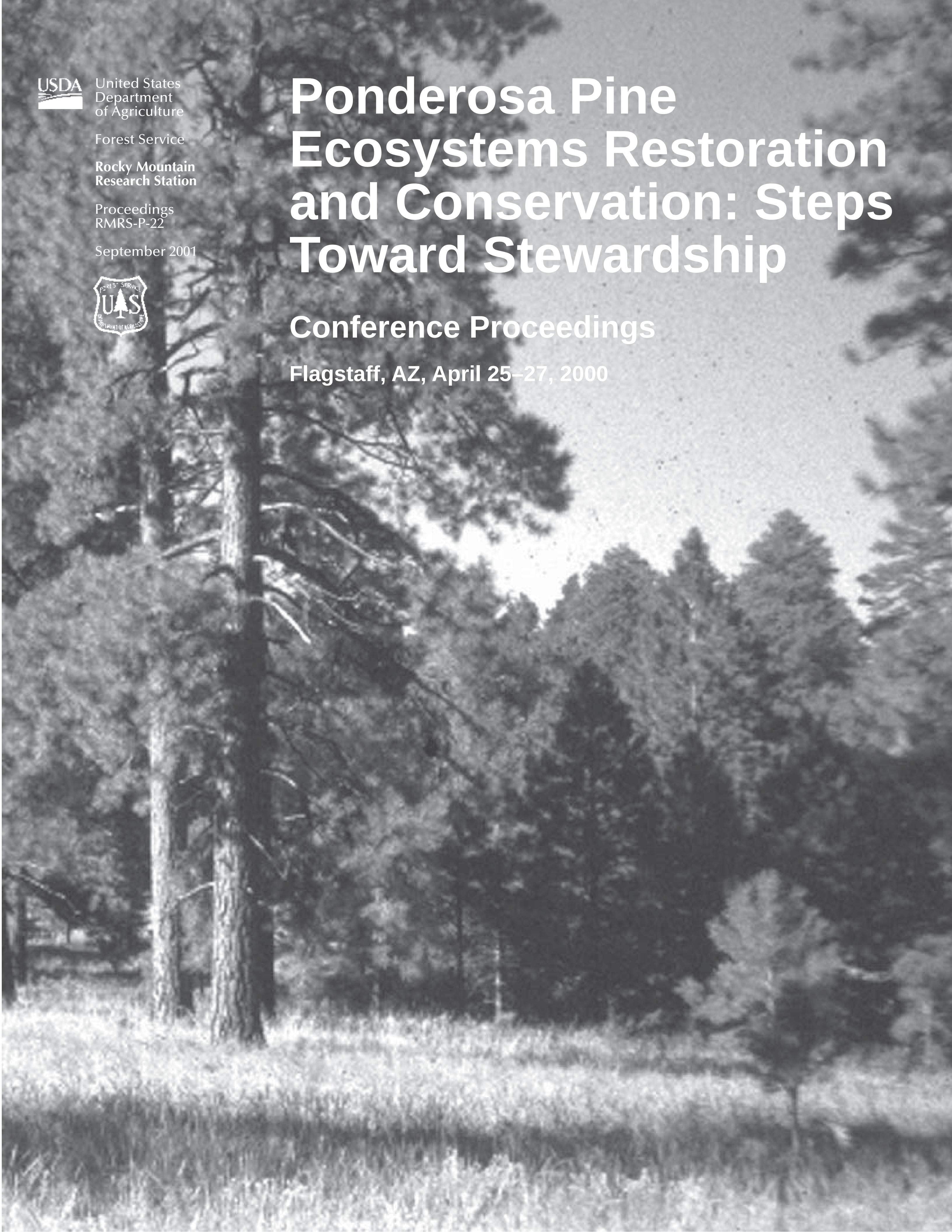

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Ponderosa Pine Ecosystems Restoration and Conservation: Steps Toward Stewardship

Conference Proceedings

Flagstaff, AZ, April 25–27, 2000

Abstract

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This volume is divided into three sections: (1) Ecological, Biological, and Physical Science; (2) Social and Cultural; and (3) Economics and Utilization. Effective ecological restoration requires a combination of science and management. The authors of the first section exemplified this integration in the course of addressing a broad range of topics, from detailed microsite and small-scale changes in fungal, plant, and animal communities, up through landscape, regional, and subcontinental scales. Although the themes were diverse, papers were linked by underscoring the relationship between restorative management actions and ecological effects. Social sciences play a key role in ecosystem restoration because collaboration, development of common goals, and political and economic feasibility are essential for success. The authors of the second section focused on public attitudes, partnerships, and the relationship between social and ecological factors. In the third section, the economics and utilization of products from forest restoration were compared in several Western locations. Both the markets for these products and the range of utilization opportunities—from small-diameter logs to energy creation—will surely evolve rapidly as society moves to address the fire hazards and other problems caused by stressed and weakened ecosystems. The turn of the century is an appropriate point to capture dramatic changes in perspective: consider how attitudes toward Western forests have evolved between 1900 and 2000. The papers in this volume chronicle adaptive research that continues to deepen our understanding of restoration in ecosystems and social systems.

Keywords: ponderosa pine, ecosystem management, landscape management, restoration, conservation, fire behavior, cost effectiveness analysis

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Ponderosa Pine Ecosystems Restoration and Conservation: Steps Toward Stewardship

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Flagstaff, AZ, April 25–27, 2000

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Preface

Contemporary ponderosa pine forests and associated grasslands in the Southwest have changed dramatically since Euro-American settlement in the 1870s. Intensive grazing, selective harvesting of large trees and fire suppression have led to changes in forest structure and composition that are unprecedented in the evolutionary history of these frequent fire ecosystems. The consequences of the changes include: increased risk of stand-replacing crownfire, decreased biological diversity, increased vulnerability to disease and insect outbreaks, and increasing ecosystem health problems that will compromise the long-term viability of ponderosa pine forests.

Local, State, Federal, and Tribal land management agencies have begun to address the problems of degraded ponderosa pine forests throughout the Intermountain West. Stakeholders from communities at risk of catastrophic fire and private landowners have also stepped forward to work collaboratively with land management agencies to design forest restoration treatments that will not only protect lives and property but also protect and restore the values and benefits provided by a healthy forest.

However, crafting scientifically valid and socially acceptable treatments is not simple. It requires commitment and constructive dialogue among people with diverse backgrounds and interests. They must be willing to listen to each other and become informed about ecology, economics, and management challenges, as well as the philosophical and social sides of forest restoration. Most important, it requires those people to build a common vision for forest restoration that can be translated into specific actions and applied by land managers. Given the scope of forest degradation and the need to act quickly it is essential that the information and lessons learned actively shared, applied, and adapted to implement the best treatments possible.

The **Steps Toward Stewardship: Ponderosa Pine Ecosystems Restoration and Conservation** conference, held in Flagstaff, AZ on April 25-27, 2000, was designed to share lessons and emerging research and information critical to successful ecologically based forest restoration. A diverse group of organizations representing a broad spectrum of expertise and interests sponsored this first national conference. The audience included researchers, academics, land managers, citizens, policy makers and other interested stakeholders from across the nation. Presenters were encouraged to identify critical indicators and benchmarks of success or failure in ponderosa pine ecosystem restoration and conservation, and the methods for evaluating these indicators. In addition, panels were designed to clarify points of agreement and disagreement across interests and disciplines so that future conservation/restoration research and experiments can advance understanding of these issues. Finally, participants were encouraged to identify missing elements—such as research, monitoring, or other factors—that are critical to develop effective conservation and restoration practices for ponderosa pine forest ecosystems.

Contents

Ecological, Biological, and Physical Science

Social and Cultural

Economics and Utilization

List of Conference Participants

Ecological, Biological, and Physical Science

Ponderosa Pine Forest Reconstruction: Comparisons With Historical Data

David W. Huffman
Margaret M. Moore
W. Wallace Covington
Joseph E. Crouse
Peter Z. Fulé

Abstract—Dendroecological forest reconstruction techniques are used to estimate presettlement structure of northern Arizona ponderosa pine forests. To test the accuracy of these techniques, we remeasured 10 of the oldest forest plots in Arizona, a subset of 51 historical plots established throughout the region from 1909 to 1913, and compared reconstruction outputs to historical data collected. Results of this analysis revealed several distinct sources of error: (1) After about 90 years, 94 percent of the recorded trees were relocated and remeasured, but approximately three trees/ha were missing in the field due to obliteration by fire or decay; (2) sizes of trees living in 1909 were overestimated by an average of 11.9 percent; (3) snag and log decomposition models tended to underestimate time since tree death by an undetermined amount; and (4) historical sizes of cut trees were difficult to estimate due to uncertainties concerning harvest dates. The aggregate effect of these errors was to overestimate the number of trees occurring in 1909–1913. Sensitivity analysis applied to decomposition equations showed variations in reconstructed sizes of snags and logs by ± 7 percent and stand density estimates by 7 percent. Results suggest that these reconstruction techniques are robust but tend to overestimate tree size and forest density.

Introduction

Ponderosa pine (*Pinus ponderosa* Douglas ex P. Lawson & Lawson) forest ecosystems in northern Arizona have undergone dramatic physiognomic changes over the past 120 years (Covington and Moore 1994a; Covington and others 1997; Fulé and others 1997). Logging, fire suppression, and overgrazing in the latter part of the 19th century created conditions suitable for a population explosion of pine regeneration. Open parklike stands, maintained by frequent surface fires prior to Euro-American settlement (about 1876) of the region, have been replaced by closed canopied forests with resulting deleterious effects on biological diversity and ecological function (Covington and Moore 1994a). Associated biomass accumulation and the development of fuel ladders represent extreme fire hazards and expose these forests to an increased potential for stand-replacing crown fires. Restoration of ecological processes and structure holds promise for reestablishing indigenous levels of biological diversity and ecological function in northern Arizona ponderosa pine forests (Covington and Moore 1994b; Covington and others 1997).

Treatments designed to restore ponderosa pine ecosystems are based on an understanding of presettlement structural and compositional characteristics that are collectively known as *reference conditions*. Forest reconstruction is one tool used to estimate reference conditions. Techniques for reconstruction include dendrochronological measurement of fire scars and increment cores from stumps, logs, and living trees, direct measurement of remnant woody evidence, and backwards radial growth modeling (Fulé and others 1997). Forest structural information generated by reconstruction includes past diameter distribution and stand density estimates. The precision of these analyses is highly dependent on field identification of presettlement evidence, dendrochronological proficiency, and relationships utilized in "reverse" growth and decay modeling. Our objective in this study was to use data from historically measured forest plots to test the precision of our forest reconstruction techniques. Specifically, we wanted to (1) test our ability to identify historically measured trees in the field, (2) compare reconstruction model outputs such as stand density and tree sizes to historical data, and (3) identify key sources of error associated with the reconstruction process.

Background

Between 1909 and 1913, a series of 51 permanent plots were established within the ponderosa vegetation type throughout Arizona and New Mexico (Woolsey 1911, 1912). The purpose of these plots was to increase understanding of western yellow pine (now ponderosa) growth, regeneration, and management. These are the oldest known ponderosa pine sample plots in the Southwest. Plots of around 1 to 6 ha (2 to 14 acres) were established on areas where up to two-thirds of the standing overstory volume had recently been harvested. The permanently marked plots were subdivided into 20 m (66 ft) grids wherein all trees greater than 10.16 cm (4 inches) in diameter at breast height (d.b.h.) (1.4 m) were mapped. Measurements for these trees included d.b.h.,

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height, vigor, and age class. The USDA Forest Service remeasured these plots every 5 years until the studies were abandoned around 1934–1939. In the mid-1990s, we began to uncover the historical data and maps associated with the "Woolsey" plots. In 1996, we initiated a project to relocate and remeasure as many of the historic plots as possible. By 2000, we had remeasured and applied reconstruction analysis to 15 plots. This paper describes the results of analysis of 10 plots located near Flagstaff, Arizona.

Methods

Study Area

The 10 historical plots used in this study are within a 24-km radius of Flagstaff (35°8'N latitude, 111°40'W longitude), Arizona, on the Coconino National Forest (fig. 1). Elevation at the sites ranged from around 2,100 to 2,200 m. Average annual precipitation in the area is 50.3 cm with about half falling in winter as snow and the other as rain associated with a mid-summer monsoon pattern. Mean annual temperature is 7.5 °C. The soil of the area is a stony clay loam of basalt derivation.

Ponderosa pine is the dominant overstory species of the area often occurring in pure stands or mixed with Gambel oak (*Quercus gambelii* Nutt.). Important understory species include grasses, *Festuca arizonica* Vasey, *Muhlenbergia montana* (Nutt.) Hitchc., *Blepharoneuron tricholepis* (Torr.) Nash, and *Sitanion hystrix* (Nutt.) J.G. Smith, and forbs, *Achillea millefolium* var. *occidentalis* D.C., *Pseudocymopterus montanus* (Gray) Coult. & Rose, *Erigeron divergens* Torr & Gray, and *Potentilla crinita* Gray. Shrubs are not common but include scattered populations of *Ceanothus fendleri* Gray and *Rosa woodsii* Lindl.

Remeasurement of Historical Plots

The 10 plots, originally established in 1909 (eight plots) and 1913 (two plots) were remeasured in 1997–1999. We used copies of the stem maps drawn in 1915 to relocate plot corners and grid intersections. Although the historical plots ranged in size, we standardized our methods to remeasure subplots of 1.02 ha (2.5 acres), systematically originating from the northwest corners of the original plots. After consulting historical maps to determine plot orientation, we used a transit, staff compass, and tape to reestablish the original grid system as truly as possible. Subplots consisted of 25 grid cells of approximately 404 m^2 each. After subplots were established, 1915 maps were not referenced again until after remeasurement had been completed. Due to the size of the subplots, results presented here are interpreted in terms of trees per hectare.

Grid cells within subplots were thoroughly searched and all structures equal to or greater than 1.4 m in height, either presently or at some past time, were numbered and tagged. Tree structures included live trees, snags, logs, stumps, and stump holes. Diameter at breast height (d.b.h.; measured at 1.4 m above the ground) and/or diameter at stump height (d.s.h.; measured at 40 cm), total tree height, condition class (1–9), and age class (presettlement, "preplot", or postsettlement; see below) were recorded for all tree structures. For stump holes, d.s.h. was estimated. Tree condition classes followed a classification system commonly used in ponderosa pine forests (Maser and others 1979; Thomas and others 1979). The nine classes were as follows: (1) live, (2) fading, (3) recently dead, (4) loose bark snag, (5) clean snag, (6) snag broken above breast height, (7) snag broken below breast height, (8) dead and down, and (9) cut stump.

In the field, age classifications were based on tree size and bark characteristics and then verified when possible in the laboratory. Tree ages were grouped in the field into three categories: presettlement, preplot, and postplot. Structures greater than 37.5 cm d.b.h., clearly yellow-barked, or dead, large, and highly decayed, were presumed to be greater than 100 years old (White 1985) and classified as presettlement in age. We classified trees as "preplot-aged" (in other words, established prior to historical plot measurement) if they did not meet presettlement classification criteria (for example, did not have yellow bark) but were larger than a predetermined size. Preplot size was determined for individual subplots by measuring diameter and field aging a sample (10) of nearby trees. Ponderosa pine with bark appearing transitional in age between yellow barked and black barked trees were located just outside the subplot boundaries and cored at 40 cm. Ring counts were made and the results were compared with the plot establishment date. The average diameter of trees within 10 years of the plot establishment date was used as the preplot size cutoff. Live trees of this size or larger without yellow bark were classified as preplot-aged. In some cases, preplot size was not different than presettlement (37.5 cm at d.b.h.) and classification was made based on bark characteristics. Trees with black bark and smaller than the minimum presettlement and preplot diameters were classified as postplot.

Diameter at stump height and crown radius (average of two measurements) was measured for all presettlement and preplot ponderosa pine trees. For species other than ponderosa pine, d.s.h. was measured on all trees. Increment cores were collected at 40 cm for all trees greater than 37.5 cm d.b.h. and for oak and juniper species greater than 17 cm. Additionally, increment cores were collected and d.s.h. and

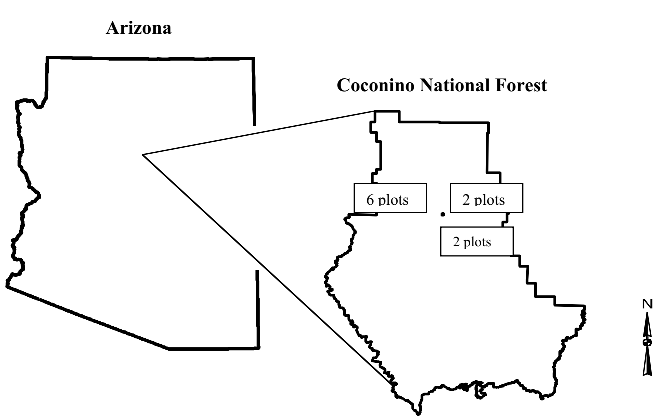

Figure 1—Study area showing general location of historical plots. Shown are the number of subplots remeasured in each location. Coconino National Forest boundary is outlined and Flagstaff, AZ, is shown as the point in the center.

crown radius was measured for a 20 percent random subsample of live postplot (all species) trees. Increment cores were stored in paper straws until denrochronological analysis could be done in the laboratory. Dendrochronological analysis involved cross-dating cores (Stokes and Smiley 1996) against known annual ring patterns (Graybill 1987) and measuring radial increments from the year prior to core collection (1997–1999) to the year of plot establishment (1909–1913). Radial increments to fire exclusion date (1876; Fulé and others 1997) were also measured. Our age classification scheme was designed to assure that detailed data, including increment cores, were collected for all trees that were historically measured or presettlement in age. It also allowed comparisons to be made regarding changes in age structure on these plots since establishment.

Analyses

Field Identification of Historical Trees—Historical trees were identified in the field by noting tree locations displayed on 1915 maps and examining structures measured on subplots. Historical trees not measured on subplots (in other words, missed during remeasurement) were noted and the error rate was calculated as follows: Error = (number missed/number on historical map) * 100. A weighted average for error incorporating all 10 plots was calculated as follows: Error = (total number missed over all subplots/total number of historical trees over all subplots) * 100.

Reconstruction Modeling—Field measurements and increment core data were entered into a computerized stand reconstruction model (Covington and others, unpublished). The model applies a series of mathematical functions to field data in order to estimate tree diameters (d.b.h.) and death dates for a particular point in time that is defined by the user. Growth functions employed were gleaned from empirical growth (Myers 1963) and decay studies (Rogers and others 1984) of Southwestern ponderosa pine as well as other species. Sizes of live trees for which increment cores were collected were reconstructed by subtracting twice the radial increment from field recorded tree diameters. Sizes of trees for which no cores were collected or for which core data were unusable were estimated by applying "reverse" growth functions. Dead trees were moved backward through decay classes (Maser and others 1979; Thomas and others 1979) until an estimated death date was reached, before which the reverse growth function was applied. Separate equations were used for blackjack and yellow pine age classes. For cut trees (stumps) measured in the field, trees were grown in reverse prior to a cut date that we defined in the model.

For our analyses, we compared stand density and tree diameter output from the reconstruction model with data available from the historical ledgers. Because data in historical ledgers was restricted to trees greater than or equal to 10.16 cm d.b.h., we limited our analysis of stand density to trees of this size. To test reconstruction sensitivity, we varied cut date and decay rate parameters in the model. Cut dates of 1910 (1914 for the two plots established in 1913), 1945, and 1980 were tested for field-measured stumps, and dead/down trees were moved backward through decay classes at rates corresponding to the 25th, 50th, and 75th percentiles found in other studies (Roger and others 1984). Thus, five model iterations were done for each subplot reconstructed to historical establishment dates. Size errors were calculated as follows: Error = (Reconstructed d.b.h. - Historical d.b.h./ Historical d.b.h.) * 100.

Results

Field Identification of Historical Trees

Nearly all the trees mapped and recorded in 1909 and 1913 were found on the 10 subplots (table 1). The number of trees missed ranged from 0 to 9 and the overall error rate was 5.7 percent. Historical trees were found in all condition classes from highly decomposed remnant structures to live trees still bearing 90-year-old tags. Frequently, missed trees were highly decomposed and little evidence was observable. A high rate of identification was possible although dense stand conditions existed at the time of subplot remeasurement (1997–1999); there was an average of 1,379 total structures across the 10 subplots. No clear relationship was observed between the number of trees missed and the number that were historically measured. However, the greatest number of trees were missed on the subplot with the greatest total number of structures at remeasurement.

Reconstruction Model

Historical Tree Diameter—Diameter reconstruction of historical trees that were still alive at remeasurement overestimated d.b.h. by an average of 11.9 percent (table 2). We found a slight trend of increased error for smaller trees (fig. 2). There were no subplots on which tree diameter was underestimated. Unexpectedly large errors (greater than 60 percent) resulted for diameter reconstruction of unusually fast- or slow-growing trees for which increment data were unavailable (rotten tree centers, incomplete cores, and so forth).

Table 1—Number of live trees on Woolsey subplots in Arizona at establishment (1909–1913) and the number missed during remeasurement (1997–1999).

Subplot	No. trees on subplot[a]	No. trees missed	Error
			percent
S1A	26	1	3.8
S1B	25	2	8.0
S2A	82	5	6.1
S2B	72	2	2.8
S3A	47	9	19.1
S3B	58	2	3.4
S4A	87	8	9.2
S4B	61	0	0.0
S5A	20	1	5.0
S5B	83	2	1.2
Average[b]			5.7

[a]Number of historical trees on subplot recorded on 1915 maps.
[b]Overall average = (S No. trees missed/S No. Trees on subplot) * 100.

Table 2—Errors for reconstruction estimates of historical tree diameters (≥10.16 cm d.b.h.) and number (n) of live trees in analysis.

Plot	Error	n
S1A	9.2	17
S1B	6.4	9
S2A	6.6	29
S2B	10.5	44
S3A	19.3	8
S3B	13.3	29
S4A	22.1	37
S4B	5.6	23
S5A	20.9	14
S5B	9.7	67
Average[a]	11.9	

[a]Average weighted by sample size (n).

Table 3—Error rate (percent) and sample size (in parentheses) for diameter (d.b.h. ≥10.16 cm) reconstruction of historical trees dead and down at time of subplot remeasurement (1997–1999).

	Decay percentile		
Subplot	25th	50th	75th
S1A	60.2 (2)	40.0 (2)	26.8 (2)
S1B	–2.6 (1)	–2.1 (2)	–10.2 (2)
S2A	–16.9 (22)	–25.2 (25)	–26.3 (28)
S2B	35.0 (6)	31.6 (6)	11.1 (6)
S3A	52.1 (4)	40.4 (6)	26.1 (6)
S3B	–8.2 (1)	–10.8 (5)	–10.8 (6)
S4A	NA (0)	22.9 (1)	29.4 (3)
S4B	34.5 (1)	18.4 (1)	7.5 (1)
S5A2	NA (0)	8.0 (1)	1.0 (1)
S5B3	70.7 (1)	32.2 (3)	20.5 (3)
Average[a]	6.9	–0.6	–6.7

[a]Average weighted by sample size.

Error for diameter reconstruction of historical trees that were dead and down at remeasurement averaged –0.6 percent (table 3). Sensitivity analysis showed that dead and down were most accurately reconstructed when moved through decay classes at the 50th percentile rate. Varying decay rate to the 25th percentile slowed movement through decay classes, resulting in earlier estimated death dates and a greater overestimate of historical diameters. Conversely, decay rate set to the 75th percentile sped movement through decay classes, resulting in later estimated death dates and an underestimate of historical diameters. Thus, varying decomposition rates altered size estimates by approximately 7 percent. Due to limited sample sizes for most plots, the overall average was heavily influenced by subplot S2A.

Diameter reconstruction of historical trees that had been cut since original plot establishment (1909–1913) was most accurate when the cut date in the model was set to 1980 (table 4). This was true for every subplot except S4A for which diameter estimates were most accurate when the cut date was set to 1945. Averaged over all subplots, diameter reconstruction of cut trees overestimated d.b.h. at plot establishment by 11.4 percent.

Tree Density on Subplots—Overestimation of tree diameter lead to reconstructed tree densities higher than those recorded on historical maps (table 5). Total number of reconstructed trees on subplots represented the sum of (1) historically measured trees, (2) trees historically existing on subplots yet too small (less than 10.16 cm) at plot

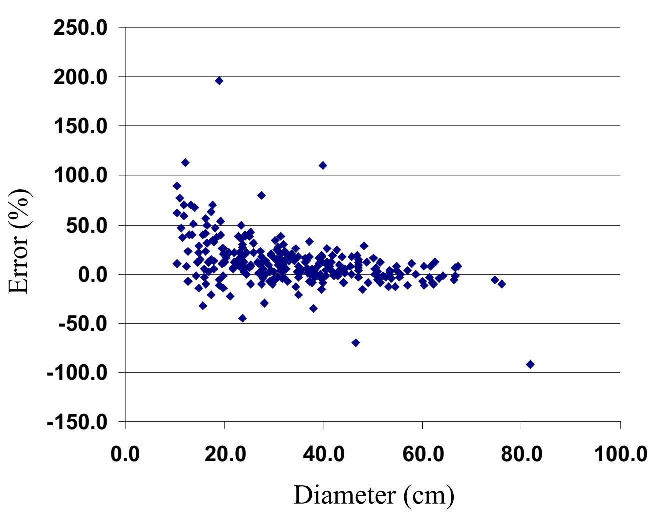

Figure 2—Plot of error (%) versus diameter of historical trees alive at remeasurement. Error compares reconstructed diameter d.b.h. to that recorded in historical (1909–1913) data.

Table 4—Error rate (percent) and sample size (in parentheses) for diameter (d.b.h. ≥10.16 cm) reconstruction of historical trees that had been cut subsequent to original plot establishment (1909–1913) and were stumps at time of remeasurement (1997–1999).

	Cut date		
Subplot	1910[a]	1945	1980
S1A	40.6 (6)	28.4 (6)	15.0 (6)
S1B	22.1 (14)	10.2 (14)	–3.1 (14)
S2A	57.5 (14)	43.1 (14)	26.8 (14)
S2B	42.0 (14)	29.1 (14)	14.7 (14)
S3A	61.1 (18)	45.9 (18)	28.9 (18)
S3B	33.3 (21)	19.1 (21)	2.8 (21)
S4A	54.4 (36)	36.2 (36)	14.7 (36)
S4B	43.8 (33)	26.8 (33)	6.8 (33)
S5A2	7.5 (4)	–0.02 (4)	–9.4 (4)
S5B3	24.7 (10)	14.3 (10)	1.2 (10)
Average[b]	43.7	28.7	11.4

[a]Earliest cut date tested for subplots S5A2 and S5B3 was 1914; these plots were historically established in 1913.
[b]Average weighted by sample size.

Table 5—Numbers of trees (≥10.16 cm d.b.h.) for subplots reconstructed to plot establishment date (1909–1913). Comparison shows number of historically measured trees (Estbl), total number of trees reconstructed (Total), and the number of trees after reconstruction totals were adjusted (Adjst1 and Adjst2). Error (No. trees is equal to (Adjst2 - Estbl).

Subplot	Estbl	Total	Adjst1[a]	Adjst2[b]	Error (No. Trees)
S1A	26	213	140	44	18
S1B	25	71	55	44	19
S2A	82	248	223	133	51
S2B	72	299	262	119	47
S3A	47	112	83	74	27
S3B	58	105	80	80	22
S4A	87	111	98	97	10
S4B	61	90	73	71	10
S5A2	20	81	42	37	17
S5B3	83	157	124	119	36

[a]Number adjusted by removing cut trees that were not originally tagged. These trees were likely cut prior to plot establishment or large regeneration thinned such as on S3A and S3B.

[b]Number adjusted by removing trees not in original ledger data but having increment core center dates of less than or equal to plot establishment date (1909–1913). These trees were likely large regeneration that were less than 10.16 cm d.b.h. in 1909–1913.

establishment to be mapped, (3) trees that had died or been cut prior to plot establishment, and (4) large trees for which no increment core data existed. Tree density estimates were also affected by variations in decay functions used in the model. Varying the rate at which dead and downed material moved through decay class by 25 percent altered estimated tree densities by approximately 7 percent.

To more clearly evaluate reconstructed tree densities (1945 cut date and 50th decay percentile) we subtracted large cut trees that were not historically recorded, as well as trees for which increment core center dates were less than plot establishment dates. We presumed these to represent trees cut prior to plot establishment and trees too small to be historically measured, respectively. Subtraction resulted in tree density overestimates of 10–51 trees per subplot (table 5). Trees reconstructed in excess of historical densities were mainly dead and down trees for which death dates were inaccurately estimated.

Conclusions

Evaluation of reconstruction techniques revealed several key sources of error. These included field identification of historical trees, size reconstruction of live trees, and determination of tree death dates. Forest structures on the subplots were readily identified in the field after ± 90 years. Missed trees resulted in an underestimate of stand density by 5.7 percent or about three trees per hectare. Factors not addressed in this analysis but that may affect success rate of identification include disturbance such as fire, time, and experience level of personnel. Very little disturbance, outside of individual tree selection harvest, occurred on the subplots in this study. Intense fire had not occurred on any of the subplots since establishment and precommercial thinning had occurred only in S3A and S3B. No clear pattern related to initial or present density of forest structures emerged to affect identification success. Identification of presettlement-aged structural evidence is important for implementing ecological restoration prescriptions in northern Arizona ponderosa pine forests (Covington and others 1997).

The reconstruction model tested in this analysis tended to overestimate tree diameters for live trees (11.9 percent), slightly underestimate (–0.6 percent) dead and downed trees, and overestimate trees that had been cut (11.4 percent). For live trees, slightly greater inaccuracies were produced for smaller size classes. Possible explanations for live-tree errors include model equations used to predict bark thickness and d.b.h. from d.s.h. (Myers 1963; Hann 1976), unusable increment cores, particularly for trees that were especially fast- or slow-growing, and eccentricity of tree bases.

Although prediction of tree death date is difficult due to factors affecting snag longevity and condition, the d.b.h. estimates provided by the reconstruction model were relatively accurate. Accuracy here was likely affected by the interaction of death date estimates and reverse growth functions.

In our analysis, global cut dates were set in the model, although in reality, trees on the subplots were not all cut in the same year. This undoubtedly affected d.b.h. reconstruction errors. Cut dates for harvested “cohorts” could be individually coded in the model using additional information not examined in this study such as harvest records and stump decay classes.

