Ecological Applications, 7(3), 1997, pp. 895–908

DETERMINING REFERENCE CONDITIONS FOR ECOSYSTEM MANAGEMENT OF SOUTHWESTERN PONDEROSA PINE FORESTS

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Abstract. The fire disturbance regime and forest structure prior to Euro-American settlement (AD 1883) of a southwestern ponderosa pine (*Pinus ponderosa*) landscape were quantified in order to establish reference conditions as a baseline for ecosystem management. The mean presettlement fire interval between 1637 and 1883 was 3.7 yr for all fires and 6.5 yr for widespread fires, but fire has been excluded from the study area since 1883. Forest density increased under fire exclusion from an average of 148 trees/ha in 1883 (65 pines, 80 oaks, three other species), an open forest dominated by relatively large ponderosa pines, to 1265 trees/ha in 1994/1995 (720 pines, 471 oaks, 74 others), a dense forest characterized by relatively small and young trees. Species composition has shifted toward greater dominance by Gambel oak (*Quercus gambelii*) and conifers less adapted to frequent fires: white fir (*Abies concolor*) and Douglas-fir (*Pseudotsuga menziesii*). The reference presettlement conditions can be applied to management of this ecosystem in two ways. First, reference conditions are a benchmark against which to evaluate contemporary conditions and future alternatives. The comparison shows that the contemporary forest is well above the range of presettlement variability in forest density, and both live and dead fuel structures have developed that can support high-intensity wildfire. Second, reference conditions can serve as a goal for ecological restoration treatments.

Key words: Arizona; ecological restoration; ecosystem management; fire regime; Pinus ponderosa; *ponderosa pine; presettlement forest reference conditions.*

INTRODUCTION

We present an integrated method for determining reference conditions of key ecosystem functional and structural components in a southwestern ponderosa pine forest prior to Euro-American settlement; then we apply the reference data in an ecosystem management context. Reference conditions are the range of historic (or natural) variability in ecological structures and processes, reflecting recent evolutionary history and the dynamic interplay of biotic and abiotic conditions and disturbance patterns (Morgan et al. 1994, Swanson et al. 1994). These conditions form the basis for comparison with contemporary ecosystem processes and structures and are a frame of reference for designing ecological restoration treatments and conservation plans (Covington and Moore 1992, 1994*a*, Grumbine 1992, Kaufmann et al. 1994, Morgan et al. 1994, Swanson et al. 1994). Almost no ecosystems remain undisturbed by industrial human society, and early historical documentation of past conditions is commonly quite limited, so quantifying reference conditions is a key initial challenge for ecosystem management (Swanson et al. 1994, Kaufmann et al. 1994, Christensen et al. 1996).

Fires recurred every 2–20 yr in southwestern ponderosa forests prior to Euro-American settlement around 1870 to 1890 (comparison of 63 fire chronologies in Swetnam and Baisan 1996). Fire has been a keystone ecological process (sensu Holling 1992) regulating the density, species composition, and amount and structure of dead biomass, maintaining open forests, and promoting nutrient cycling in presettlement ponderosa pine ecosystems (Cooper 1960, Covington and Sackett 1990, Covington and Moore 1994*a*, *b*). Although we recognize that Native Americans ''settled'' North America long before Euro-Americans, for the sake of brevity the term *presettlement* will be used here to refer to conditions prior to Euro-American settlement and the term *postsettlement* to refer to conditions after Euro-Americans began the intensive alteration of the landscape that resulted in fire exclusion.

The century-long exclusion of frequent, low-intensity fires has led to striking and rapid changes in ecosystems that evolved under frequent disturbance: trees less adapted to frequent fire (e.g., *Abies, Pseudotsuga*) have invaded at the expense of other plants, and pine (*Pinus*) tree biomass, both live and dead, has steadily accumulated, contributing to progressively declining herbaceous productivity, increasing susceptibility to insect and disease epidemics, and supporting a shift from frequent, low-intensity surface fires to increasingly larger crown fires (Cooper 1960, Swetnam 1990, Covington and Moore 1994*a*, *b*, Kolb et al. 1994, Swetnam and Baisan 1994). Similar deleterious changes seen in related long-needled pine ecosystems throughout western North America (Covington et al. 1994) form much of the impetus behind increasingly urgent calls for bi-

Manuscript received 6 May 1996; revised 4 November 1996; accepted 26 November 1996.

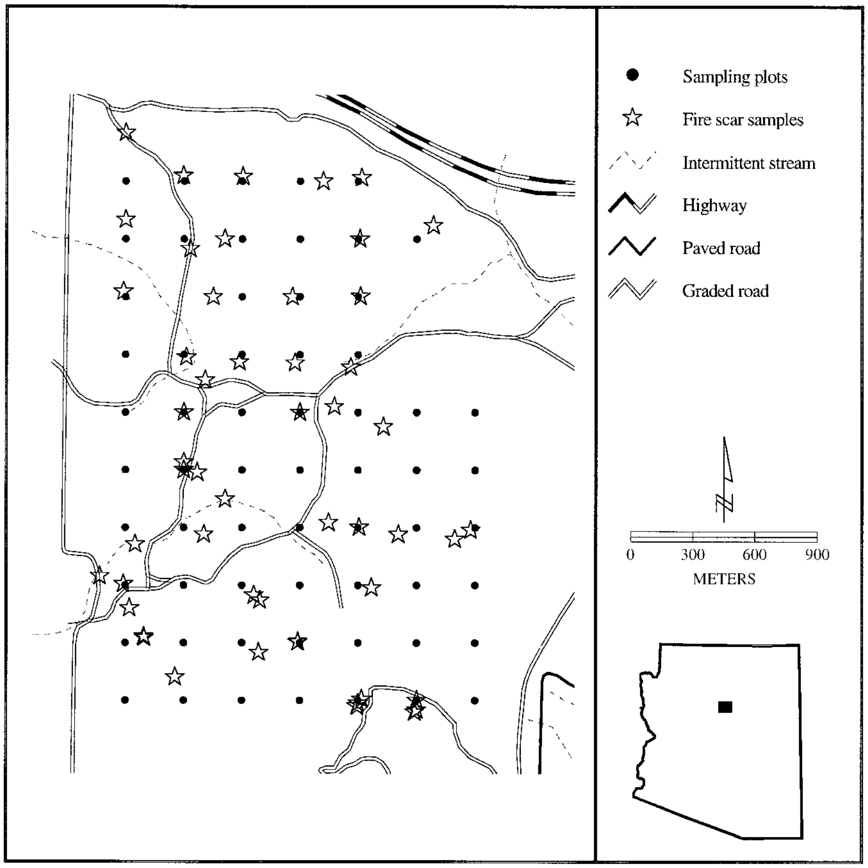

FIG. 1. Study area map showing fire scar and forest structure sampling locations.

ological conservation, ecological restoration, and ecosystem management (Leopold 1937, Grumbine 1992, Covington et al. 1994, Kaufmann et al. 1994, Arno et al. 1995*a*).

