively, Figure 3). In contrast to the time consuming manual digitizing method, the faster index-based approach more often misclassified large non-burned areas (Figure 3). If we for a moment assume that the hand-digitized perimeter as the true fire perimeter, the fire atlas data had errors of commission and omission of 12.4% and 26.8%, respectively, for the 1985 fire and 20.4% and 16.2% for the 1993 fire (Table 3).

Discussion

Given the limited dataset used in this analysis these results may not generalize to other areas, as two fires from the same environment (albeit different fire seasons) will not capture variations in topographic, fire, and environmental characteristics. However, our results indicate that timing of the post-fire image acquisition affected the ability of the spectral indices to identify area burned. Indeed, the separability of the NDVI and NBR indices were greatly increased in the post-fire image acquired only three weeks after the 1993 fire, compared to the image acquired more than two months following the 1985 fire.

The ability of spectral techniques to discriminate between burned and unburned areas likely depends on the vegetation type, the fire severity (i.e. the vegetation mortality), the speed of vegetation recovery, and the temporal availability of the imagery. In this region of the southwestern United States, the decrease in spectral separability with acquisition time after the fire could be due to the onset of the monsoon rains, which often signal the end of the fire season. The resulting new vegetation growth and recovery that follows the rain can be rapid and would quickly act to reduce brightness temperatures. The ability to detect burned forested areas might be improved in fires that are largely ‘stand-replacing’ and where vegetation recovery is slow. As the rise in surface brightness temperature is likely due to the reduction in evapotranspiration, reduced surface albedo and increased soil cover (i.e. due to vegetation removal) (Eva and Lambin, 1998), the brightness temperature could potentially be considered a measure of the vegetation recovery. As such, the NBRT indices could provide pertinent information on the burn severity (i.e. post-fire ecological effects) beyond that provided by the non-thermal NBR index.

The correspondence between the burned areas derived from Landsat and mapped in the fire atlas was high. Clearly, considering either fire atlases or imagery-derived perimeters as “truth” is questionable. The digitized Gila Wilderness fire atlas used in this study was produced retrospectively using 12-year old-field maps and educated guesses on where the perimeter was likely to be based on geographic features and local expert knowledge (Rollins, pers. comm.). Errors in fire atlas mapping are particularly likely for older fires that occurred prior to the widespread application of GPS technology in resource and fire management. Despite such limitations, several land management agencies in the United States have begun developing fire atlases from satellite sensor imagery and field maps as part of fire management efforts. As yet, no standardized protocol has been developed for building digital fire perimeter layers and the quality and accuracy of potential data sources are highly variable.

Conclusions

The apparent errors present within the fire atlas perimeter data indicate the need for alternative methods to refine the accuracy of current fire perimeter maps. The proposed new $NBRT_1$ index series shows promise for identifying burned areas within environments where fire-induced vegetation mortality is low. Future research should evaluate how timing of post-fire image acquisition influences discrimination between burned and unburned areas using these and similar two or three-dimensional indices. The indices demonstrated herein should also be tested across a range of vegetation types, fire regimes, ranges of fire size, and geographic areas to assess their utility and accuracy. Although the degree of NBRT commission errors observed in this study suggests that Landsat data should be manually digitized to produce historical fire perimeters at the quality needed by natural resource managers, the temporal dependence of the surface temperature to parameters such as the vegetation recovery and fractional pixel cover warrant further assessment.

Acknowledgements

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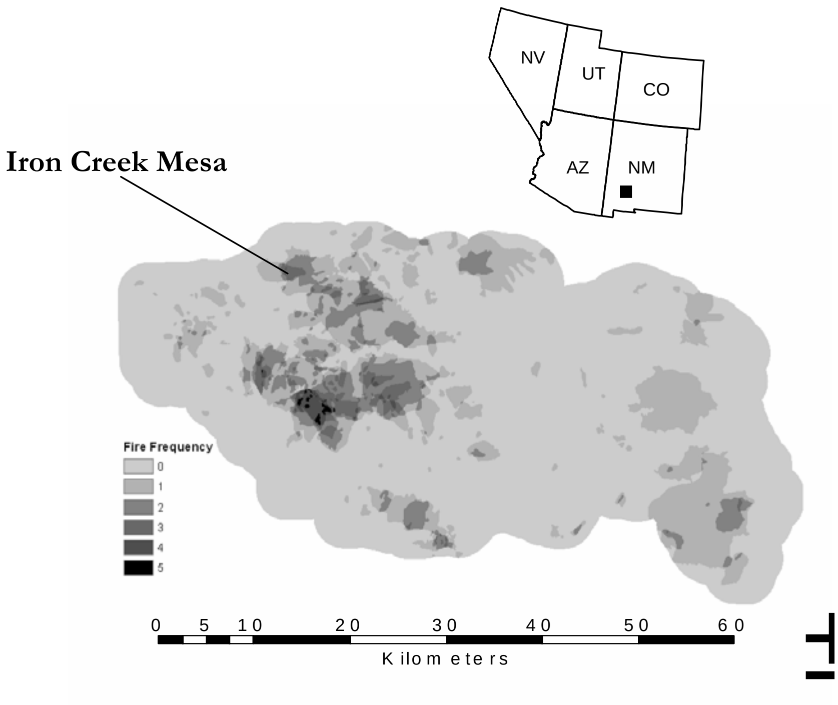

Figure 1. Study area within the Gila Aldo Leopold Wilderness fire atlas

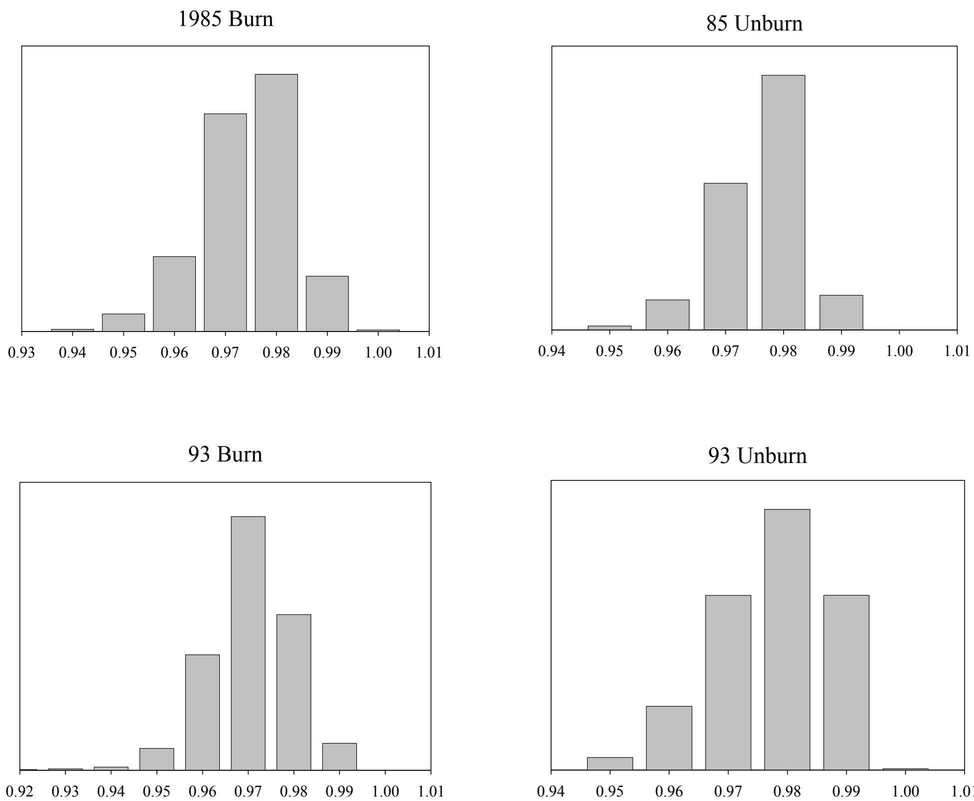

Figure 2. Histograms of NBRT index image region of interest data for assessment of normality.