Overestimation of d.b.h. coupled with death date and cut date uncertainties lead to overestimates of past tree density. However, our reconstruction techniques allowed reductions of these estimates based on increment core data and sizes of cut trees. Refinement of decay functions may allow better accuracy in stand density reconstruction.

The reconstruction techniques evaluated in this study appear to be robust and useful for estimating past forest structural characteristics. Although the model was used generally, adjustments could be made for specific sites using relationships developed from site-specific field data. The reconstruction techniques allow a better understanding of reference conditions in ponderosa pine forests of the Southwest. Further analysis will be done to reconstruct presettlement structural conditions on these plots.

Acknowledgments

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Cheesman Lake—A Historical Ponderosa Pine Landscape Guiding Restoration in the South Platte Watershed of the Colorado Front Range

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Abstract—An unlogged and ungrazed ponderosa pine/Douglas-fir landscape in the Colorado Front Range provides critical information for restoring forests in the South Platte watershed. A frame-based model was used to describe the relationship among the four primary patch conditions in the 35-km^2 Cheesman Lake landscape: (1) openings, (2) ponderosa pine forest, (3) ponderosa pine/Douglas-fir forest, and (4) persistent old growth. Each condition is possible over time at any nonriparian site, with fire and tree recruitment the primary processes causing changes from one condition to another. The Forest Vegetation Simulator model was used to estimate forest conditions at Cheesman Lake in 1900, prior to fire suppression effects. These results and 1896 Cheesman Lake photographs indicate that more than 90 percent of the historical landscape had a crown closure of 30 percent or less, compared with less than 50 percent of current nearby forests affected by logging, grazing, tree planting, and fire suppression. The historical fire regime was mixed severity, and passive crown fire was probably more common than active crown fire. Currently, surrounding forests have almost no openings, little old growth, high tree density, and increased Douglas-fir. Fire behavior has switched to a crown fire regime with sometimes catastrophic results. Historical Cheesman Lake forest landscape conditions are being used to guide restoration of surrounding forests.

Introduction

Restoration of ponderosa pine forests to ecologically sustainable conditions requires knowledge of historical conditions and processes that shaped natural landscapes. Until recently, studies from the Southwestern United States provided the majority of scientific literature describing ponderosa pine stand and landscape structure and the natural processes affecting these forests (Cooper 1960; Covington and Moore 1994; Fulé and others 1997; Mast and others 1999; Moore and others 1999; Pearson 1933; Swetnam and Baisan 1996; Swetnam and others 1999; White 1985; Woolsey 1911). Pine forests in the Southwest historically were characterized by a low-intensity surface fire regime, with fires typically occurring over large areas every 2 to 10 years or more. Stands were described as open and parklike, with productive grassy understories and few small trees. Several reports suggest that the fire frequency in other regions was lower but maintained similar open stands in less productive northern systems (Arno and others 1995; Goldblum and Veblen 1992). Reconstructions of historical stand structures in both the Southwest and other regions have been attempted, but little information on structure of historical ponderosa pine forests at a landscape scale has been available.

Research is being conducted on a 35-km^2 forest area at Cheesman Lake in the Colorado Front Range. Land around Cheesman Lake, a reservoir owned by Denver Water, has never been logged and has been protected from grazing since 1905, when the dam for the reservoir was completed. This research provides a description of a ponderosa pine landscape that evolved with a mixed severity fire regime (Brown and others 1999; Kaufmann and others 2000a,b). Mixed severity fires differ from the frequent low intensity surface fire regime of the Southwest in that individual fire events generally are less frequent and have a patchy crown fire component along with surface fire. They differ from crown fire regimes because the crown fire component is smaller and much more patchy. The Cheesman Lake climate appears to be less influenced by the El Niño pattern and is drier than the climate in the Southwestern United States. Fire history data at Cheesman Lake span eight centuries, with the earliest fire scar sampled dating to 1197 (Brown and others 1999). Maps of fire scar locations shown by Brown and others and additional unpublished data of other fire scar locations indicate that large fires (5–10 km^2 or larger) occurred in 1496, 1534, 1587, 1631, 1696, 1723, 1775, 1820, 1851, and 1880. The only significant fire since 1880 was in 1963, when a fire in the southeast portion of the Cheesman Lake landscape was suppressed after reaching a size of around 25 ha. A moderate number of more localized fires occurred between the larger fires. In addition, extensive studies of tree age and size indicated that tree recruitment occurred in pulses, each about 10 years long (Kaufmann and others 2000b). The recruitment pulses tended to coincide

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with the larger fires; most of the major fires occurred during these pulses, but not necessarily in a cause and effect sequence.

Kaufmann and others (2000a) suggested that relatively infrequent fire and gaps between tree recruitment pulses reflect unfavorable interim climatic conditions for both production of herbaceous vegetation (needed for fuel) and tree seed germination and establishment. In many of their plots, the age of the oldest tree at specific locations in the Cheesman Lake landscape indicated that all trees postdated certain fires identified in the study area by the fire history analysis (Brown and others 1999). Similar results were observed in western Montana (Arno and others 1995). These data strongly suggest that the crown fire component of past fires killed all trees in certain areas and created openings. The current oldest trees reflect the dates trees were recruited into each opening. An expansion of this analysis is given in another paper in these proceedings (Huckaby and others).

Fire and tree recruitment patterns are key processes affecting landscape structure and change over time in the Cheesman Lake landscape. In this paper we describe the four primary types of forest patches that make up the upland landscape. These four components are interchangeable in that they potentially can occur at any upland site over time. The components are useful for describing not only the historical landscape, but also forests that were altered following European settlement. While riparian areas are critical components occupying 1–2 percent of the landscape, they are separate from the upland sites and are not included in our analysis of the upland areas.

We use a frame-based model to discuss the factors that cause transitions of the upland conditions from one state to another (Starfield and others 1993). In addition, we use a forest growth model to "degrow" the Cheesman Lake landscape to assess its condition prior to the effects of fire suppression during the 20th century. These "degrown" conditions are compared with historical photographs of the Cheesman Lake landscape. Finally, we discuss implications of presettlement fire behavior and landscape structure on current restoration activities in the South Platte watershed.

Primary Upland Components

Ponderosa pine forests in the historical Cheesman Lake landscape and elsewhere in the Colorado Front Range are relatively simple in that only two species, ponderosa pine and Douglas-fir, comprise the vast majority of trees (Jack 1900; Kaufmann and others 2000b; Peet 1981; Veblen and Lorenz 1986). While fire suppression during the last century is believed to have increased forest density at Cheesman Lake (discussed later), diversity in age and diameter class distributions, tree density, proportions of the two species, and patch size make the landscape structure very complex, driven by both topography and natural disturbance history. In an adjacent logged landscape, other species (Colorado blue spruce, aspen, narrow-leaf cottonwood, and Rocky Mountain juniper) are more significant, especially in lowland areas (Kaufmann 2000b). We mapped and classified the entire 35-km^2 landscape using an overstory classification template created from extensive plot data (data summarized in Kaufmann and others 2000b) in conjunction with 1:6000 color infrared photographs. Over 3,000 polygons in the historical landscape were classified by percent crown closure in six distinct diameter distribution classes. While polygon characteristics are still being analyzed in GIS, the mapping and classification effort provides a basis for understanding landscape dynamics discussed here.

Examination of plot and polygon data and extensive field observations indicate that four basic stand conditions are found in the Cheesman Lake landscape (fig. 1). These are roughly based on stand development stages and successional trajectories. The conditions are (1) openings vegetated primarily with grasses and shrubs, (2) patches that are pure or nearly pure ponderosa pine, (3) patches having both ponderosa pine and Douglas-fir, and (4) patches having very old trees, which we term "persistent old growth." Openings presently existing in the Cheesman Lake landscape have coarse woody debris dated to 1851, a year in which a large fire occurred (Brown and others 1999). Only one opening of 49 was found having no coarse woody debris (Tobler 2000), indicating that nearly all openings are a transient patch condition in the landscape. The pine and pine/Douglas-fir patches have a specific characteristic distinguishing them from persistent old growth: a cap or upper limit on the age of the oldest trees in the patch. This suggests that they developed following a stand-replacing natural disturbance, most likely fire (Kaufmann and others 2000b). The oldest trees may be quite old (more than 400 years), but even the oldest trees in these patches postdate known fires. In contrast, the persistent old-growth patches appear to be regulated primarily by microscale disturbances that kill only one or a few trees at a time, such as heartrot, insect attack, windthrow, or very small fires. Trees in these patches have a wide age distribution (some may be more than 500 years old), with varying states of health, and a large amount of coarse woody debris is common.

Frame Model of Landscape Components

Model Description

The relationships among the forest patch conditions shown in figure 1 and the key processes affecting the transitions between them are shown in a frame-based model, where each frame represents a potential condition that can exist on a site (fig. 2). Extensive field data on fire history and tree recruitment indicate that the following processes affect transitions among states. First, tree recruitment converts openings created by fire to forest (fig. 2, arrows with small trees). Ponderosa pine patches develop when recruitment is limited to this single species. Age data for a large number of field plots suggest that historically, pure ponderosa pine stands developed on most east, south, and west slopes. In contrast, mixtures of ponderosa pine and Douglas-fir trees often were recruited on north aspects. Tree age data indicate that in the 1900s (during which fire suppression occurred), patches became much more dense, and many pure ponderosa pine patches were converted to mixed patches through ingrowth of Douglas-fir.

Figure 1—Four major patch components of the ponderosa pine/Douglas-fir landscape of the South Platte watershed in the Colorado Front Range. (Upper left) openings; (lower left) persistent old growth; (upper right) ponderosa pine; and (lower right) ponderosa pine/Douglas-fir. Each condition may occur on all upland sites. Parameters of each component are found in table 1.

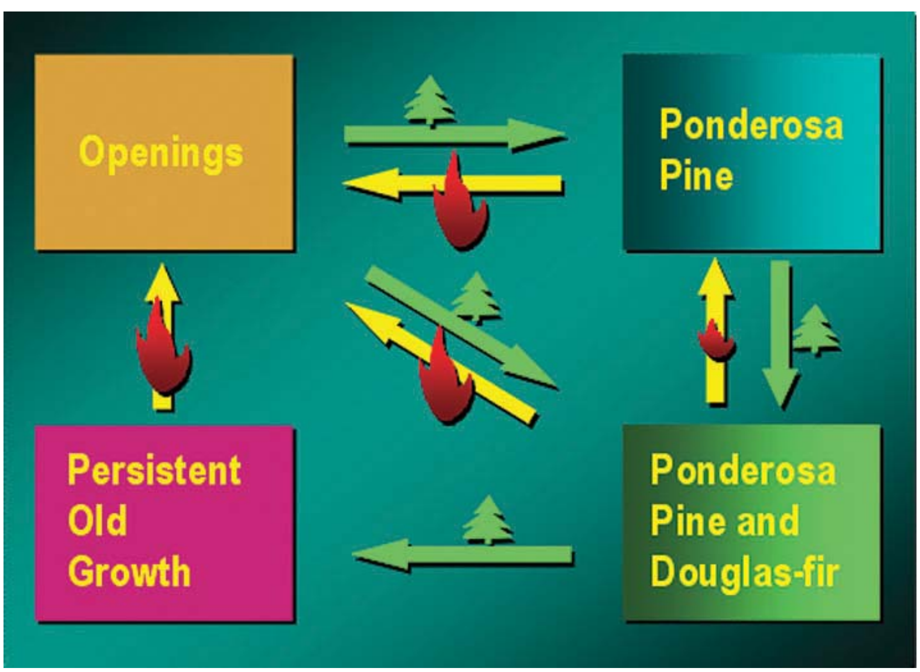

Figure 2—Frame model showing the interrelationships among openings and patches of ponderosa pine, ponderosa pine/Douglas-fir, and persistent old growth in the South Platte watershed of the Colorado Front Range. Arrows with trees represent tree recruitment processes, and arrows with flames represent either crown fire creating openings or surface fire reducing ingrowth of Douglas-fir.

Persistent old-growth patches in our model require very old trees, which we define to be the condition in which trees die predominately from natural causes other than stand-replacing fires. Recruitment occurs where small openings provide enough resources for new trees to be incorporated into the stand. While the oldest trees generally are ponderosa pine, older Douglas-fir trees are found in many persistent old-growth patches (Huckaby and others, this proceedings). We have observed a few pine trees more than 600 years old, some over 500, and many exceeding 400 years. Trees recruited after the fire in 1496 could be 500 years old, and we are not confident in suggesting that these trees reflect recruitment after a stand-replacing fire in 1496 because many trees die for reasons other than fire before reaching an age of 500 years. We are more confident suggesting that trees postdating the 1531 fire reflect a stand-replacing event (Huckaby and others, this proceedings). A caveat in suggesting that polygon reforestation postdates specific fires, however, is that there always is the possibility that older trees predating the indicated fire may have died of other causes. Nonetheless, extensive data over a large portion of the landscape suggest that age caps related to stand-replacing fires are common, and these age caps are useful for understanding landscape dynamics (Arno and others 1995; Kaufmann and others 2000a,b; Huckaby and others, this proceedings).

Fire is almost certainly the primary process creating openings of 1 ha or more in the landscape (fig. 2, arrows with flames). There is little likelihood that other natural disturbance factors such as insects, disease, or wind could have resulted in complete mortality in this system (Kaufmann and others 2000b). Stand-replacing fires were probably more common in ponderosa pine and pine/Douglas-fir forest patches and less common in persistent old growth. Characteristics of stand-replacing fires in the historical Cheesman Lake landscape are discussed later. All fires undoubtedly had a surface fire component, and under certain conditions surface fires may have reduced ingrowth of younger Douglas-fir, thereby converting mixtures of pine and Douglas-fir back to pure or nearly pure ponderosa pine.

Model Parameters

Tree age, density, species composition, and fire history data provide insight into probabilities of changes among components in the frame model shown in figure 2. Table 1 provides criteria we have adopted to distinguish among the four patch conditions. We believe these criteria are suitable for most of the ponderosa pine forest area of the South Platte basin. Openings are defined as areas that have 10 percent or less crown closure. We distinguish between pure ponderosa pine and ponderosa pine/Douglas-fir patches based upon basal area or density of trees. Patches having more than 10 percent of the basal area or more than 20 percent of the trees breast height or taller in Douglas-fir are considered ponderosa pine/Douglas-fir patch types.

Information about the fire and tree recruitment processes that change the condition from one patch type to another is summarized in table 2. This information reflects data for 4 centuries before Euro-American settlement, which occurred in the late 1800s. Current analyses indicate that large fires (more than 5 km^2, listed in the introduction)

Table 1—Characteristics of patches in ponderosa pine/Douglas-fir forests in the unlogged Cheesman Lake and adjacent logged Turkey Creek landscapes. Patch types are those shown in figure 1. BA = basal area.

Patch type	Crown closure	Condition	Tree age structure
Openings	≤10 percent	May be mixed species	Few, generally young
Ponderosa pine	>10 percent	≤10 percent BA Douglas-fir, and ≤20 percent of trees Douglas-fir	Age cap evident May be old growth
Ponderosa pine/Douglas-fir	>10 percent	>10 percent BA Douglas-fir, or >20 percent of trees Douglas-fir	Age cap evident May be old growth
Persistent old growth	>10 percent	Mixed ponderosa pine and Douglas-fir	No age cap evident Very old

Table 2—Fire and tree recruitment characteristics affecting changes in ponderosa pine/Douglas-fir patch structure in the Cheesman Lake landscape. Processes are those depicted by arrows in figure 2. Data summarized from Brown and others (1999), Kaufmann and others (2000b), and additional analyses of fire scars and tree ages.

Process	Mean interval	Range
	Years	
Fires >5 km^2 in 35-km^2 landscape, 1496–1880	42.7 ± 12.7	27–65
Fires in 0.5-2 km^2 areas, 1496–1880	50.0 ± 17.2	29–83
Tree recruitment, 1588–1885	45.3 ± 23.5	18–82

occurred with a mean interval of about 43 years in the 35-km^2 landscape. This estimate is shorter than the 59 years estimated by Brown and others (1999) because it includes three fires (1775, 1820, and 1880) not included in the earlier analysis. These fires were included here based upon additional evidence of fire scars not available earlier. When the fire scar sampling area was divided into 11 smaller areas of 0.5-2 km^2 each based upon sampling clusters and topography, the mean fire interval for all areas was 50 years, with a range of 29 to 83 years. Studies are under way to assess fire intervals in other portions of the South Platte watershed.

Tree age data for a 4-km^2 portion of the Cheesman Lake landscape were obtained by dating cores collected 30–40 cm above the ground, and correcting the pith age for the estimated time required after germination to reach the coring height (Kaufmann and others 2000b). These data indicated that most tree recruitment occurred in pulses lasting about a decade each (see fig. 6 in Kaufmann and others 2000b). For a 421-year period between the 1560s and 1980s, half of the surviving trees in 25 0.1-ha plots in the Cheesman Lake landscape were recruited in these pulses, which accounted for only one-fourth of the time. The mean interval between recruitment pulses was about 45 years, which agrees closely with the MFI for the area (table 2).

Current and Historical Landscape Crown Closure

A frame model is useful for assessing how important processes and conditions affect patch dynamics over time and space at a landscape scale. We are addressing the application of a frame model at a landscape scale by examining landscape structure at several points in time. At the initial stages of this application, we are focusing on diameter class distribution and crown closure, which we can estimate using 1:6000 color infrared photographs. The infrared photographs cannot be used to map and classify patches based upon species composition, but they provide a basis for mapping by diameter class distribution and crown closure. The map of percent crown closure in figure 3 (left) was developed from photographs taken in 1996. This map indicates that current crown closure varies widely, with many areas having very open forest (10–30 percent crown closure) and others dense forest (more than 45 percent).

A tree growth model, Forest Vegetation Simulator (FVS, USDA Forest Service 1999), was used with polygon data on tree diameter class distribution and density to "degrow" patches back to 1900, the time just before fire suppression efforts may have begun on the landscape. FVS cannot be run backward in time; however, through a trial and error process

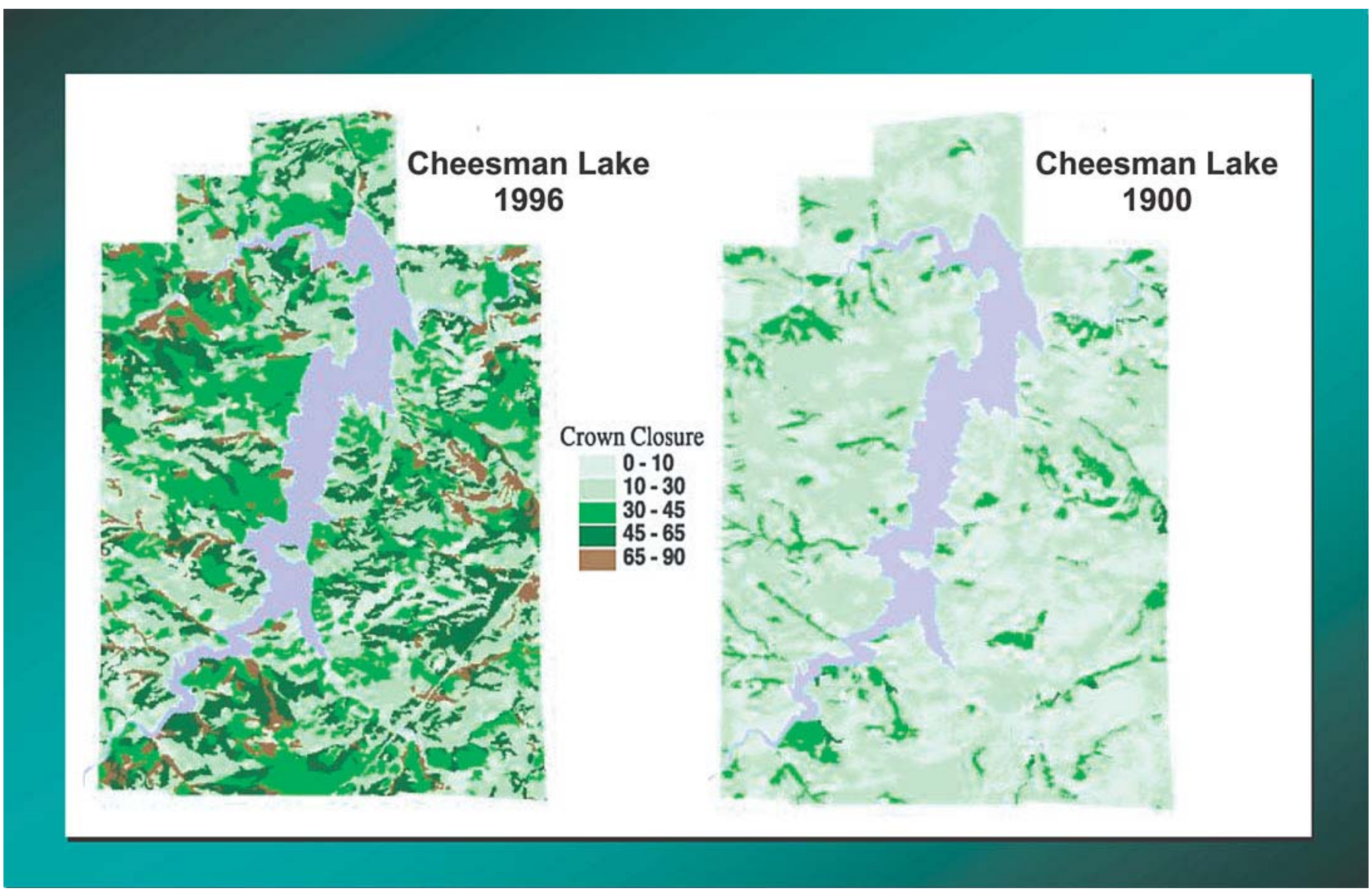

Figure 3—Crown closure percent for the Cheesman Lake landscape. The 1996 map was developed from 1:6000 color infrared photographs and a patch classification template based on plot data. The 1900 map was developed using the Forest Vegetation Simulator to estimate conditions that, projected forward, yielded conditions observed in 1996.

average stand conditions can be identified for earlier periods that, when grown forward, are comparable to current stand conditions. Stand conditions for 1900 were accepted when FVS projections to 1996 agreed well with current (1996) estimates of stand size class distributions and densities. While a number of simple assumptions about tree recruitment rates and mortality will be reexamined, an initial version of the landscape condition in 1900 derived from these calculations is shown in figure 3 (right). Differences between 1996 and 1900 are dramatic, yet the very low forest densities estimated for 1900 agree very well with age structure data for plots with 20th century trees removed and with the open forest structure shown in historical photographs.

Historical photographs taken in 1896 near the site being selected at the time for the Cheesman Lake dam (Denver Water archives, Denver, Colorado) augment our understanding of forest structure at several locations in the Cheesman Lake landscape prior to the effects of 20th century fire suppression. While some grazing undoubtedly had occurred late in the 19th century (primarily along the river), effects on forest structure in the photographed areas are believed to be negligible. Nor was logging a factor, based upon archived records of logging history in the area (South Platte District Office of the Pike/San Isabel National Forest). However, fire suppression limited burning to a single wildfire smaller than 100 ha in 1963. The photograph in figure 4 includes an area in the foreground known to have burned in the 1880 fire, probably resulting in the dead trees on the ground. Across the valley (which is presently part of the reservoir), the far left background shows openings that we believe were created by a fire in 1851. The area in the center across the valley presently has many old trees, indicating it was largely free of stand-replacing fire for at least several centuries, even though fires occurred there in 1631, 1696, 1820, and 1880 (Brown and others 1999). Figure 4 and other historical photographs of unlogged areas show a generally very low historical forest density, a limited number of small trees, and an understory dominated by grasses and shrubs.

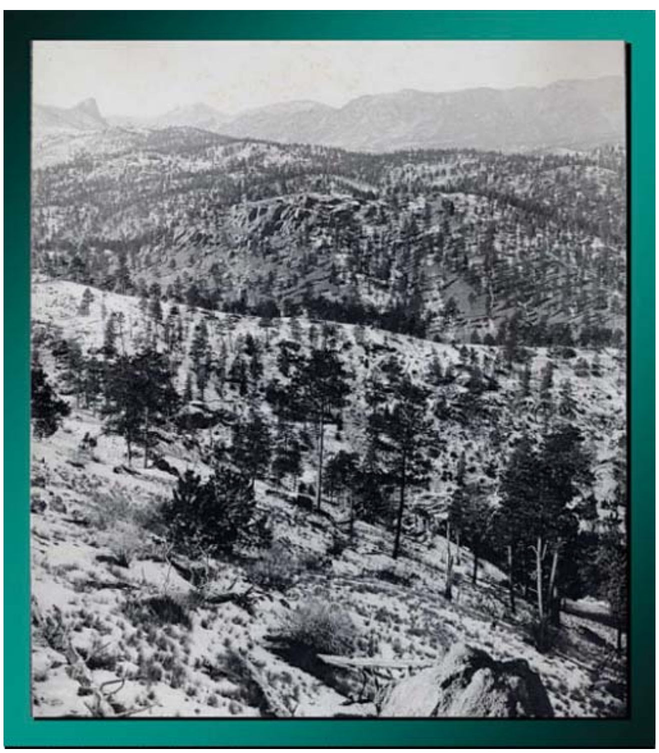

Figure 4—Historical photograph illustrating forest density of the unlogged Cheesman Lake landscape in 1896. Photograph from archives of the Denver Water Department.

Since fire suppression, however, stands have become denser both in the Cheesman Lake landscape and in all nearby areas where logging and grazing occurred. Ingrowth of Douglas-fir is common on all aspects and has converted many ponderosa pine stands into mixtures of the two species. Paired photographs in the Cheesman Lake landscape from 1903 and 1999 (fig. 5) illustrate the change in forest density and loss of openings (right background) along the South Platte River just below the Cheesman Lake dam. A notable exception to the increased forest density in the Cheesman Lake landscape is the area burned in 1963. This area, which had previously burned in 1851, was thinned dramatically by the 1963 fire, leaving a forest structure similar to that found in figure 4.

Polygon crown closure data were compared in GIS for the 1996 Cheesman Lake and adjacent Turkey Creek landscapes, and for the Cheesman Lake landscape degrown to 1900. The Turkey Creek polygon data were for a 4-km^2 study area just outside the historical landscape. This landscape was logged in the 1890s and typifies other forests in the South Platte watershed, though other areas logged earlier may be denser. The landscape area for low, medium, and high crown closure is shown in figure 6. These data suggest that in 1996 only 55 percent of the Cheesman Lake landscape and 47 percent of the Turkey Creek landscape had crown closures of 30 percent or less. However, we estimate that in 1900 over 90 percent of the Cheesman Lake landscape had crown closures no greater than 30 percent. The amount of the landscape having a crown closure of 30 percent or less is significant, because at these tree densities fires usually do not spread from crown to crown. Though preliminary, these estimates and historical photographs support the widely held view that earlier ponderosa pine forests in the Colorado Front Range were much less dense than presently found.

Frame Model Component History

The proportion of the overall landscape we estimated to be occupied by each of the four patch conditions in figures 1 and 2 is shown in figure 7. The upper portion (Historical to Settlement) depicts proportions that we believe occurred the last several centuries before the effects of settlement. The illustration refers only to upland areas and does not include riparian areas (1–2 percent of the landscape) or meadows existing along the South Platte River before the reservoir was completed. The proportions are based on examinations of tree ages for ponderosa pine and Douglas-fir trees over a range of topographic conditions and over a large portion of the landscape (Kaufmann and others 2000b; Huckaby and others, this proceedings). As noted earlier, data indicate that

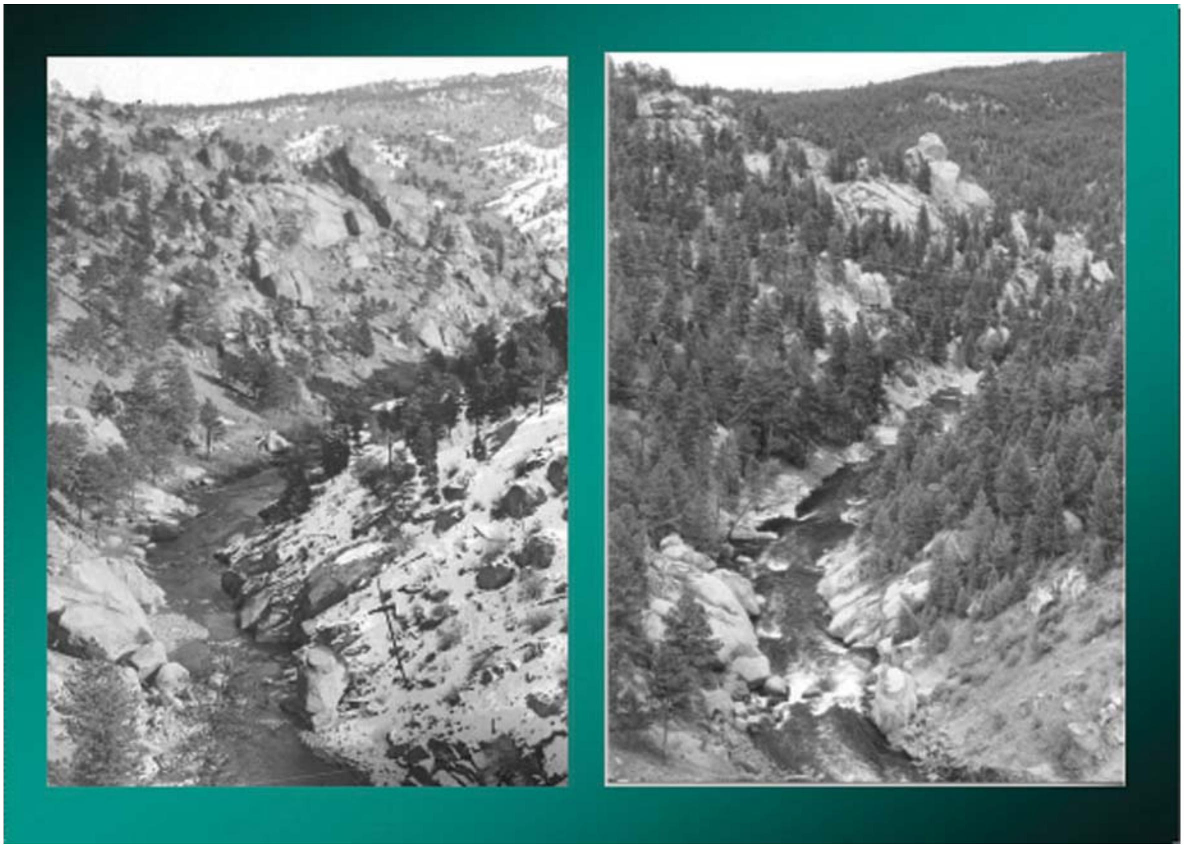

Figure 5—Paired photographs showing forest condition along the South Platte River just below the Cheesman Lake dam in 1903 (left) and 1999 (right). The 1903 photograph is from archives of the Denver Water Department.