Keystone ecosystem variables for presettlement ponderosa pine reference conditions include the autotrophic organisms (trees, shrubs, and herbaceous plants) and the predominant contagious disturbance process, fire. These variables regulate their own environment through positive feedback interactions (sensu Perry 1995) and in turn constrain other components of the ecosystem (sensu Holling 1992); therefore they form the basis for management intervention.

The questions addressed by this study are: what were the temporal and spatial patterns of presettlement fire in a Southwestern ponderosa pine forest? What was the presettlement forest structure and how has it changed following fire exclusion and other human-caused perturbations? Finally, how can this information be applied to support sustainable management of Southwestern ponderosa pine ecosystems?

STUDY AREA

The study area is ≈700 ha of ponderosa pine (*Pinus ponderosa* Dougl. ex. Laws.)–Gambel oak (*Quercus gambelii* Nutt.) forest at Camp Navajo, an Arizona Army National Guard facility 16 km west of Flagstaff, Arizona (Fig. 1), latitude 35°15′ N, longitude 111°52′ W. Average annual precipitation at the Flagstaff weather station, elevation 2134 m, is 502.9 mm, with a distinct dry period in May and June (average 31 mm), followed by the summer monsoon in July and August (average 134.8 mm) (Sellers and Hill 1974). The study area has a gently rolling topography with a single large hill, Volunteer Mountain; elevations range from 2175 to 2453 m. Most soils are of volcanic origin but soils derived from underlying limestone and sandstone formations occur at lower elevations (Soil Conservation Service 1970). Ponderosa pine and Gambel oak are the dominant trees, with scattered alligator juniper (*Juniperus deppeana* Steud.). On mesic, north-facing slopes, Douglas-fir (*Pseudotsuga menziesii* (Mirb.)) and white fir (*Abies concolor* (Gord. & Glen.) Lindl.) are also encountered. New Mexican locust (*Robinia neomexicana* Gray) thickets occur in the southern part of the study area.

Evidence of human occupancy dates from the archaic period (≈5500 BC–AD 100) (Grenda 1993). Large-scale timber extraction became profitable when the transcontinental railroad arrived in 1882 and logging for ties and pilings began around 1885 (Timber Atlas, on file at Kaibab National Forest Supervisor's Office, Williams, Arizona). Livestock grazing began in the same period. The area was transferred from mixed National Forest and private ownership to military use in 1942 and has formed part of the undeveloped buffer zone around the military facilities to the present.

METHODS

Quantitative procedures were selected to reconstruct past fire regimes and forest structures. Comparative studies of fire scar analysis methods concur that the most reliable fire histories in forests with low-intensity fire regimes are based on crossdated fire-scarred samples from trees, stumps, and logs (Madany et al. 1982, Baisan and Swetnam 1990). In ponderosa pine forests where direct evidence (stumps, logs, old-growth trees) of past forest structure still exists due to the very slow wood decomposition rate in the absence of fire, direct measurement of presettlement-era remnant wood together with contemporary forest structure is possible (Habeck 1990, Covington and Moore 1994*a*, *b*, Arno et al. 1995*b*). We applied dendroecological reconstruction (Fritts and Swetnam 1989) in a hierarchical sampling framework scaled to different forest structural components (e.g., presettlement pine clumps, regeneration, coarse woody fuels) to determine reference conditions and compare them to contemporary conditions across a relatively large forest landscape.

Field sampling

A 300 × 300 m systematic sampling grid was established in the study area and sample points were assessed for suitability according to the following criteria: (A) suitable soil type for ponderosa pine forest based on Soil Conservation Service (1970) soil survey; (B) at least 10% forest cover in presettlement times, with ponderosa pine a dominant species, based on the presence of presettlement-era trees, snags, stumps, or logs; and (C) at least 20 m from a road to minimize disturbance to living trees and dead woody material. A total of 62 plots covering 558 ha was sampled in 1994 and 1995.

Sample plots were laid out as a series of nested plots of different sizes. At the largest scale, fire scar samples were collected and their locations mapped (Fig. 1) throughout each 9-ha area centered on the sampling grid points, effectively searching the entire study area. Cross section samples of fire-scarred tree catfaces (a lesion on the tree bole caused by repeated fire scarring) were collected from trees, stumps, or logs apparently recording the oldest scars and/or the best preserved record of multiple fires (Baisan and Swetnam 1990).

Overstory vegetation was sampled on circular plots 400 m^2 in area (11.28 m radius) centered on each sample point. Species, condition, diameter at stump height (dsh, 40 cm above ground level), and a preliminary field classification of presettlement or postsettlement origin, were recorded for all live and dead trees $>$1.37 m (breast height) in height, as well as for stumps and downed trees that surpassed breast height while alive. Tree condition classes were assigned based on a tree, snag, and log classification system (Maser et al. 1979, Thomas et al. 1979) widely applied in ponderosa pine forests (Covington and Moore 1994*a*, *b*, Lundquist 1995). The nine condition classes (1, living; 2, declining; 3, recent snag; 4, loose bark snag; 5, clean snag; 6, snag broken above 1.37 m; 7, snag broken below 1.37 m; 8, downed; 9, cut stump) were used to determine dead tree structure and to estimate the death date of presettlement-era snags and logs, as described below. Potentially presettlement ponderosa pine trees were identified based on size (dsh $>$ 40 cm) or yellowed bark (White 1985). Trees of all other species were considered as potentially presettlement if dsh $>$ 40 cm, or dsh $>$ 20 cm for oaks and junipers (Barger and Ffolliott 1972). All potentially presettlement trees, as well as a random 20% subsample of other trees, were cored with an increment borer at 40 cm above ground level to determine age and past size, as described below. Diameter at breast height (dbh) was recorded for all cored trees.

Seedlings or sprouts of tree species were tallied on nested 100-m^2 (5.64 m radius) plots. Stems $<$ 1.37 m in height were tallied on the nested 100-m^2 plot by species and condition in three height classes: (1) $\leq$ 40 cm; (2) 40.1–80 cm; and (3) 80.1–137 cm. Herbaceous and shrubby species were measured on two 1-m^2 subplots per plot. The frequency (presence/absence) of dominant plants and percent herbaceous cover were recorded. Forest floor fuels and litter and duff depths were measured along 15-m planar intersect transects (Brown 1974) established in random directions from each plot center. Woody fuel biomass was calculated using procedures in Brown (1974) and Sackett (1980).