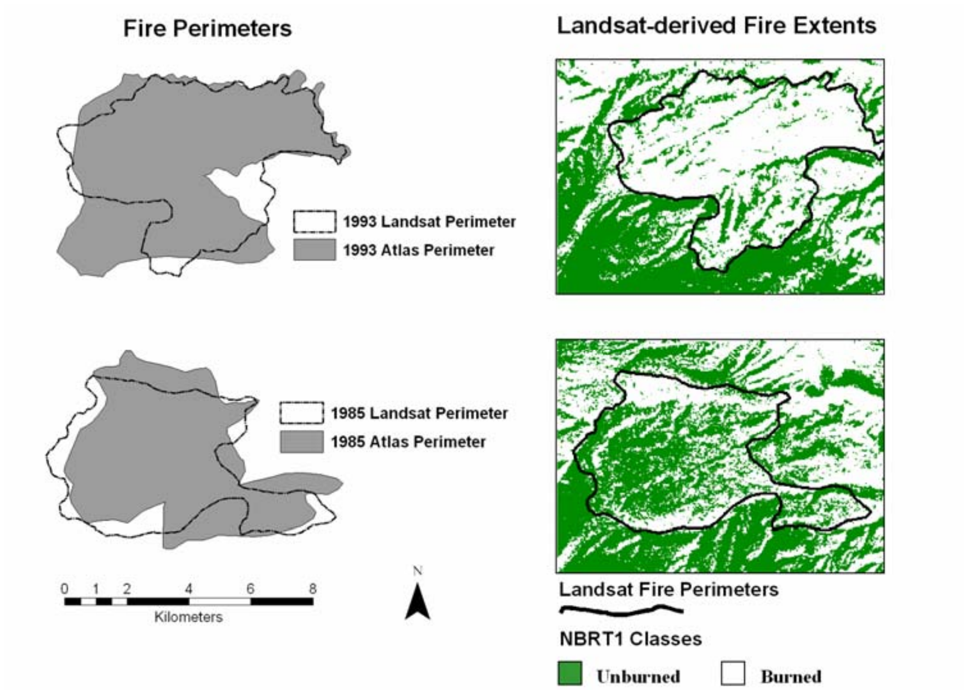

Figure 3. Comparison of 1985 and 1993 fire perimeters derived from fire atlas data and hand-digitized from Landsat TM imagery.

Table 1 Spectral Indices Applied to Determine the Area Burned

Method	Equation(s)	Reference(s)
NDVI[a]	$\left(\dfrac{\rho_2 - \rho_1}{\rho_2 + \rho_1}\right)$	Pereira (1999) Smith *et al.* (2002)
VI6(T)[b]	$\dfrac{(\rho_2 - SB_{TM6})}{(\rho_2 + SB_{TM6})}$	Smith et al (Submitted)
NBR	$\left(\dfrac{\rho_4 - \rho_7}{\rho_4 + \rho_7}\right)$	Chuvieco and Congalton (1988) Key and Benson (2002)
$NBRT_1$	$\left(\dfrac{(\rho_4 - (\rho_7 * SB_{TM6}))}{(\rho_4 + (\rho_7 * SB_{TM6}))}\right)$	
$NBRT_2$	$\left(\dfrac{\left(\dfrac{\rho_4}{SB_{TM6}}\right) - \rho_7}{\left(\dfrac{\rho_4}{SB_{TM6}}\right) + \rho_7}\right)$	
$NBRT_3$	$\left(\dfrac{(\rho_4 - SB_{TM6}) - \rho_7}{(\rho_4 - SB_{TM6}) + \rho_7}\right)$	

[a] ρ denotes the top-of-atmosphere reflectance of band X, where X is given by the Landsat sensor.
[b] SB_{TM6} denotes scaled brightness temperature of the Landsat band 6 thermal band. In this study the band 6 brightness temperature was divided by 10000.

Table 2. Spectral separability of Landsat TM-derived indices for mapping burned and unburned areas.

TM band or Spectral Index	Iron Fire(1985)	Gilita Fire (1993)
Visible – SWIR Bands		
TM1	0.02	0.41
TM2	0.04	0.13
TM3	0.07	0.37
TM4	0.43	0.78
TM5	0.12	0.67
TM6 (1985 only)	0.71	0.99
TM7	0.28	1.07
NDVI	0.75	1.09
NBR	1.08	1.66
Inclusion of Thermal Bands		
VI6T	0.65	1.10
$NBRT_1$	1.18	1.76
$NBRT_2$	0.86	1.57
$NBRT_3$	1.05	1.59

Table 3. Accuracy of fire atlas perimeter data compared to image-derived fire perimeters

	Iron Fire (1985)	Gilita Fire (1993)
Overlap (%)	84.0	89.0
Commission	12.4	20.4
Omission	26.8	16.2

CHAPTER 2

PONDEROSA PINE SNAG DENSITIES FOLLOWING MULTIPLE FIRES IN THE GILA WILDERNESS, NEW MEXICO

Abstract

Fires create and consume snags (standing dead trees), an important structural and ecological component of ponderosa pine forests. The effects of repeated fires on snag densities in ponderosa pine forests of the Southwest have not been studied. Line intercept sampling was used to estimate snag densities in areas of the Gila Wilderness that had burned 1, 2 and 3 times under Wildland Fire Use for Resource Benefit (WFURB), a fire management policy implemented since 1974 and designed to restore natural fire regimes. Seventeen randomly located transects were measured in areas burned since 1974, six in once-burned areas, six in twice burned areas and five in thrice-burned areas Three additional transects were measured in an area that burned in 1946 and twice more since 1974. The mean density $\pm$ standard errors of large (>47.5 cm dbh) snags for areas that burned once, twice and thrice was 7.0 $\pm$ 2.7, 4.4 $\pm$ 1.1 and 4.8 $\pm$ 1.3 snags/ha, respectively. Differences in snag densities between once and multiple burned areas were significant (ANOVA, n=6, $p<0.05$). There was no significant difference in density of large snags between twice and thrice burned areas ($p = 0.51$). We observed significantly lower snag densities on Johnson Mesa (2.8 snags/ha, $p < .05$) which burned in 1946 and then again in 1992 and 2003. Proportions of type 1 snags (recently created) were higher in once and twice burned areas than in areas that burned 3 times, likely reflecting high tree mortality and snag recruitment resulting from an initial entry fire. Type 3 snags (charred by previous fire) were more abundant in areas that burned multiple times. The lack of differences in snag densities between areas that burned two and

three times suggests that repeated fires leave many snags standing. The increasing number of type 3 snags with repeated fires supports this conclusion. These results have important implications for ecological restoration and management of fire in southwestern ponderosa pine forests.

Introduction

Ponderosa pine (*Pinus ponderosa var. Laws.*) is one of the most broadly distributed tree species in North America, extending from Mexico to southern Canada. As a keystone disturbance agent, fire has played a critical role in shaping the structure and function of ponderosa pine forests (Cooper 1960; Covington and Moore 1994a). Historically, frequent surface fires carried by grasses and fine fuels maintained open stands of mixed size and age that were “park-like” in appearance, with an understory characterized by dense, tall grasses and forbs (Cooper 1960; White 1985). Grazing, road development and an intense campaign of fire suppression all contributed to the exclusion of fire from many ponderosa pine forests in the western United States. After long periods without fire, many ponderosa pine forests have undergone significant changes, including increased surface fuel loading and higher densities of trees than were thought to have occurred historically (Covington and Moore 1994b; Savage 1991). These changes are particularly well documented in the southwestern United States (Friederici 2003; Mast and others 1999; Moore and others 1999; Moore and others 2004).The need for ecological restoration of ponderosa pine forests is now widely acknowledged by researchers and managers (Allen and others 2002; Covington and Moore 1994a; Weaver 1951). A major goal of ecological restoration of ponderosa pine ecosystems is restoring fire as a natural ecological process (Society for Ecological Restoration 1996).