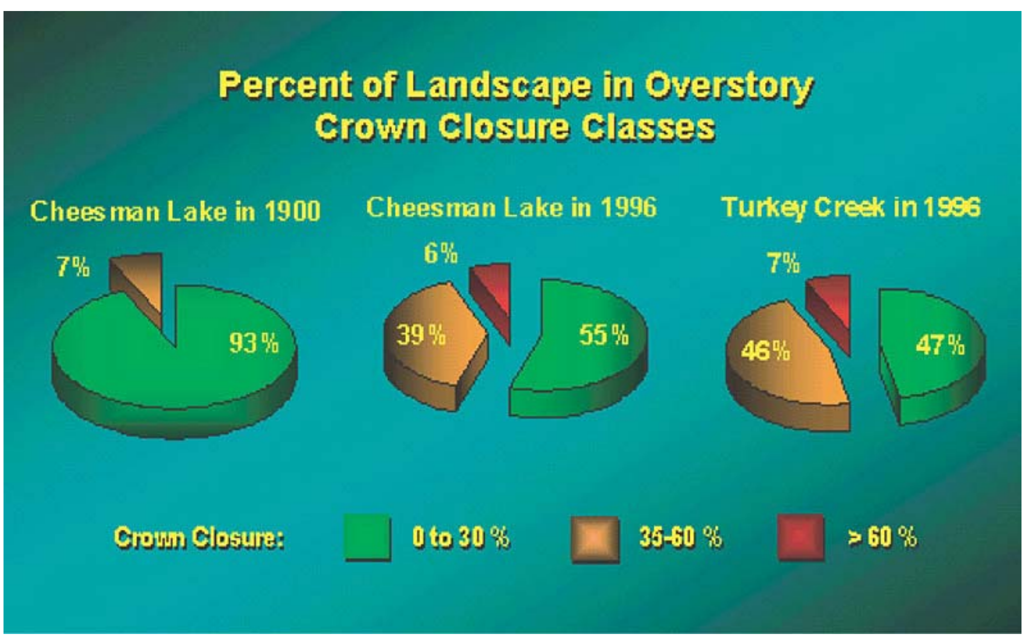

Figure 6—Proportion of the landscape covered by forest in crown closure classes, showing dramatic increases in crown closure since 1900. Cheesman Lake data for 1900 and 1996 are the same as shown in figure 3. Turkey Creek data for 1996 are for a logged area adjacent to the Cheesman Lake landscape.

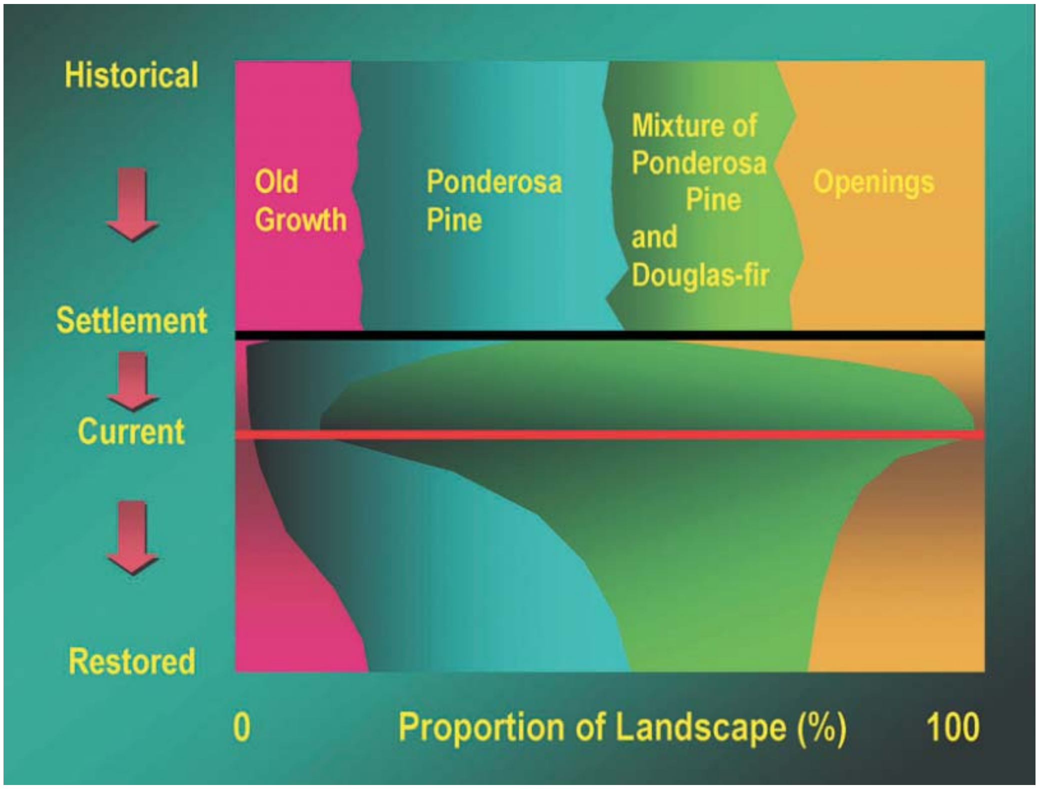

Figure 7—Estimated proportions for the four components of the ponderosa pine/ Douglas-fir forest landscape in the South Platte watershed. The period from Historical to Settlement reflects information from Cheesman Lake in the absence of logging, grazing, or fire suppression effects. The period from Settlement to Current reflects changes in forest conditions found in much of the logged and grazed South Platte watershed. The period from Current to Restored reflects landscape changes that might occur under a restoration scenario.

before settlement most trees on east, south, and west slopes were ponderosa pine, whereas Douglas-fir and ponderosa pine occurred in roughly equal proportions on north slopes.

We estimate that about 15 percent of the landscape had persistent old-growth patches (fig. 7). Pure ponderosa pine patches (see table 1 for characteristics) probably accounted for 35–50 percent of the landscape, primarily on east, south, and west slopes. Ponderosa pine/Douglas-fir patches on north slopes and portions of upper ridges may have accounted for 20–30 percent of the landscape, and at least 25 percent of the landscape was open, with no more than 10 percent tree crown closure. Undoubtedly these proportions varied over time, especially when fires created openings, reduced tree densities, or killed young Douglas-fir trees invading patches. Furthermore, tree densities (represented by shading in fig. 7) and the amount of Douglas-fir are likely to have increased within each category as time since fire increased.

The middle portion of figure 7 indicates general shifts we estimate occurred in the forest components of areas outside the historical landscape. These estimates are based on measurements in the adjacent logged Turkey Creek landscape and on observations of forest structure throughout the South Platte watershed. The shifts in patch proportions reflect the effects of logging, grazing, fire suppression, and transplanting, all of which are likely to increase forest density. Logging decreased the amount of persistent old growth. Grazing probably reduced understory competition and allowed the establishment of new seedlings, and the lack of fire allowed the seedlings to survive. The result was a sharp increase in forest density, expansion of the area having a significant Douglas-fir component, and the loss of most openings that temporarily increased during logging. The Current to Restored portion of figure 7 is discussed below.

Implications About Fire Behavior

The current condition of ponderosa pine/Douglas-fir forests in the Colorado Front Range favors a crown fire regime, with a high risk of catastrophic stand-replacing fires. The Buffalo Creek fire in 1996, about 10 miles north of Cheesman Lake, burned 4,800 ha (11,900 acres), of which 3,000 ha (7,500 acres) was a crown fire that traveled 11 miles in $4\frac{1}{2}$ hours. Forest conditions here are similar to those throughout much of the West, with high tree densities favoring crown fires (Covington and Moore 1994).

Strong evidence suggests that the historical fire behavior at Cheesman Lake followed a mixed severity fire regime, which is characterized by a combination of surface fire and patchy crown fire occurring within the fire perimeter. Data suggest that many forest patches have an age cap, indicating regeneration following a stand-replacing disturbance (fire), while other patches nearby retained much older trees (Huckaby and others, this proceedings; Kaufmann and others 2000b). We originally hypothesized that the mixed severity fire pattern creating the openings resulted from long intervals between fires, during which stand density and fuel ladder conditions increased. These conditions would have led to forest patches burning by active crown fire (fire spreading both on the surface and tree-to-tree). We now suspect that low stand densities (crown closure under 35 percent) were the general condition in the historical landscape, and we hypothesize that the mixed severity fire regime included passive crown fire (ignition from a surface fire and not from adjacent crowns) as a major component, because trees were generally too far apart to spread fire in the crowns.

More open forests would likely have had a grassy understory, perhaps with a strong but patchy shrub component including primarily mountain mahogany and currant. Local fire behavior specialists suggest that the transition between grassy understory vegetation and a litter surface fuel condition occurs between 30 and 40 percent crown closure (Henry Goehle and Doug Steven, personal communication). While coarse-textured granitic soils and low precipitation result in low overstory and understory productivity in the study area, particularly when the climate is in dry cycles, more open stand conditions would favor better (though patchy) understory production. Furthermore, fires are most common during the decade-long pulses of tree recruitment, when climatic conditions probably favor both tree recruitment and understory production favorable for surface fire (Kaufmann and others 2000b). If shrubs, low limbs, or coarse woody debris accumulate beneath individual tree crowns, large numbers of trees may be killed by passive crown fire. In addition, we suspect that small patches of young trees may have occurred in some areas, and these patches may have been burned by localized active crown fires. Thus passive crown fire and very localized active crown fires in young trees were probably major causes of openings and low density forest in the South Platte ponderosa pine/Douglas-fir system.

Implications for Landscape Restoration

Research at Cheesman Lake on characteristics of historical ponderosa pine/Douglas-fir landscapes provides a sound basis for landscape restoration in surrounding areas (Culver and others, this proceedings). Clearly the current forest density and amount of Douglas-fir in these areas are too high. The 11,900 acre Buffalo Creek fire in 1996 near the Cheesman Lake landscape illustrated the huge risks of wildfire and postfire erosion in these dense forests.

An ecologically sustainable landscape would have a forest structure similar to that found historically, presuming that climatic conditions have not changed substantially. The lower portion of figure 7 suggests how changes in the four primary components of the South Platte basin might be returned to the landscape structure conditions found historically. The amount of persistent old growth, reduced sharply by logging of old trees, requires considerable time for restoration. However, many trees over 200 years exist in logged areas examined in the Turkey Creek landscape adjacent to Cheesman Lake and in logged forests northeast of Deckers in potential restoration areas, placing the age structure well on the trajectory for reestablishing the persistent old-growth condition (Kaufmann 2000b). Restoration activities should protect all old trees.

Openings existed historically but were lost since settlement. They can be recreated by removing trees, perhaps through a combination of mechanical treatment and prescribed burning. The ponderosa pine component can be restored by selective removal of Douglas-fir, and thinning would reduce densities to levels found historically. Similar treatments would reduce the spatial extent of the ponderosa pine/Douglas-fir component.

A landscape assessment for the South Platte watershed identified several large subwatersheds at high risk of wildfire and postfire erosion (Culver and others, this proceedings). These areas are close to the Cheesman Lake landscape, and they have similar topography, soils, and climate. Restoring historical conditions is likely to address the two major issues of the Upper South Platte Watershed Protection and Restoration Project (Culver and others, this proceedings): (1) restoration of the landscape to an ecologically sustainable condition, and (2) mitigation of the risk of large-scale crown fires and postfire erosion. The changes in landscape components in the lower portion of figure 7 address both of these issues.

It is unrealistic to expect the entire South Platte watershed to be restored to historical conditions in a short amount of time. Furthermore, in some places the historical condition may not be the desirable outcome. However, with priorities established by the landscape assessment for the South Platte watershed, restoration of portions of the overall watershed can be staged to create fuel breaks that reduce the risk of large-scale crown fires such as that in Buffalo Creek in 1996.

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Landscape Patterns of Montane Forest Age Structure Relative to Fire History at Cheesman Lake in the Colorado Front Range

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Abstract—Lack of Euro-American disturbance, except fire suppression, has preserved the patterns of forest structure that resulted from the presettlement disturbance regime in a ponderosa pine/Douglas-fir landscape at Cheesman Lake in the Colorado Front Range. A mixed-severity fire regime and variable timing of tree recruitment created a heterogeneous forest age structure with considerable old growth. Surrounding forests subjected to human alteration since the late 1800s are younger, denser, and more continuous. We present preliminary data from a study of fire history and age structure. We mapped forest patches based on tree size and density using color-infrared aerial photos, then randomly sampled 10 percent of these patches across the 35 km^2 landscape for the ages of the five apparent oldest trees. Trees older than 200 years were found in 70 percent of sampled stands. Trees older than 400 years were found in 30 percent of sampled stands, suggesting that old growth was common and widespread in historical landscapes in the Front Range. We compared the stand ages with locations of known fire dates derived from fire scars. Concentrations of trees that postdate known fires indicate a past stand-replacing fire. Such postfire cohorts are discernible as far back as 1531 A.D. Of 21 fires recorded by scars between 1531 and 1880, 16 appear to have had a stand-replacing component, and seven known fires predate 71 percent of the postfire cohorts. Time between stand-replacing disturbance and tree establishment varied considerably between sites, but generally ranged from 20 to 50 years. Some openings began to regenerate within 10 years after fire, while others remain unforested 150 years later.

Introduction

Most scientists and managers acknowledge that old-growth forest was more common in Western ponderosa pine landscapes before Euro-American settlement than it is now (Regan 1997; Baker 1992; Kaufmann and others 1992; Moir 1992; Veblen and Lorenz 1991). Old-growth ponderosa pine forests with large, open-grown trees generally older than 200 years were more heavily impacted by logging and grazing during and immediately after the settlement period than younger stands (Covington and Moore 1994; Cooper 1960). Managers attempting forest restoration in ponderosa pine landscapes of the West face several questions: what constitutes old-growth forest in their ecosystems? How much old growth existed historically, and how was it distributed? What were the disturbance regimes that regulated landscape structure? How much old growth remains on the present landscape, and how can we recreate a more historical and sustainable forest structure? In many locations, these questions are largely unanswerable. No old growth remains that developed under historical disturbance regimes. Where old growth does exist, it is often small, isolated stands that were preserved by their inaccessibility or lack of productivity, and does not necessarily reflect the historical distribution of old-growth forest on the landscape.

Reconstructing historical disturbance regimes is essential for understanding the processes that regulated forest structure and the distribution of old growth across the landscape (Veblen and others 2000; Mast and others 1999; Fulé and others 1997; Brown and Sieg 1996; Swetnam and Baisan 1996; Brown 1995; Veblen and Lorenz 1986). The spatial and temporal scale of historical disturbances determined the distribution, extent, character, and persistence of old-growth stands. Using the age structure of present forests, particularly in relatively undisturbed stands, is one way to reconstruct historical forests (Mast and others 1999; Arno and others 1995; Duncan and Stewart 1991). However, because remaining old stands are often small and isolated, it is difficult to extrapolate to a larger scale.

We have been fortunate to study a relatively undisturbed ponderosa pine landscape of 35 km^2 in the Colorado Front Range. The land surrounding Cheesman Lake, on the South Platte River, experienced only minimal, localized logging and grazing before 1900, unlike most montane forests in the Front Range. While fire suppression since 1900 has allowed for some ingrowth of trees and increased prevalence of young Douglas-fir, the lack of other anthropogenic disturbance has preserved the distribution of presettlement trees that resulted from the historical disturbance regime. The cool, dry climate has preserved remnant wood, allowing us

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to reconstruct the fire history back to 1197 A.D. (Brown and others 1999). The historical mixed-severity fire regime and variable timing of tree establishment in the ponderosa pine/Douglas-fir forest created an older and more heterogeneous age structure than that found in surrounding areas that have experienced more human alteration since the mid-19th century (Kaufmann and others 2000a). The size of the protected area has allowed us to study a historical landscape on a scale similar to that at which we believe historical disturbance regimes actually operated, and gives us some sense of the scale of patterning in the landscape. We present preliminary data from a landscape-scale study comparing forest age structure and fire history.

Background and Methods

The Cheesman Lake Landscape

The Cheesman Lake landscape is 3,040 ha of land area, excluding the lake, located in the montane forest zone, about 60 km southwest of Denver. Elevations range from 2,100 m at the water level of the reservoir to 2,400 m on the west side of the lake. The vegetation at Cheesman Lake is dry ponderosa pine/Douglas-fir forest (*Pinus ponderosa*/*Pseudotsuga menziesii*) (Peet 1991). Understories are typically grassy or shrubby or both; denser forests have very sparse understories. Most of the botanical diversity in this landscape is concentrated in the relatively small riparian areas around streams, many of which are intermittent. These forests were historically open, with the Douglas-fir concentrated on northerly slopes (Kaufmann and others 2000b). Aspen (*Populus tremuloides*), blue spruce (*Picea pungens*), Rocky Mountain juniper (*Juniperus scopulorum*), and narrow-leaf cottonwood (*Populus angustifolia*) are typically found in riparian areas, but are more common outside the study area. Soils are gravelly coarse sandy loams derived from weathered Pikes Peak granite (USDA Forest Service 1992). They are very well drained and erosive. Average annual precipitation is around 40 cm (USDA Forest Service 1992), and there is no persistent winter snowpack. Summer precipitation comes mainly in the form of erratic thunderstorms that produce much lightning.

The native inhabitants since at least 1700 were nomadic Utes (Cassells 1983), whose activity was probably confined to the river corridor and other reliable sources of water before 1880 (A. Kane, Pike NF Archaeologist, personal communication; De Lay 1989). Bison may have grazed in the area before 1880 (De Lay 1989). Euro-American settlement began in the Front Range after 1859, but did not become heavy until the 1870s–1890s. Logging and ranching began downstream from the Cheesman property during that period, and reached forests adjacent to Cheesman in the late 1890s. The Cheesman Lake property is presently owned and managed by Denver Water. The dam on the South Platte River that created the reservoir was completed in 1905 (Denver Water archives). At the same time, a six-strand barbed wire fence was erected around the entire property to exclude grazing. The forest on the property was not cut except for logging below the present water line. Grazing has been excluded since 1905, and was probably light before that time. There was little mining in the area, though placer claims did exist along the river where the reservoir now lies (Denver Water archives). Fire has been suppressed on the Cheesman Lake property since the time of dam construction. The only fire that burned significant acreage within the landscape since 1880 occurred in 1963, burning about 25 ha on the south end of the property before it was extinguished (Bill Newberry, Denver Water, personal communication).

Sampling Scheme

We mapped the Cheesman Lake landscape from 1:6000 color-infrared aerial photographs, drawing polygons around visually ecologically distinct areas. We classified polygons based on tree density and size distribution as either forested or nonforested (rocks, water, grass, riparian shrubs, less than 10 percent tree canopy cover), and estimated forest canopy cover by 10 percent classes. We were unable to differentiate between ponderosa pine and Douglas-fir on the color-infrared photographs. We scanned the polygon maps into digital form and georectified them by overlaying them onto the digital orthoquarterquad using ArcInfo.

We randomly selected 10 percent of the mapped forested polygons from our GIS database for sampling. Sampled polygons varied in size, from less than 0.1 ha to 33 ha; the average size of sampled forested polygons was 3.3 ± 2.0 ha. The total area of the sampled polygons, 680 ha, represents 22 percent of the total land area. Within each polygon, we selected the five living trees that appeared to be the oldest, or to represent the oldest cohort, and cored them at approximately 35 cm above the ground. Where trees were rotten, or there appeared to be more than one older cohort, we took more than five cores and included those ages in our analysis. In some stands, all of the trees appeared to be old; in other stands, a few old trees were present among younger trees. Where there were no old trees at all, we selected the dominants. The correlation between size and age on our sites is very poor. Some of the largest trees on favorable sites are only around 150 years old, while some of the oldest trees on dry slopes range from 25 to 35 cm d.b.h.

Ponderosa pine trees begin to take on distinctive characteristics at around 200 years of age (Kaufmann 1996). We used these characteristics to identify the old trees in stands (fig. 1). Old ponderosa pines have smoother, lighter colored bark, smaller live crown ratio, and more flattened crowns than young trees. Dead tops are common, as are fire scars, lightning scars, and other injuries. Old Douglas-fir trees also have distinctive characteristics, including thick, deeply fissured bark and relatively small live crown ratios; crowns tend to be sparser, rounder, and less conical than those of young trees. As with pines, dead tops and injuries become more common over time.

Tree-Ring Dating

We used standard dendrochronological cross-dating techniques to age our sampled trees (Swetnam and others 1985; Stokes and Smiley 1968). Cores were surfaced with progressively finer grits of sandpaper to 400 grit and cross-dated under a stereomicroscope, using the chronology developed

Figure 1—This ponderosa pine germinated around 1447 A.D. It exhibits classic old-growth characteristics. The other trees sampled on this dry, south-facing slope dated to the mid-18th century.

for dating the Cheesman Lake fire history (Brown and others 1999). It extends from 1107 A.D. to 2000 A.D.

We cored trees as close as possible to the pith. We determined the approximate germination date of sampled trees by estimating rings to the pith using the concentric circle method (Applequist 1958), and we estimated the number of years required for a tree to grow to coring height of 35 cm based on aging destructively sampled small trees on good and poor sites (Kaufmann and others 2000a; Mast and others 1998). Growth times to 35 cm ranged from 5 years for ponderosa pines on good sites to 18 years for Douglas-fir on poor sites.

Our fire history was constructed from fire-scarred living trees and remnant material sampled at sites subjectively located across the Cheesman lake landscape, and is reported in Brown and others (1999). Fire scars were sampled by cutting a cross-section or partial cross-section through the fire-scarred face of a tree, snag, or log with a chainsaw. Sections were surfaced and cross-dated using the chronology developed for Cheesman Lake.

Old Growth and Fire History

Historical Fire Regime

We believe that individual historical fire events at Cheesman Lake burned with variable intensity. Under less extreme conditions, surface fires consumed grassy fuels and scarred but did not kill mature trees. Buried fire scars on some samples are evidence of fires light enough to scar but not kill trees 2–5 cm in diameter. Where fuels were heavy or in high winds, fires could be intense enough to kill all of the trees in an area. Stands on south- and west-facing slopes may have been so sparsely forested that fires killed trees only under extreme conditions. Modeling with FARSITE indicates that considerable wind is required to advance a fire in this environment even after decades of fire suppression, and is particularly necessary for crown fire (unpublished data; see Kaufmann and others, this proceedings). Fuel buildups from a series of wet years followed by dry weather probably also contributed to large, intense fires (Veblen and others 2000).

Seasonality of scars from the more widespread fires encompasses the entire growing season (mid-June to early September) and the dormant season, which could be either fall or spring, both usually dry periods (Brown and others 1999). Historically, fires that started early in the season may have burned across the landscape throughout the summer until snowfall put them out. Lightning is very common from April through September and per year ignites 10 to 12 fires, which are presently suppressed (Bill Newberry, Denver Water, personal communication). The Native American influence on historical ignitions is unknown.

The mean fire interval (MFI) at Cheesman Lake depends on the scale at which it is observed. The length of fire-free interval is inversely proportional to the size of area over which the intervals are assessed; larger areas have shorter intervals. Any given location on the Cheesman landscape might historically have gone as much as 128 years between fires. For the full period of analysis (1285 to 1963 A.D.) over the entire landscape, the MFI was 9.2 years ±7 years (Brown and others 1999). However, this includes fires of all intensities and many localized fires. When the fire scar sample area was divided into smaller areas of 0.5 to 2 km^2, the MFI for all individual areas between 1496 A.D. to 1880 A.D. was 50.0 years, ±17.2 years (Kaufmann and others, this proceedings). When we consider only the larger scale fires that burned more than 5 km^2, based on the distribution of fire scars, the MFI between 1496 and 1880 was 42.7 years, ±12.7 years, with a range of 27 to 65 years between fires (Kaufmann and others, this proceedings). Large-scale fires tended to alternate across the landscape; for example, the northern area that burned in 1696 did not burn in 1723. However, many areas that had burned in 1723 also burned in 1851, but not in 1820. Only the fire in 1631 scarred trees at every sampling location.

Figure 2 shows generalized minimum extents of the widespread fires based on locations where fire scars from that year were collected (Brown and others 1999). Many other fire years were recorded as far back as 1197 A.D., but sample depth declines dramatically before 1500 A.D. Note that fire scar sampling did not extend beyond the area mapped for the 1631 fire and was generally confined to

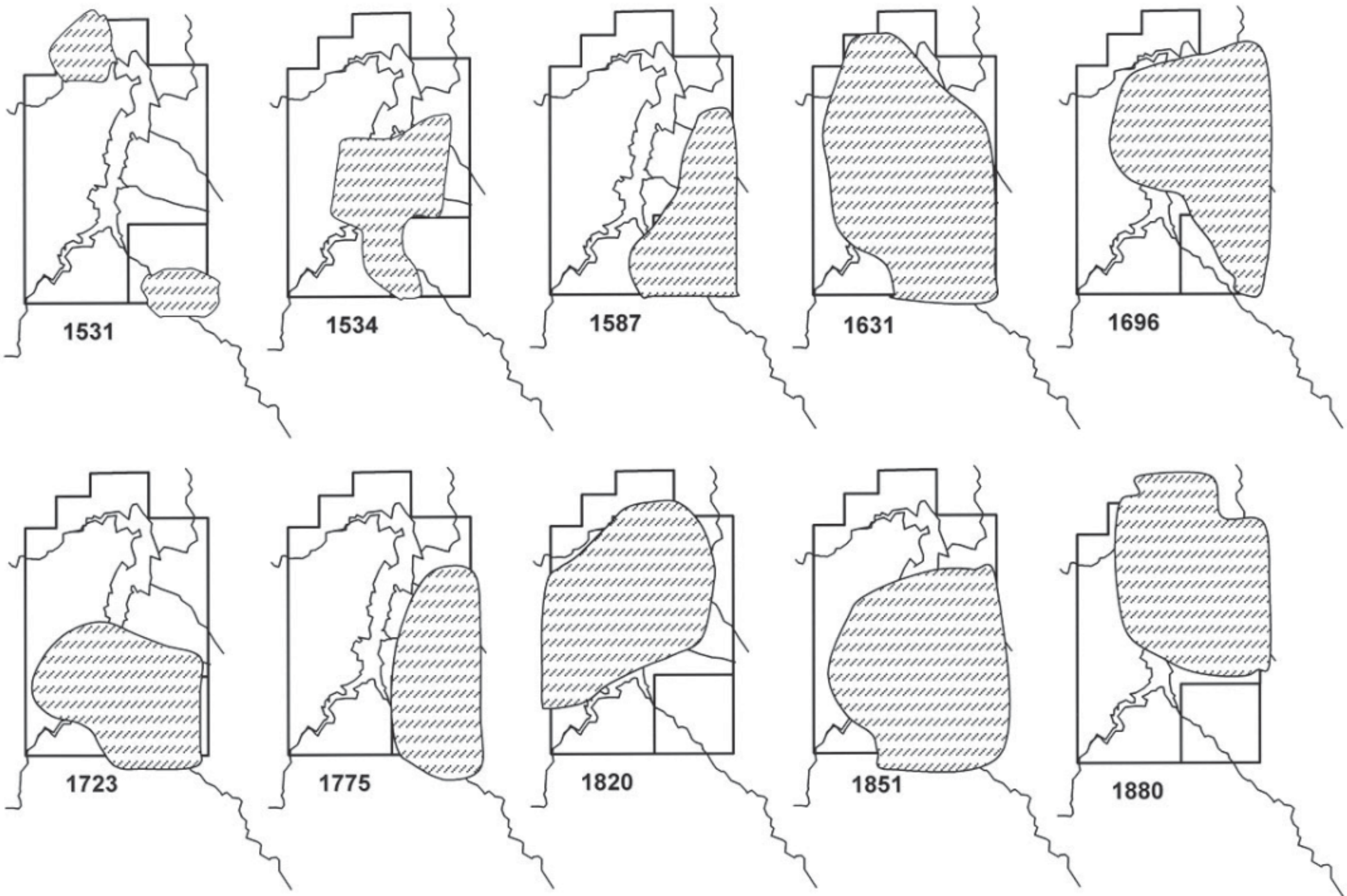

Figure 2—Generalized extents of large fires with stand-replacing components, based on the locations of trees sampled for the fire history. Compare these with polygons postdating these fires in figure 5. Because fire scar sampling was confined to the Cheesman property, the actual areas of the fires may be larger than shown. (Maps are developed from Brown and others 1999.)

the Cheesman property, except for an area southeast of the Cheesman boundary, so actual fire areas were probably larger than those shown. The Cheesman Lake landscape seems to encompass the total area of some historical fires and not of others. The South Platte River does not seem to have been a significant firebreak, as fires predating the reservoir were recorded on both sides of the river. Historical photos and predam survey maps (courtesy of Denver Water archives) show that the valley where Cheesman Lake currently lies was fairly flat and grassy, so fire probably spread rapidly across it. Suppression of spreading fires outside of the property may have had as much effect on the decline in fire frequency in the 20th century as suppression within the property.

Age Distribution and Old Growth

Old growth is a complex ecological concept, based not only on the ages of the oldest trees, but also on stand structural and functional characteristics (Kaufmann and others 1992; Moir 1992). Old-growth stands contribute to biodiversity by providing habitat for wildlife and late successional plant species. Ponderosa pine stands begin to take on old-growth characteristics around 200 years of age. Old stands tend to have more snags and coarse woody debris, large trees for the site conditions, old trees, and a variety of trees of younger ages. Such stands evolved under presettlement disturbance regimes. Even in relatively undisturbed places such as Cheesman Lake, fire suppression has changed the structure of old stands by allowing ingrowth of young trees, especially Douglas-fir. Where logging has occurred, the distribution and prevalence of old-growth stands in presettlement times is difficult to determine (Mast and others 1999; Regan 1997).