Dendrochronology–fire disturbance history

In the laboratory, fire scar samples were surfaced and crossdated (Stokes and Smiley 1968) with master tree-ring chronologies from the region (D. A. Graybill, *unpublished tree-ring chronologies AZ521 and AZ547,* National Geophysical Data Center, World Data Center-A for Paleoclimatology, Boulder, Colorado). Ring widths of all samples were measured and dating of each sample was checked with the COFECHA program (Grissino-Mayer and Holmes 1993). A 20% subset of samples was also checked through independent visual crossdating by another dendrochronologist. To identify the season of occurrence of fires (Baisan and Swetnam 1990), the relative position of each fire scar within the ring was recorded using the following categories: EE (early earlywood), ME (middle earlywood), LE (late earlywood), L (latewood), and D (dormant season, scar occurring between the cessation of latewood growth and the beginning of the next year's earlywood growth). Dormant scars are considered to represent spring fires in the Southwest (Baisan and Swetnam 1990).

Fire history was reconstructed from analysis of 473 fire scars from 51 ponderosa pine samples and 1 white fir sample. The majority of the samples were cut from dead trees, but samples from several live trees brought the fire history up to 1995.

Age structure

Tree increment cores were surfaced and crossdated (Stokes and Smiley 1968) with local tree-ring chronologies (D. A. Graybill; see *Dendrochronology–Live disturbance history*). Rings were counted on cores that could not be crossdated. The accuracy of ring counting was tested with a set of counted and crossdated presettlement cores, finding a mean error of 0.9 ± 2.1 yr (mean ± 1 SD). We concluded that grouping tree ages into 10-yr classes was reasonable. Additional years to the center were estimated with a pith locator (concentric circles matched to the curvature and density of the inner rings) for cores that missed the pith. Tree center dates are at the 40-cm sampling height; as few as 3 yr (Sackett 1984) or as many as 8–10 yr (Cormier 1990) may be required for ponderosa pine seedlings to reach 40 cm in the Flagstaff area.

Reconstruction of presettlement forest structure

The year 1883, in which the final fire of the presettlement frequent-fire regime occurred, was chosen as the date for reconstruction of presettlement forest structure. Reconstruction of forest structure at an earlier date would present greater difficulties because the frequent fires during this era probably consumed much rotten woody material.

The field determination of presettlement or postsettlement tree status was confirmed or rejected using the age data. Several large trees turned out to be of postsettlement origin and the data were adjusted accordingly. However, no trees field-classified as ‘‘postsettlement’’ turned out to be of presettlement age. For each cored presettlement tree, the 1883–1994 radial growth increment was measured on the core and the 1883 diameter calculated. Species-specific equations developed by Myers (1963) and Hann (1976) in the Southwest were used to estimate bark thickness and to predict dbh based on the inside-bark diameter of stumps. Site-specific regressions were developed from the data set to predict dbh from dsh for all species.

The year of death of presettlement snags and logs was estimated based on tree diameter and condition class. Dead trees were not crossdated in this study, but a companion study in which all dead material was crossdated at the Gus Pearson Natural Area, ≈15 km northeast of Camp Navajo, found that all dead wood classified in the field as ‘‘presettlement’’ based on size and decay was of presettlement origin (J. N. Mast and others, *unpublished data*). However, direct determination of death date was precluded by sapwood rot on almost all of those samples. Harvest dates for stumps were determined from timber harvest and forest inventory records of the Tusayan Forest Reserve (on file in the Supervisor’s Office, Kaibab National Forest, Arizona). The records were spatially detailed and corresponded well with field observations of the high stumps (up to 1 m) in the initially logged areas (1885) and lower stumps in areas harvested later (1925 and 1933).

Rates of snagfall and movement from class to class were summarized by Rogers et al. (1984) who combined data from Cunningham et al. (1980) and Avery et al. (1976) to calibrate for northern Arizona the tree decomposition model developed by Thomas et al. (1979). Because of high variability among substrates and environmental conditions, as well as the extremely long time span required for research, the dynamics of tree decomposition are poorly understood (Harmon et al. 1985) and mathematical models based on observed decomposition classes are likely to be highly imprecise. To assess the effect of such variability on the presettlement forest reconstruction, a sensitivity analysis was carried out by using three different decomposition rate percentiles, 25, 50, and 75%, to examine the effect of slower or faster decomposition on the estimates of death date and 1883 structure. For example, a 65-cm dbh snag in condition class 5 would be assigned a death date of 1950, assuming the 50th percentile of the distribution of decomposing trees. At the 25th and 75th percentiles, the assigned death dates would be 1906 and 1976, respectively. Presettlement forest structure based on each of the three percentiles was calculated to determine the relative effect of model imprecision.

To determine the 1883 diameter of dead trees, growth estimates for the period from 1883 to death date were subtracted from the measured diameter, adjusted for the loss of bark where appropriate. In the case of ponderosa pine, where an increment core data set adequate for modeling was obtained in the study, a site-specific predictive regression relationship between diameter and basal area increment was developed for presettlement trees ($r^2 = 0.70$). As is often the case, pine diameter proved to be an inadequate predictor of radial increment ($r^2 = 0.07$). For other species, published diameter-dependent growth regression equations were applied to determine 1883 diameter (Rogers et al. 1984).

A potential source of error in this reconstruction approach is missing presettlement-era woody material, which would lead to an underestimate of presettlement forest density. Fire consumption of wood was controlled as much as possible by selecting 1883, the year of the last widespread fire on the site, as the reconstruction year. Complete decomposition of heartwood or other resinous woody material in the absence of fire is likely to take longer than 112 yr (1883 to 1995) due to the very slow decomposition rate of ponderosa pine (Jenny et al. 1949, Hart et al. 1992), as evidenced by the many sampled and crossdated fire-scarred stumps, harvested ≈1885, collected in this study. Juniper and Douglas-fir wood is also highly resistant to decay. Gambel oak and white fir may decompose more quickly, often beginning with heart rot in the living trees (Barger and Ffolliott 1972, Burns and Honkala 1990), but old oak and fir stumps and logs were also found on sample plots. Some wood removal may have occurred in slash disposal, but this rarely affects the stumps and root crowns needed for presettlement reconstruction.

Because small-diameter pines rot quickly (Harmon et al. 1985, Arno et al. 1995*b*), it could be argued that small trees or even doghair thickets that were alive in the settlement year but died soon thereafter would be missed. However, there is no evidence in historical accounts, photographs, or early inventories for the existence of numerous small trees in presettlement northern Arizona—quite the contrary (Cooper 1960). Nor does it seem likely that such thickets could have become established under frequent fire regimes (White 1985). Finally, there is no reason to believe that such trees would have died in high numbers following fire exclusion, especially in light of the remarkable persistence of small trees in stressed postsettlement doghair thickets (Schubert 1974, Avery et al. 1976, Ronco et al. 1985, White 1985). In sum, while the possible underestimation of some small presettlement trees that left no trace cannot be dismissed, such trees are unlikely to have formed a substantial component of presettlement forest structure.