In 1974, The Gila National Forest, NM formally implemented one of the first Wildland Fire Use for Resource Benefit (WFURB) programs in the nation. Wildland Fire Use fires are those naturally ignited fires which are managed to continue burning to accomplish resource benefits as long as no critical resources are threatened, adequate personnel and other resources are available for suppression and an approved fire management plan is in place

(USFS 1996). The goal of the WFURB program is to restore natural fire regimes, and ultimately to allow fires to burn with a minimum of human intervention. The initiation of WFURB is a milestone in fire management, representing a formal acknowledgement of the natural role played by fire in ecosystems of the southwestern United States and elsewhere. More than 7000 ha in the Gila-Aldo Leopold Wilderness Complex have now burned three or more times since 1974, 82% of those areas in ponderosa pine and Douglas-fir potential vegetation types (Rollins 2000) (Figure 1). The Gila Wilderness has been touted as a model for restoration of natural fire regimes to forested ecosystems (Boucher and others 2000). The cumulative and landscape-scale effects of the last 30 years of WFURB fires have been little studied. We are conducting research to examine the effects of multiple fires on tree mortality and stand structure in ponderosa pine forests, and the degree to which WFURB fires have restored ecosystems. WFURB is destined to become the new paradigm in fire management, especially in remote areas (Government Accounting Office 2002). While one of the primary goals of WFURB is to restore fire as a natural disturbance process, such broad-scale fire management decisions will significantly impact wildlife and other resources. In the Gila Wilderness, these include the northern goshawk, Gila trout, Gila chub and numerous snag-dependent bird species (Monzingo pers. comm.). It is critical that we understand the effects of fires and repeated fire treatments on stand structure and snag dynamics in ponderosa pine forests. Given its rich, well documented history of frequent, landscape-scale fires that burn through a variety of environmental conditions, the Gila Wilderness is an ideal natural laboratory in which to study these effects.

Snag Management

Tree death and resulting snags and downed logs play a vital ecological role in forests, affecting tree population and community structure, resource dynamics and geomorphology (Franklin 1987; Maser 1984). Until the late 1970's, snags were considered unsightly fire and safety hazards and were often removed; many are still removed during fire suppression activities. However, snags have gained recognition as an important forest structural component, providing important habitat for small mammals, amphibians, reptiles, bats and insects (Bull 1997). Snags also provide nest substrates, roosts, perches, sites for vocalization, food storage and feeding for a variety of bird species (Balda 1975; Cunningham and Balda 1980; Thomas 1979). Throughout North America, 85 species of birds use cavities in dead or decaying trees (Scott and others 1977). Cavity-nesting species make up 25-30% of the breeding bird species in many northern Rocky Mountain coniferous forests (Raphael 1987). In Arizona, Balda (1975) reported that approximately 50% of the breeding bird populations in ponderosa pine stands containing decayed wood are secondary cavity nesters. Insectivorous birds dependent on snags may play an important role in regulating insect populations that influence forest health (Cline et al., 1980). Primary cavity-nesting species in the southwestern United States include the hairy woodpecker (*Picoides villosus*), acorn woodpecker (*Melanerpes granivorus*) and northern flicker (*Colaptes auratus*). Secondary cavity nesters include the western bluebird (*Sialia sialia*), house wren (*Troglodytes aedon*) and violet green swallow (*Tachycineta thalassina*). When snags are removed as part of intensive forest management, it is very detrimental to cavity nesting birds (Scott 1979, Scott and Oldemeyer 1983), so snag density and snag management is a significant ecological concern.

Fire kills trees, creating new snags and removing existing snags. However, the effects of multiple fires on snag densities have never been measured. Balda (1975) established 4.2 snags/ha as the minimum snag density required to sustain cavity-nesting bird species in southwestern ponderosa pine forests. This number has since been adopted by many land management agencies in guidelines for management of snags (U.S. Department of Agriculture 1996). However, historical snag densities in ponderosa pine forests reported by Lang (1909) are very low (0.1 snags/ha) and recent work in the southwestern United States suggests that Balda's (1975) guidelines may be unrealistically high (Ganey 1999). Boucher et al. (2000) suggested that current snag densities in ponderosa pine forests in the southwestern US may be artificially high due to fire exclusion. The first fire after a long period of fire exclusion kills many trees through crown scorch and crown kill, or through cambial damage to stems and roots as that occurs when the heavy duff and litter that accumulated in the absence of fire finally burns, thereby recruiting large numbers of snags (Sackett and others 1996). However, snag densities are likely to decrease with repeated fires. Prescribed burning in ponderosa pine forests in Arizona resulted in a 50% decline in snag density (Horton 1988). Several researchers have observed that continuous understory vegetation and grasses grow near the base of snags and large trees (Boucher and others 2000; Fulé and Covington 1995). Understory vegetation and fine fuels may carry fire to the base of snags, which then burn and fall to the ground. According to this model, repeated fires would lead to low snag densities.

The objective of our study was to compare snag densities in ponderosa pine forests of the Gila Wilderness that had burned 1, 2 or 3 times since the WFURB program was implemented in 1974. We were unable to locate unburned areas large enough to permit snag density sampling and to serve as controls for this natural experiment. We defined snags as all dead ponderosa pine and Douglas-fir *(Pseudotsuga menzesii),* trees greater than 47.5 cm dbh and 5 m tall. Other studies have measured snags of all size classes. We measured only large-diameter snags because of their longer retention time and preferred use by cavity-nesting bird species (Bull 1983; Bull 1997). Based on previous research that measured the removal of snags by prescribed fire (Horton 1988), and observations by fire managers that snags frequently burn and fall during fire events, we hypothesized that snag densities would be significantly higher in areas that had burned only once and would continue to decrease with repeated fires.

Methods

Study Area

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 4). 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. Annual precipitation varies from 250 mm to 760 mm at higher elevations (Beschta 1976), falling as snow in the winter and as rain in the early-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, leading to favorable fire conditions. Historically, low-severity (low tree mortality) surface fire regimes dominated ponderosa pine forests of the Gila Wilderness (Swetnam 1983). Mixed severity fire regimes can be found at higher elevations (Abolt 1996). Pinõn-oak juniper woodlands occur at lower elevations of the wilderness and cover 23% of the GALWC (Rollins and others 2001). At middle elevations (2250-2550 m), extensive stands of ponderosa pine (approximately 21% of the GALWC) cover mesa tops above the Gila River. Upper elevations 2550-3300 m) support mixed Douglas-fir, southwestern white pine (*Pinus strobiformis*), Engelmann spruce (*Picea engelmannii*), subalpine fir (*Abies lasiocarpa*) and aspen (*Populus tremuloides*) forests.

Field Sampling and Analysis

Snag densities in the Gila Wilderness were estimated in May-August of 2004 using line transect sampling methods (Anderson 1979). We measured the diameter at breast height (dbh) and perpendicular distance to all ponderosa pine and Douglas-fir snags 47.5 cm dbh or greater along each transect. Twenty snag transects were randomly located on mesa tops in the locations described below across approximately 60,000 ha of the northern part of the Gila Wilderness (Figure 4). Transects were located in areas dominated by ponderosa pine, although Douglas-fir trees occur on more mesic aspects and occurred infrequently in all sample transects.

Six transects were located in areas that had burned once, six in areas that had burned twice, and five in areas that burned three times since 1974. Three additional transects were located on Johnson Mesa, an area that burned in the mid-twentieth century (1946) and then twice more during the WFU-era (1974-2003). We analyzed these three transects separately in order to further contrast the effects of different fire treatments (fire timing and frequency) and stand structure on snag densities. Sampling was stratified using historical fire perimeter records compiled in a digitized fire atlas. Fire atlases are valuable records of landscape fire frequency (Rollins and others 2001) but are prone to inaccuracy because they often exclude small fires and assume that the entire area within a perimeter burned (Morgan and others 2001). This may have affected the accuracy of our sampling and contributed to variation in our results. Several geographically separated mesas had burned only once while others burned two and three times from 1974 to the present. Transects were dispersed among these areas in an effort to encompass variation that likely occurred between different fire events and variation in stand structures across the landscape, as well as to broaden the inferences of our results.

Transect start points and azimuths were selected randomly within each fire frequency strata. A minimum of 40 snags were measured along each transect and the number of observations collected along each transect varied from 41 to 79. Transect lengths varied from 500 to 1800 m. We discontinued sampling along an individual transect if it approached an area with a different fire frequency, or if the transect reached the edge of a mesa. Snags were grouped into 3 categories, following methods described by Boucher et al. (2000). Type 1 snags appeared to have been dead for less than 6 yr and retained needles and/or small branches. Type 2 snags had probably been dead for more than 6 yr and had lost all twigs and small branches. Type 3 snags were charred snags that had clearly experienced a recent fire (extensive visible char around circumference of the bole) but remained standing. All type 3 snags appeared to have been dead for at least six yr.

Snag densities were estimated using the program DISTANCE (Laake 1996). Uniform, half normal and hazard rate models for detection distances of each transect were compared. Best fitting models of snag detection were selected by comparing Akiake Information Criterion (AIC) values between the different models. Statistical differences in snag densities between areas with different fire frequencies were assessed for normality and compared using analysis of variance procedures in the program R, an open source statistical environment (R Development Core Team 2003).