Our data indicate that old stands were common and widely distributed in unlogged montane forests around Cheesman Lake. Figure 3 shows the locations of our sampled polygons classified into age categories based on the age of the oldest tree collected. At Cheesman Lake, where the historical age structure persists, 70 percent of the sampled polygons had trees 200 years old or older; 30 percent of sampled polygons had trees more than 400 years of age. In 30 percent of polygons, all trees sampled were less than 200 years old, indicating more recent stand-replacing disturbance in a

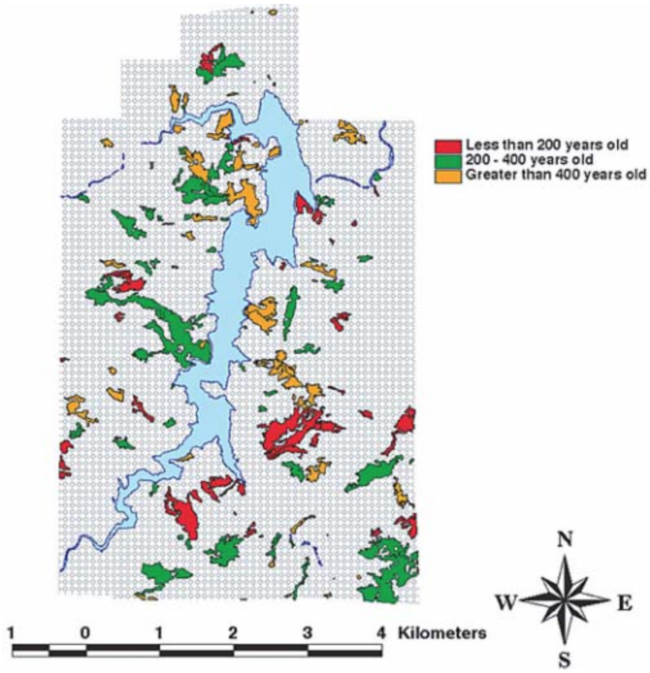

Figure 3—Distribution of old growth among sampled stands, based on the age of the oldest tree cored in the polygon. Thirty percent of sampled stands had all trees younger than 200 years; 40 percent had the oldest trees between 200 and 400 years old, and 30 percent had the oldest trees older than 400 years.

third of the sampled landscape. These younger stands were concentrated in the southeast part of the property where the 1851 fire burned. The oldest stands were concentrated in the northwest quadrant of the property, though stands with old trees were fairly evenly distributed over the landscape.

Though old trees were widespread among our samples, very old trees were relatively uncommon, reflecting the fact that many stands have a few old trees surrounded by trees of younger cohorts. Only 10 percent of the sampled trees were more than 400 years old; 40 percent of trees were 200–400 years old. Trees less than 200 years old constituted 50 percent of the trees sampled as the oldest in the stands. Douglas-fir has always been a component of the landscape, though it is probably more abundant and widespread now than in the past. Of 1,279 cores sampled and dated from the Cheesman Lake landscape, 86 percent (1,108) were ponderosa pine and 13 percent (171) were Douglas-fir. The estimated germination year of the oldest living ponderosa pine collected was 1395 A.D. The estimated germination year of the oldest Douglas-fir collected was 1459 A.D. The oldest trees in most polygons were ponderosa pines, but 25 polygons out of 224 had a Douglas-fir as the oldest tree.

The oldest trees in most stands appear to have an upper age limit postdating a documented fire in the area. These stands with an age cap are the most common stand structure in the landscape (Kaufmann and others, this proceedings). However, some stands seemed to have no identifiable initiating event. The ages of sampled trees in these stands were continuous over a long time, with no concentrations of ages (for example, truly uneven-aged). Some were more than 600 years old. We call these stands persistent old growth (Kaufmann and others, this proceedings; fig. 4). Such stands apparently have not experienced a stand-replacing event in the lifetimes of the trees. Their structure seems to be regulated by microsite events such as heartrot, windthrow, individual tree insect infestation, or mistletoe. Fire scars are usually present in these stands, indicating periodic surface fire, but age distributions do not suggest a stand-initiating event that coincides with any recorded fire in the area. Such stands constituted 16 percent of sampled polygons. Persistent old growth occurs in all parts of the landscape, but is concentrated in the northwestern quadrant. Some stands classified as persistent old growth may in fact postdate early fires for which no scar record exists. Further analysis is required to determine what factors allow the persistence of such stands.

Figure 4—Persistent old growth. These stands have very old trees with no evidence of a stand-initiating event. They are characterized by a continuum of tree ages, including trees more than 400 years of age, old trees dying from microscale disturbances such as mistletoe infestation, windthrow, or lightning strike, and coarse woody debris on the ground. Fire scars are sometimes present, indicating that surface fires have burned in the stands without killing all of the trees.

Age Structure and Fire Regime

Determining the ages of the oldest trees in a stand and comparing them to known fire dates in the area gives us some idea of where stand-replacing fire burned in individual fire years. Age structure has been used to approximate the dates of stand-replacing fires elsewhere (Mast and others 1999; Arno and others 1995; Goldblum and Veblen 1992; Duncan and Stewart 1991). Concentrations of trees that postdate known fires suggest the forest in that location burned in a stand-replacing fire. In an attempt to determine the locations of stand-replacing components of known fires, we compared the estimated germination dates of the oldest trees sampled in the polygons with the dates of fire scar records in the vicinity. Of 21 fires recorded over the entire landscape from 1531 to 1880, 16 appeared to have a crown fire component detectable in the age structure of sampled trees. Seventy-one percent of sampled polygons clearly postdated one of seven major fire events over the last 470 years (1531, 1534, 1587, 1631, 1723, 1851, 1880), suggesting that those events had a significant stand-replacing component (fig. 5).

This approach has limitations that become more restrictive to interpretation the further back in time we go. We can only assess the residual polygons, in other words, those that have survived subsequent fires. For example, a polygon in which the forest appears to postdate the 1851 fire might also have burned as a stand-replacing fire in 1631, but evidence of the stand that initiated after 1631 has been destroyed by more recent stand-replacing fire. Therefore, figure 5 is a conservative estimate of the number of polygons that postdate known historical fires. Only polygons burned in 1851 and 1880 have not been burned over by subsequent fires. Similarly, evidence of surface fire in the form of scarred trees may have been obliterated by subsequent disturbances, so the mapped extents of the fires based on scars is also conservative. It is also certain that the mosaic of forest stands of different age and size structure (which we mapped as polygons) has shifted over time as the result of disturbances. The polygon map of 1630 probably would have looked very different from the one of today.

Locations of sampled polygons that appear to postdate known stand-initiating fires are shown in figure 6. The earliest scar-recorded fire that clearly predated some sampled stands was in 1531 A.D. (figs. 2 and 6). This fire burned in both the far northern and southern parts of the Cheesman Lake landscape. Evidence of this fire in the central portion of the landscape may have been obliterated by subsequent fires. Nine sampled polygons located within the fire boundary (as defined by the locations of fire-scarred samples) postdated this fire, yet initiated before the next known fire in the area. A fire in 1534 A.D. burned the central and south-central two-thirds of the Cheesman Lake landscape. The oldest trees in 22 polygons postdated this fire (fig. 5). A fire in 1587 burned the southeastern part of the Cheesman Lake landscape. Ten polygons clearly postdated this fire.

The fire in 1631 was recorded by scars at all of the sampling locations throughout the landscape, and 36 polygons appear to postdate this fire, mostly in the central and northern part of the landscape (figs. 5 and 6). Scars from a fire in 1696 were widespread over the northern and central

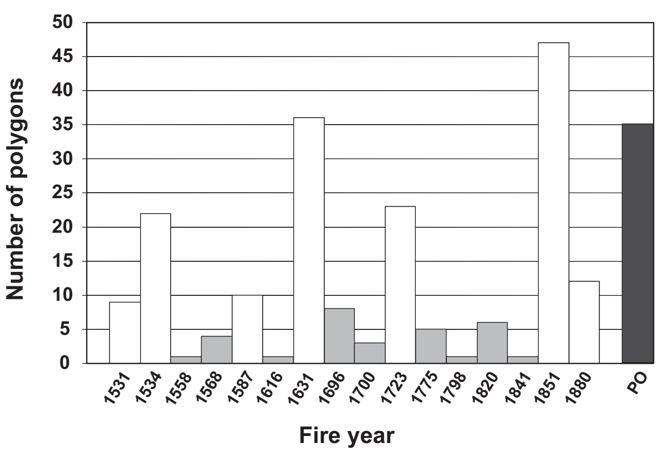

Figure 5—The number of sampled polygons where the oldest tree postdated a known fire in the area, by fire year. Seventy-one percent of sampled polygons clearly postdated one of seven major fires over the last 470 years (1531, 1534, 1587, 1631, 1723, 1851, 1880), indicating that those events had a significant stand-replacing component. A total of 21 fires were recorded by fire scars during that period, of which 16 seem to have had some crown fire component. Fifteen percent of the polygons sampled were classified as persistent old growth, with no clear stand-initiating event evident in the ages of the oldest trees.

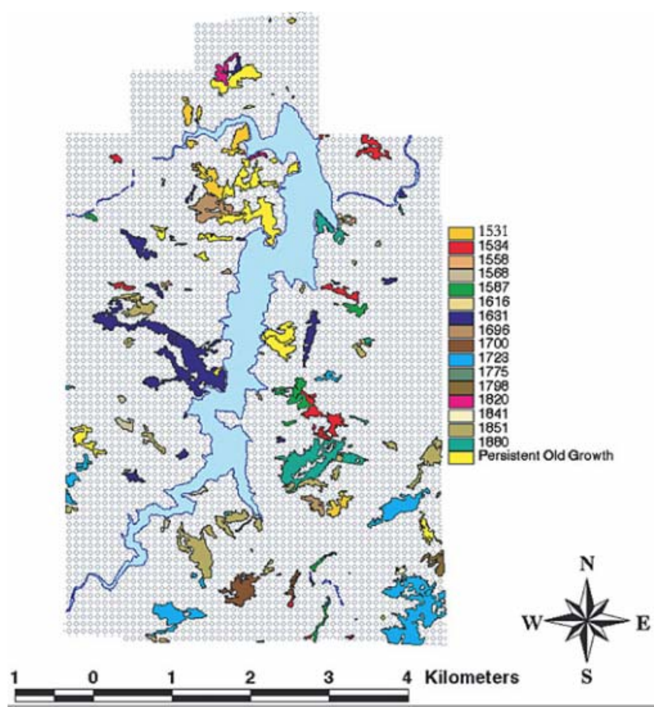

Figure 6—We mapped the distribution of apparent stand-replacing fire events in the sampled polygons. This map shows the recorded fire that probably predated the oldest trees in each sampled polygon. Stands that did not appear to have a clear initiating event are labeled *persistent old growth*.

part of the landscape, but relatively few stands (except for some on the northwest side of the river) postdate this fire. The fire in 1631 may have left large areas unsusceptible to stand-replacing fire 1696. A fire in 1723 burned the southern half of the landscape, on both sides of the river. Much of the evidence of stand initiation after this fire may have been obliterated by the fire in 1851, which burned much of the same area. A fire in 1775 scarred widely distributed trees on the east side of the river, but apparently had only a small stand-replacing component. Similarly, a fire on the north half of the landscape in 1820 scarred widespread trees but initiated few stands. These may have been less intense surface fires in the very open landscape left in the wake of fires in 1631 and 1723. This was also a fairly cool and wet period (P. Brown, personal communication).

Fire-created openings persisted for variable lengths of time, depending on when conditions favored tree establishment in subsequent years (Kaufmann and others 2000a), and whether the location experienced another disturbance before the trees were large enough to survive it. Tobler (2000) sampled vegetation in openings at Cheesman Lake and found coarse woody debris in all but one of them, indicating that they had been forested in the past. Death dates on the down logs indicate that the openings were created by a fire in 1851, and remained unforested 149 years later. Fire scars found nearby suggest that the northern part of the area may have burned again in 1880, slowing tree regeneration.

When we compared the ages of the oldest trees sampled in the polygons with the dates of nearby fire scars, we saw highly variable intervals between fire and tree establishment. Time to regeneration in sampled polygons ranged between 0 and 95 years, but generally occurred within 15 to 30 years, with an overall average of 18 years (table 1). Tree ages collected in other plots in the southeast part of the property were also compared to fire scar dates in the vicinity. Some of them appear to have regenerated immediately, while others seem to have remained unforested for as many as 107 years (Kaufmann and others 2000b). We believe this phenomenon is due to climate and episodic periods of regeneration (Kaufmann and others 2000a). If a fire occurred at the beginning of a period favorable for tree establishment, even south-facing slopes could regenerate quickly, but if the fire occurred at the end of such a period, regeneration would be limited until the next favorable period.

The mean regeneration period after the seven major fires with stand-replacing components was 22 years. Regeneration was slower in the northwest quadrant of the landscape, where the oldest stands were concentrated. Time to regeneration there averaged 30 years after a fire, with a range of 0 to 93 years, while the southeast quadrant averaged only 12 years to tree regeneration, with a range of 0 to 57 years. Interpretation of these data is complicated by how long ago some of the recruitment occurred. Establishment time for both species was highly variable, but where Douglas-fir was among the oldest trees, it established more quickly on average, within 15 years, while ponderosa pine averaged 21 years to establish following fire. However, Douglas-fir was more often found on north- and east-facing slopes where establishment conditions were probably better.

The fire in 1851 burned the southern two-thirds of the landscape. Openings created by this fire still persist on south- and west-facing slopes (fig. 7). Vegetation in these openings is mainly bunchgrass, or bunchgrass and shrubs, primarily mountain mahogany (*Cerocarpus montanus*) and wax currant (*Ribes cereum*). Trees began to invade these openings between 1880 and 1920. The northern part of

Table 1—Years to tree regeneration following historical stand-replacing fires.

Stand-replacing fire year	Mean years to regeneration	Standard deviation
1531	32.9	21.4
1534	25.6	18.7
1558	16.0	0
1568	16.8	14.7
1587	18.3	10.7
1616	11.0	0
1631	32.5	17.4
1696	24.6	27.3
1700	14.7	8.5
1723	20.4	17.6
1775	25.6	20.8
1798	20.5	2.5
1820	6.2	20.6
1841	0	0
1851	13.6	11.6
1880	10.1	7.3
Average time between fire and regeneration (all stand-replacing years) range: 0 to 33	18.0	9.3

Figure 7—These openings were created by the 1851 fire, based on death dates from the coarse woody debris that is scattered throughout. Vegetation in openings may be grassy or a combination of grass and shrubs, primarily *Cerocarpus montanus* and *Ribes cereum*. Parts of these openings also may have burned in the 1880 fire.

these openings may have burned again in 1880, as scars from this fire spatially overlap the ones from 1851. Areas adjacent to the present openings also appear to have burned in stand-replacing fires in 1851, but have now regenerated to young forest on north- and east-facing slopes. Shrubs common in the openings persist in the understories.

Fire, Old Growth, and Restoration Efforts

Our data suggest that old trees and old stands were historically common in the montane zone of the Colorado Front Range. The presettlement fire regime was spatially and temporally a mixture of surface fire and stand-replacing fire, which maintained open stands interspersed with areas that remained treeless for decades at a time. Logging, grazing, and fire suppression have altered this condition in much of the Front Range, creating a forest that is younger, denser, and more continuous than the presettlement norm and converting fire behavior from a mixed severity fire regime to a crown fire regime. Recent rapid growth of the human population has created an extensive urban-wildland interface in the Front Range that is at risk from wildfire, and water sources are also in danger from postfire erosion (Culver and others, this proceedings).

Restoring the landscape to an ecologically sustainable condition by recreating a more historical forest structure may mitigate the risk of intense, large-scale wildfires and subsequent erosion (Kaufmann and others 2000b). Restoration efforts should consider the role and distribution of old growth on the landscape, and its relationship to disturbance regime. It is important to conserve the old trees that still exist on the landscape and to select trees for retention that will grow into an old-growth condition in a reasonable amount of time (Mast and others 1999; Fulé and others 1997; Kaufmann and others 1994). It is also important to restore a disturbance regime that allows the old trees to persist, for example, periodic surface fire that removes ingrowth without killing the old trees. Mechanical thinning may be required to allow for prescribed fires of appropriate intensities, especially in areas with considerable young ingrowth. However, restoring an ecologically sustainable landscape with a historical structure requires creation and maintenance of unforested openings, so treatments including both thinning and prescribed fires must be locally intense enough to replace some stands at longer intervals.

Front Range forests are not as productive, in terms of trees or understory, as those in the Southwest, and their historical fire regime was less frequent and more variable in intensity, so the Southwestern model cannot be correctly applied here (Veblen and others 2000; Brown and Sieg 1996; Swetnam and Baisan 1996; Goldblum and Veblen 1992). Because fire and tree regeneration historically occurred at long, often coincident intervals, the timing of restoration efforts may be important for successful maintenance of openings.

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Potential Fire Behavior Is Reduced Following Forest Restoration Treatments

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Abstract—Potential fire behavior was compared under dry, windy weather conditions in 12 ponderosa pine stands treated with alternative thinning prescriptions in the wildland/urban interface of Flagstaff, Arizona. Prior to thinning, stands averaged 474 trees/acre, 158 ft^2/acre basal area, crown bulk density 0.0045 lb/ft^3, and crown base height 19.2 ft. Three thinning treatments differing in residual tree density were applied to each of three stands (total of nine treated, three control). Treatments were based on historic forest structure prior to Euro-American settlement and disruption of the frequent fire regime (*circa* 1876). Thinning reduced stand densities 77–88 percent, basal areas 35–66 percent, crown bulk densities 24–48 percent, and raised crown base height an average of 11 ft. Before thinning, simulated fire behavior under the 97^{th} percentile of June fire weather conditions was predicted to be intense but controllable (5.4 ft flame lengths). However, active or passive crownfires were simulated using crown base heights in the lowest quintile (20 percent) or winds gusting to 30 mph, representing the fuel ladders and wind gusts that are important for initiating crown burning. Under the identical conditions after thinning, all three treatments resisted crown burning. The degree of resistance was related to thinning intensity. It is crucial to remove thinning slash fuels through prescribed burning or other means. If not removed, slash fuels can cause crownfire behavior in the thinned stands under severe wildfire conditions. Finally, the crownfire resistance achieved through thinning will deteriorate over time unless maintenance burning and/or thinning is continued.

Introduction

Flagstaff, Arizona, is located at the northwestern end of the largest contiguous ponderosa pine forest in the world. Increased fire intensity and severity are major concerns around Flagstaff and generally in southwestern ponderosa pine forests (Swetnam and Betancourt 1998), due to the regional increase in surface and canopy fuels following a century or more of fire exclusion and other human-caused disruptions of ecological processes (Cooper 1960; Covington and others 1994). In 1996, the two largest wildfires in the history of the Coconino National Forest burned a few miles north of Flagstaff. Seeking to prevent such fires from burning into developed areas, a collaborative group called the Grand Canyon Forests Partnership was formed to restore ecosystem health, reduce catastrophic fires, and improve economic benefits and management on public lands (GCFP 1998).

Tree thinning, prescribed burning, and/or other fuel reduction methods can reduce the hazard of intense fires (for example, Van Wagtendonk 1996; Graham and others 1999; Agee and others 2000). Using these techniques to restore a regime of frequent, low-intensity fires and tree structures approximating the relatively open presettlement forest stands should, in theory, simultaneously address the Partnership's goals. These treatments have potential for improving ecosystem health (Kolb and others 1994), reducing fire hazard (Covington and others 1997), and offering some economic benefits through forest product removal (Larson and Mirth 1998). Actually achieving this array of outcomes in complex ecosystems and social systems is difficult, requiring choices among competing interests. For example, Scott (1998a) compared the economic, aesthetic, ecological, and fire behavior tradeoffs of a set of alternative fuel treatments in a western Montana ponderosa pine forest. Kalabokidis and Omi (1998) carried out a similar analysis in a Colorado lodgepole pine forest.

The Grand Canyon Forests Partnership's initial wildland/urban interface experimental treatments were started in 1998 in cooperation with the Coconino National Forest and Rocky Mountain Research Station. The experiments had multiple objectives, but our focus in this paper is only on the treatment effects on potential fire behavior. The greatest concern in the wildland/urban interface is crownfire, both "passive" crownfire (tree torching) and "active" crownfire (fire spreading through the canopy). Crownfires spread rapidly (Rothermel 1991), resist control by hand crews and often mechanical or aerial equipment (Pyne and others 1996), and threaten structures with intense heat and firebrand showers (Cohen 2000).

Several complementary actions can improve the ability of communities to resist fire hazards to lives and property, including enhanced firefighting resources, improved access routes and rural address systems, heightened public awareness, reduction of structure flammability (Cohen 2000), and reduction of forest susceptibility to crownfire. The forest treatments discussed here address this latter factor. Local

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initiatives are under way to enhance other fire-resistance factors in the Flagstaff wildland/urban interface.

Until recently, fire behavior modeling tools such as BEHAVE (Andrews 1986) simulated only surface fire behavior. New tools such as FARSITE (Finney 1998) and Nexus (Scott 1999) have greatly increased the ease with which many aspects of crownfire behavior can be modeled and compared. It is important not to attach too much specificity to crownfire behavior predictions: the fundamental reason that crownfire modeling has advanced slowly is that crownfires are rare and occur in extraordinarily complex weather and fuel environments (Rothermel 1991). With caveats, however, simulations provide useful insights into the relative differences between treatments and the relative sensitivity of crownfire behavior to different variables.

From previous simulations with FARSITE and Nexus, as well as from literature results (Van Wagtendonk 1996; Scott 1998a,b), we recognized that simulations often resulted in outputs that appeared contrary to actual wildfire experience. In particular, simulated fires using our fuel and weather conditions proved nearly impossible to crown using realistic data, even though real fires had crowned under similar or even less severe conditions. One possible solution was to manipulate model output with adjustment factors. However, this method is unsatisfactory for modelers and their audiences, who would prefer to use well-supported numbers.

We tried a different approach. Both with weather and fuel data, we reasoned that "average" conditions were a misrepresentation of the real forest situation. For instance, to cross the threshold into tree torching, surface flame lengths must preheat the branches and leaves close to the bottom of the crown. Achieving this transition in simulations has been difficult because the average crown base height is often a relatively high value (15–30 ft). The fuel ladders, surface fuel jackpots, and wind gusts that facilitate the transition to the crown in real fires are not accounted for when uniform averages are used.

Taking the variability of the data into account could help simulate more realistic fire behavior, but which fraction of the variability is important? A single low crown is probably insufficient to initiate a crownfire, but crownfires can start and be sustained in strong winds even with much less than 50 percent of the stand in a crownfire-susceptible condition. We chose to rank the data by quintiles—20 percent groups—and compare fire behavior and treatment effects on both the stand averages and the susceptible quintiles, suggesting that the fire behavior in the vulnerable quintiles may be important in triggering intense fires.

Methods

Treatments

The Grand Canyon Forests Partnership chose to compare three treatments differing in residual tree density. All treatments were based on the presettlement pattern of tree structure as inferred from: (1) living trees of presettlement origin, characterized by larger size and yellowed bark (White 1985; Covington and Moore 1994), and (2) remnant material from snags, logs, and stumps of presettlement origin, which were well-conserved in the dry environment in the absence of fire (Dieterich 1980; Fulé and others 1997; Covington and others 1997; Mast and others 1999). All living presettlement trees were retained. In addition, wherever evidence of remnant presettlement material was encountered, several of the largest postsettlement trees within 30 ft were retained as replacements. If suitable trees were not found within 30 ft, the search radius was extended to 60 ft. The three thinning treatments each had a different replacement tree density:

- 1.5-3 replacements: replace each remnant with 1.5 trees (in other words, 3 replacements per every 2 remnants) if the replacements were 16 inches d.b.h. or larger, otherwise replace each remnant with 3 trees. Because relatively few greater than 16 inches postsettlement trees were encountered in any of the sites, all the thinning treatments tended to retain the higher number of replacement. The 1.5-3 treatment, called "full restoration," reduced tree density most closely to presettlement levels.
- 2-4 replacements: replace remnants with 2 trees greater than 16 inches d.b.h., otherwise 4 trees.
- 3-6 replacements: replace remnants with 3 trees greater than 16 inches d.b.h., otherwise 6 trees.
- Control treatment: no thinning, no burning.

Study Sites

The treatments were tested on three experimental blocks in or adjacent to the Fort Valley Experimental Forest, approximately 15 km NW of Flagstaff, Arizona (fig. 1). Each block contained a 35-acre replicate of each of the three thinning levels and a control. The study area is at 7,400 ft elevation with gentle topography and a cool, subhumid climate (Avery and others 1976). Mean annual precipitation is 57 cm, with approximately half occurring as snow. The remainder occurs as summer monsoonal rains following the spring/early summer drought. Soils are of volcanic origin, a fine montmorillonitic complex of frigid Typic Argiboroll and Mollic Eutroboralf (Mast and others 1999). Experimental

Figure 1—Prescribed burning in Fort Valley treatment area, May 12, 2000.

blocks were laid out in cooperation with Forest Service staff, subject to constraints of other experimental studies and wildlife habitat. As a result, the treatment units in experimental blocks 1 and 2 could not be contiguous. All treatments were randomly assigned.

The timing and method of treatment differed in the experimental blocks due to economic constraints, primarily the very low value of the material removed, and to the Partnership's intention to make the site available to different operators. Thinning of the blocks began in November 1998 and was completed in September 1999. Blocks 1 and 2 were thinned with a mechanical feller and limbed at the tree, resulting in broadcast slash fuels. Block 3 was thinned in a whole-tree harvesting operation, resulting in slash piles. Piles in block 3 were burned in February 2000. All blocks were scheduled for broadcast burning in the spring or fall 2000.

Measurements

Twenty experimental block (EB) plots were established on a 60-m grid in each of the 12 units. Plot centers were permanently marked with iron stakes at ground level and slope and aspect were recorded. Overstory trees over breast height (bh, 4.5 ft) were measured on a 0.1 acre (37 ft radius) circular fixed-area plot. Species, condition (1-living, 2-declining, 3-recent snag, 4-loose bark snag, 5-clean snag, 6-snag broken above bh, 7-snag broken below bh, 8-downed, 9-cut stump), and d.b.h., were recorded for all live and dead trees over breast height, as well as for stumps and downed trees that surpassed breast height while alive. Tree heights and average crown base height per plot were measured. Trees below breast height and shrubs were tallied by condition class and by three height classes (0–15.7, 15.8–31.5, and 31.6–54 inches) on a nested 0.025 acre (18.5 ft radius) subplot. Shrubs over breast height were also measured. Herbaceous plants and canopy cover (vertical projection) were measured along a 164-ft line transect oriented up- and down-slope. Point intercept measurements were recorded every 11.8 inches along each transect. Dead woody biomass and forest floor material were measured on a 50 ft planar transect in a random direction from each plot center. Fuels were measured by diameter/moisture timelag classes (1H timelag = 0–0.25 inch diameter, 10H = 0.25–1 inch, 100 H = 1–3 inches, 1000H = over 3 inches, sound (S) and rotten (R) categories). Woody debris biomass was calculated using procedures in Brown (1974) and Sackett (1980). Forest floor depth measurements were converted to loading (Mg/ha) using equations from Ffolliott and others (1976). Plots were originally measured from August through November 1998. After thinning, preburn fuels were measured on the same transects in October 1999.

Modeling

Fire behavior was modeled with the Nexus Fire Behavior and Hazard Assessment System (Scott and Reinhardt 1999). As described by Scott (1998a, 1999), Nexus integrates models of surface fire behavior (Rothermel 1972) with crown fire transition (Van Wagner 1977) and crown fire spread (Rothermel 1991). Nexus is similar to the landscape fire behavior modeling program FARSITE (Finney 1998) in that both link the same set of surface and crownfire models. However, Nexus is better suited for comparing fire hazards under alternative conditions because environmental and fuel factors are kept constant for each simulation, rather than changing continuously with time and location, as in FARSITE.

Custom fire behavior fuel models were developed and tested with the NEWMDL and TSTMDL modules of BEHAVE (Andrews 1986). Pretreatment fuel models were modified from the standard fire behavior fuel model 9, "hardwood litter" (Anderson 1982). Postthinning fuel models were modified from standard model 11, "light slash." Future fuels, after thinning and burning, are likely to have reduced woody fuel loads and increased herbaceous fuels. A hypothetical future fuel model was developed by modifying standard model 2, "timber (grass and understory)." The predicted future herbaceous fuel load was 200 lbs/acre, based on a basal area/herbaceous production relationship developed in northern Arizona (Brown and others 1974).

Crown fuels were estimated with locally developed allometric equations for ponderosa pine shown in table 1. Crown volume was estimated using averages of maximum tree height (top of the canopy) and crown base height (bottom of the canopy). Crown bulk density was calculated as crown biomass divided by crown volume. This procedure is straightforward and appears to adequately represent the canopy fuels actually available in a ponderosa pine crown fire. Alternative methods of crown fuel estimation can lead to substantially different numerical values, so density values in different studies may not be directly comparable. The situation is further complicated by the relatively high sensitivity of crownfire behavior modeling to canopy bulk density.

Fire weather extremes representing the 90^{th} and 97^{th} percentiles of low fuel moisture, high winds, and high temperature were calculated from 30 years of data on the Coconino National Forest using the FireFamily Plus program (Bradshaw and Brittain 1999). Weather values were calculated for the entire fire season (April 23 to October 16) as well as for June, historically the month with the most severe fire weather (table 2). Fire behavior information from the two largest wildfires on the Coconino, the 1996 Horseshoe (May) and Hochderffer (June) fires, was used to estimate wind gusts during periods of extreme fire behavior

Table 1—Allometric equations for ponderosa pine foliage and fine branches.

Variable	Equation	R^2
Total foliage	ln(biomass, kg) = –3.9274 + 1.9654 ln(d.b.h., cm)	0.96
Needle-bearing twigs	ln(biomass, kg) = –4.5478 + 1.7352 ln(d.b.h., cm)	0.85
0-0.63 cm branches	ln(biomass, kg) = –4.3268 + 1.4172 ln(d.b.h., cm)	0.57

Table 2—Fuel moisture, wind, and temperature for the Coconino National Forest, 1970–1999. The 90th and 97th percentiles are shown for the entire fire season (April 23 to October 16) and for the month of June.

Variable	Fire season 90th percentile	Fire season 97th percentile	June 90th percentile	June 97th percentile
1 H moisture (percent)	3.2	3.0	2.3	2.2
10 H moisture (percent)	4.4	4.0	3.0	3.0
100 H moisture (percent)	7.2	6.5	5.0	4.7
Wind speed (mph)	17.7	22.4	20.0	25
Temperature (°F)	82	82	89	89

(McCoy 1996). Wind gusts to 40 mph and sustained winds of 30 mph were observed on these fires. The 30-year fire weather record also shows that winds of 30 mph or more were recorded in the 1,300 hours observation on approximately 1 percent of June days.