Statistical analysis

Statistical analysis of the fire history data was carried out with the FHX2 fire history analysis software developed by Grissino-Mayer (1995). The areal extent and intensity of presettlement fires are difficult or impossible to reconstruct in ecosystems characterized by frequent, low-intensity fire where forest stand boundaries do not correspond with fire edges (Swetnam and Dieterich 1985). However, relative fire extent and/or fire intensity within the sampled area can be inferred to some degree from the percentage of samples scarred and their spatial distribution. Therefore, fire return intervals were analyzed statistically in different subcategories. First, all fire years, even those represented by a single scar, were considered. Then only those fire years were included in which 10% or more, and 25% or more, respectively, of the recording samples were scarred.

Statistical comparison of the 1883 and contemporary forest structure was carried out with the SYSTAT software (Wilkinson 1988). Alpha level for all analyses was 0.05. Multivariate analysis of variance (Tabachnick and Fidell 1983:222–291) was used to determine differences in mean density, basal area, and quadratic mean diameter on the entire set of tree species. If a significant difference was found between the presettlement and contemporary forests (Wilks' lambda statistic), then Bonferroni-adjusted univariate *F* statistics were compared by species.

RESULTS

Fire disturbance history

Fire was frequent in the period 1637–1883, with a mean fire return interval (MFI) of 3.7 yr, but only a single fire scar, in 1951, was recorded after 1883 (Fig. 2). The year 1637 was selected as the initial year for fire history analysis because it was the earliest fire year deemed to have an adequate sample depth (5 recorder trees, ≈10% of the total sample size) to confidently evaluate fire occurrence (Grissino-Mayer et al. 1994).

The statistical analysis of fire return intervals in Table 1 includes the mean fire interval (average number of years between fires), the median, and the Weibull median probability interval (WMPI), used to model asymmetrical fire interval distributions and to express fire return intervals in probabilistic terms (Johnson 1992, Grissino-Mayer 1995). All three measures of central tendency over the 1637–1883 period were in close agreement and distributions were asymmetrical, positively skewed with a clumped distribution of short fire return intervals and fewer long intervals.

Spatial patterns of fire were inferred from the similarity of MFI values from both the all-scar and 10%-scar distributions, 3.7 and 4.6 yr, respectively, and identical median value of 4 yr for both (Table 1). The measures of central tendency in the 25%-scar distribution, possibly representing more widespread fires, are only ≈1.5 times greater than the all-scar and 10%-scar distributions (Table 1) and no significant difference in fire return intervals or percentage of scarring for the fire interval distributions between the north and south halves of the study area over the period 1684–1883. Fire years were not statistically independent in the north and south and, in all but the 25%-scar distribution, the number of synchronous dates was significantly greater than the number of asynchronous dates. Overall, these results support the inference of a spatially homogeneous fire regime, consistent with the lack of topographic or vegetative barriers to fire spread on the site and the fact that fires could have persisted for extended periods in the absence of fire suppression activities.

The presettlement fire regime was also temporally homogeneous. No statistically significant differences were found in fire interval means, variances, or distributions in the periods 1637–1759 and 1760–1883, although the fire history overlapped with the major regional climatic trend, the Little Ice Age, a period of warming beginning from a cold extreme in 1600 until 1850 (D'Arrigo and Jacoby 1991). The lack of significant long-term fire regime change appears to correspond with the short-term nature of climatic fluctuations within this period (Meko et al. 1995). On a shorter time scale, however, fire occurrence appeared to be linked to annual climatic factors, with fires occurring in the study area during some of the largest presettlement regional fire years in the Southwest, based on a

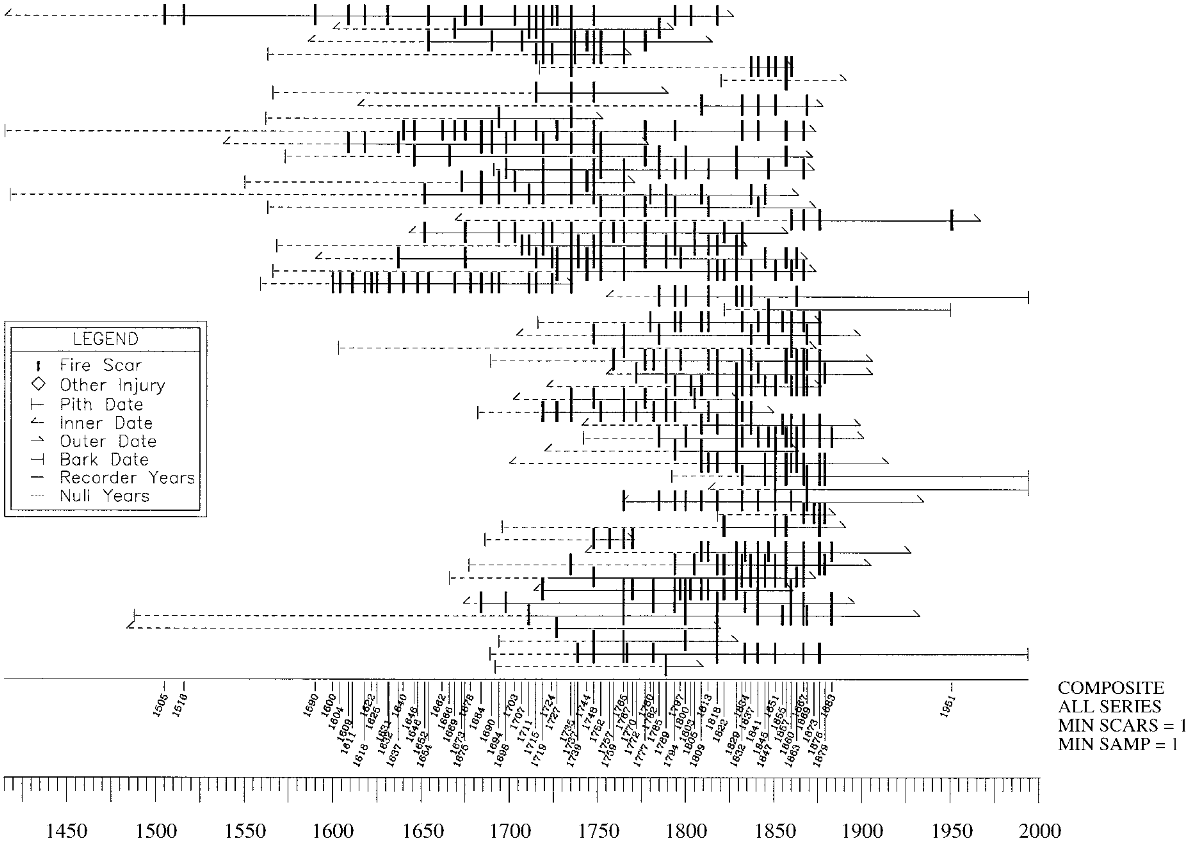

FIG. 2. Composite master fire chart for the Camp Navajo study area. Each horizontal line in the graph corresponds to a fire-scarred sample. Dashed lines show where the sample is not considered a recording tree (e.g., before the first fire scar forms), and continuous lines show recording periods. Fire dates are indicated by short vertical lines, and the dates are displayed along the lower axis. Frequent fires were interrupted after 1883, the year in which the transcontinental railroad was completed across the northern edge of the study area. The top to bottom order of samples runs roughly from north to south.

comparison of 63 southwestern fire chronologies for the period 1700–1900 (Swetnam and Baisan 1996). Fires burned at Camp Navajo in 8 of the 20 major regional fire years; in the top two regional years, 1748 and 1851, fires scarred 63 and 44%, respectively, of the recording sample trees. Climatic influences over broad geographic regions are considered responsible for this synchrony (Swetnam and Baisan 1996).