Results

Mean snag densities in areas that burned 1, 2 and 3 times were 7.0 $\pm$ 2.7, 4.4 $\pm$ 1.6 and 4.8 $\pm$ 1.7 snags/ha respectively (Table 4). The mean snag density on the twice-burned and mid-century burned Johnson Mesa was 2.8 $\pm$ 1.4. Snag densities were significantly lower in areas that had experienced two or three fires compared to areas that burned only once ($p < 0.05$) (Table 4, Figure 5). Snag densities did not differ significantly between areas that burned two and three times. Snag densities on Johnson Mesa were significantly lower than other twice and thrice-burned areas (mean = 2.8, $p = 0.02$). Areas burned multiple times had significantly higher numbers of type 3 snags than areas burned only once ($p<.001$). Type 3 snags (standing snags charred by a previous fire) were two to four times more abundant in twice and thrice-burned areas than areas that burned only once. Differences in the number of type 2 snags between fire treatments were small but significant ($p<.01$). Type 1 snags occurred more frequently in once and twice-burned areas than areas that burned three times ($p <.001$).

Discussion

As expected, snag densities in areas that burned multiple times were significantly lower than in areas that burned only once. An initial fire after long periods of fire exclusion is likely to cause high tree mortality and hence snag recruitment if there is an excessive accumulation of fuel, including litter and duff that burns around the tree's base. Severe fires can kill fine tree roots near the surface of the soil (Swezy and Agee 1991), alter microbial and mycorrhizal dynamics at the soil surface (Grogan and others 2000), and cause cambial heating sufficient to kill trees directly (Harrington 1996; Ryan 1988; Sackett and others 1996). With surface fuel loading reduced after the first fire, a second fire should kill fewer trees, but could fell many of the snags created by the first fire if enough wood at the snag's base is consumed, or if some of the snags are rotten at the base. Subsequent fires could continue to fell remaining snags without killing many additional large live or dead trees.

Snag densities were significantly lower on Johnson Mesa (Table 4, far right column) than in other areas sampled that had also burned twice during the WFURB era (1974-2003). According to fire perimeter records, this area burned in 1946. The stand structure of this area reflects the effects of that mid-century fire (Holden et al., unpublished data). Trees are generally larger in diameter and sapling and pole-sized trees are virtually absent. We sampled only three transects on Johnson Mesa. However, the relatively low snag densities on Johnson Mesa provide an interesting contrast to areas from which fire had previously been excluded for more than 100 years. We are currently analyzing age structure of stands on Johnson Mesa to confirm that the open stand structure we observed there is in fact the result of the mid-century fires. If this is the case, these results will have important implications for the ecological restoration of ponderosa pine forests. Regardless, snag density data for this area suggests that stand density (number of trees/ha) will influence snag density. This is intuitive, given that stands containing fewer living trees before a fire will contribute fewer potential snag recruits following a fire.

We were surprised by the lack of differences in snag densities between areas that burned two and three times. We expected snag density reduction to continue with repeated fires. However, we rarely observed grass and understory vegetation growing directly at the base of snags. Instead, snags typically had a lot of bark and woody debris at the base, which probably inhibited the establishment of grass and forbs. Several scenarios could explain the lack of apparent differences between areas that burned two and three times. One possibility is that with multiple fires, fire-caused snag recruitment and snag reduction reach equilibrium, such that new snags created by a fire are roughly equal to the number of snags felled. However, we did not detect differences in the number of large-diameter living trees between areas with different fire frequencies, which suggests that mortality of large diameter trees following an initial fire is low (Holden et al., unpublished data). We suggest the following simple conceptual model to explain the lack of differences between twice and thrice-burned areas. An initial surface fire after years of fire exclusion kills some trees, including some large overstory trees. Mortality may be direct, with crown scorch and cambial or root damage sufficient to kill trees outright, or indirect, with trees suffering from stress and dying several years after the fire. When these trees die, their remaining needles drop around their base. As the new snag continues to decay, bark scales and chunks begin to slough off, accumulating around the base of the tree. With a second fire, the needles that have fallen off the newly recruited snag mixed with the fallen bark may carry fire to the tree's base, felling some snags and charring others. The condition of the wood at the snags base will influence the probability of the snag being felled by the fire (Harrington, pers. comm.). Snags that develop rotten bases are likely to fall from any contact with flame or embers, while snags with a solid base will be more resilient to fire. The snags that remain standing after this second fire should be less susceptible to felling by a subsequent fire. While the needles dropped after the tree's death may have been largely consumed by the second fire, some of the bark scales and pieces should still remain and additional bark scales added, limiting the amount of understory vegetation growing near the base of the tree and insulating the tree against future fires. While the debris remaining at the snag's base may smolder and partially burn during a fire, it may not burn enough of the snag's base away and thus leaves them standing. Large snags often stand longer than small-diameter snags (Morgan, unpublished personal observation).

Proportions of type 3 snags in each fire treatment support our snag density results. Type 3 snags are the charred standing snags or carbon cores of previous snags burned by a subsequent fire. Sixteen percent of snags observed in areas that burned 3 times were type 3, compared to 9% in twice burned areas and only 3% in areas that burned only once (Table 4). This suggests that while a second fire may fell some snags, some are charred and remain standing, while an even higher proportion of snags survive a third fire. Those that survive the first and second fires may become relatively fire resistant and persist on the landscape. Type 1 snags (recently created) were more frequent in the once and twice burned areas, while proportions of type 2 snags were nearly the same for each fire treatment. The proportion of each snag type depends on the timing of a fire. Fires in many of the once-burned areas occurred more than 10 years ago. Therefore, many of the type 1 (recently created) snags we would expect to see after an initial entry fire would have already decayed into type 2 snags. However, on Jackass Park, which burned for the first time in 2002, we observed a high proportion (70%) of type 1 snags, supporting our assumption that there is significant mortality of large trees following initial entry fires.

Snag quality as well as quantity determines whether snags are used by cavity-dependent bird species (Bull 1997). Causes of tree mortality may influence suitability of snags for use by wildlife (Shea 2002). We did not carefully evaluate the condition of individual snags (e.g. decay characteristics, presence of insects or cavity-dependent bird species). However, large type 1 and 2 snags are likely to provide the conditions (e.g. wood density, level of decay) preferred by cavity nesting species (Bull 1997; Farris 2002). Relative decay rates of type 2 and type 3 snags are unknown. It is possible that once charred, decay rates of type 3 snags will decrease, increasing their longevity. However, charring could result in significant hardening of a snag, making it less suitable for use by bird species. Many studies indicate that cavity-nesting birds prefer snags with some degree of decay as nesting or feeding sites (Harris 1983). Wakkinen (1992) also found that snags with a relatively large number of intact limbs, indicating a recent death, received significantly more use by cavity-nesting birds than trees with reduced numbers of limbs (i.e., snags that have been dead longer). Several studies support the idea that newer snags or partially dead snags (as opposed to 100% dead) are the most important foraging substrates. Cunningham et al. (1980) noted that birds that used snags as a feeding substrate frequently selected those which were recently dead (1-5 years) and had a high percentage of bark cover. They suggested that this is the result of invasion by insect larvae following tree death, peaking in numbers at about two years after death and declining thereafter.

Snag densities in the Gila Wilderness are consistent with those reported elsewhere. Boucher et al. (2000) reported snag densities ranging from 1.7 to more than 80 snags/ha in areas that varied widely in past management and fire histories. Horton (1988) reported pre-fire snag densities of 12.9 snags/ha (all size classes) in ponderosa pine forests of Arizona in which fire had been absent for many decades. Ganey (1999) reported snag densities ranging from 0 to 45 snags/ha (all size classes) in ponderosa pine forests across six National Forests in Arizona. These studies highlight the broad range of stand conditions that exist in the southwestern United States, and the potential variation in snag densities that can exist, depending on stand history, site quality and past management.