Results

Prior to treatment, forest structural conditions were similar across the study sites (table 3). Basal area ranged from 148.5 to 167.7 ft^2/acre, while tree density was more variable (386.7 to 603.9 trees/acre). Average stand heights were within 7 ft of each other across the sites (67.2 to 73.9 ft) and average crown base heights were within approximately 4 ft (17.4 to 21.5 ft). Crown bulk density values averaged 0.064 to 0.083 kg/m^3, similar to values reported by Scott (1998a) in a Montana ponderosa forest. Thinning reduced tree density and biomass most strongly in the full restoration (1.5-3) treatment and least in the 3-6 treatment, as expected. Postthinning densities ranged from 56.8 to 98.3 trees/acre, an average reduction of over 396 trees/acre (77 percent to 88 percent of trees removed). Because the largest trees were retained, however, basal area and crown biomass decreased by much smaller proportions. Postthinning basal area ranged from 44 percent to 65 percent of pretreatment values. Thinning reduced crown bulk density to 52 percent to 76 percent of pretreatment values. Crown base height was raised an average of 11 ft and the lowest quintile (20 percent) of crown base height was raised an average of 10.6 ft from 8.5 ft before thinning to 19.1 ft after thinning. Pretreatment surface fuels averaged 25.4 tons/acre, but the quintile (20 percent) of plots with the heaviest loading of less than 1000H fuels averaged 38.2 tons/acre (table 4). Postthinning fuels were surprisingly similar between the broadcast slash blocks (11 tons/acre of less than 1000H fuels) and the whole-tree harvested block (7.2 tons/acre of less than 1000H fuels). However, the primary difference was an extra 3.9 tons/acre of 1H fuels in the broadcast blocks, the fuel component most strongly influencing fire behavior. The broadcast blocks did have 80 percent more heavy fuel (more than 1000H and duff) loading, 18 versus 10 tons/acre. Burnout of these heavy fuels would be expected to lead to increased canopy and soil heating in the broadcast blocks after the passage of the flaming front.

Fires modeled in pretreatment conditions using the average stand values for crown bulk density and crown base height remained surface fires (table 5) even under the severe

Table 3—Forest stand structure and crown fuels at the Fort Valley study sites. See text for description of treatments. Prior to thinning, the lowest quintile of crown base heights averaged 8.5 feet. After thinning, the lowest quintile of crown base heights in the treated units averaged 19.1 feet.

	Control	Full restoration (1.5-3)	Intermediate (2-4)	Intermediate (3-6)
		Pretreatment		
Basal area (ft^2/acre)	164.3	151.5	167.7	148.5
Trees/acre	480.6	386.7	603.9	422.3
Crown bulk density (lb/ft^3)	0.0052	0.0040	0.0044	0.0042
Average crown base height (ft)	21.5	19.1	17.4	18.9
Minimum crown base height (ft)	11.5	8.2	6.6	8.4
Crown fuel load (ton/acre)	5.2	4.8	5.3	4.6
Stand height (ft)	67.2	73.9	73.2	69.7
		Postthinning		
Basal area (ft^2/acre)	164.3	67.8	77.7	97.2
Trees/acre	480.6	56.8	68.8	98.3
Crown bulk density (lb/ft^3)	0.0052	0.0021	0.0026	0.0032
Average crown base height (ft)	21.5	29.1	31.9	27.4
Minimum crown base height (ft)	11.5	12.6	17.0	18.0
Crown fuel load (ton/acre)	5.2	2.0	2.3	2.9
Stand height (ft)	67.2	73.9	73.2	69.7

Table 4—Surface fuel characteristics. Fuels were measured on the study sites except for the “hypothetical posttreatment fuels” (see text).

Description	1 H	10 H	100 H	Live	SAV	SAVLive	Depth	Moist. Ext.	Heat	1000 HS*	1000 HR*	Duff*
	ton/ac				*1/ft*		*ft*	*percent*	*BTU/lb*	*ton/ac*		
Pretreat average	2.9	0.8	2.3	0	2500	500	0.4	25	8000	5.9	4.8	8.7
Pretreat top 20 percent	4.3	1.7	5.9	0	2500	500	0.5	25	8000	11.3	4.8	10.2
Postthinning (broadcast slash)	7.2	1.2	2.6	0	1500	500	1.0	15	8000	7.1	3.9	7.0
Postthinning (whole-tree harvest, piled slash)	3.3	1.2	2.7	0	1500	500	1.0	15	8000	2.2	0.8	7.0
Hypothetical posttreatment fuels: grass and understory, modified FBFM 2)	2.0	1.0	0.5	0.1	3000	1500	0.5	15	8000	N/A	N/A	N/A

*These variables are not included in fire behavior fuel models.

Table 5—Fire behavior outputs using the average pretreatment fuel loads under the June 97th percentile weather conditions with 97th percentile winds (top), 30-mph winds and lowest quintile crown base height (center), and posttreatment crown fuels with 30-mph winds and lowest posttreatment quintile crown base height (bottom). Foliar moisture content was held constant at 100 percent, wind reduction factor was 0.3, and slope was 7 percent (study site average) for all simulations.

	Control	Full restoration (1.5-3)	Intermediate (2-4)	Intermediate (3-6)
	Pretreatment (June 97th percentile weather)			
Fire type	Surface	Surface	Surface	Surface
Crown percent burned	0	0	0	0
Rate of spread (ft/min)	28	28	28	28
Heat/area (BTU/ft^2)	491	491	491	491
Flame length (ft)	5.4	5.4	5.4	5.4
Crown fire outputs				
Torching index (mph)	54	49	45	48
Crowning index (mph)	28	34	32	33
	Pretreatment (June 97th percentile weather, 30-mph winds, lowest quintile crown base height)			
Fire type	Active	Passive	Passive	Passive
Crown percent burned	100	58	74	65
Rate of spread (ft/min)	128	90	105	97
Heat/area (BTU/ft^2)	2331	1473	1876	1569
Flame length (ft)	31.3	20.2	25.2	21.6
Crown fire outputs				
Torching index (mph)	23	23	23	23
Crowning index (mph)	28	34	32	33
	Posttreatment (June 97th percentile weather, 30-mph winds, lowest quintile crown base height)			
Fire type	Active	Surface	Surface	Surface
Crown fraction burned	100	0	0	0
Rate of spread (ft/min)	128	37	37	37
Heat/area (BTU/ft^2)	2331	491	491	491
Flame length (ft)	31.3	6.2	6.2	6.2
Crown fire outputs				
Torching index (mph)	23	49	49	49
Crowning index (mph)	28	55	47	40

fire weather conditions represented by the June 97th percentile (table 2). Fire behavior outputs were virtually identical across treatments prior to treatment, with only slight differences in the torching index (an estimate of the windspeed required to initiate tree torching or "passive" crown fire behavior) and the crowning index (an estimate of the windspeed required to support "active" fire spreading through the crown). The minor fluctuations in these two indices reflected the small differences in crown base height (important for the transition from surface fire to torching) and canopy bulk density (important for sustaining active crownfire). The torching index showed that a wind of at least 45 mph would have been needed to cause passive crownfire. If fire were already in the crown or entered from outside the stand, a windspeed of 28–34 mph would have sufficed to sustain active canopy burning. However, both indices were above the modeled 25 mph windspeed.

The simulated flame lengths, 5.4 ft, would have precluded direct attack by firefighters but mechanized equipment or indirect attack would have a high likelihood of successful suppression (Pyne and others 1996). The fact that modeled fires were amenable to suppression even under severe wildfire conditions is an accurate reflection of reality: the overwhelming majority of wildfires on the Coconino are contained below 10 acres (99.6 percent, fire records 1970–1999).

With 30 mph winds and/or the lowest quintile of crown base height, however, crownfire was simulated in the pretreatment sites. Keeping the crown base height at the average values but increasing wind to 30 mph led to conditional crownfire behavior (crownfire won't start, but could be sustained if it entered from outside the stand) in the stand with the highest crown bulk density. Lowering the crown base height to 8.5 ft, the average of the lowest pretreatment quintile, caused active or passive crownfire in all the sites at both the 25 and 30 mph windspeeds (table 5). Because 30 mph or higher wind gusts occur, and because at least one-fifth of the modeled forest is vulnerable to crownfire, these results may bridge the apparent contradiction between observed crownfire behavior and the unrealistically high windspeeds required for simulated crownfires using average stand characteristics.

Thinning treatments substantially reduced fire behavior under the same environmental circumstances. As shown in table 5, with the identical 30 mph wind and the lowest quintile of posttreatment crown base height, the simulated fire did not achieve any category of crown burning. All three treatments had the same torching index (49 mph) but the crowning index differed with canopy bulk density. The modeled 3-6 treatment could support conditional crownfire at windspeeds as low as 40 mph, while the modeled 1.5-3 treatment required 58 mph, 45 percent higher.

Although the comparison in table 4 shows a clear change in fire behavior due to the restoration treatments, the postthinning fuels are different than the pretreatment fuels. As the treatments progress, the slash fuels created by thinning are scheduled to be removed by prescribed burning. Mechanical means could also be used. But as long as these fuels remain in the stand, they present a threat of intense fire behavior. Active or passive fires crowned in all the simulated stands, including the treated sites, using either the broadcast or the whole-tree harvest slash fuel models in table 4. With standard fuel model 11, however, the control had active crownfire but the treated stands had only surface fires.

Fire behavior in future fuels, after removal of the slash, will probably be influenced by a higher herbaceous component. Under the hypothetical model presented in table 4, with 30 mph winds and the lowest quintile of crown base height, conditional crown fire was predicted for the control stands and surface fire for all the treated stands.

Discussion

Model results should always be applied cautiously. Current models that link surface and crownfire behavior are highly sensitive to crown base height, windspeed (or wind reduction factor), fuel moisture, and surface fuel model variables (1H fuel loading, herbaceous fuels, surface-area-to-volume ratio, fuel bed depth). We held slope constant at 7 percent (the average slope of the experimental blocks) but similar fuels on steeper slopes would exhibit higher fire intensity. There are a number of uncertainties in the models integrated in Nexus, reflecting the complexity of fire behavior (Scott 1998b). The actual numerical values used for model inputs produced realistic predictions but in some instances the differences between crown and surface fire behavior were separated by only a few miles/hr of windspeed (table 5). If wind gusts of higher speeds or higher surface fuel loadings were encountered, portions of the stands would be more likely to exhibit crownfire behavior. The behavior of real fires in these stands would be affected by roads, meadows, surrounding forest fuels, landscape topography, and suppression activities.

The purpose of the modeling analysis was not to accurately estimate the behavior of a real fire but rather to compare the treatment alternatives. All three thinning treatments tested by the Grand Canyon Forests Partnership substantially reduced the potential for passive and active crownfire. All the treatments increased crown base height to nearly 30 ft, making passive crownfire initiation difficult. However, the different thinning levels in the three treatments created differences in crown bulk density that were reflected in the potential for active crownfire. Prior to treatment, the crowning indices of all the stands were separated by only 6 mph (top section of table 5). After treatment (bottom of table 5), the crowning index ranged from 28 mph (control stands), 40 mph (3-6 treatment), 47 mph (2-4 treatment), to 55 mph (1.5-3 treatment). In relative terms, taking the control crowning index as unity, the 3-6 treatment required 43 percent more windspeed, the 2-4 treatment required 68 percent more windspeed, and the 1.5-3 treatment required nearly double (96 percent) more windspeed, for active crownfire.

The restoration treatment is not complete when the thinning is finished. Slash fuels increase the fire hazard as long as they remain on the ground, so prompt treatment with prescribed fire or mechanical means is important. Over time, vegetation in the treated units will change as both herbaceous plants and trees respond to the thinning. The potential intensity of grass-fueled fires should not be underestimated. Stand basal area even in the full restoration stands remained high enough to limit predicted herbaceous production to approximately 200 lbs/acre. If herbaceous production in the treated stands reached the 1,000 lbs/acre in the standard fire behavior fuel model 2, passive crownfire was predicted in the lowest crown base quintile for all treatments under severe weather conditions. However, grass fuels would be unlikely to have reached full productivity or to be fully cured in June. Even with a high fireline intensity, grass fires are of short duration with few heavy fuels and are more amenable to control than timber fires.

Strictly from a fire control perspective, therefore, a balance of relatively more trees and relatively less grass, such as the 2-4 or 3-6 treatments, might be useful in areas close to homes. On the other hand, future fire behavior will also be influenced by the growth of residual trees and new regeneration. Treatments with relatively high residual density might more rapidly grow back into a hazardous condition. Maintenance burning and/or further thinning can be used to regulate growth and keep the stands relatively crownfire-resistant. The failure to carry out these management activities would eventually eliminate the original treatment effects on fire behavior.

Potential fire behavior is an important consideration in the design of wildland/urban interface forest treatments, but it is not the only consideration. Fire hazard tradeoffs should be recognized and evaluated against many other forest values. In the present analysis, we have incorporated some of the variability in fuels and weather. A more complete analysis, however, could include spatial variability within stands and across landscapes, temporal variability (diurnal to seasonal change), successional change (years to centuries), and predicted changes in land use. In addition to modeling intensity and behavior of the flaming front, the effects of fuel burnout and smoke production should be considered. Many of the tools and components of such a comprehensive analysis are being rapidly improved.

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Effect of Prescribed Burning on Mortality of Presettlement Ponderosa Pines in Grand Canyon National Park

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Abstract—Ponderosa pine (*Pinus ponderosa*) trees established before Euro-American settlement are becoming rare on the landscape. Prescribed fire is the prime tool used to restore ponderosa pine ecosystems, but can cause high mortality in presettlement ponderosa pines. This study uses retrospective techniques to estimate mortality from prescribed burns within Grand Canyon National Park (GCNP). Live and recently dead presettlement ponderosa pines were sampled in four prescribed burns and three adjacent unburned areas. Presettlement ponderosa pine mortality (not including areas of crownfire) was higher than that of control sites in all four burns, although control areas showed elevated mortality rates compared to presettlement times. The highest mortality (23 percent) was found on a prescribed natural fire converted to a wildfire, the second highest (17 percent) on a site with extremely heavy mistletoe, the lowest (10 percent) on a spring burn. Bole scorch height and bole char severity were higher on dead trees than live trees, and may be useful in predicting postfire mortality. GCNP management objectives for overstory mortality are probably being met, but these guidelines do not account for the possibility of mortality delayed more than 5 years.

Introduction

Prescribed fire has become the primary tool in attempting to reverse the degradation of fire-adapted ecosystems due to past fire exclusion and other management activities. However, some researchers have raised doubts about reintroducing fire without otherwise treating the area (Bonnicksen and Stone 1985; Covington and others 1997; Fiedler and others 1996; Sackett and Haase 1998). Of special concern are high mortality rates in presettlement-aged, "old-growth" trees (hereafter presettlement trees). These trees harbor centuries of genetic diversity, provide important wildlife habitat (Thomas 1979), and are an important aesthetic feature (Brown and Daniel 1984). Past logging has dramatically reduced the presettlement ponderosa pine population over the landscape, and surviving trees are more susceptible to pathogens, drought, and injury because of increased stress due to competition from high densities of postsettlement trees (Covington and others 1994; Feeney and others 1998; Sackett and others 1996; Mast and others 1999). Because they would take longer to replace than any other living feature of the ecosystem (200–400 years), preserving a healthy population of these trees should be an important long-term management objective (Moore and others 1999; Mast and others 1999).

Some studies focusing on prescribed fire effects on presettlement pines have found substantial mortality (20–40 percent), some of which can be delayed for a decade or more (Sackett and Haase 1998; Swezy and Agee 1991; Thomas and Agee 1986). A major cause of this type of mortality is thought to be the extended smoldering of the thick duff layer that has accumulated around large trees in the absence of fire. Some researchers and managers now recommend raking this duff away from the bases of the presettlement trees before burning (Covington and others 1997; Sackett and Haase 1998; Taylor 1996). Many fire managers regard such intensive preburn mechanical treatment as impractical to apply over large areas (A. Farnsworth, Prescribed Fire Specialist, Coconino National Forest, K. Kerr, Prescribed Fire Manager, Grand Canyon National Park, personal communications). They argue that basal damage can be controlled by adjusting the burn prescription and that the few documented cases of unacceptably high mortality were due to extenuating factors.

Grand Canyon National Park (GCNP) encompasses one of the largest areas of unlogged ponderosa pine forests remaining in the Southwest. Fire managers have been conducting prescribed burns in GCNP since 1971 (GCNP, unpublished data). Over the past few years, fire managers have greatly increased the area treated with prescribed fire (K. Kerr, Prescribed Fire Manager, GCNP, personal communication).

More surveys of postburn mortality are needed to determine if high levels of mortality in presettlement trees are widespread or limited to special circumstances. Because full mortality may take a decade or more to be expressed, conventional fire effects monitoring studies may not provide information in time to be relevant. This study uses retrospective techniques to estimate long-term mortality of presettlement trees on prescribed burn sites within GCNP, and to relate high mortality to site, burn, and/or tree characteristics. The merits and limitations of this type of study and management implications of results are discussed.

Methods

Study Area

This study examined four prescribed burns (and adjacent unburned control areas) in three distinct areas of Grand

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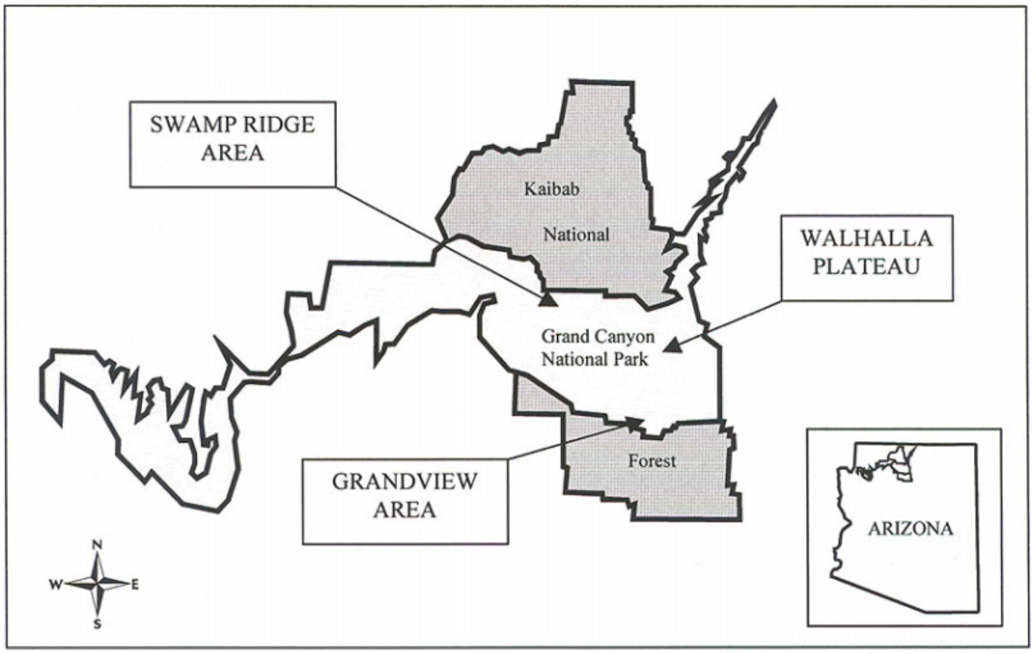

Figure 1—Study areas in Grand Canyon National Park, AZ.

Canyon National Park in Arizona (fig. 1; table 1). Information on the burning conditions and fire behavior during these burns was not available. All sites are within 4 km of the rim of the Grand Canyon and average 2,200 m in elevation. Terrain is dominated by karst topography, mostly level or rolling but deeply dissected by drainages. Most of the soils in this area belong to the Soldier series (Bennett 1974).

Table 1—Prescribed burns sampled in Grand Canyon National Park. All information comes from the Grand Canyon National Park fire history database and prescribed fire program summary reports. Sites are named for the name of the burn or, for controls, the burn unit in which they are found. Sizes are not given for control areas because only portions of these areas were sampled. PNF = prescribed natural fire; MIPF = management-ignited prescribed fire; CTRL = unburned control area.

Site	Type	Size	Ignition date	Sampling date
		ha		
Grandview area:				
Grapevine	MIPF	293	5/95	7/99
Hance	MIPF	267	8/95	7/99
Grandview	CTRL	—	—	7/99
Walhalla Plateau:				
Matthes	PNF	400*	7/95	8/98
Walhalla	CTRL	—	—	8/98
Swamp Ridge area:				
Northwest I/II	MIPF	92	9/92	7/99
Northwest V	CTRL	—	—	7/99

*Area burned before conversion to wildfire.

Average annual precipitation is 57.9 cm on the North Rim and 36.8 cm on the South Rim (GCNP, unpublished data). Precipitation is bimodal, divided between winter snow and late summer rain, with a distinct dry period in May and June.

Although ponderosa pine is the dominant overstory species at all three sites, the associated species differ. At Swamp Ridge, white fir (*Abies concolor*), Douglas-fir (*Pseudotsuga menziesii*), and aspen (*Populus tremuloides*) are present, and Engelmann spruce (*Picea engelmannii*) dominates north facing slopes. White fir and aspen also are present on the south end of the Walhalla Plateau, but are confined to mesic sites—remaining areas are mostly pure ponderosa pine. In the Grandview area, ponderosa pine-Gambel oak (*Quercus gambelii*) forests grade into piñon pine (*Pinus edulis*)-Utah juniper (*Juniperus osteosperma*) woodlands. Big sagebrush (*Artemesia tridentata*) and cliffrose (*Cowania neomexicana*) are common in openings.

The earliest evidence of human presence in this area is 3,000 to 4,000 years old, with further evidence indicating more or less continuous habitation through the present (Altschul and Fairley 1989). The historic fire regime was disrupted by heavy livestock grazing associated with Euro-American settlement (Altschul and Fairley 1989). Fire regime disruption dates are 1879 for the western North Rim and 1887 at Grandview (Fulé and others, in review). Fire regime disruption was found to be less abrupt on the Walhalla Plateau, with mean fire intervals not lengthening significantly until the early 1900s (Wolf and Mast 1998). Active fire suppression began with the designations of the Grand Canyon Forest Reserve in 1893 and Grand Canyon National Park 1919. Livestock were fenced out of GCNP by about 1938 (Fulé and others, in review; Verkamp 1940). Most of GCNP has never been logged.

Field Methods

Burns greater than 40 ha in size and 3–7 years old were selected for this study. A minimum time since fire of 3 years was necessary because trees injured by fire often take several years to die (Sackett and Haase 1996; Wagener 1961). The maximum time of 7 years increased the probability that outer ring dates could be obtained from increment cores for most of the trees that died since the fire. Unburned areas nearby were sampled as controls to determine background mortality rates (in other words, mortality from old age).

Within a selected site, sampling was limited to areas dominated by ponderosa pine and showing no evidence of logging or widespread crown damage from fire. The sites on the Walhalla Plateau were sampled in 1998 and all other sites were sampled in 1999. Minor changes in methods between field seasons are noted. Sites were sampled with belt transects arranged to capture as much spatial variation as possible. Transects were 40 m wide. At least 12 ha in each burn and each control area were surveyed.

Presettlement ponderosa pines judged to have been alive at the time of the burn were tallied in three categories: green trees, snags that torched or were severely scorched in the burn, and recent snags that did not torch. Presettlement trees were identified in the field using characteristics developed by Keen (1943) and modified for use in the Southwest (Covington and Moore 1994; White 1985). Characteristics included bark characteristics (large plates, light color) and crown architecture (large, drooping branches, columnar crown). At each site, several borderline presettlement trees (in other words, trees whose age was questionable) were cored and ring-counted in the field to calibrate these characteristics. Additional borderline trees found on the transects were cored and cross-dated in the lab to evaluate the accuracy of this technique.

Tallied trees were randomly sampled at varying rates (2–20 percent of green trees and 50 or 100 percent of recent snags), depending on relative densities of green trees and snags, to optimize transect length and number of trees sampled. Nominal sampling percentages were held constant for each transect, but actual percentages varied due to the random number generator. The total number of green trees and the total number of recent snags in each transect were recorded.

For each sampled green tree, samples of cambium from four quadrants (uphill, downhill, and both sides at 40 cm above the forest floor) were extracted to determine cambium condition (Ryan 1983). Cambium sampling was initiated in response to ambiguous data from the 1998 season, and therefore was not conducted at Walhalla Plateau sites. Increment cores were taken at 40 cm above the forest floor and through bark plates instead of through furrows (Ryan 1983). Unless the first core taken from a recent snag was completely sound, a second core was taken. The following measurements were made for each cored tree:

- Diameter of the tree or snag at height of core to the nearest cm.
- Condition class according to a tree, snag and log classification system based on that of Thomas and others (1979) and used widely in ponderosa pine forests of the Southwest (Cunningham and others 1980; Fulé and others 1997). Green trees were further classified according to relative health and vigor (Thompson 1940). This system has five vigor classes, which are determined by live crown ratios (LCRs) as follows: AA ("wolf" trees), LCR 70 percent or more; A (full vigor), LCR 55–70 percent; B (good to fair vigor), LCR 35–55 percent; C (fair to poor vigor), LCR 20–25 percent; and D (very poor vigor), LCR less than 20 percent. Recent snags (except those on the Walhalla Plateau sites) were further classified as to needle and fine branch retention as follows: A (retains most needles); B (retains some needles); C (no needles but retains most fine branches); and D (no needles but retains some fine branches). Snags lacking fine branches or having loose bark were assumed to have died before the fire and were not tallied or sampled.
- Obvious damage and tree abnormalities were recorded (for example, dwarf-mistletoe, insect attack, lightning strike, fire scar, fork top, and so forth).

The following measurements were made only on burn sites:

- Bole scorch height (from ground level to the highest point of scorched bark) was measured to the nearest 0.25 m.
- Basal char severity was quantified using the following categories (after Ryan 1983): 0 = no visible charring of bark; 1 = some blackening around base; 2 = bark plate surface uniformly blackened; 3 = some depth of char; 4 = deep char.

Dendrochronology

Cores were mounted, sanded, and crossdated (Stokes and Smiley 1968) using chronologies established for nearby areas (Fulé and others, in review). A systematic subsample of 40 percent of the cores was selected for analysis (all cores from the Walhalla sites were analyzed). Inner ring dates (IRD, just outside the pith) and outer ring dates (ORD, just inside the cambium) and were determined. IRD is an estimate of the year of tree establishment. ORD is an estimate of the year of growth cessation (in other words, imminent death). If crossdating was adequate for more than 100 rings but then became undatable, rings were counted to establish the IRD. For cores in which pith was not included (more than 90 percent of the cores), the IRD was estimated using a pith locator consisting of concentric rings printed on a transparency. IRDs were grouped by 25 year age classes (Mast and others 1999). Undatable cores were classified according to the reason they could not be dated.

Mortality Estimates

Two types of mortality estimates were made: a "field" estimate and a "lab" estimate. The field estimate was made using the tallies of live and recently dead trees on the transects and the visually determined snag decay classes. All trees with green needles were assumed alive, and all snags retaining at least some needles and with tight bark were assumed to have died within the past 5 years (Cunningham and others 1981). Trees that torched in the fire are included in this estimate.

Because this method estimates mortality over approximately the past 5 years, it may slightly underestimate mortality for the 7-year-old NW I/II burn and slightly overestimate mortality for the other (4-year-old) burns. Because condition classes of dead trees and numbers of torched trees were not recorded in the first field season, field estimates using all tallied dead trees (but not including torched trees) are substituted for the Walhalla sites. These numbers overestimate mortality by including many snags that died before the prescribed burn.

"Lab" estimates were based on dendrochronological data. Green trees were assumed alive if the ORD was equal to or 1 year less than the date of sampling. They were assumed dead or dying after the burn if the core showed no growth for 2 years or more before the date of sampling, but were still growing in the year of the burn. It is well established that ponderosa pines showing no basal area growth for 2 years or more have little chance of survival (Rogers and others 1981; Mast and Veblen 1994). Recent snags were assumed to have died after the fire if their ORD was in or after the year of the burn. Snags with earlier ORDs were assumed dead or dying at the time of the burn.

The field estimates are expressed as the percentage of the trees that were alive at the time of the burn but had died by the date of sampling. The lab estimates include some near-future mortality. Neither type of estimate can be directly compared among sites because of the variation in time lags between the burn dates and the sampling dates (3 to 7 years—table 1).

Other Data Analyses

Mean tree diameters were compared between live and dead trees for each site with Bonferroni-corrected approximate two-tailed t-tests for independent samples with unequal variances (corrected $p = 0.0071$; Ott 1993). Mean bole scorch heights were compared similarly for each burn (corrected $p = 0.0125$).

Bole char severity distributions for live and dead trees on each burn site (excepting Matthes, which was discarded from this analysis due to inconsistent data collection methods) were analyzed using a Bonferroni-corrected Chi-squared contingency test (corrected $p = 0.0167$, df = 4; Ott 1993). Vigor class distributions for live and dead trees were also compared with this test, but results were grouped for all sites ($p = 0.05$, df = 3). Due to the small number of "AA" trees, this category was combined with "A."

Results and Discussion

Sampling

Several issues about the way trees were sampled may influence the conclusions from the data. First, because different proportions of trees were sampled in different transects, some sampled trees represent a greater proportion of the tree population than others do. Second, the crossdating of cores from borderline presettlement trees revealed that most unsampled borderline trees in the Grandview area were in fact of presettlement age. No presettlement trees sampled in 1999 were found to be of postsettlement age. Additionally, few trees tallied in the transects were under 41 cm diameter at 1.37 m, the cutoff for what GCNP fire managers consider overstory trees (K. Kerr, Prescribed Fire Manager, GCNP, personal communication). Presettlement trees smaller than this were almost all highly suppressed members of old-growth clumps. Taken together, these factors suggest that the tree population examined in this study is comparable to the population specified in the GCNP overstory mortality objective. However, the youngest class of presettlement trees (120–150 years old) might be relatively underrepresented, and the highly suppressed presettlement trees relatively overrepresented.

The total number of cores for which an ORD could be determined varied from 31 to 67 per site. The number of cores from green trees varied from 22 to 64. The number of cores from dead trees whose ORD was in or after the year of the fire ranged from 11 to 21 for burned sites and only three to five for unburned control sites. The Grandview sites have smaller numbers of useable cores than the North Rim sites.