Seasonal data on fire occurrence (Table 2) showed that ≈40% of presettlement fires occurred in the spring (approximately late April to June [Baisan and Swetnam 1990]) and 60% in the summer (July to early September), without evidence of burning during latewood formation. This seasonal pattern is consistent with regional fire weather and lightning ignition patterns (Swetnam and Baisan 1996). Fire season could not be determined on ≈40% of the scars (Table 2), often due to narrow rings, possibly biasing the seasonal determination toward years in which better conditions for tree growth led to wider rings.

Changes in forest structure

Pine and oak density increased significantly and pine quadratic mean diameter decreased significantly from 1883 to 1994 (Table 3). The average density of ponderosa pine was 65.3 trees/ha with quadratic mean di-

TABLE 1. Fire return intervals (in years) over the period 1637 to 1883.

Percent of recording trees scarred	Number of intervals	MFI† (yr)	Median	Standard deviation	Min.	Max.	Skewness	Kurtosis	WMPI‡
All fires	66	3.7	4	1.42	2	8	0.94	0.92	3.7
Over 10% scarred	53	4.6	4	2.19	2	13	1.94	4.41	4.6
Over 25% scarred	37	6.5	6	3.45	2	15	0.76	−0.46	6.2

† MFI = Mean fire interval.
‡ WMPI = Weibull median probability interval.

TABLE 2. Seasonal distribution of fires based on the position of the fire injury within the ring. Total number of scars = 457.

	Season determined	Season undetermined	Dormant	Early earlywood	Middle earlywood	Late earlywood	Latewood	Dormant + early (= spring fires)	Middle + late (= summer fires)
Number	277	180	52	59	125	41	0	111	166
Percentage	60.6	39.4	18.8	21.3	45.1	14.8	0	40.1	59.9

ameter of 41.6 cm in 1883, as compared with 720.2 trees/ha and quadratic mean diameter of 22.8 cm in 1994; pine basal area increased from 11.7 to 18.2 m^2/ha. Thus the pine population has changed from fewer, larger trees to numerous, smaller trees (Fig. 3). Density and basal area of oaks both increased (Table 3). Other species had such low and variable occurrence that no statistically significant changes were observed, although all trended toward increasing density. Living presettlement-era trees comprise only 7% of contemporary tree density but make up 33.9% of contemporary basal area despite their sparse distribution.

New Mexican locusts found on the study area were small and young, so no estimation of presettlement locust density could be made and locust was excluded from the totals in Table 3. However, New Mexican locust does sprout readily following removal of overstory pines (Gottfried 1980) and dense locust thickets were observed surrounding old pine stumps, suggesting that locust density has increased since 1883.

The sensitivity analysis on modeling of tree decomposition showed that estimates of 1883 tree density were robust. Only under the slowest modeled decomposition conditions, the 25th percentile, were a few trees assigned death dates prior to 1883 (3 out of 54 dead presettlement trees, 5.6%) and no death dates prior to 1883 were calculated using the 50th and 75th percentiles, leaving the estimated 1883 forest density unchanged. Estimates of tree size in 1883 were also relatively robust. The 25th percentile model estimated the 1883 basal area at 13.7 m^2/ha, a 7% increase above the 50th percentile model, while the 75% percentile model estimated the 1883 basal area at 12.4 m^2/ha, a 4% decrease. The narrow range of differences in estimated 1883 structure, compared with the wide differences in decomposition rate percentiles and associated tree death dates, indicates that the 1883 reconstruction is relatively insensitive to decomposition model imprecision.

Distributions of size and age (Figs. 3 and 4) show

TABLE 3. Presettlement forest structure reconstruction in 1883 compared with contemporary forest structure from 1994/1995 inventory at Camp Navajo, Arizona. Presettlement New Mexico locust shrub structure could not be reconstructed so locust is not included in the totals (data enclosed in brackets).

Species	Presettlement 1883		Contemporary 1994	
	Mean	SEM	Mean	SEM
Trees/ha				
Ponderosa pine	65.3*	7.0	720.2*	129.0
Douglas-fir	0	⋯	3.6	2.1
White fir	0.8	0.8	62.5	42.3
Juniper species	2.4	1.7	7.7	3.4
Gambel oak	79.4*	13.6	471.4*	77.0
New Mexican locust			[489.9]	[178.2]
Total	148.0	14.3	1265.3	173.1
Basal area (m^2/ha)				
Ponderosa pine	11.7*	1.4	18.2*	1.6
Douglas-fir	0	⋯	0.2	0.1
White fir	0.2	0.2	1.0	0.8
Juniper species	0.02	0.01	0.2	0.2
Gambel oak	1.0*	0.2	6.6*	1.0
New Mexican locust			[0.1]	[0.1]
Total	12.9	1.4	26.2	1.7
Quadratic mean diameter (cm)				
Ponderosa pine	41.6*	2.7	22.8*	1.3
Douglas-fir	0	⋯	1.2	0.7
White fir	0.8	0.8	0.8	0.4
Juniper species	0.3	0.2	1.6	0.7
Gambel oak	6.7	1.0	10.6	1.2
New Mexican locust			[0.3]	[0.1]

Note: Sample size, $n = 62$. Within-row means followed by asterisks are significantly different ($P < 0.05$). SEM = standard error of the mean.

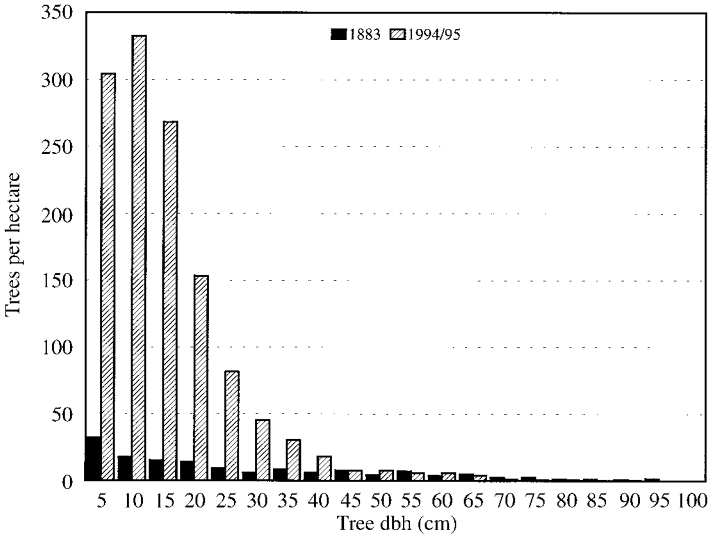

FIG. 3. 1883 and 1994/1995 diameter (dbh) distributions of all trees.

that the contemporary forest is dominated by relatively smaller, younger trees. Large pulses of tree establishment (Fig. 4) occurred in the late 19th century for oaks (1885–1914) and in the early 20th century for pines. Given 3–10 yr to reach the 40-cm sampling height, many of the pines in the largest center-date class (1935–1944) may have originated in the 1920s. Contemporary woody seedling density (Table 4) is highly variable but averages several thousand stems per hectare, mostly oak and locust.