The snag densities we report for areas of the Gila Wilderness fall near or within the snag management guidelines currently used by National Forests in the Southwest (U.S. Department of Agriculture 1996). However, it is important to note that management in the Gila Wilderness has differed dramatically from most areas of the southwestern United States. While the Gila National Forest, including remote areas of the wilderness, was grazed intensively until 1950, the wilderness was never logged. Consequently, large-diameter trees and trees that would have later been recruited into larger size classes were retained. In contrast, selective timber harvest in other areas of the southwestern United States has reduced the number of large-diameter trees. Recent analysis of U.S. Forest Service inventory data suggests that more than 90% of all trees in southwestern ponderosa pine forests are 30 cm dbh or smaller (Southwest Forest Alliance 1996).

Implications for Snag Management

Results from our study suggest that many large ponderosa pine snags in the Gila Wilderness are retained on the landscape, even after exposure to repeated fires. To our knowledge, this is the first study to quantify the effects of repeated fire treatments on snag densities. These results have important implications for ecological restoration and management of fire in ponderosa pine forests in the southwestern US and elsewhere. Scientists and managers have long recognized the need to restore fire to forests that are significantly departed from their historical fire regime (Allen and others 2002; Covington and Moore 1994a). The major challenge to restoring natural fire regimes will be to alter stand structure characteristics and surface fuel loads sufficiently to promote surface fires while minimizing severe stand-replacing events. However, one fire may not sufficiently reduce fuels and stand density, and fire hazard often remains high because tree injury and mortality after an initial fire increases surface fuels and changes the characteristics of crown fuels (Harrington 1981; Harrington 1982; Sackett and others 1996). Therefore, repeated fires will be needed to achieve ecological restoration or fuel management objectives. Once a forest or stand has been restored with respect to surface fuel loading, follow-up treatments will be necessary to maintain those conditions. Given the importance of snags as habitat for wildlife species, and the current guidelines advocating management for relatively high snag densities, understanding the effects that repeated burning has on snag densities will be critical in determining additional management actions that can be taken to recruit and retain snags as an ecologically important part of forest structure.

Conclusions

Snags and downed logs play an important ecological role in forest dynamics. Pervasive alterations of fire regimes across many forested regions of the United States have altered these dynamics. These changes are most pronounced in areas that experienced historically frequent surface fires, where relatively brief disruption of historical fire regimes can lead to significant departures in vegetation from historical conditions. Ponderosa pine forests are perhaps the best-known and most extensively studied examples of such areas. Fire is now frequently used as a management tool to manage fuels and to restore and maintain structural and ecological conditions in many areas. However, questions still remain about the effects of repeated fires on stand structure and snag densities. Our results suggest that repeated fire treatments leave many large-diameter snags standing. Both recruitment of new snags and retention of snags of a variety of stages of decay will be important to cavity-nesting birds. To the extent that restoration burning promotes growth and survival of large-diameter trees, such burning will promote a supply of large-diameter snags.

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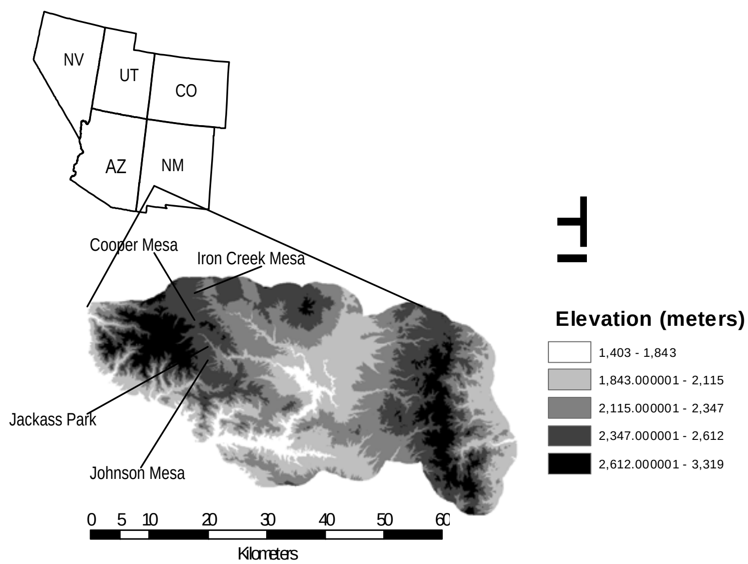

Figure 4. Study areas in the Gila Wilderness, New Mexico.

Table 4. Mean snag densities (snags/ha) with 95% confidence intervals for transects burned one, two or three times under WFURB (1974-2004).

Fire Occurrence	**1x Burn (a)**	**2x Burn (b)**	**3x Burn (c)**	**Mid-century +2 WFU-era fires (d)**
Mean	7.0	4.4	4.8	2.8
95% C.I.	4.25 - 9.65	2.8 - 5.0	3.1 - 5.9	1.4 - 4.2
Significance test (α = .05)	a vs. b	a vs. c	b vs. c	b vs. d
Anova (n= 6; F-test)	p=0.006	P= 0.05	p=0.49	p = 0.02
% Type 3 snags	3%	09%	16%	22%
% Type 2 snags	78%	72%	77%	69%
% Type 1 snags	19%	19%	7%	9%

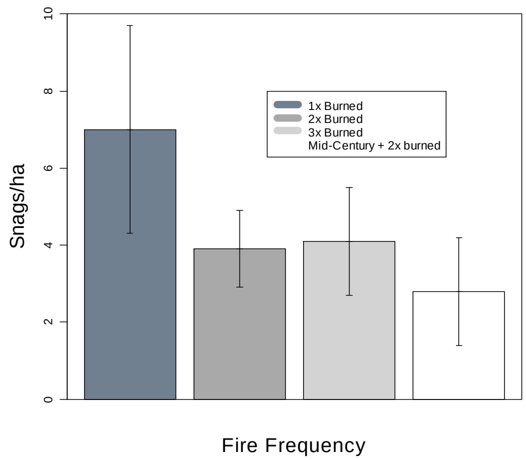

Figure 5. Mean snag densities by fire frequency. Vertical bars represent standard errors.

Figure 6. Photographs of the Gila Wilderness. Johnson Mesa (left) burned in 1946, 1992 and 2003. Langstroth Mesa (right) is unburned in the 20th century according to fire perimeter records.

CHAPTER 3

THIRTY YEARS OF WILDLAND FIRE USE: EVALUATING THE EFFECTS OF MULTIPLE FIRES ON PONDEROSA PINE MORTALITY AND STAND STRUCTURE IN TWO SOUTHWESTERN WILDERNESS AREAS, USA

Keywords: Ponderosa pine, Wildland Fire Use for Resource Benefit, restoration

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), tree diameter-class distributions, basal area (BA) and stand density index (SDI) were compared in 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 RMW and Gila Wilderness, significantly fewer small-diameter (0-22.5 cm) trees occurred in areas that burned multiple times than areas that were unburned ($p < 0.05$). The density of large diameter (45-90 cm+) trees in Gila Wilderness was highly variable and did not differ significantly between fire treatments ($p > 0.32$). In the RMW, 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 increment growth rates (BAI) from tree cores were compared for areas in the Gila Wilderness that did and did not burn mid-century (1946) before burning during the WFU era. These were similar pre-fire but diverged significantly post-fire ($p<0.05$ for decades

since 1946 fire). Thus stand density and tree diameter growth reflect the thinning effect of that mid-century fire. Forests in the Gila Wilderness and RMW are structurally diverse and resilient to fires burning under dry, hot, windy conditions, suggesting that repeated WFU fires have restored forest resilience to fire.

Introduction

Southwestern ponderosa pine (*Pinus ponderosa*) ecosystems evolved under a regime of frequent, low-intensity fires that limited fuel accumulation and maintained low densities of small trees (Cooper 1960; Swetnam and Baisan 1996; Swetnam and Dieterich 1985; Touchan and others 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 reduced understory vegetation that carried low-intensity fires, limiting their size and frequency (Savage 1991). Fire suppression and other land uses have excluded all but the 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 (Arno and others 1995; Covington and Moore 1994b). 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 and others 1997). Habitat quality for key wildlife species has also declined (Long and Smith 2000).

Field Code Changed

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 result from fire exclusion (Rollins and others 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 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 (Allen and others 2002; Weaver 1951). 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 have 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 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 (Allen and others 2002; Covington and others 1997).