For many cores, especially those from dead trees, the ORD could not be determined due to breakage, rot, extreme suppression, or other reasons. These cores are hereafter referred to as "unusable." The proportion of unusable cores from recent snags ranged from 17 percent to 72 percent, and tended to be higher in control areas. Because of the relatively small numbers of dead trees found on these sites, older classes of snags were more likely to be sampled. The number of broken and rotten cores increased along the xeric-mesic gradient between sites (most broken cores were also due to rot). The average proportion of unreadable dead cores is comparable to that reported by Mast and Veblen (1994). The sometimes high proportion of unusable cores may bias lab estimates due to the undersampling of rot-infected trees.

Estimates of Presettlement Tree Mortality

One GCNP fire management objective is to keep overstory mortality, as measured 5 years postburn, below 20 percent over the entire vegetation type in the Park (K. Kerr, Prescribed Fire Manager, GCNP, personal communication). Because the main purpose of this study was to document mortality due to factors other than crown damage, mortality due to widespread crown damage was not documented, although estimates do include trees that torched individually. Therefore, landscape level mortality is higher than numbers reported here.

Field estimates show mortality of presettlement trees in all burned areas is many times greater than in respective controls (table 2). This result supports the idea that noncrown damage from prescribed fires can lead to increased mortality of presettlement trees. The Swamp Ridge area control site estimate may be artificially low due to faster rates of snag decay at this more mesic site. The other control area estimates show mortality rates somewhat lower (0.5–0.75 percent/year) than other studies measuring contemporary background mortality in mature ponderosa pine (1–2 percent/year—Avery and others 1976; Hamilton and Edwards 1976; Mast and others 1999). However, these rates are still much higher than estimates of mortality rates in presettlement times (Mast and others 1999).

The field estimate shows 27 percent mortality on the Hance burn, more than twice that of any other site. This site has very high levels of dwarf-mistletoe infection (70 percent of the trees on transects were infected, compared to 40 percent on the Grandview control area and 20 percent on the Grapevine burn). Infected trees are more likely to torch due to low witches' brooms and have reduced vigor (Harrington

Table 2—Presettlement ponderosa pine mortality estimates. These numbers represent the proportion of trees alive at the time of the prescribed burn that died between the burn date and time of sampling. The field mortality estimate counts all snags retaining needles as dead since the burns. The estimates in parentheses include snags with no needles but most fine branches due to the older age of the NW I/II burn. Because of different data collection methods, the field estimates for Walhalla sites (in italics) include all recent snags retaining needles and/or fine branches. For the lab estimates, a tree is considered dead/dying if its core shows that it has not put on a growth ring for 2 years or more. This estimate can be thought of as including some near-future mortality. All South Rim sites were sampled 4 years postburn. NW sites were sampled 7 years postburn, and WP sites 3 years postburn.

Site	Field estimate	Lab estimate
	- - - - - - - - - - *Percent* - - - - - - - - - - -	
Grandview area:		
Grapevine	10	12
Hance	27	17
Grandview (control)	2	10
Walhalla Plateau:		
Matthes	*13*	23
Walhalla (control)	*3*	5
Swamp Ridge area:		
Northwest I/II	2 (12)	15
Northwest V (control)	1 (2)	5

and Hawksworth 1991) making them more susceptible to other damage. The field estimate for this site may be inflated because of the tendency for witches' brooms to stay alive many years after most of the tree canopy has died (Mathiasen 1996). This could cause many trees that were effectively dead before the prescribed burn to retain needles until the date of sampling.

The Grapevine and Matthes burns show intermediate mortality rates around 10 percent, while the field estimate for the Northwest I/II burn is extremely low (2 percent), even lower than some estimates of background mortality. If snags of condition class C are included, the estimate is 12 percent. True mortality on this site is probably somewhere between these numbers.

The lab estimates, which count trees with cores showing zero growth for 2 years or more as dead or dying, include some near-future mortality. They show background mortality rates roughly comparable to those in other studies (about 1–2 percent/year—Avery and others 1976; Hamilton and Edwards 1976; Mast and others 1999). These numbers translate to the death of approximately 2–7 presettlement pines/ha/decade, depending on the presettlement tree densities at the different sites. The Grandview control site shows a higher mortality rate than the two North Rim control sites, possibly because of higher stress levels of on the former, more xeric site.

The lab estimates show the highest mortality (23 percent) for the Matthes burn, which was a prescribed natural fire that was converted to wildfire status and suppressed due to increasing fire intensity and a lack of resources. The second-highest mortality was found on the dwarf-mistletoe-infested Hance burn. The Grapevine burn, which had the lowest lab mortality estimate (comparable to the rates in control areas) was a spring burn. This suggests that burning in the spring, when forest floor moisture is high, might reduce basal damage. However, spring burning may have other undesirable consequences. For the burns in the Grandview and Swamp Ridge Areas, these lab estimates translate to the death of approximately five presettlement pines/ha. The Matthes burn, occurring in the study area with the highest density of mature pines, had a much higher per-unit-area rate. According to the lab estimate for that burn, approximately 20 presettlement pines/ha died.

If these mortality estimates are accurate, and mortality due to crown damage is not widespread, GCNP management objectives (less than 20 percent overstory mortality 5 years postburn) are probably being met. However, the GCNP management objective does not address the possibility of significant mortality after 5 years. Sackett and Haase (1998), studying presettlement ponderosa pine mortality at the Chimney Spring site in northern Arizona, found that mortality may take more than a decade to be fully expressed. Five years after the initial burns at Chimney Spring, "old-growth" mortality averaged approximately 28 percent in the burned areas. Eighteen years after the burns, the overall old-growth mortality had almost doubled to 50 percent, and old-growth trees were still dying at higher rates than those in the unburned control area were. However, the severe drought in 1996 (the year after most of the sites in this study were burned) probably was a contributing factor in the deaths of many trees. Trees were killed that otherwise may have lingered for years in a weakened state. It is therefore likely that these mortality estimates account for a high proportion of the fire-induced mortality on the burn sites.

Tree, Site, and Burn Characteristics

No statistically significant differences were found in comparisons of diameter or age between live and dead trees on burn sites. Vigor was lower for trees in burned areas than for trees in control areas, but this was not statistically significant, either.

Although not statistically significant, mean bole scorch height (excluding values above 10 m) was higher for dead trees than for live trees for all burns. Basal char severity was higher on recently dead trees on all three burns analyzed. These results support the hypothesis that basal and/or root damage is a contributing factor to postburn mortality. Measures of fire intensity at the bases of trees must be considered in addition to crown scorch when predicting postfire mortality of large trees (Ryan 1983; Regelbrugge and Conard 1993). The failure to identify more significant differences in tree and burn characteristics may be due to the relatively small sample sizes of recent dead trees from some sites.

Conclusions

Estimates show that mortality levels of presettlement ponderosa pines in Grand Canyon National Park are higher in areas treated with prescribed fire compared to unburned areas, even when discounting mortality due to crown damage. Higher mortality is associated with high infection levels of dwarf-mistletoe on one site, and prescribed natural fire that was converted to wildfire on another site. The lowest postburn mortality rate (little higher than control area mortality) occurred on a spring burn, possibly because higher forest floor and soil moisture levels reduced basal damage. Because of the small sample size (four burns), it is not possible to generalize these associations between heightened mortality and other factors. More postburn mortality surveys are needed.

Postburn mortality as measured on these four prescribed burns is probably within management objectives of the GCNP fire program. Because of the 1996 drought, future mortality on these burns is predicted to be relatively low. However, the reduced vigor of trees in the burns in this study, as well as results from other studies, suggest that mortality rates can remain abnormally high long after 5 years postburn in some cases. Methods of predicting future mortality (for example, from annual growth increment data) should be used to supplement studies such as this one.

Mortality estimates based on dendrochronological data in unburned control areas agreed closely with background mortality estimates from other studies. These may be much higher than historical rates, probably because of increased competitive stresses due to higher tree densities in contemporary forests. The 1996 drought might also have elevated the mortality rates in the control areas. The unburned area surveyed on the xeric South Rim shows especially high mortality (more than 2 percent/year). These figures highlight the need to reinvigorate presettlement trees as well as merely protecting them.

This study could not identify tree characteristics associated with increased risk of postfire mortality. Studies measuring tree characteristics both before and after burns are probably better suited than retrospective surveys to gather this type of information. Measures of basal fire intensity were higher on dead trees than live trees in all burns. This agrees with results from other studies and argues that these measures can be important in predicting postburn mortality of mature trees.

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Effects of Low Intensity Prescribed Fires on Ponderosa Pine Forests in Wilderness Areas of Zion National Park, Utah

Henry V. Bastian

Abstract—Vegetation and fuel loading plots were monitored and sampled in wilderness areas treated with prescribed fire. Changes in ponderosa pine (*Pinus ponderosa*) forest structure tree species and fuel loading are presented. Plots were randomly stratified and established in burn units in 1995. Preliminary analysis of nine plots 2 years after burning show litter was reduced 54.3 percent, duff was reduced 34.7 percent, ponderosa pine tree density in the 10.2 cm to 30.5 cm d.b.h. (diameter at breast height) size class was reduced 18 percent, and ponderosa pine tree density greater than 61.2 cm d.b.h. increased in Zion National Park.

Introduction

Historically, lightning and human caused fires influenced vegetation structure. Vegetation is constantly growing and changing and fire provides a natural means of checks and balances for many landscapes. Fires reduced fuel accumulations and maintained open, grassy forest stands. However, human settlement and land use practices resulted in fewer fires and those that did ignite were quickly suppressed, whether in wilderness or not. As a result, forest structure has changed with increased fuel loads and thicker tree and shrub densities.

Fire is a natural process and a critical component in nutrient cycling and vegetation structure maintenance of many ecosystems. "Entire ecosystems can be drastically altered or maintained by periodic burning" (Kozlowski and Ahlgren 1973). Fire is a tool that can be used in restoring many forested and nonforested areas.

West and Madany (1981) cited Alter (1942) with a description from Priddy Meek's journal. This description is of the land just to the northeast of what is presently Zion National Park. In June of 1852, Priddy described the area as "…Rich soil, plenty of grass and timber… so that a team and wagon might be driven any place…. We traveled three days amongst this timber, which is of the best quality and clear of underbrush." What caused this open vegetation structure? Many studies have shown that ponderosa pine vegetation systems have evolved with fire and require it for growth and recruitment. West and Madany (1981) researched the fire history of Zion National Park and stated that, "From the fire scar record we can safely state that any location within a ponderosa pine forest burned at least once, and more likely twice, every decade in the time before white settlement." This conclusion fits comparable descriptions from early settlers of the West and demonstrates that many fire cycles have been missed in this fire regime. Using prescribed fire in wilderness areas may help restore this scene of open forests, as well as promote nutrient cycling and vegetation structure maintenance at Zion and across the West. "The inescapable conclusion of fire effects in ponderosa pine is that the land manager responsible for these communities should be doing a lot of burning" (Saveland and Bunting 1987).

Objectives

Specific objectives for an initial prescribed fire treatment of ponderosa pine were: Reduce needle/litter and duff layers by 40–60 percent, and attain a 30–60 percent decrease in ponderosa pine of size class 10.2–30.5 cm diameter at breast height (d.b.h.). The objective for ponderosa pine greater than 61.2 cm d.b.h. was to maintain the density of trees. However, fires and effects can be unpredictable so mortality was required to be limited to 20 percent for fire prescription parameters.

Study Area

Zion National Park is located in southwestern Utah. The park covers 59,490 ha and ranges in elevation from 1,127 to 2,659 m. Temperatures vary with elevation, with lows below 0 °F in the upper elevations during winter, and exceed 100 °F at lower elevations during summer. Afternoon thunderstorms are common in July and August. Peak precipitation occurs December through March with an average of 15 inches annually at park headquarters. Ponderosa pine occurs from approximately 1,980 to 2,400 m in Zion National Park. The random plots were located in the Stave Spring, Echo Canyon, East Boundary, and Goose Creek prescribed burn units.

Methods

Data Collection

Ponderosa pine vegetation monitoring plots were established according to the Western Region Fire Monitoring

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Handbook Protocols (USDI NPS 1992). Nine plots were stratified randomly within the prescribed burn units. The plots, established before a controlled burn, consist of a 20 x 50 m area. All overstory trees (>15.1 cm d.b.h.) were recorded in the 20 x 50 m area. Pole sized trees (2.51 cm–15.0 cm d.b.h.) were sampled in a 10 x 25 m section and seedling sized trees (<2.50 cm d.b.h.) were sampled in a 3 x 10 m section. Brush and herbaceous plants are also sampled in the plot but not analyzed for this poster. Forest fuel load conditions were measured on four transects. Fuel load data are calculated for litter, duff, 1, 10, 100, and 1,000-hour fuels (Brown 1974). Plots were sampled immediate postburn, 1, and 2 years after the burn to monitor the effects. Plots will be sampled at 5, 10, and 20 years after the burn or until the area has fire introduced again and the sampling starts over for a second burn event sampling all components described above. Plots in this study were burned between 1996 and 1997 during the months of September through November.

Immediate postburn sampling was done within 1 month of the burn and sampled burn severity, overstory tree postfire assessment, and fuel loading transects. Burn severity was assessed on a scale of one to five, and done in conjunction with the fuel loading transects. At each sample point, severity was evaluated in a 4 square decimeter area. For example, if litter and duff were consumed leaving white ash and all plant parts were consumed, the area would be rated one (1). If the area was not burned, it would be rated five (5) for unburned. The data were then averaged to give a percent for organic substrate and vegetation categories and to estimate burn severity for the area.

Overstory tree postfire assessment characterizes the amount of heat received by the trees, via sampling char height, scorch height, and percent crown scorch. Char height was measured from the ground to the maximum height on the bole of the tree. Scorch height was measured from the ground to highest point where foliar death was evident. Percent scorch was an estimate of the entire crown of the tree that was scorched.

Prescribed Burn Conditions

The ponderosa pine fuels were classified as a timber fuel model 09 (Anderson 1982). Fuel models were used to predict fire behavior and to assess the potential to control a fire and protect resources. Plots were burned between 1,100 and 1,600 hours. Weather conditions during prescribed fires included: ambient air temperatures of 55 °F to 70 °F, relative humidity of 13–33 percent, mid-flame wind speeds of 0–18 km per hour, and 0–70 percent shade. Fire behavior observations included: flame lengths of 0.2 to 0.6 m, flame zone depths of 0.4–0.8 m, rates of spread for backing fires of 5 to 17 m per hour (m/hr), flanking fires of 5 to 15 m/hr, and head fires of 11 to 18 m/hr.

Results

Average burn severity was 3.7 for organic substrate giving a “lightly burned” rating and 4.0 for vegetation for a “scorched” rating (fig. 1). Overstory tree postfire assessment averages for char and scorch heights were 1.4 m and 2 m, respectively. The average canopy scorch was 10.7 percent for 254 trees.

Litter was reduced 55.3 percent from 1.03 kg/m^2 to 0.46 kg/m^2 at the immediate postburn sample. Duff was reduced 34.7 percent from 4.52 kg/m^2 to 2.97 kg/m^2 immediate postburn (fig. 2). Total fuel loading was reduced 34.5 percent from 6.89 kg/m^2 to 4.50 kg/m^2 immediate postburn (fig. 3).

Ponderosa pine overstory in the 10.2–30.5 cm d.b.h. size classes was reduced 18 percent from 138.9 trees per ha to 114.4 trees per ha and trees greater than 61.2 cm increased from 1.1 to 7.8 trees per ha within 2 years after the burn (fig. 4).

Discussion

This early analysis indicates that prescribed fire may be used to reduce fuels and small trees without mortality of large trees (figs. 5–10). These initial prescribed fires were designed to treat the areas multiple times with low to moderate fire intensities and to reduce the potential of fire starts, natural or human-caused, that occur under both extremely dry conditions and heavy fuel accumulations from years of fire suppression. Thinning was considered but due to the area being wilderness, prescribed fire was chosen. After some of the areas have been treated with prescribed fires, it is hoped that fire will be restored to occur naturally.

Ecosystems are complex, dynamic, and difficult to sample and analyze. With many of these environmental factors and others (soil composition, precipitation, topography, vegetation composition and structure, live vegetation fuel moisture, fire behavior, air temperature, and relative humidity), sample error needs to be discussed. It can be seen from the data that standard error varied. What may be represented here may not be what actually occurred. Due to many variables in observing and recording data, it is difficult to accurately sample without some error.

Minimum plot calculations were done using an 80 percent confidence level with an R-value of 25. Eight plots were needed for total fuel loading and seven plots were needed for overstory tree density to meet the 80 percent confidence level. A paired t-test was done on litter fuel loading, duff fuel loading, total fuel loading, overstory (10.2 to 30.5 cm), and overstory (61.2 cm) tree densities. The t-test showed the following values, respectively: 5.246, 2.779, 3.948, 2.204, and –1.250, showing in general a significant change from pre- to postburn conditions.

Objectives and target conditions are continually being developed and redefined from knowledge of the past and present. There are many factors that need to be considered when dealing with fire and vegetation succession. “Fire history has a decided influence on the particular successional status of vegetation and in what direction and at what rate it is changing” (West and Loope 1977). Thus, it is not surprising that prescribed fire has become a useful tool to meet a variety of management objectives (Biswell and others 1973).

Summary

Needle/litter fuel load layer was reduced 54 percent. This meets the objective to reduce it by 40 to 60 percent. Duff fuel loading was reduced 35 percent. This is close to meeting the objective. Pole sized trees were reduced 18 percent, which is

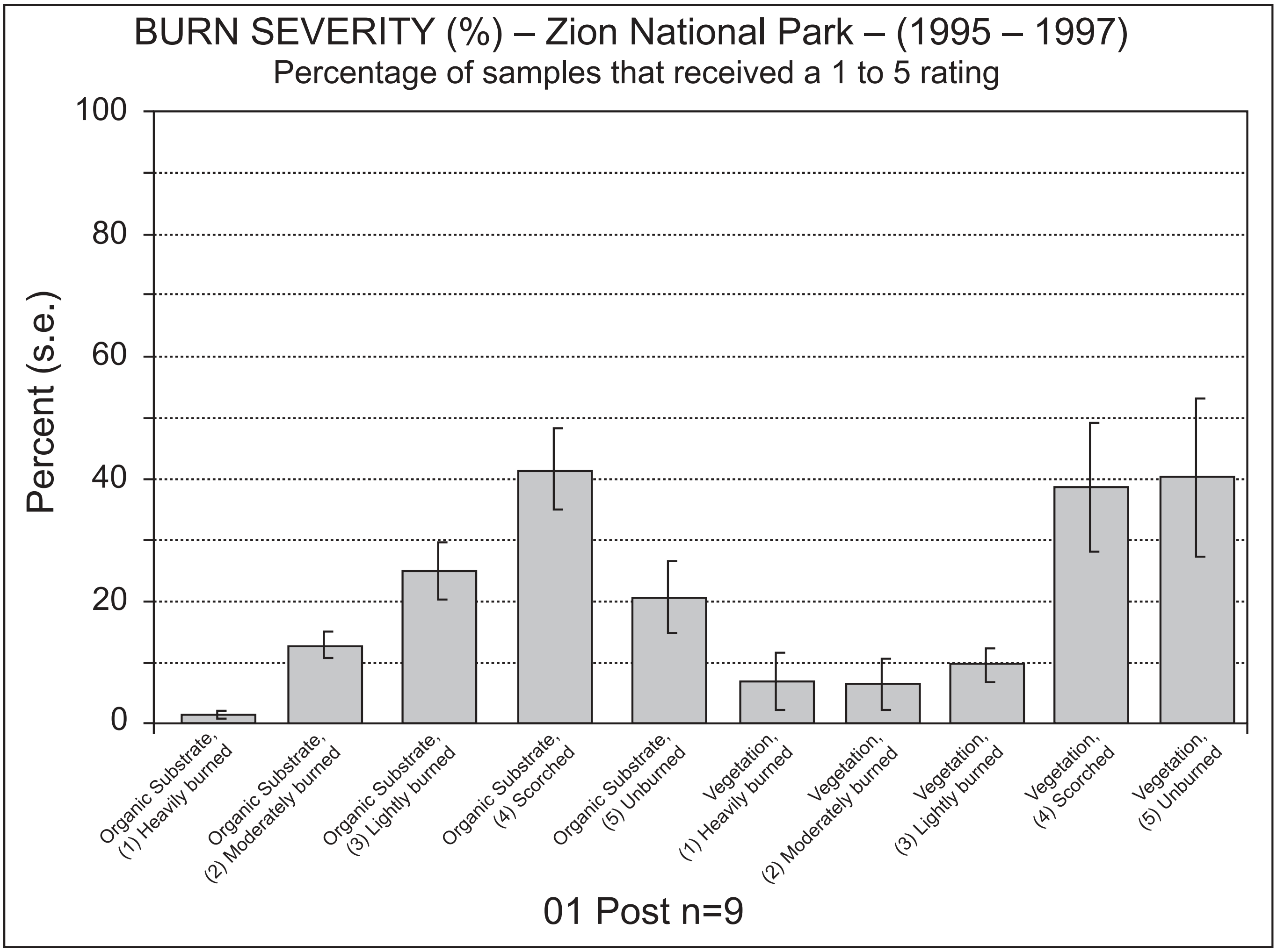

Figure 1—(USDI NPS 1999). Percentage of burn severity categories on organic substrate and vegetation. 01 = the first time the area has been treated with prescribed fire, Post = sampling done within 1 month of completion of burn, and n=9 = sample size of nine plots.

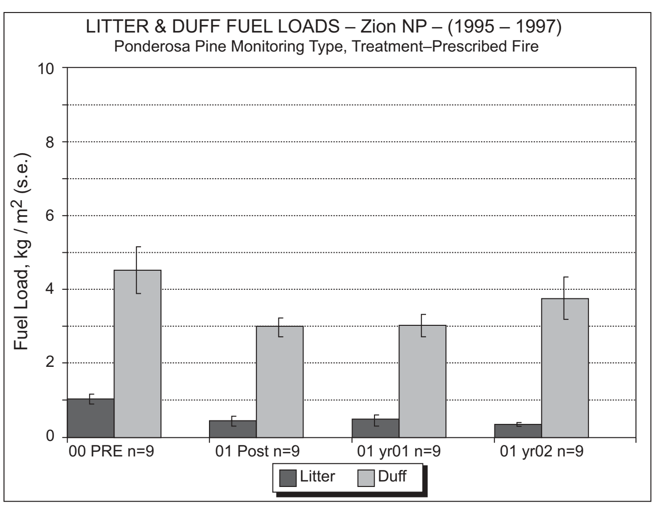

Figure 2—(USDI NPS 1999). Litter and duff fuel loading by sample period.

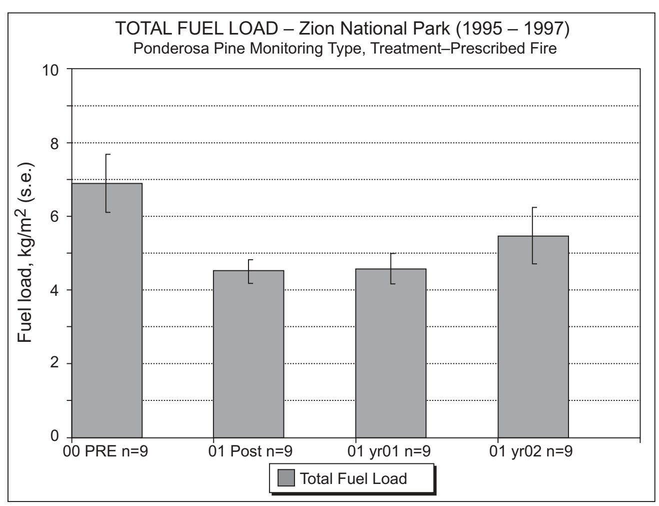

Figure 3—(USDI NPS 1999). Total fuel load (litter, duff, and woody fuels) kg/m^2 from preburn sample to 2 years after the burn.

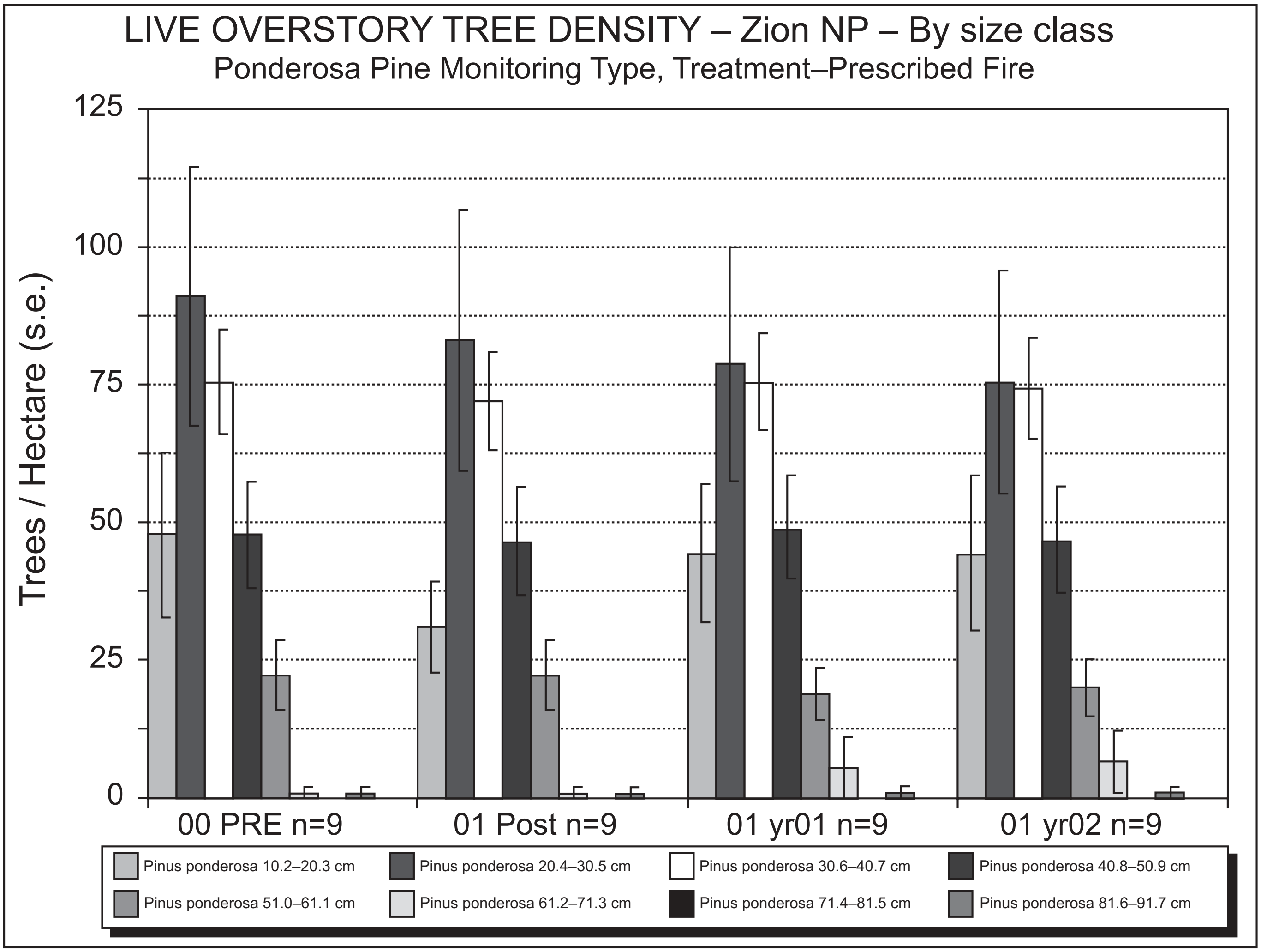

Figure 4—(USDI NPS 1999). Ponderosa pine overstory trees per ha divided into size classes by sample period.

Figure 5—Preburn Echo Canyon photograph of ponderosa pine forest at Zion National Park.

Figure 6—Postburn Echo Canyon picture, same view as figure 5. Smaller trees have been reduced a little toward target objective.

Figure 7—Same view from figure 5, 1 year after the burn. Brush has generally decreased.

Figure 9—Postburn picture, same view as figure 8.

Figure 8—Preburn Goose Creek picture of ponderosa pine forest at Zion National Park.

Figure 10—One year after the burn, same view as figure 8. Understory has been opened up.

well below the objective of attaining a 30 to 60 percent decrease in this size class. These results provide some evidence that prescribed burning can be an important step in restoring some vegetation structure described in the past (figs. 5–10). Continued burning and monitoring may help restore the natural range of variability in these vegetation communities to a self-sustaining state. “One of the primary and unique missions of the National Park Service is to perpetuate natural ecosystems in a state approximating the pristine” (Stone 1965; Houston 1971).

Acknowledgments

A special thanks to the people who assisted in this project: A. Latterell, J. Bradybaugh, P. Reeberg, W. Sydoriak, Zion Prescribed Fire Module, Bandelier Prescribed Fire Module, Zion Fire Crew, and Utah/Great Basin Fire Effects Crew.

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The Effects of a Low Intensity Fire on a Mixed Conifer Forest in Bryce Canyon National Park, Utah

Henry V. Bastian

Abstract—Prescribed fire was used to reduce fuel loading and tree densities. Permanent vegetation and fuel loading plots were randomly established within prescribed burn units. The plots were established in 1995 and were sampled, immediately postburn (within 1 month of the fire), 1 year after the burn, and 2 years after the burn. The prescribed burns were implemented in August of 1995. Preliminary analysis of 11 plots shows fuel loading was reduced from 31.9 tons/acre to 11.4 tons/acre immediate postburn. White fir (*Abies concolor*) overstory was reduced 35 percent, poles 52 percent, and seedlings 71 percent by the second year following the burn.

Introduction

Many factors influence vegetation succession and how a fire burns across the landscape, but both are mainly affected by land use or management practices. A few other factors include: soil composition, precipitation, topography, vegetation composition and structure, live vegetation fuel moisture, fire behavior, air temperature, and relative humidity. All of these factors combine to produce varied effects on successional vegetation mosaics. "A comparison of today's landscape at Bryce Canyon National Park with the landscape shown in historic photographs indicates that a major change has occurred in the Park's vegetative mosaic" (Roberts and others 1993). "Journals from early settlers in Garfield County [Bryce Canyon area] describe open forests, where visibility was several hundred yards. These early ranchers and farmers tell of being able to take a wagon and team of horses through the forests on top of the Paunsaugunt Plateau" (Bryant 1995). What caused the change in the vegetation and landscape as described above? Land use practices and an active role in suppressing fires are the main factors that helped influence the change. Prescribed fire may be used in restoring the natural vegetative mosaic.

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Henry Bastian is a Fire Effects Specialist with Zion National Park. He attended Utah State University and received a B.S. degree in Fish and Wildlife Biology. He continues to monitor and study the effects of using prescribed fire.