Grasses were the dominant herbaceous understory plants in the contemporary forest, followed by composites, legumes, and sedges (Table 5). All identified species were native to northern Arizona. Percentage cover by herbaceous species averaged 19%.

Changes in tree and grass structure over the mid-twentieth century are illustrated in Fig. 5. The 1942 photograph, the earliest available scene from Camp Navajo, a few kilometers southeast of the study area, shows a very open forest structure with many stumps after railroad logging in the 1920s. Herbaceous plants were dense but numerous patches of pine seedlings are evident, corresponding with the large increase in pine establishment during this period (Fig. 4). By 1995 the same site was dominated by small-diameter pines with an apparent decrease in herbaceous density (Fig. 5, bottom).

Contemporary dead woody biomass was highly variable on the study site (Table 6) but forest floor depth was more consistent. In terms of potential fire behavior,

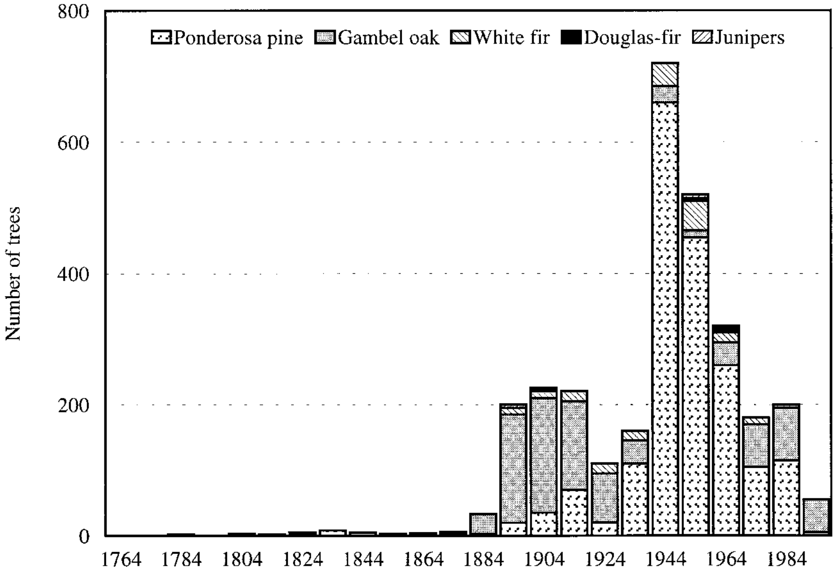

FIG. 4. Age distribution of living trees in 1994/1995 showing tree center date at the 40-cm aboveground sampling height (labeled dates are the endpoints of 10-yr classes).

TABLE 4. Regeneration density (stems/ha) by species and height class.

	Height class (cm)					
	0–40		40.1–80		80.1–137	
Species	Mean	SEM	Mean	SEM	Mean	SEM
Trees						
Ponderosa pine	66.1	21.7	62.9	32.1	45.2	27.9
Douglas-fir	11.3	9.8	1.6	1.6	0	
White fir	9.7	8.2	4.8	3.6	3.2	3.2
Juniper species	4.8	2.8	4.8	2.78	1.6	1.6
Gambel oak	3966.1	1149.8	582.3	197.6	80.7	33.6
Shrubs						
New Mexican locust	714.5	236.1	1085.5	357.7	517.7	179.2
Serviceberry	4.8	4.8	1.6	1.6	0	···
Total	4777.4	1204.59	1743.6	427.1	648.4	195.2

Note: Sample size, $n = 62$, 100-m^2 plots. SEM = standard error of the mean.

these fuel data imply that continuous fuels to carry the flaming front are found throughout the study area, while heavy fuels that may support intense fire behavior occur in scattered patches.

DISCUSSION

Context of changes in fire disturbance regime and forest structure

The beginning of effective fire exclusion in the study area after 1883 is coincident with the construction of the transcontinental railroad across northern Arizona from 1882 through 1884 and the initiation of heavy commercial logging in the region (Cline 1994). The date is also consistent with the cessation of frequent fires throughout most of the Southwest by 1900, with a substantial decline in fire occurrence beginning in the 1870s and 1880s (Swetnam 1990, Swetnam and Baisan 1996). Heavy livestock grazing was associated with fire exclusion at Chimney Spring, ≈15 km east of the study area (Dieterich 1980). The date of initiation of livestock grazing at Camp Navajo is not known, but nearby Rogers Lake was impounded to water livestock in 1878 (Grenda 1993).

Comparing presettlement forest structure to that of nearby areas, presettlement pine and oak densities on the study area were greater than the 43 pines/ha and 4 oaks/ha found by Covington and Moore (1994*a*) at Bar-M Canyon, but pine density was close to the 61 pines/ha at the Gus Pearson Natural Area (Covington et al., *in press*). Presettlement pine density at the study area also falls within the range of 7 to 116 pines/ha reported in several early National Forest inventories and other studies in the region (see Covington and Moore [1994*b*] for comparison). In particular, Woolsey (1911) reported a mean of 26 pines >15.2 cm dbh per hectare on ‘‘average’’ stands in timber-sale areas on the Tusayan National Forest (which then included the study area) and a mean density of 85 pines >10.2 cm dbh per hectare on ‘‘maximum’’ stands in the adjacent Coconino National Forest. The 1883 Camp Navajo reconstruction is within this range with 61 pines/ha >15.2 cm dbh and 63 pines/ha >10.2 cm dbh. There is no evidence in the reconstructed forest structure data that dense stands existed in the study area: the *maximum* presettlement tree density was 500 trees/ha, <40% of the *mean* contemporary tree density of 1265 trees/ha (Table 3).

Relatively high densities of small-diameter oaks were observed in the 1883 forest, comprising 94% of all trees under 10 cm dbh. Because Gambel oak is a prolific sprouter but highly susceptible to fire, es-

TABLE 5. Frequency of the 20 most encountered herbaceous species; frequencies of all other herbaceous species were <1%. The mean herbaceous cover for 59 plots was 18.8%, with a range of 0–78%, and standard error of the mean of 2.5%.