Structural restoration, especially removal of small understory trees is an important component of ponderosa pine forest restoration prescriptions (Covington and others 1997). However, in some cases, fire may not kill small trees, and reducing the density of small trees by fire or mechanical means may not be possible without also damaging large overstory trees (Lynch 1959; Swezy and Agee 1991). Mechanical thinning of some areas may be necessary before fire can be safely reintroduced. 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 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 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; Harrington 1982; Sackett and others 1996). However, the effects of multiple fires on tree mortality are not well studied (but see Sackett 1996). Previous studies on the effects of prescribed fires have shown that ponderosa pine trees 5-15 cm dbh trees are frequently killed by surface fires while larger trees generally survive (Harrington and Sackett 1990). However, prescribed burning in ponderosa pine forests is often done in the early spring or fall when temperatures are cool and fuel moistures are high in order to minimize extreme fire behavior. In contrast, wildland fires typically occur when weather conditions are extreme and fuel moisture levels are low.

Fire behavior under these conditions, especially after years of fire exclusion, will likely be more severe than prescribed fires 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 results 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 trees by an initial entry fire may result in heavy post-fire fuel loading, increasing the intensity of a second 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). Large portions of the Gila Wilderness and RMW have burned repeatedly since this policy was implemented in 1975. Comprehensive fire atlases within geographic information systems document the spatial extent and year of all fires. In both areas, fires have burned repeatedly in many places (Rollins and others 2001). The rich history of well documented fires and extensive spatial data available on the size and timing of fires provides a unique natural experiment with which to evaluate the effects of one and multiple fires on stand structure characteristics of ponderosa pine forests. In both the Gila Wilderness and RMW, occurrence of mid-century fires prior to implementation of the WFU program allows us to evaluate the effects of prior treatments 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. To our knowledge, this is the first research study to evaluate the effects of repeated, natural fires at broad spatial scales on southwestern ponderosa pine forest structure. 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. For these reasons, 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 7). 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. Annual precipitation varies from 250 mm to 760 mm at higher elevations (Beschta 1976), falling as snow in the winter and as rain in the early-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, leading to favorable fire conditions. Historically, low-severity (low tree mortality) surface fire regimes dominated ponderosa pine forests of the Gila Wilderness (Swetnam 1983). Mixed severity fire regimes can be found at higher elevations (Abolt 1996). Pinõn-oak juniper woodlands occur at lower elevations of the wilderness and cover 23% of the study area (Rollins and others 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 until the 1950's but has never been logged.

Rincon Mountain Wilderness Area

The 29,700-hectare Rincon Mountain Wilderness area of Saguaro National Park (Figure 7) 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. Elevation of the Rincon Mountains ranges from 870 meters at their base to 2600 meters at the peak of Mica Mountain. Annual precipitation in the Rincon Mountains varies by elevation. Lower elevations receive 330 mm while upper elevations receive from 640-760 mm, some of which falls as snow. Precipitation peaks occur in the winter and summer months, with the majority falling as rain in the summer. Orographic lifting of moist air from the gulf results in frequent summer thunderstorms. These storms bring lightning, and with it, fire. Fire season in the Rincon Mountains typically begins in

April, peaking in July at the height of the monsoon storm season. Historically, ponderosa pine forests in the RMW were dominated by low-severity surface fire regimes (low mortality of large trees) with fire frequencies ranging from 3-19 years (Baisan 1983). The area's steep elevation gradient and diverse conditions give rise to a remarkably rich variety of 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 with increased elevation. We focus this study in 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.

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 by WFU-era (1974-2003) fire frequency. Strata included unburned areas and areas that burned once and two or more times from 1974-2003 according to fire perimeter data from digitized fire atlases records (Rollins et al 2001). Additional plots were located in areas that burned in the early and mid-century (1946) prior to implementation of Wildland Fire Use and then again during the WFU era. In both 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). Sampling of unburned areas in the Gila Wilderness was limited to one relatively small mesa and is unreplicated. 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 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 terrain and great topographic variability.

Fire Atlas Accuracy Assessment

In the Iron Creek Mesa area of the Gila Wilderness, sample plots were sometimes located near the boundary of two intersecting fire perimeters. Due to concerns about the accuracy of fire atlas perimeters with which we stratified our sampling, we used Landsat TM images to map burned area perimeters of two fires sampled in the Gila Wilderness (Holden and others 2005). Correspondence between the atlas-derived and imagery-derived fire perimeters was high, especially at the eastern and western edges that defined critical 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 (figure 8). 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 historical fire perimeters confirms that fires in that area, including older fires, were mapped with a high degree of accuracy.

Field Code Changed

Stand Structure Sampling

Stand structure data was collected within 11.3 m (400 m^2) fixed-radius plots following FIREMON fire effects monitoring protocol (www.fire.org/firemon). 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 (FMH) plots (for 64 plots total in RMW) (http://www.nps.gov/fire/public/pub_und_fireeffectsmonitor.html). FMH forest plots measure DBH and height of all trees >15cm DBH within 1000m^2 plots and all trees <15cm DBH within a 250m^2 subplot. These data were scaled to match the FIREMON plot data. The nested plot design of FMH data made it impossible to calculate mean tree diameter and to fit Weibull density distributions to these data that were calculated for data collected in the Gila Wilderness using FIREMON sampling.

Age Structure Sampling

Age data were collected within a randomly selected subset of sample plots in order to establish relationships between tree age and size. 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.

Despite the presence of false rings throughout the increment cores, we chose not to cross-date samples (Stokes and Smiley 1968). While cross dating may have added additional precision to our estimates of tree age, we felt this precision was unnecessary for the purposes of evaluating general tree growth patterns. When they were available, one to three additional trees near but outside the plot were cored to obtain ages using the same methods for trees larger than those on the plot.

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. Radial increment growth of 18 trees from each treatment (2x and 2x+preWFU) were 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

Two analysis methods were used to compare stand structure across different fire treatments. This was necessary because the National Park Service fire effects monitoring data used for the RMW was incompatible with our own data for calculating mean DBH as well as generating tree diameter distribution functions. For the Gila Wilderness, shape and scale parameters extracted from two-parameter Weibull functions fit to tree density by size data for each plot were analyzed as a bivariate response variable using multivariate analysis procedures with fire treatment as a predictor variable. Four fire treatments were defined: unburned, once burned (1x), burned multiple times from 1974-2003 (2x), and burned in the mid-century (1946) and again during WFU (2x+preWFU). The Hotelling-Lawley test was used to test for differences in Weibull parameters between individual fire treatment levels. The Weibull function is highly flexible, and commonly used to model forest stand structure (Avery 1994; Bailey 1973). 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.

For both the RMW and Gila Wilderness, parameters associated with tree size class distributions, including mean stem diameter at breast height (DBH), mean density (trees/ha), basal area (BA m^2/ha) and stand density index (SDI), were compared between fire treatments. We compared differences in mean density of 0-22.5 cm trees (size classes likely to be affected by fire) and trees 47.5 cm or larger (large diameter trees important for wildlife species and of interest to managers) between fire treatments. The small tree diameter cutoff was selected based on previous research on the effects of prescribed burns on tree mortality (Harrington 1993; Sackett and others 1996). Data were analyzed using two way analysis of variance procedures and Tukey's test for honest significant differences in program R, an open source statistical environment (R Core Development Team 2003). 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.

Results

Tree Density and Stand Structure

Stand structure differed significantly between fire treatments. Mean tree size (DBH) was higher and density (TPH) was lower in areas subject to multiple fires in the both the Gila Wilderness (Table 5, Figure 9) and the RMW (Table 6, Figure 9). Mean DBH was significantly higher and tree density was significantly lower in the 2x+preWFU than in 2x (areas burned twice in WFU but not burned in 1946) ($p < 0.01$) (Tables 5 and 6). There were significant differences in the density of small diameter trees (0-22.5 cm) between fire treatments ($p < 0.05$) but no statistically significant ($p = 0.32$) differences in the density of large diameter (>45 cm) trees (figures 4 and 5). SDI and BA did not differ significantly (p values >0.05) between treatments in either the Gila Wilderness or the RMW.

In the Gila Wilderness we were able to fit the Weibull distribution to the data. There, shape and scale parameters did not differ significantly between unburned and once burned areas ($p = 0.06$), but were significantly different between all other fire treatments ($p < 0.01$) (Table 5).

Comparison of mean 10-yr basal area increment growth of trees from areas that did (2x+preWFU) and did not burn mid-century before burning during the WFU era suggest that differences in stand structure between these areas reflects the effects of that early fire. Growth rates of trees from both areas are similar up until the mid-19th century. While growth rates continue to follow the same relative pattern, they diverge after the 1940's (figure 12).