Objectives

The main objectives of the project were to use prescribed fire to reduce hazardous fuel conditions and to burn the area with a low fire intensity. Specific objectives were to reduce total fuel load 20–50 percent immediate postburn, reduce white fir (*Abies concolor*) poles 10–20 percent, and white fir seedlings 20–40 percent within 2 years of the burn. Using prescribed fire may reduce fuels and vegetation densities to diminish the risk of a catastrophic wildfire. "[A park management goal]…is to restore the park ecosystem to a condition typical of pre-European settlement and prior to the establishment of fire exclusion policies" (Bryant 1994).

Study Area

Bryce Canyon National Park, in south/central Utah on the Paunsaugunt Plateau, covers 35,852 acres and ranges in elevation from 6,000 ft to 9,000 ft. Mixed conifer forests range above 8,200 ft. The prescribed burns occurred in the County Line and Yovimpa burn units in the south end of the park. General weather patterns include temperature ranges from –30 °F to 90 °F. Precipitation peaks are in January/February and July/August.

Methods

In 1995, 11 mixed conifer (*Abies concolor*/*Pinus ponderosa*) vegetation-monitoring plots were established according to the Western Region Fire Monitoring Handbook Protocols (USDI NPS 1992). All plots were randomly located within the prescribed burn units. The plots, established before a controlled burn, consist of a 20-m by 50-m area. All overstory trees (>15.1 cm diameter at breast height, d.b.h.) are recorded in the 20 m x 50 m area. Pole trees (2.51 to 15.0 cm d.b.h.) are recorded in a 10 m x 25 m area. Seedling trees (<2.5 cm d.b.h.) are recorded in a 5 m x 10 m area. Forest fuel is calculated on four, 50 ft transects. Calculations measure litter, duff, 1, 10, 100, and 1,000-hour fuels following the methodology of Brown (1974). The plots were sampled immediate postburn, 1, and 2 years after the prescribed burns.

Burn severity is determined within 1 month of the burn. It is assessed on a scale of 1 to 5 (fig. 1) and conducted in conjunction with the Brown's fuel loading transects. At each sample point (1, 5, 10 ft, and so forth), severity is evaluated in a 4 square-decimeter area. For example if litter and duff is consumed leaving white ash and all plant parts are consumed the area would be rated one (1). If the area was not burned, it would be rated five (5) for unburned. This information is then calculated to give an average burn severity.

Burn Severity Data

BRCA	1 = Heavily Burned	2 = Moderately Burned	3 = Lightly Burned	4 = Scorched	5 = Unburned
Organic Substrate [Litter and Duff] (Standard Error)	19.9 % (s.e. 4.7)	4.1 % (s.e. 1.1)	7.6% (s.e. 2.2)	7.2% (s.e. 1.8)	61.2% (s.e. 7.5)
Vegetation (Standard Error)	5.5% (s.e. 5.5)	9.5% (s.e. 9.1)	4.0% (s.e. 2.5)	16.4% (s.e. 5.0)	55.5% (s.e. 11.5)

Average Severity was 3.9 Organic substrate and 3.8 Vegetation

Figure 1—Percentage of severity categories and average severity. The data average gives a "lightly burned" severity rating.

Results

Burn severity ratings showed that 19.9 percent of the sample received a one (1) heavily burned rating, 18.9 percent received a two to four rating, and 61.2 percent receiving an unburned rating (fig. 1) creating a patchy burn with varied intensities across the landscape. The average burn severity ratings were 3.9 for organic substrate and 3.8 for vegetation for a "lightly burned" rating. Burn severity for the 11 plots was lower than expected given the mortality seen in the overstory, pole, and seedling trees especially with a significant amount receiving an unburned rating (N = 11 for this sample).

Figure 2 shows the mean total fuel load and standard error of the sample. The preburn fuel loading was 31.9 tons/acre. Fuel loading was reduced 64 percent to 11.4 tons/acre immediate postburn. Total fuels have achieved 52 percent of the pre fuel loading level to 16.5 tons/acre 2 years after the burn.

Figures 3 through 5 show the mean overstory, pole, and seedling tree densities with standard errors. White fir overstory trees had a density of 81.7 trees/acre preburn and ponderosa pine had a density of 22.8 trees/acre. Two years after the burn, overstory white fir was reduced 35 percent to 53 trees/acre and overstory ponderosa pine was reduced 16 percent to 19.1 trees/acre. White fir poles decreased 52 percent from 169.2 trees/acre to 80.9 trees/acre 2 years after the burn and ponderosa poles decreased 50 percent from 5.9 trees/acre to 2.9 trees/acre. Seedling white fir decreased 71 percent from 1,604.1 trees/acre to 463 trees/acre 2 years after the burn, while ponderosa pine seedlings decreased 40 percent from 36.8 trees/acre to 22.1 trees per acre. It is interesting to note that there were no quaking aspen (*Populous tremuloides*) seedlings preburn, but 2 years after the burn aspen was regenerating (fig. 5). At the 2-year reading, aspen seedlings were recorded at 117.7 trees/acre, but this occurred in only one plot.

Discussion

Results demonstrate that this fire produced small to medium changes to the total fuel load, overstory trees, pole trees, and seedling trees. These low intensity fires ("lightly burned") can result in some mortality to overstory, pole, and

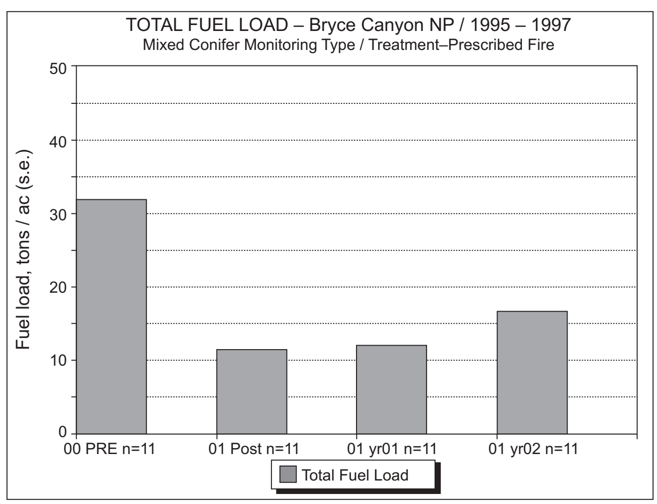

Figure 2—(USDI NPS 1999). The mean total fuel load in tons per acre before a controlled burn, immediate postburn, 1 year, and 2 years after the burn.

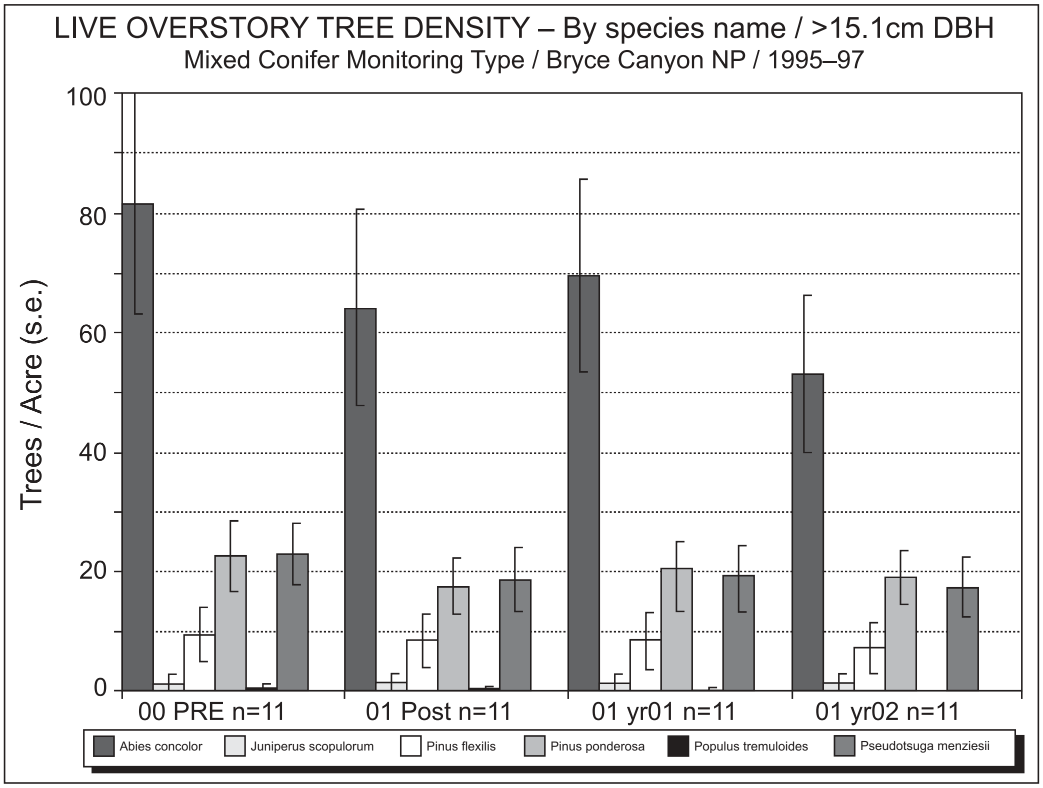

Figure 3—(USDI NPS 1999). The mean density of overstory tree species (>15.1 cm d.b.h.) before a controlled burn, immediate post, 1 year after, and 2 years after the burn.

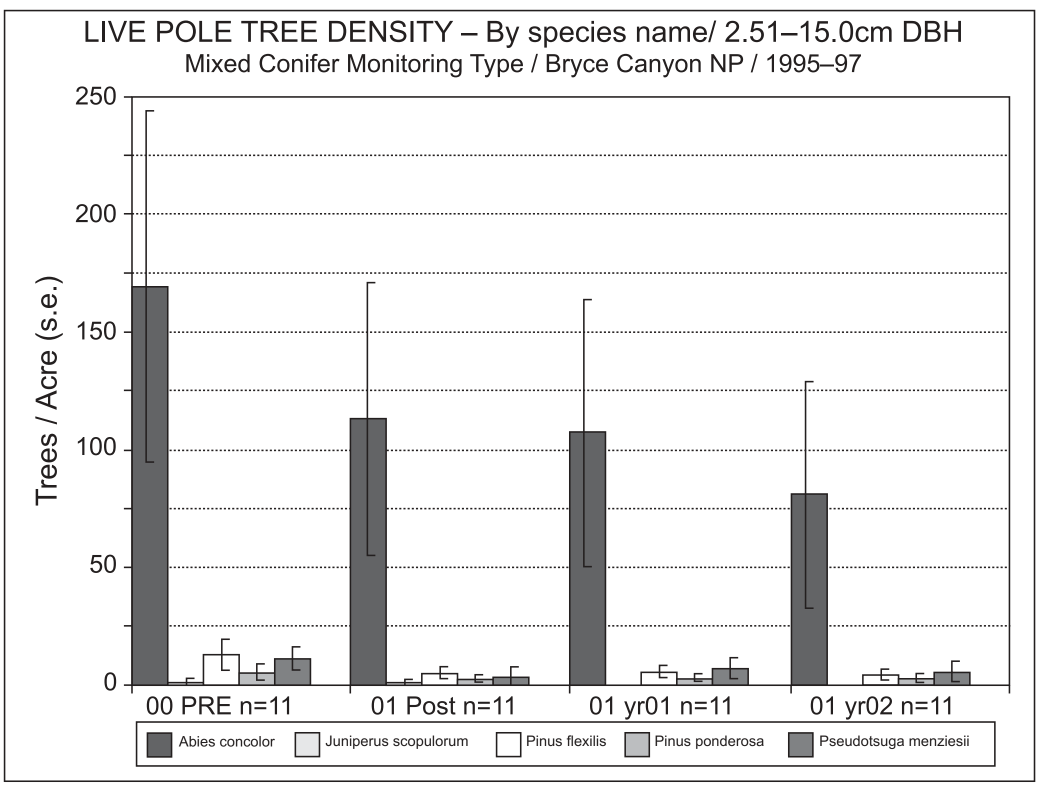

Figure 4—(USDI NPS 1999). The mean density of pole sized trees (2.51–15.0 cm d.b.h.) before a controlled burn, immediately post, 1 year after, and 2 years after the burn.

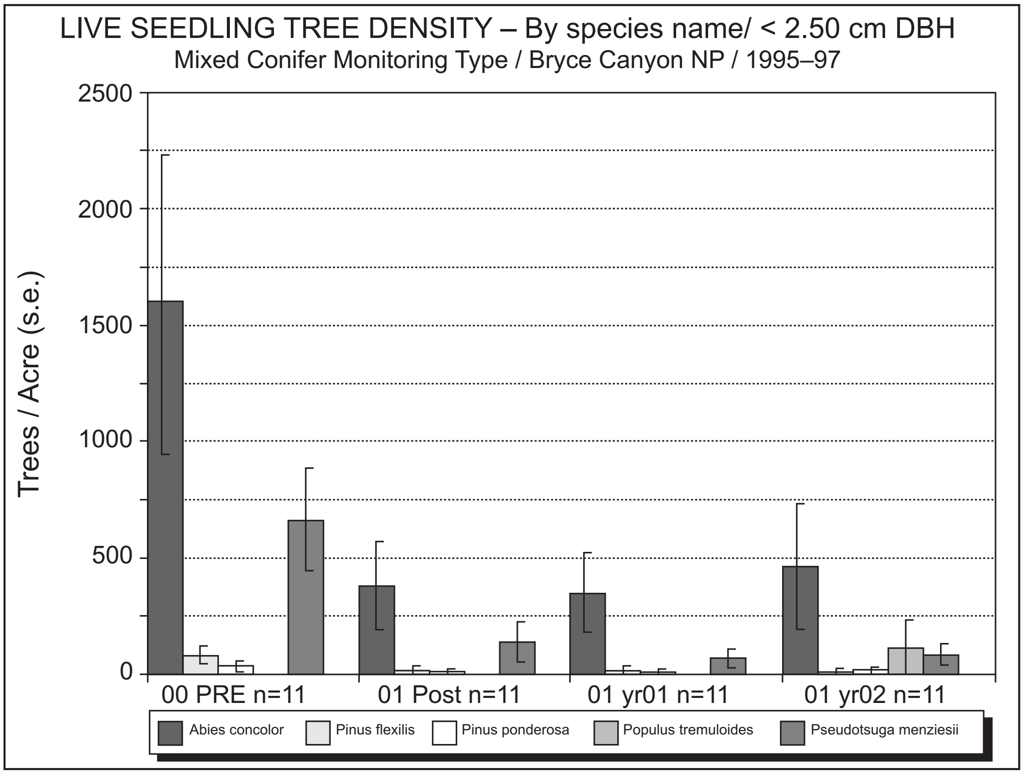

Figure 5—(USDI NPS 1999). The mean density of seedling sized trees (<2.50 cm d.b.h.) before a controlled burn, immediate post, 1 year after, and 2 years after the burn.

seedling-sized trees. Monitoring shows changes in preburn and postburn forest fuel conditions (fuel loading) and overstory, pole, and seedling trees densities. "In mixed conifer forests where white fir dominates the understory due to years of fire suppression, prescribed low-intensity surface fires will kill large numbers of white fir" (FEIS 1996). " This reduces the hazard of white fir providing a fuel ladder to ignite the crown of overstory trees and also restores tree species composition closer to that of pristine conditions" (FEIS 1996).

Remnant populations of quaking aspen were scattered throughout the mixed conifer forests and it was hoped that through using prescribed fire, the aspen would be stimulated and regenerate in openings created by the fire. The results demonstrate that this may be occurring but, it is too early to report any significance of aspen regeneration.

In 1995 Jenkins examined the mixed conifer forest near the south end of the park and concluded that the mean fire return interval was 7.5 prior to the 1900s. "Studies concluded that the accumulation of woody vegetation and down and dead woody fuels have increased significantly in the present century" in Bryce Canyon National Park (Jenkins 1995).

Ecosystems are complex and dynamic. It is difficult to accurately sample them without error. Figure 3 demonstrates this point. Overstory white fir trees were reduced from 81.7 trees/acre preburn to 64.0 trees/acre postburn. However, at the 1 year reading there were 69.5 trees/acre. This is due to the difficulty in accurately determining if the tree is dead or alive immediate postburn due to scorching of the tree canopy. Observers introduce error regardless of their knowledge, experience, and expertise on the job. Due to many variables in observing and recording data, it is difficult to accurately sample without some error.

Minimum plot calculations were done using an 80 percent confidence level with an R value of 25. Eleven plots were necessary for total fuel load to meet the 80 percent confidence level. Overstory trees required six plots, poles 35 plots, and seedlings needed 44 plots to meet 80 percent confidence. It is important to remember that the sample size must be large enough to infer that the result has occurred across the entire landscape, when evaluating the data.

Conclusion

The results indicate that ("lightly burned") low intensity prescribed burns will create small to medium changes creating openings in the forest. These results meet the project and program objectives at Bryce Canyon. Total fuel loading was reduced a little over half. Overstory, pole, and seedling densities were reduced with moderate changes. With all of the factors that influence vegetation succession and fire severity in the environment, a mosaic of patterns and intensities from fire can be expected. Fire behavior and intensity can vary from burn to burn, and will vary across the landscape producing different effects. These changes move the forest vegetation structure toward a less crowded forest where future prescribed or natural fires will function in maintaining an open vegetation mosaic. This is an example of what prescribed fire can do in this vegetation type. "Prescribed fires can be used as a means of reducing hazardous fuel loads and reestablishing pre-settlement fire regimes" (Jenkins 1995).

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Fire Process Research Natural Areas: Managing Research and Restoration of Dynamic Ecosystem Processes

Timothy Ingalsbee

Abstract—Since 1992 a collaborative group of fire scientists, forest conservationists, and Federal resource specialists have been developing proposals for a Research Natural Area (RNA) in the Warner Creek Fire area on the Willamette National Forest in Oregon. Inspired by these proposals, the Oregon Natural Heritage Plan created the new category of "Fire Process RNAs" in order to protect landscape-scale areas for dynamic ecosystem disturbance and succession processes resulting from wildland fires. Fire Process RNAs have many values for basic wildland fire research, ecosystem restoration, biodiversity conservation, and public education. They are especially suited to frequent-fire ecosystems, such as the ponderosa pine ecosystems of the Southwest and could serve as important control areas for measuring the effects of managed restoration activities and broad-scale environmental change due to global warming. This paper will: (1) present some of the history and theory of the evolving Warner Fire Process RNA proposals; (2) discuss some of the social issues, scientific controversies, and management challenges involved with designing and managing Fire Process RNAs; and (3) present the Warner proposal and its public support as a model to advocate for a network of Fire Process RNAs throughout fire-prone ecosystems of the West.

Introduction

In October 1991, arsonists ignited the Warner Creek Fire inside a protected Habitat Conservation Area (HCA) managed for the northern spotted owl (*Strix occidentalis caurina*). The wildfire burned across 8,900 acres of public wildlands, including the entire 6,800 acre Cornpatch Inventoried Roadless Area. The Warner Creek Fire became the second largest, costliest wildfire in the history of the Willamette National Forest. Fearing that the fire-killed snags and logs posed a risk of potential future catastrophic wildfire, the Forest Service proposed extensive salvage logging in order to reduce fuel loads and construct fuel breaks. This sparked a firestorm of controversy among conservationists who feared copycat arson-for-salvage incidents would occur in other protected habitat reserves.

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Over the next 4 years, the Warner Salvage Sale became one of the most controversial, contested timber sales in the country. The Clinton Administration finally withdrew the timber sale in 1996, and no salvage logging ever occurred inside the wildfire area. Currently, the Warner Burn represents one of the rarest landscapes in the Cascadia bioregion: a roadless, mid-elevation, largely unmanaged burned forest containing both young natural stands and high-mortality old-growth stands.

During the conflict over the Warner Salvage Sale, a collaboration among fire scientists, forest conservationists, and Forest Service resource specialists proposed that the Warner Burn be managed as a Research Natural Area (RNA) in order to research and restore natural fire disturbance and succession processes. This idea was first articulated in "Alternative EF: Ecology of Fire" published in the Warner Fire Recovery Project's Final Environmental Impact Statement. The Warner Creek Fire and Alternative EF inspired a formal revision of the Oregon Natural Heritage Plan that eventually created the new "Fire Process" RNAs. These were intended to protect large areas of public land for research, education, and restoration of dynamic ecosystem disturbance and recovery processes associated with wildland fire.

The vision of the Warner Fire Process RNA has been expanded in a more recent proposal that utilizes Conservation Biology principles to link the Warner Burn with five adjacent Inventoried Roadless Areas and two designated Wilderness Areas. This would form a landscape-scale fire ecology research complex suited to the fire regime of the westside Cascades. Anticipating future progressive developments in fire management philosophy and policy, proponents seek to develop a model fire management plan for the RNA that would facilitate research and restoration-oriented prescribed burning, wildland fire monitoring, innovative minimum-impact suppression techniques, and showcase fire ecology education in various interpretive trails and displays. However, as the Warner RNA Proposal evolves, it encounters increasingly complex social, scientific, and management issues that challenge not only our desires to "learn from the burn," but also our abilities to live with wildland fire.

This paper discusses the development of the citizen-initiated Warner Fire Process RNA Proposal out of the Warner Creek Fire and the Warner Fire Recovery Project. I hope that this paper will inspire efforts to develop additional Fire Process RNA proposals in other regions, especially the ponderosa pine ecosystems of the Southwest. I also hope that this paper will reinforce the growing understanding among the fire community that basic fire ecology field research and education are strategic needs that can provide essential management guides and critical public support for forest ecosystem restoration programs.

Warner Creek Fire

Ignited by arsonists during extreme drought conditions, the Warner Creek Fire entered the tree canopy almost from the point of ignition, and surged rapidly upslope. During a blow-up event, nearly 3,000 acres of prime spotted owl habitat were severely burned in a single afternoon. Stands of old-growth Douglas-fir, western hemlock, and western red cedar were completely scorched from ground to crown, affecting 42 percent of the burned acres. Another 25 percent of the area had some intermediate or mixed mortality ranging from 30–70 percent of the overstory trees. Approximately 33 percent of the area experienced a low-intensity underburn, removing most of the grasses, forbs, shrubs, and saplings, but causing little or no mortality of the overstory trees (Brown and others 1998: 4). This includes 512 acres that were completely unburned within the interior of the Warner burn, inside the designated old-growth grove located at the bottom of the Kelsey Creek basin. As the wildfire backed down the steep ridges, the fire was naturally extinguished when it entered the moist primeval forest in the valley bottom. Forest Service scientists later determined that fire has not burned in Kelsey Creek Basin for the last 850 years.

Although the Warner Creek Fire was ignited and propagated by unnatural ignition sources—criminal arsonists—the resulting mosaic of fire effects was representative of the natural fire regime of the westside Oregon Cascades. The fire was especially effective in restoring the ridgetop complex of dry meadows. These meadows were previously maintained by frequent lightning and Indian burning along the historic trail corridor atop Bunchgrass Ridge. Until the Warner Creek Fire, though, the meadows were declining due to fire exclusion. The elimination of Indian burning and aggressive suppression policies had both caused the meadows to shrink in size and species diversity. The wildfire killed many of the dense patches of young firs that were encroaching upon the meadows, and increased the vitality of the native bunchgrasses. For these and other reasons, the news media's depiction of the Warner Creek Fire as "catastrophic" contradicted the ecological effects of the fire.

Warner Fire Recovery Project

Two weeks after the Warner Creek Fire was declared out, the Forest Service initiated the Warner Fire Recovery Project. The stated purpose and need for the project was to "recover spotted owl habitat" and "increase knowledge about owl habitat recovery." The secondary underlying need—to increase knowledge—later proved vital in legitimizing an RNA alternative. The Forest Service's Inter-Disciplinary Team (IDT) working on the recovery plan originally assumed that the Interagency Scientific Committee's standards and guidelines for HCAs would prohibit salvage of any downed or standing trees (Thomas and others 1990: 325). Therefore, the IDT determined that the Project provided "an opportunity to set aside all or a portion of the fire area for studies of how both natural and managed landscapes respond to large scale fires (USDA-FS 1992: S-5)."

The category of Research Opportunities thus became a significant issue for analysis in the EIS. The idea of setting aside the Warner Burn for the study of natural fire recovery processes was quickly adopted by the public, including members of a special citizen advisory group organized by the Willamette National Forest to give regular input on the Recovery Project. A majority of the Warner Public Participation Group formally requested that a new alternative be created that would designate the burn as a Research Natural Area (RNA), and develop a new fire management plan that allowed prescribed natural fires (PNFs). Unfortunately, the Willamette National Forest declared that both RNAs and fire planning were issues "outside the scope of the project," and excluded these from the Draft EIS.

Not satisfied with that decision, the citizen advisory group networked with fire scientists and forest ecologists from across the Pacific Northwest to develop their own proposal for an RNA-based fire recovery plan, and submitted this during the public comment period. The proposal was called "Alternative EF: Ecology of Fire." Later, when the Willamette's Draft Preferred Alternative was withdrawn by the Forest Service's Owl Oversight Committee because it was deemed inconsistent with the agency's owl conservation strategy, Alternative Ecology of Fire was authorized to be fully developed, analyzed, and published in the Final EIS. This set in motion a truly collaborative effort of agency specialists, conservationists, academics and independent researchers to further develop the concept of a Fire Process RNA for the Warner Burn.

Alternative EF: Ecology of Fire

The goal of the authors of Alternative EF (Alt. EF) was to propose a recovery strategy that centered around research and restoration of wildland fire processes. Alt. EF proposed allowing natural succession processes to recover owl habitat, and a fire management strategy using PNFs to help protect owl habitat from future severe wildfires. The entire burn was divided into four zones for PNF prescriptions that ranged from low to moderate intensity in the zone containing spotted owl nest sites, and higher intensity in the ridgetop meadow zone. Along with PNFs, an active program of fire effects monitoring and research was proposed in order to study both natural fire disturbance and recovery processes.

Alt. EF prioritized PNFs, but also allowed some management-ignited prescribed fires, especially in the ridgetop meadow zone. Such fires could occur only after intensive fire history research was completed that included anthropological research on the frequency, locations, and methods of historic Native American burning along Bunchgrass Ridge. In a move that preceded the Federal Wildland Fire Management Policy and its concept of "Appropriate Management Response," Alt. EF allowed limited and modified suppression activities to occur in order to keep natural fires within their prescription. In the event that some fires exceeded their prescribed fire intensity, Alt. EF mandated that only minimal-impact suppression tactics (MIST) could be used within the burn. The concern was that aggressive suppression actions would not only damage the environment, but might destroy ongoing research projects and monitoring plots.

Rationale for Prescribed Underburning in Spotted Owl Stands

The project's decisionmaker and IDT believed that both prescribed and wildland fires would adversely affect owl nesting and roosting habitat. However, unbeknownst to the authors of Alt. EF at that time, the fire and fuels management team working on the California Spotted Owl conservation plan (a.k.a. the "CASPO" Report) urged that prescribed underburning be used to prevent stand-replacing wildfires in spotted owl Protected Activity Centers (Verner and others 1992: 254). Proponents of Alt. EF argued that, logically, spotted owls evolved with natural succession processes and recurring fire disturbances, and observed that most existing owl stands exhibit some evidence of past fires. Proponents added that fires are a prime agent creating forest structure such as multi-storied canopies and large snags and logs that were vital components of superior spotted owl habitat.

Furthermore, RNA proponents rejected the agency's strategy of protecting owl habitat reserves with fire exclusion—a policy that had already essentially been "vetoed" by the arsonists who ignited the Warner Creek Fire. Conservationists took comfort in the fact that the resident population of spotted owls continued to inhabit and successfully reproduce in the burn. Indeed, the continued existence of the owls inside the burn challenged the assumption that intensive management, especially salvage logging, was necessary to recover the burned area.

Alt. EF was endorsed by prestigious academic members of the research community, including some of the scientists who helped design the Northwest Forest Plan. The student governments of Oregon's two largest universities passed official resolutions in support of Alt. EF and later sent these to Forest Service Chief, Jack Ward Thomas. Then, the Forest Service's Regional RNA Coordinator and ecologists from the Pacific Northwest Research Station drafted their own RNA proposal for the Warner Burn. Finally, the Oregon Natural Heritage Advisory Board used the Warner Creek Fire and Alt. EF as inspiration to establish a new "Fire Process" cell for their network of RNAs. This created a qualitatively new kind of RNA aimed to protect areas for their dynamic ecosystem processes rather than static species composition or geologic features. Consequently, although the Willamette National Forest's final recovery plan was to construct fuelbreaks with salvage clearcuts, the decisionmaker also set aside a 4,200 acre portion of the burn as a "Natural Succession Area" (NSA) to be later considered for designation as an RNA.

Warner Fire Recovery Plan Voided

Implementation of the Willamette National Forest's final recovery plan was repeatedly delayed first by a dozen administrative appeals and then by a lawsuit that resulted in a permanent injunction against the salvage sale. All of these delays occurred during a time of rapid change in forest and fire management policies initiated by the Northwest Forest Plan and the Federal Wildland Fire Management Policy and Program Review. In the midst of a nationwide protest campaign whose slogan was "Stop the Warner Salvage Sale; Save the RNA!" President Clinton ordered the timber sale to be withdrawn in August 1996. No salvage logging ever occurred inside the Warner burn, and the Willamette National Forest essentially selected the No Action Alternative by default.

Warner Fire Process RNA Proposal

The group of "citizen-scientists" who had drafted Alt. EF developed a new, more expansive RNA proposal, and submitted it to the Regional RNA coordinator in September 1997. Called the "Warner Proposal" for the sake of brevity, it was no longer confined to the 9,000 acre wildfire perimeter. New boundaries were drawn along suitable topographic features and landforms that would aid minimal-impact fire confinement strategies and minimize the need for aggressive suppression.

Using Conservation Biology principles, the Warner Proposal linked the burn with a cluster of four other Inventoried Roadless Areas (RAs). These RAs, in turn, were adjacent to two contiguous Wilderness Areas along the Cascade Crest that had recently developed PNF plans. The original proposal was a 48,000 acre RNA, but this has recently been reduced to approximately 31,000 acres in accordance with the results of a 1999 symposium on Fire Process RNAs. The 31,000 acre Warner RNA connected with 336,000 acres of designated Wilderness would form a landbase where it is hoped that large-scale wildland fire processes could be managed for research and restoration purposes.