Species	Frequency (%)
Grasses	
Sitanion hystrix	17.1
Muhlenbergia montana	16.6
Festuca arizonica	14.5
Bouteloua gracilis	4.4
Poa fendleriana	2.8
Aristida sp.	1.3
Composites	
Cirsium sp.	7.8
Unknown composite sp.	2.8
Erigeron sp.	2.3
Senecio sp.	1.8
Achillea lanulosa	1.6
Lactuca sp.	1.6
Erigeron flagellaris	1
Antennaria sp.	0.8
Legumes	
Vicia sp.	3.1
Astragalus sp.	1.8
Lathyrus arizonicus	1
Sedge	
Carex sp.	3.9
Other	
Unknown forb sp.	3.1
Ceanothus fendleri	1

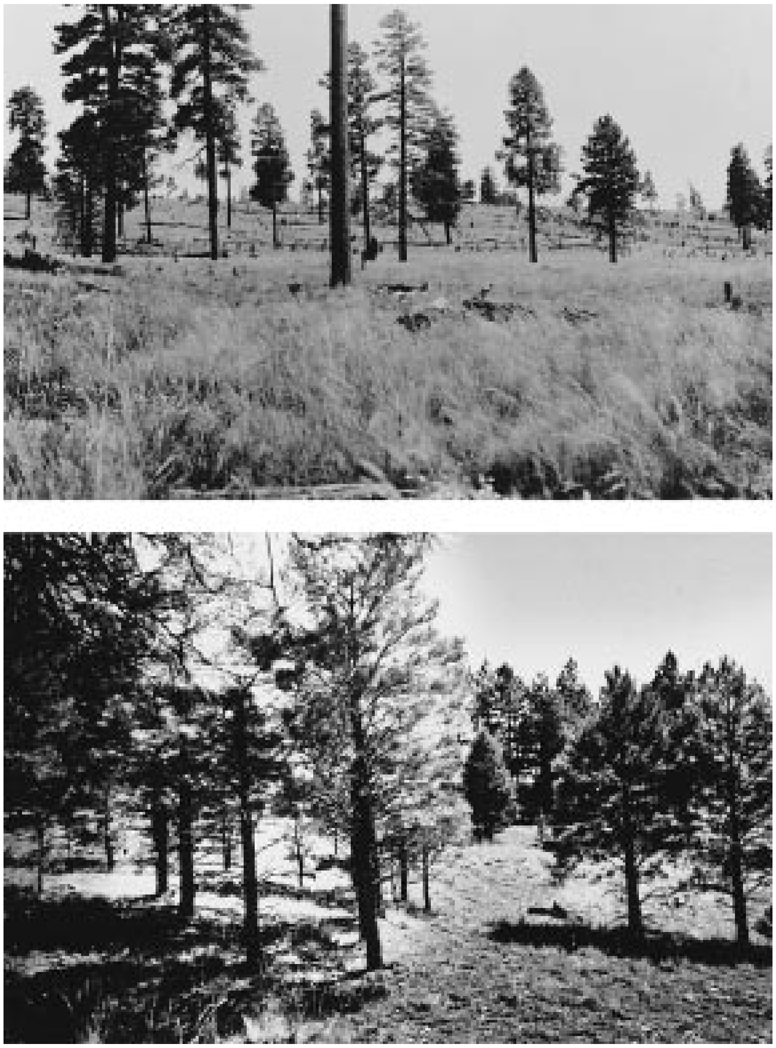

FIG. 5. Top: 1 September 1942, forest scene 3.5 km south of the study area (center of township 21 north, range 5 east, section 20, view east). This scene is typical of many in the 1942 photo archives: a very open residual ponderosa pine stand with numerous scattered stumps and logs from railroad logging in the 1920s. In the background, especially at right center, dense patches of small seedlings can be seen. The full crowns of the residual trees suggest that the stand was probably not very dense even prior to logging. (Navajo Army Depot Collection, Cline Library, Northern Arizona University). Bottom: 15 August 1995, scene from approximately the same location. A dense, sapling- to pole-sized ponderosa pine stand dominates the area. Grassy openings remain, but herbaceous density appears to be substantially less than in 1942.

TABLE 6. Dead woody biomass and forest floor depth.

	Mean	Minimum	Maximum	SEM
Woody fuels (Mg/ha)				
0.0–0.6 cm diameter	0.19	0	2.18	0.04
0.6–2.5 cm diameter	0.95	0	5.57	0.14
2.5–7.6 cm diameter	2.16	0	16.87	0.42
>7.6 cm diameter sound	3.57	0	114.67	1.92
>7.6 cm diameter rotten	7.89	0	99.05	2.44
Total	14.76	0	214.27	3.88
Forest floor depth (cm)				
Litter	1.48	0.31	3.18	0.09
Duff	2.59	0.16	8.65	0.24
Total	4.06	0.48	11.11	0.30

Note: Sample size, $n = 62$, 15-m planar intersect transects. SEM = standard error of the mean.

pecially small-diameter trees (Harrington 1985), it is likely that many or most of these small trees would have been thinned had fires continued after 1883. The density of oaks >5 cm dbh, 48 oaks/ha, or the density over 10 cm dbh, 33 oaks/ha, may be more representative of the long-term average oak density under a continuing frequent fire regime.

The increase in density of all tree species at the study area by 1994 is consistent with the structural changes in ponderosa pine ecosystems since Euro-American settlement in Arizona (Cooper 1960, White 1985, Covington and Moore 1994*a*, *b*) and in related long-needled pine ecosystems throughout western North America (Covington et al. 1994, Arno et al. 1995*b*). The contemporary density of ponderosa pine is very close to the 1985 Arizona average of 776 trees/ha (Garrett et al. 1990), but less than the 1900 ponderosa pines/ha reported by Covington and Moore (1994*a*) at Bar-M Canyon, and well below the 3098 pines/ha at the Gus Pearson Natural Area where logging, thinning, and firewood cutting have been excluded (Covington et al., *in press*). Contemporary oak density (Table 3) is over three times greater than the 156 oaks/ha at Bar-M Canyon (Covington and Moore 1994*a*) and higher than oak densities averaging 138–182 oaks/ha in a central Arizona ponderosa forest (Barger and Ffolliott 1972). Juniper density at the study area is lower than the 27 junipers/ha at Bar-M Canyon (Covington and Moore 1994*a*) and lower than the average of 88 junipers/ha in central Arizona (Barger and Ffolliott 1972).

In the northern part of the study area, adjacent to the railroad, the impact of early logging may have been a factor in limiting pine establishment. On the 15 northern plots, presettlement pine stump density averaged 33 stumps/ha, while living presettlement trees averaged 3 trees/ha. The heavy pine removal at the end of the 19th century may have eliminated seed sources and precluded the regeneration flushes that followed good regional seed years such as 1919 (Schubert 1974). However, the period of favorable moisture conditions in the Southwest extended from 1905 to 1928 (D'Arrigo and Jacoby 1991). Later logging in the southern part of the study area (≈1933), could have damaged seedlings, contributing to the relatively late pine regeneration pulse (Fig. 4). These effects of pine logging may explain the relatively slow increase in ponderosa pine basal area from 1883 to 1994 despite the manyfold increase in tree density as well as the relatively high proportion of oak in the contemporary forest.