Discussion

Our results suggest that repeated WFU-era fires have generally reduced the density of small-diameter trees without significantly affecting the density of larger trees. These results are consistent with mortality patterns observed in prescribed fire studies in ponderosa pine forests elsewhere in the southwestern US (Sackett and others 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 acres over weeks and months. We expected to see higher mortality rates for trees of all sizes, including those of large diameter, especially in areas of high tree density and in locations where fires had not burned frequently.

Tree size-class distributions in many stands in both the RMW and Gila Wilderness are skewed toward small to medium diameter (15-30 cm) trees (Figures 9 and 12), even in sites burned three or more times in the 20th century. Such distributions likely reflect pulses of regeneration during favorable environmental conditions (Savage and others 1996) and subsequent 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 (Moore and others 2004; Oliver 2001; White 1985).

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. Small diameter trees were uncommon. In the Gila Wilderness, large diameter (82.5-112.5 cm) trees were present in areas burned mid-century and largely absent from areas that did not burn in mid-century. Biophysical characteristics at these sites, or long-term history, such as fires unrecorded in the fire atlas, may have contributed to the difference in stand structure we observed. 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 12). 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 in the face of reduced competition. Studies that have evaluated the effects of thinning and prescribed burning 7-9 years post-treatment generally support this conclusion (Sala and others 2005; Wallin and others 2004). However, this study is the first to retrospectively evaluate the effects of early (mid-century) fires on resulting stand structure and growth patterns in a ponderosa pine forest.

Timing of repeated fires 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 interesting patterns that highlight the impacts of fire exclusion on stand development in the Gila Wilderness. Assuming that the fire that burned across Johnson Mesa in 1946 is responsible for the low tree densities and larger tree diameters compared to other areas of the wilderness, we can infer that the timing of fires during the last hundred years has been critical in determining present stand structure. The 1946 fire may have occurred early enough to kill many of the small diameter trees, 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 Southwest, is much denser than Johnson Mesa. 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 such that continued burning, even under natural conditions and at historical frequencies may thin recently recruited small understory trees but will not significantly affect mid-sized and larger trees that established during a period of fire exclusion. As such, achieving late-successional characteristics at these sites may take many years.

Limitations

This study has several important limitations. First, the accuracy of fire atlases are unknown, and probably low, especially in areas characterized by low-severity fire regimes because it is difficult to map fire perimeters when few of the larger trees are killed (Morgan and others 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 and others 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 (figure 8), 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. Second, our primary unburned (control) area in the Gila

Wilderness burned in 2002, months before we planned to sample there, leaving us with a paucity of large areas that have not burned during the WFU era. We did locate one area that had not burned in the 20^{th} century. However, it was small relative to other mesas (approx 6 km^2), and the relatively small number of plots collected there (n=12) may not represent the variety of stand conditions one might expect to see across a broader area. 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 (fire frequency, site productivity, etc.) 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. Forest stand structure is highly variable across all of our study areas, even in areas that have burned multiple times (Figure 13). Repeated fires continue to reduce density of small trees while not significantly reducing the density of large trees. This effect is likely to be more pronounced wherever stand density is very high and fires have been more effectively excluded as the result of fire suppression and livestock grazing, as is the case outside of large wilderness areas with active WFU programs. Fire-caused canopy openings ranging in size from 0.5-20 ha have been observed across both study areas. Comparison of historical aerial photographs with modern imagery indicates that an initial fire created most of these openings after a long period of fire exclusion (Holden et al., unpublished data). Subsequent fires wrought fewer changes in the forest. Thus, most of the risks, in terms of severe, canopy-replacing fires are associated with the first fire after long periods of fire exclusion. This suggests that multiple treatments may influence stand dynamics via further reduction of fuel loads and small diameter trees without significantly impacting large diameter trees.

Historical accounts of early settlement in what is now the Gila Wilderness recall extensive grazing by sheep and cattle from the 1870's through the 1950's (Woodrow Unknown). Grazing allotments still exist in the wilderness today. There are also accounts of fires in the early 20th century that may not be accurately represented in modern archived fire perimeter data. However, the size and relative inaccessibility of the Gila Wilderness and its early establishment (1924) as a wilderness area may have limited the intensity of grazing and the associated changes in understory vegetation that limit fire spread and promote ponderosa pine seedling establishment. They have also spared it from the widespread logging that occurred across much of the southwestern U.S. Similar patterns of disturbance occurred in the RMW. Grazing was widespread in the Rincon Mountains, reaching its peak at the end of the 19th century (Abolt 1996). While fuel accumulation during nearly a century of fire exclusion in many areas of the RMW and Gila Wilderness would have increased the intensity and severity of initial entry fires, ecological and structural change may not have been as severe in the Gila as in other areas of the southwestern US. Tree densities reported in the literature for southwestern ponderosa pine sites are generally higher than those in the Gila Wilderness and RMW (Covington and Moore 1994b; Moore and others 1999). This limits the extent to which we can extend inferences we draw about the Gila Wilderness to other areas of the Southwest.

Conclusions

Forest structure in the RMW and Gila Wilderness reflects a history of recent and historic fires. Stand structure in the areas we sampled were remarkably diverse, even in areas that burned multiple times, with small-diameter trees present at both sites. Recent fires have contributed to the resilience of many stands against continued burning. In some areas, early mid-century fires have contributed to relatively open stands of larger-diameter trees. Comparisons between these sites and areas that burned repeatedly, but only during the WFU-era suggest that the timing of these fires may have been critical in determining successional trajectories of forest stand structure. While resilience to burning 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 reduced densities of small diameter trees, but many stands are probably still denser than in pre-settlement times. These data suggest that repeated fires, even under natural conditions may not remove pole-sized 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 on the physiological responses of individual trees.

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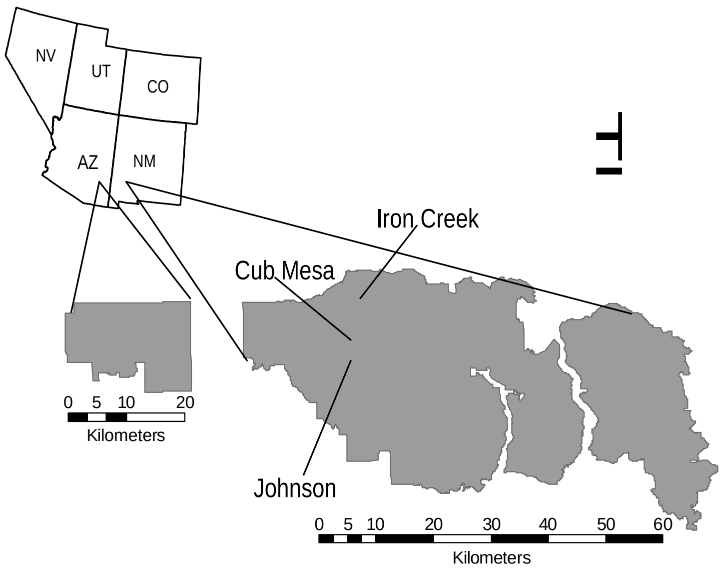


Figure 7. Study areas in the Rincon Mountain Wilderness, AZ (29,500 ha) (left) and the Gila-Aldo Leopold Wilderness Complex, NM (230,800 ha) (right)