Ecosystem management

1. *Comparison to contemporary conditions.*—The reference presettlement conditions can be applied to management of this ecosystem as (1) a benchmark against which to evaluate contemporary conditions and future alternatives (Kaufmann et al. 1994, Swanson et al. 1994), and (2) a goal for ecological restoration treatments (Covington et al., *in press*). Taking the 1883 forest structure as the reference condition, the contemporary structure has tree densities far higher than the mean and increased variability in both absolute and relative terms (Table 3). The mean 1994 density of 720.2 pines/ha is well outside even the 99th percentile of the 1883 pine distribution (1883 mean + 3 standard deviations = 231 pines/ha) and pine densities on the sample plots in 1883 ranged from 0 to 250 pines/ha, as opposed to the broader range of 0 to 6250 pines/ha in 1994.

Mesic conifer density has increased. Although some presettlement-era mixed conifer forest existed on the north-facing slopes of Volunteer Mountain, no presettlement Douglas-fir and only two presettlement white firs were encountered on the sampling plots. By 1994 these species made up 5% of the total tree density and dominated the north-facing slopes, with white fir reaching a maximum density of 2225 firs/ha.

Contemporary forest floor fuel loading is probably higher than in presettlement times. With fires occurring every 3–6 yr in presettlement times, small-diameter woody fuels would have been consumed quickly. Larger diameter rotten woody fuels would have been consumed as well (Covington and Sackett 1984). In contrast to presettlement conditions, currently there is ample fuel to support high-intensity fire behavior, including torching through live fuel ladders and crown fire, in hot, dry, windy weather. However, herbaceous fuel loading is probably greatly reduced in the contemporary forest.

Presettlement regeneration patterns may be inferred to some extent from the age structure of living presettlement trees (Fig. 4). White (1985) argued that presettlement pine regeneration would most likely have required ''safe sites,'' such as in the ash bed left by a completely consumed log, until seedlings grew above the lethal scorch height of a fast-moving surface fire. Ponderosa pine regeneration is also linked to sporadic periods of favorable climatic conditions, even if favorable establishment sites exist (Savage and Swetnam 1990). Under such limitations, the relatively normal distribution of presettlement pine diameters may be explained by the few establishment opportunities for seedlings, but the high survivorship among young trees, which grew above the lethal effect of the flames and benefited from uncrowded growing conditions (Fig. 3).

Postsettlement woody regeneration patterns (Table 4) are likely to differ from presettlement patterns on the study area, because herbaceous competition has been reduced or eliminated, fire has not played a role in thinning, harvesting has preferentially removed pines, and the relatively dense canopy may favor shade-tolerant species.

2. *Ecological restoration.*—Although restoration to conditions resembling those of presettlement times is not necessarily a feasible or desirable management goal on all forest lands (Kaufmann et al. 1995), managers

of the Camp Navajo study area have agreed that an ecosystem management approach based on restoration of ecosystem function and structure similar to that characteristic of recent evolutionary history (Covington et al. 1994) is consistent with the training uses and buffer zone function of this site. Detailed description of ecological restoration in southwestern ponderosa pine forests requires an extended treatment (Covington et al., *in press*; W. W. Covington, *unpublished manuscript*) but key points are summarized here.

The challenge for management is to conserve existing presettlement-era resources, design treatments that restore ecosystem structure, and permit the reintroduction of frequent, low-intensity fires. These are not simple tasks. Even conservation of living presettlement trees presents problems: growth rates of old ponderosa pine trees at the Gus Pearson Natural Area have slowed sharply, despite the exclusion of logging and grazing and without evidence of changing climate (Sutherland 1983, Kolb et al. 1994, Biondi et al. 1994, *in press*). These authors suggested that a primary cause of mortality of old trees is increased stress from competition with dense thickets of younger trees following fire exclusion. Reintroduction of fire without fuel treatment can also kill old-growth trees. Experimental prescribed fires beginning in 1976 at Chimney Spring, ≈15 km northeast of Camp Navajo, led to mortality of >35% of the living presettlement pines, attributed to the lethal temperatures produced at the root collars of old trees by slow smoldering consumption of accumulated fuels (Harrington and Sackett 1990).

The underlying premise of restoration ecology is that ecosystems function best under the conditions to which they adapted over evolutionary time (Swanson et al 1994). In the case of southwestern ponderosa pine, the open, park-like forests, productive herbaceous understory, and frequent fire regime are key characteristics of this long-term evolutionary environment. Applying the reference condition data as a guide, we suggest that restoration treatments (Covington et al., *in press*) should include (1) establishment of permanent monitoring plots for evaluation and modification of treatments in an ''adaptive ecosystem management'' framework (Walters and Holling 1990); (2) restoration of overstory structure by conserving living presettlement trees and replacing dead presettlement trees; (3) protection of old trees from excessive cambial heating by reducing fuels around the boles; (4) reintroduction of fire disturbance through prescribed burning; and (5) restoration of the herbaceous and shrub understory through seeding or planting of native species where natural regeneration is inadequate.

We expect the balance of fire and vegetation in the restored ecosystem to ultimately return to the self-regulating patterns and rhythms of the presettlement forest because the keystone natural mechanisms controlling structure and function will be able to operate. However, while extensive information exists on the small-scale (plot-level) response of woody and herbaceous plants to thinning and burning, the effectiveness of an integrated large-scale (landscape-level) restoration treatment in ponderosa pine forests has only begun to be examined (Covington and Moore 1994*a*, Arno et al. 1995*a*, Covington et al., *in press*). Continued monitoring of these treatments is essential and on-going management intervention and adaptive new approaches will be necessary over time as we evaluate and learn from our first attempts at restoration and ecosystem management of landscapes.

ACKNOWLEDGMENTS

This project was supported by a grant from the Arizona Army National Guard. The authors thank SFC Donald Hack, Jr., CPT John Morrow, and CPT Elizabeth Gilman, of the Arizona Army National Guard. Field and laboratory work were carried out with the help of Tom Heinlein, Joy Nystrom Mast, John Paul Roccaforte, Judy Springer, and Amy Martin Waltz. Forest management history was provided by Teri Cleeland of the Kaibab National Forest. At Northern Arizona University, we thank the staff of the Deaver Herbarium for assistance in identifying plant specimens and the Special Collections and Archives staff of the Cline Library, particularly Bruce Hooper, for help with the Navajo Army Depot collection. Tom Kolb, Tom Whitham, Tom Swetnam, and an anonymous reviewer provided helpful comments.

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