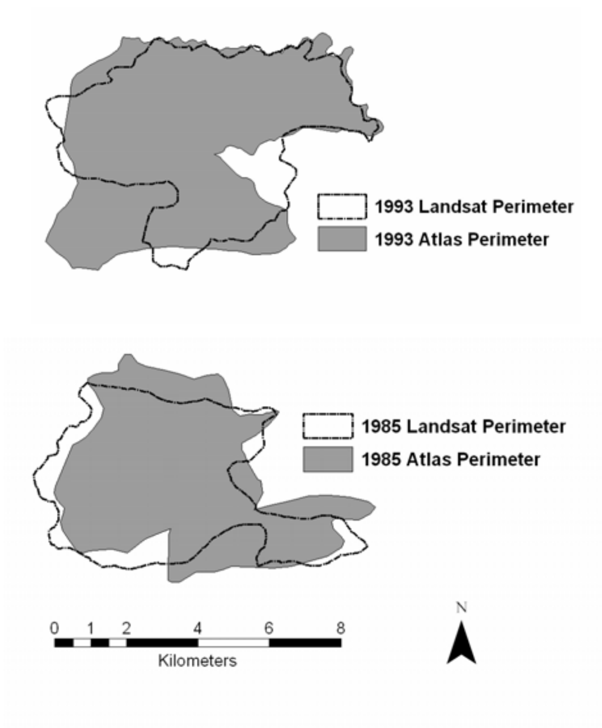
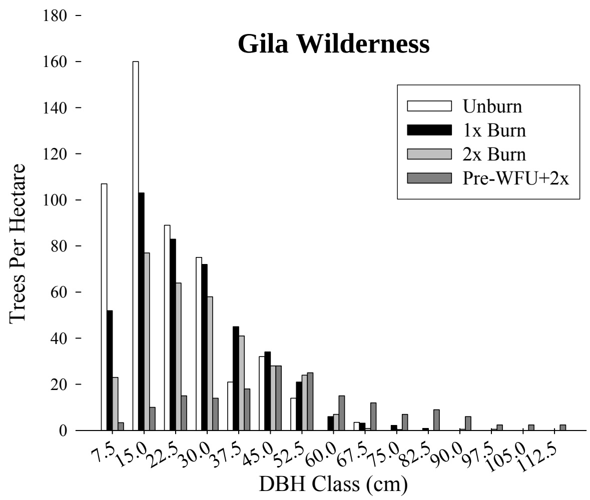


Figure 8. Correspondence between fire perimeters derived from satellite imagery (clear outline) and fire atlas perimeters (gray) for two fires sampled on Iron Creek Mesa.



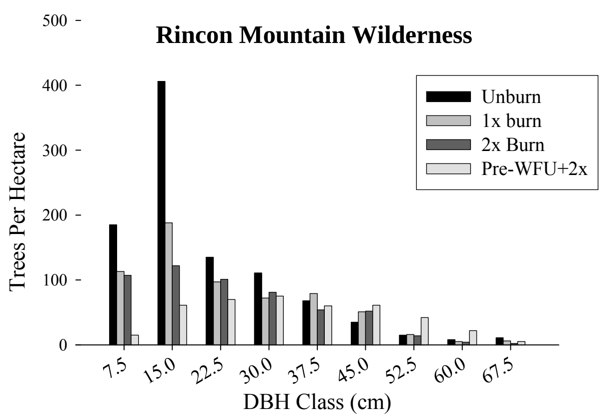


Figure 9. Gila Wilderness and RMW trees/ha by 7.5 cm size class.

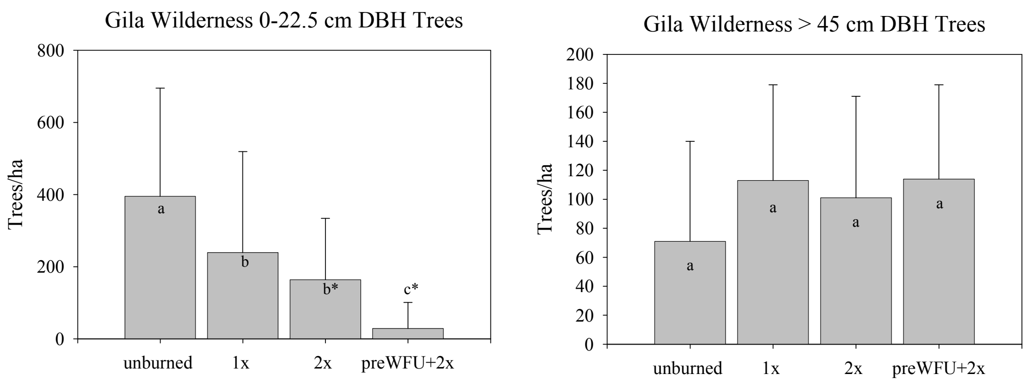


Figure 10. Gila Wilderness bar plots comparing small diameter (0-22.5 cm) and large diameter (>45 cm) trees by fire treatment. Vertical bars represent 1 SD. Different letters represent significant differences at α = 0.10. Different letters followed by * indicate differences significant at α = 0.05.

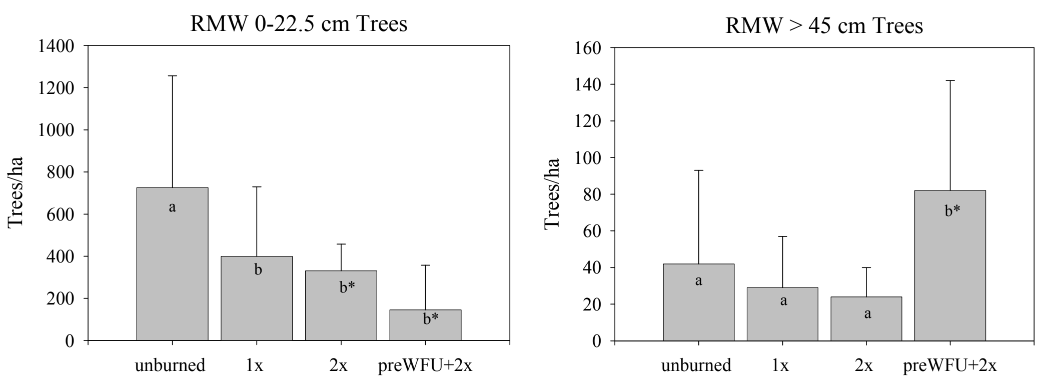


Figure 11. Mean density of small (0-22.5 cm DBH) and large diameter (>45 cm DBH) trees for fire treatments in RMW. Vertical bars are standard deviations. Different letters represent significant differences at α = 0.10. Different letters followed by * indicate differences significant at α = 0.05.

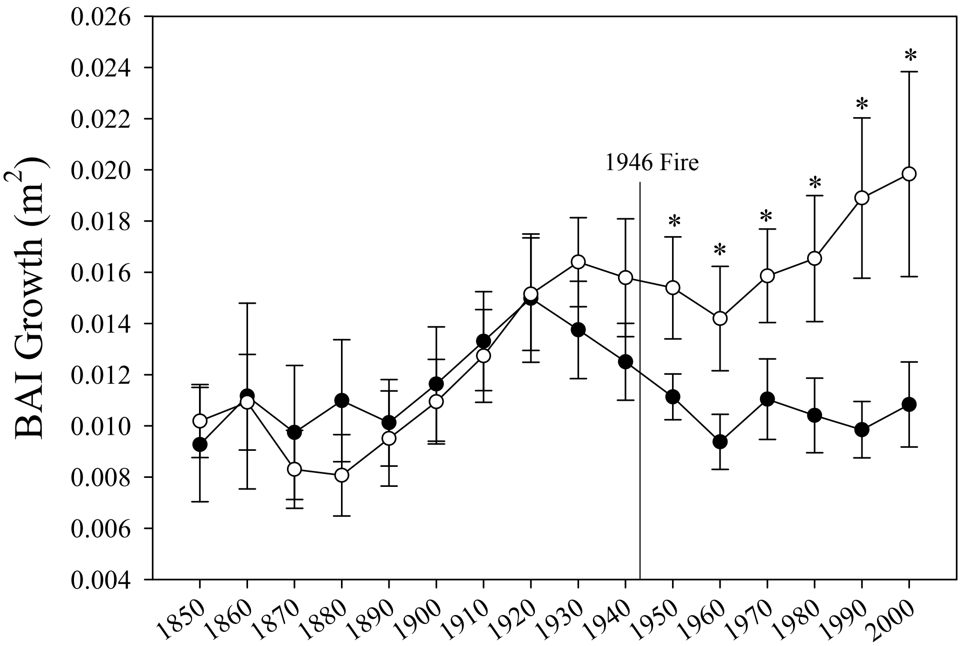


Figure 12. 10-yr mean periodic basal area increment growth of trees that burned in 1946 and then again during WFU (2x+preWFU) with stands that burned only during the WFU era. Years represent midpoint of 10-year mean. Error bars represent 1 SD. * indicates significant differences at α= 0.05.

Figure 13. Photographs of unburned stands (upper right photo) and areas burned multiple times (upper left photo) in the Gila Wilderness and unburned stands (lower right) and areas burned multiple (lower left) times in the RMW.

Table 5. 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 (0-22.5 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 6. 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 * indicate 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*
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
0-22.5 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*