The core of the Warner Proposal includes the entire 9,000 acre Warner Burn. This is one of the rarest landscapes in the westside Oregon Cascades: a roadless, mid-elevation, relatively unmanaged, recently burned forest containing both young natural stands and high-mortality old-growth stands. A wide range of disturbance intensities, plant associations, and stand conditions currently exist. Early and late seral stages following large-scale fire disturbances are not adequately represented within existing RNAs. Thus, the Warner Burn has the potential to capture early seral stages of plant communities protected in other RNAs. The Warner Burn presents opportunities to study not only single elements or communities encompassed within it, but also the complex array of fire intensities and communities, their arrangement, connections, and interrelationships through time. The relatively large area and diversity of topography, vegetation, microclimates, habitats, and fire effects make the Warner Burn and its surrounding unburned area well suited for a broad range of research and educational uses.

Agency Responses to the Citizens' Warner RNA Proposal

The Willamette National Forest responded to the Warner Proposal by resurrecting the decisionmaker's 4,200 acre NSA proposal minus the fuelbreaks and salvage clearcuts. However, during the public scoping period the agency received over 1,000 comment letters and every single letter rejected the NSA proposal. The chief criticism was that the NSA was too small and its boundaries too arbitrary to allow wildland fire use. Instead, the letters demanded that the citizen-scientists' Warner RNA Proposal be included in the EIS process.

The results of the scoping period prompted the Willamette National Forest and the Pacific Northwest Research Station to host a roundtable workshop of fire scientists in April 1999. The assembled scientists from Oregon, Washington, and British Columbia discussed design and management criteria for Fire Process RNAs. The scientists agreed that landscape-scale RNAs were needed in order to best capture natural fire processes at the spatial and temporal scales they function in the westside Oregon Cascades. The scientists determined that for a mid-sized Fire Process RNA, approximately 30,000 acres containing portions of at least two watersheds would be needed to maximize research opportunities. The end result of the symposium was that the scientists validated the general principles articulated in both Alt. EF and the new Warner Proposal.

Ongoing Activities to Learn From the Burn

Despite the lack of formal RNA status, the Warner Burn has attracted fire ecology research and educational activities throughout the last decade. For example, 45 monitoring plots have been established by Forest Service ecologists in order to document the early structure, composition, and regeneration of the burn. If future Forest Service budgets allow, these plots will be resurveyed as part of a long-term monitoring plan. Scientists from the University of Oregon and Oregon State University have also conducted some studies on soils, vegetation, and wildlife in the burn.

The Cascadia Fire Ecology Education Project (CFEEP), a nonprofit conservation organization, and the Northwest Youth Corps (NYC), an alternative high school for at-risk youth interested in pursuing forestry careers, have embarked on a partnership to establish long-term fire effects monitoring plots. CFEEP and NYC have initiated a unique snag longevity study to monitor the rate of fall and decay of fire-killed snags and logs over the next several decades. The methodology for the snag study was developed with the help of Forest Service ecologists and sustainable forestry consultant, Chris Maser. (Known as one of the early "gurus of old-growth" while he was a Bureau of Land Management ecologist stationed at the Pacific Northwest Research Station, Maser was the lead author of the 1988 publication, "From the Forest to the Sea: A Story of Fallen Trees" (Gen. Tech. Report PNW-GTR-229)). The nonprofit groups are now applying the National Park Service fire effects monitoring protocol to their current and future research plots in the burn.

Additionally, students from all across the country have attended special fire ecology field seminars and research outings in the Warner burn as part of curriculum for the Wildland Studies Program and the Cascade Science School. The Warner burn is conveniently located next to a major highway just one hour's drive from the city of Eugene, Oregon. Educators from local universities and school districts are becoming some of the more vocal advocates of the Warner Proposal because it has great potential as an outdoor education site and living laboratory for academic research projects. This fulfills one of the oft-neglected purposes of RNAs to foster education, as well as conduct research and conserve biodiversity (USDA-FS 1997: 2). Accordingly, it is hoped that formal protective status as an RNA would enable future generations of scientists and students continual opportunities to "learn from the burn."

Controversies and Challenges of Managing Fire Process RNAs

RNA proponents are urging the Willamette National Forest to proceed with the EIS process and include a citizen-scientist RNA Alternative in the document. This effort conforms with the recent Committee of Scientists' Report that encouraged more up-front collaboration with scientists in Forest Service planning and projects (Committee of Scientists 1999: xxv). The Federal Wildland Fire Management Policy and Program Review also lauded the role of communication and collaboration, and has called for a renewed emphasis on public participation and partnerships in all aspects of wildland fire management (USDA/USDI 1995: iii). If and when the Willamette National Forest moves forward on the proposal, the fire research community will want to review and submit substantive comments on the EIS. The document will likely raise several critically important, controversial issues that are pertinent to other fire/fuels management programs and projects. With active participation of the fire research community, RNA proponents hope that the Willamette National Forest will produce a model fire management plan focused on research and restoration of wildland fire processes. This should develop a template to help propose and establish additional fire process RNAs in other regions of the National Forest System, especially in the Southwest.

Socioeconomic Issues

An RNA designation would preclude intensive management and commodity resource extraction activities. Accordingly, persuading the citizenry that some large acreage of commercially-valuable timber lands should be set aside for the so-called "wrath of wildfire" will present a public relations challenge, to say the least. The specific Warner RNA proposal largely mitigates this issue by incorporating lands such as LSRs and RAs that make future commercial timber extraction problematic. Timber sales will be even more unlikely if President Clinton's Roadless Area Protection Initiative results in protection of RAs from further logging as well as road-building. Yet, the only commercial logging that would have been affected within the 31,000 acre Warner RNA Proposal was the Helldun timber sale, located just outside the burn perimeter. This timber sale was withdrawn in spring 2000, due to widespread public opposition to its potential adverse effects on the RNA proposal.

Research and Restoration Jobs-in-the-Woods—Too often land managers consider RNAs to be strict land "set-asides" or "lock-ups" that only benefit researchers; however, there are a number of potential socioeconomic benefits that could accrue from a large-scale RNA (Tyrrell, n.d.: 6). Just in terms of research jobs there could be substantial opportunities for workers needed to establish research plots, conduct field surveys, collect baseline data, initiate research projects, construct trails, and monitor wildland and prescribed fires. Restoration could include jobs for road obliteration, in-stream rehabilitation, invasive weed eradication, and hazardous fuels reduction with manual and prescribed fire treatments. Of course, these research and restoration jobs would have to be funded by alternative sources such as Federal appropriations, grants, endowments, and so forth rather than traditional commodity-producing projects. But given a suitable funding source, the number and kinds of research/restoration jobs and other socioeconomic benefits that could accrue from a fire process RNA over the next century or more is limited only by one's imagination.

Additionally, some have raised the idea of possibly allowing limited commercial extraction in special buffer zones that would straddle the RNA's boundaries. Thus, for example, a fuels management program to reduce hazardous fuels and construct defensible boundaries might allow extraction of some of the understory trees and dead surface fuels to supply a commercial firewood operation. Other commercial enterprises, such as mushroom harvesting, might also be permitted in these buffer zones along the RNA boundaries. Providing jobs, resources, and other socioeconomic benefits in these buffer zones might make the RNA proposal more attractive to certain sectors of the public and political representatives; however, these would have to be subordinated to the overriding goal of protecting the RNA from adverse edge effects that could potentially be caused by commercial activities.

Scientific Controversies

There are genuine unresolved questions and controversies over the relationship of northern spotted owls, "westside" old-growth forests, and wildland fires. These unknowns became the very impetus and driving rationale for the RNA alternative in the Warner Fire Recovery Project. It is probably a safe assumption that a fire process RNA will entail some degree of extra risk of potential loss of some components of owl habitat. However, it is hypothesized that given sufficient time, natural regeneration and succession processes can and will recover burned sites from the effects of fire disturbances. Given the long fire return intervals, and the long period required to develop superior spotted owl nesting habitat, the testing of this hypothesis might not come for several decades, even centuries. This is an argument in favor of RNA status that would provide long-term protection against intrusive or intensive management that would alter the natural processes and potentially invalidate the data.

Role of Early Seral/Successional Stages in Old-Growth Development—In the dominant mode of managed "recovery" and "reforestation," Forest Service managers typically prefer to skip early successional stages following fire or logging disturbances and plant nursery-bred conifers. However, it remains a mystery whether or not fire disturbances and early successional stages are necessary elements of long-term old-growth development. For example, recurring fires could stimulate pulsed regeneration and self-pruning mechanisms of trees that help develop all-aged, multistoried canopy structures that are preferred by northern spotted owls. Likewise, it is recognized that fires are a prime disturbance agent creating large-diameter snags and logs that are vital habitat structures to owls and their prey. An RNA could help reveal how natural fire disturbance and recovery processes function over the long-term, and serve as a control area to compare with managed areas undergoing quasi-experimental silvicultural treatments.

Presence of Past Management Impacts in an RNA—Another controversial issue concerns the presence of past management impacts inside a new RNA. Inside the Warner burn alone, there are 77 miles of logging roads and 28 plantations whose dense stands of young artificial regeneration were destroyed by the wildfire. These plantations caused fire severity to increase as ember-filled convection columns preheated and ignited the canopies of adjacent old-growth stands. Logging roads and plantations are not suitable for an "elemental" RNA designation, but for a "process" RNA the story is different. It should be assumed that fire behavior and effects will differ in managed versus unmanaged stands; however, it is hypothesized that fire disturbance processes would continue to function naturally on a landscape scale.

In regions such as the Pacific Northwest, it is difficult to find large expanses of intact wildlands that had escaped the Forest Service's road-building and logging program of the 1970s and 1980s. This program had specifically targeted RAs for development. Realizing this fact, the Oregon Natural Heritage Advisory Board came up with the criteria that up to 10 percent of the area of a fire process RNA could contain previously managed sites and still be deemed "natural" enough for RNA designation. The important factor is that the remaining 90 percent of an RNA should be natural, with predominantly native species composition, and lacking evidence of significant management-caused alteration.

Given the extent of forest fragmentation and ecosystem alteration from past intensive management activities, the most likely location containing the quantity and quality of natural areas suitable for fire process RNAs would be found in Roadless Areas. As occurred with the Warner proposal, Fire Process RNAs of the future will likely be proposed "opportunistically" following a wildland fire in an RA. This provides a baseline fire event to study, and prompts significant public interest that could support (or possibly oppose) RNA designation. However, in the frequent fire regimes of the Southwest, some visionaries might take the initiative to propose some RAs for RNA status in anticipation of a future fire event. This prefire proactive planning would provide a much-needed antidote to the current system of "emergency" wildfire suppression followed by "emergency" salvage logging.

Management Conflicts

Other issues that have been raised during the development of the Warner RNA proposal revolve around managing the RNA for research, education, and conservation of biological diversity, but also mixing these with "multiple uses" such as recreation. RNA proponents have argued for essentially a "passive" restoration strategy focused on PNFs to restore the fire regime. But a more "active" restoration strategy utilizing management-ignited prescribed fires could also be used as means of restoring and maintaining biodiversity.

Questions arise as to whether these prescribed fires could be considered suitable research activities, as well as maintenance tools, in the RNA.

Prescribed Fire as a Research Subject and Restoration Tool—Fire history research to date suggests that the frequency of natural ignitions may not be sufficient to restore the natural fire regime of the Warner area given nearly a century of fire exclusion and the elimination of historic Native American burning. Although there are strong advocates for wildland and prescribed fire use in RNAs specifically to maintain ecological processes, most forest managers have opted for a "hands off" approach to fire management (Johnson 1983: 39). Unfortunately, out of 79 established RNAs that are comprised of fire-dependent plant communities, only five of these RNAs undergo prescribed burning; the rest are declining in species, structural, and seral diversity due to fire exclusion (Greene and Evenden 1995: 32). Fortunately, one of these RNAs receiving periodic prescribed burning is located approximately 60 miles from the Warner burn in an eastside Cascade ecosystem on the Deschutes National Forest. In the Metolius RNA, scheduled prescribed burns are conducted as part of a long-term monitoring program serving both restoration and research objectives (Riegel and Youngblood 1999). The Metolius RNA prescribed burning program offers a working model for Willamette managers to apply to a fire process RNA in the Warner burn.

Potential Recreation Impacts on Research Projects—During the development of Alt. EF, a conflict arose within the research community over the belief by some scientists that recreational activities were inappropriate uses for the RNA. Some scientists opposed the construction of a fire ecology interpretive trail out of fear that off-trail hikers would unwittingly trample upon research plots, or vandals would deliberately destroy research sites and equipment. The Warner Proposal, however, includes sufficient acreage that the potential risk of a catastrophic loss of research plots should be minimized. And protection of research sites should be a prominent theme for interpretive and educational programs. National Parks must frequently grapple with the dilemma of providing for "recreational playgrounds and natural area preservation"; hence, National Park Service employees may want to lend some expert advice to the Forest Service on how to manage for these seemingly contradictory uses of the Warner RNA.

Systemic Lack of Funding of RNA Program—A perplexing issue concerns the current general lack of funding for the RNA program in the Pacific Northwest Region (Greene 1999: 3). The Willamette Supervisor cited "a climate of dramatically reduced budgets for RNAs" as one of the reasons for refusing to proceed with an EIS for the Warner proposal. Indeed, more demands are being placed on Forest Service researchers with less staffing and budgets available. However, the example of the privately-funded CFEEP-NYC research program demonstrates that the Federal government could tap into local community labor resources to get some of the "grunt work" of baseline data collection and field research projects started. Managers are thus being urged not to restrict themselves to their agency's internal resources of budgets, staff, and expertise, but to look to forming collaborative partnerships with the research and conservation communities to supplement limited Federal resources.

Lack of an Approved Fire Management Plan Continues Commitment to Fire Exclusion—Perhaps the most frustrating issue that affects the viability of the Warner proposal is the fact that the Willamette National Forest lacks an approved Fire Management Plan. The Willamette National Forest's current Land and Resource Management Plan as amended by the Northwest Forest Plan does not provide for wildland fire use, and commits to total, aggressive suppression in Late-Successional Reserves (LSRs). Approximately 15,000 acres or 48 percent of the citizens' proposal is currently managed as a LSR, and Willamette National Forest managers have raised the specter of "unfettered wildfires" in the RNA potentially destroying the habitat values of the LSR.

The concept of wildland fire use burning under *planned prescription* is key to agency and public acceptance. If and when a natural or management-ignited fire exceeds its preplanned prescription window, then it would be converted to a wildfire and suppressed. RNA proponents are clear about their wider policy objectives in establishing the Warner Fire Process RNA: it is intended to serve as a kind of testing ground for changing fire management policies and practices in the Pacific Northwest. It is hoped that the Warner RNA will provide a safe space far away from private property or human communities where not only can people learn from the burn, but can learn to live with fire as a vital ecosystem process.

Conclusion

At the time of this writing (September 2000) the current status and future fate of the Warner Fire Process RNA proposal is uncertain. On the one hand, Forest Service officials claim that their budget does not have enough money to fund an EIS for the RNA. On the other hand, the Warner Proposal continues to generate growing interest and endorsements from scientists, researchers, educators, and resource specialists in the Forest Service and other land management agencies.

With support from the Oregon Governor's office, Congressman Peter DeFazio has submitted a funding request for the EIS in the 2001 Appropriations Bill in order to provide the Willamette National Forest with the money needed to do the analysis and establish the RNA. Congress is gaining more interest in fire research and fuels management, as evidenced by its continuing funding of the Joint Fire Sciences Program, the Hazardous Fuels Reduction Program, and proposed massive budget increases for Federal fire management programs in the 2001 Interior Appropriations bill. There is also a growing effort by researchers and conservationists familiar with the Warner Proposal to develop citizen-initiated Fire Process RNA proposals in other locations around the West. Proponents argue that the Warner Proposal and Fire Process RNAs are ideas whose time has come, and look forward to inspiring the public, fire professionals, and political representatives on the critical need for fire ecology research and education to guide ecosystem restoration programs in fire-dependent/fire-adapted ecosystems.

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Six-Year Changes in Mortality and Crown Condition of Old-Growth Ponderosa Pines in Ecological Restoration Treatments at the G. A. Pearson Natural Area

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Abstract—Ecological restoration treatments using thinning and prescribed burning have been proposed to reverse the decline of old-growth ponderosa pines in the Southwest. However, long-term data on the effectiveness of such treatments are lacking. In 1993–1994, two ecological restoration treatments and a control were established at the G. A. Pearson Natural Area (GPNA) near Flagstaff, AZ. The thinned treatment removed many postsettlement-aged trees to create tree density and stand structure similar to pre-Euro-American forests. The thinned + burned treatment included prescribed burning of the forest floor in combination with this thinning. The control was a dense stand of pre- and postsettlement trees with no thinning or burning. Crown dieback of presettlement trees decreased by about 3 percent in both thinned treatments over 6 years since initiation of treatments, whereas dieback increased by about 4 percent in the control. Crown dieback was not related to tree age. Change in height of presettlement trees between 1994 and 2000 did not differ among treatments. Of 146 presettlement trees monitored for survival between 1994 and 2000, four died between 1997 and 2000, and all were in thinned treatments. Two of the four trees that died toppled or the stem broke in a severe windstorm in 1997. The other two dead trees died between 1997 and 2000 following the severe regionwide drought of 1996. These two dead trees had large amounts of canopy dieback prior to treatment initiation, suggesting that thinning did not contribute to their mortality. Our results indicate that heavy thinning of postsettlement trees improved the crown condition of presettlement trees at the GPNA over 6 years since treatment, but also may have increased windthrow and wind breakage.

Introduction

Ponderosa pine (*Pinus ponderosa*) forests in northern Arizona have changed dramatically in stand structure over the past century due to human alteration of the natural fire regime, overgrazing, and logging (Arnold 1950; Cooper 1960; White 1985; Covington and Moore 1994; Fulé and others 1997). In most areas of northern Arizona, fires have been suppressed for at least 87 years (Madany and West 1983; Savage and Swetnam 1990). Pre-Euro-American settlement forests were characterized by clusters of old pines separated by lush grassy areas. These open canopy conditions were maintained by low-intensity surface fires that occurred every 2–12 years (Weaver 1951; Cooper 1960; Dieterich 1980; Dieterich and Swetnam 1984; Swetnam and Baisan 1996). The present forests are currently dominated by dense thickets of sapling- and pole-sized trees and have heavy accumulations of litter and woody fuels, and poor understory grass, forb, and shrub growth (Cooper 1960; Harrington and Sackett 1992; Covington and Moore 1994; Dahms and Geils 1997).

There is currently much interest in improving the condition of Southwestern ponderosa pine forests by thinning to reduce competition among trees and by prescribed burning to reduce fuel loads. However, few studies have examined the effects of such thinning and burning treatments on presettlement ponderosa pines. Studies of historical tree growth rates of presettlement ponderosa pines in northern Arizona have suggested that the remaining presettlement trees are declining because of competition from postsettlement trees (Sutherland 1983; Biondi 1996). Two studies have reported negative effects of prescribed burning on the growth, water relations, and survival of presettlement trees (Sutherland and others 1991; Swezy and Agee 1991). In both studies, the burns were conducted in stands containing heavy fuel loads due to fire suppression, causing heat damage to the presettlement trees.

We initiated a study in 1993 designed to evaluate techniques for improving the condition of ponderosa pine forests in the Gus Pearson Natural Area (GPNA) in northern Arizona (Covington and others 1997). The GPNA is a unique, relict stand of presettlement ponderosa pines that has been protected from harvesting and heavy grazing. The treatments included thinning of postsettlement trees to create a more open stand condition, and a combination of thinning and prescribed burning. The influence of these treatments on the growth, water, carbon, and nutrient relations of presettlement trees have been reported by Stone and others (1999) for the first year following treatment, and by Feeney

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and others (1998) for the second and third years following treatment. These authors concluded that thinning increased uptake of water and nitrogen of presettlement trees, and these changes contributed to greater rates of photosynthesis and stem radial growth. Differences in growth and physiological characteristics between the thinned treatment and the thinned + burned treatment were small, except for tree resin flow, which was higher in the thinned + burned treatment than in the thinned treatment (Feeney and others 1998).

In this paper, we compare mortality and crown condition of presettlement trees 6 years after initiation of the restoration treatments at the GPNA. Measurements of these characteristics made in 1994 are compared to measurements made in 2000 to show temporal changes in each treatment area. Our general hypothesis is that the positive effects of the thinning and thinning + burning treatments on presettlement tree condition that were documented for the first 3 years after the initiation of treatments (Feeney and others 1998; Stone and others 1999) would continue through year 6. Specifically, we expected lower tree mortality and a greater improvement in crown condition in thinned and thinned + burned treatments compared with the unthinned and unburned control.

Methods

Study site

The study was conducted at the GPNA, within the Coconino National Forest approximately 10 km northwest of Flagstaff, Arizona. The study site occupies 3 ha of the 30-ha GPNA at an elevation of 2,195–2,255 m. Aspect is generally southwest, with gentle topography (slopes 0–5 percent). Soils, derived from basalt and volcanic cinders, are classified as a Brolliar stony clay loam, fine, smectitic, Typic Argiboroll. Mean annual precipitation in Flagstaff is 56.7 cm, with approximately one-half of the precipitation falling as snow from November to May and one-half falling as rain primarily during the late-summer monsoon season (July and August). Mean annual air temperature in Flagstaff is 7.5 °C. The climate is subhumid, with early summer droughts common. The average frost-free growing season is 94 days (Schubert 1974).

The vegetation community at the GPNA is a previously unharvested ponderosa pine stand that is uneven-aged with even-aged groups of pole-sized trees and uneven-aged groups of presettlement trees (Schubert 1974; White 1985). Pole-sized trees (10–37.4 cm d.b.h.) are the predominant size class. The predominant Euro-American influences at the GPNA have been livestock grazing, which occurred between 1876 and 1910, and fire suppression. The last natural fire was in 1876; prior to that time, the fire return interval averaged approximately 2 years (Dieterich 1980).

Experimental Design and Treatments

The GPNA study site was subdivided into three largely contiguous areas, each occupying approximately 1 ha (Covington and others 1997). Each area was subjected to one of three treatments: control, thinned, and thinned + burned. Each treatment area was further subdivided into five approximately 0.20-ha plots that served as experimental units. We grouped these plots into five blocks (one plot per treatment) that were used to implement all measurements.

Because our experimental design does not include true spatial replication across the landscape, we used precautions to strengthen our inferences regarding treatment effects. Specifically, an earlier analysis (Stone 1997) showed similar pretreatment levels of soil total nitrogen (P = 0.19), phosphorus (P = 0.25), organic matter (P = 0.19), predawn leaf water potential (P = 0.82 for May to June, P = 0.25 for July to August), basal area growth rate (P = 0.96), stem diameter (P = 0.32), and tree-to-tree competition index (P = 0.66) of presettlement trees among the areas to be treated. Thus, resource availability, tree growth, and tree physiological condition were similar among areas prior to treatment. Second, our interpretation of statistical results applies only to the specific locations in the GPNA where the treatments were applied.

The thinning was conducted to simulate the presettlement (circa 1876) stand structure, which was determined using dendrochronological techniques (Mast and others 1999). The average pretreatment basal area was 34.5 m^2 ha^{-1} (average of 3,100 trees ha^{-1}), which was retained in the control area. In November of 1993, two-thirds of the site was thinned to an average basal area of 13.0 m^2 ha^{-1} (average of 151 trees ha^{-1}), with all presettlement trees and all trees greater than 40 cm d.b.h. retained. Following the thinning treatment, the unthinned control area had an average d.b.h. of 16.6 cm in 1993, while both thinned areas had an average d.b.h. of 40.9 cm (Covington and others 1997). Thinning did not substantially change light availability to the presettlement trees used in the study because the crowns of the thinned postsettlement trees were lower than the crowns of the presettlement trees.

Half of the thinned area was also subjected to a low-intensity prescribed burn in 1994 (October) and 1998 (October). Prior to the first burn in 1994, the O_i (slightly decomposed organic matter), O_e (moderately decomposed organic matter), and O_a (highly decomposed organic matter) layers of the forest floor and woody debris were removed by hand raking to simulate presettlement forest floor conditions, which would have had little forest floor litter and debris because of frequent fire. The O_i layer (2–4 years of litterfall) was replaced prior to burning with dried native grass foliage from a nearby prairie (672 kg ha^{-1} dry biomass) to simulate presettlement forest floor conditions in which grasses were dominant. Fire characteristics for the 1994 burn were previously reported by Covington and others (1997); flame length averaged about 15 cm with maximum lengths of 60 cm. For the second burn in 1998, flame length averaged 11 cm with maximum lengths of 180–240 cm that occurred in fallen limbs and needles of windthrown trees.

Crown Dieback and Mortality

We measured crown dieback of 71 presettlement trees in July 1994 and March 2000. Of these trees, 27 were in control plots, 22 were in thinned plots, and 22 were in thinned + burned plots. These trees were selected for measurement in 1994 because all had good crown condition with little dieback

Table 1—Percentage classes for crown dieback.

Percentage class	Class range
0	0
5	1–5
10	6–15
20	16–25
30	26–35
40	36–45
50	46–55
60	56–65
70	66–75
80	76–85
90	86–95
100	96–100

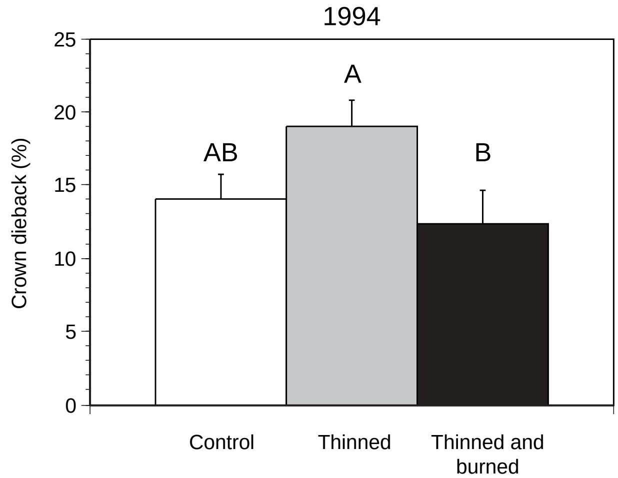

Figure 1—1994 average crown dieback of presettlement ponderosa pines at the GPNA in three treatments: control, thinned, and thinned + burned. The bars show one standard error of the mean. Means with the same letter do not differ significantly in Fisher's Protected LSD tests ($P \leq 0.01$).

prior to initiation of the treatments and were located at least 10 m from treatment boundaries, which reduced potential influences from other treatments. Crown dieback of each tree was visually assessed by at least three trained observers using a system with 12 percentage classes (table 1) developed for forest health monitoring (Millers and others 1991; Kolb and McCormick 1993). The same leader (T. E. Kolb) trained assessment crews in both 1994 and 2000, and he checked the crown condition assessment of all trees in both years for accuracy. Crown dieback was defined as the percentage of total crown volume that contained dead branches with bark or with branch tips less than 2.5 cm diameter (Millers and others 1991). We also measured the height of all trees with a clinometer at the time of crown dieback assessment in both years. Tree mortality status (live or dead) was noted in 1994, 1996, and 2000 on a total of 146 presettlement trees that consisted of the same 71 trees measured for crown dieback and an additional 75 trees.

Data Analysis

We compared differences in crown dieback and height growth among the control, thinned, and thinned + burned areas of the GPNA using a fixed-effects analysis of variance (ANOVA) on plot means. Mean comparisons among treatments were performed with Fisher's protected LSD, and a threshold P value of 0.10 was used in all tests because of the inherent large variability of a population of old trees. All statistical analyses were conducted using the SAS JMP statistical package (SAS Institute Inc., Cary, NC, U.S.A.). Only trees that were living in both 1994 and 2000 were included in the analysis of crown dieback.

Results

Crown Dieback

Crown dieback of presettlement trees differed significantly ($P = 0.073$) among treatment areas in 1994 when treatments were initiated. Dieback was significantly greater in the thinned treatment (mean = 19.0 percent) compared with the thinned + burned treatment (mean = 12.4 percent) (fig. 1). Dieback did not differ significantly between the control (mean = 14.0 percent) and either thinned treatment in 1994 (fig. 1).

Crown dieback also differed significantly ($P = 0.037$) among treatments in 2000, 6 years after the initiation of treatments. In contrast to the results in 1994, dieback was greatest in the control (mean = 18.0 percent), intermediate in the thinned treatment (mean = 14.3 percent), and lowest in the thinned + burned treatment (mean = 10.4 percent) (fig. 2). The difference in dieback between 1994 and 2000 was

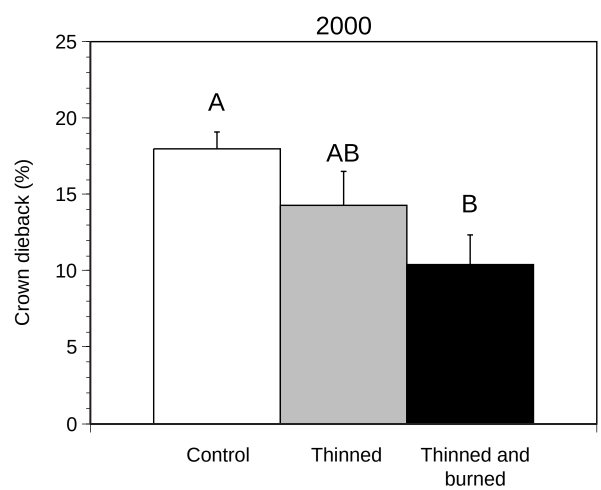

Figure 2—2000 average crown dieback of presettlement ponderosa pines at the GPNA in three treatments: control, thinned, and thinned + burned. The bars show one standard error of the mean. Means with the same letter do not differ significantly in Fisher's Protected LSD tests ($P \leq 0.01$).

caused by a difference among treatments in temporal change in crown condition. In the control, crown dieback increased between 1994 and 2000 by an average of 3.5 percent, whereas crown dieback decreased during this period in the thinned (–3.9 percent) and thinned + burned (–2.5 percent) treatments (fig. 3). This difference in temporal change in dieback between the control and thinned treatments was significant (P = 0.037).

There was no relationship between tree age and 2000 crown dieback (R = 0.153, P = 0.284) or 1994–2000 change in dieback (R = 0.103, P = 0.473) for data pooled over all treatments (figs. 4 and 5). Correlations between these variables were also not significant for data analyzed by treatment group, except for 2000 crown dieback in the thinned + burned treatment which was positively associated with tree age (R = 0.466, P = 0.059) (fig. 4).

Change in Tree Height

Tree height growth, expressed as a percentage change between 1994 and 2000, was highest in the thinned + burned treatment (15.8 percent), intermediate in the thinned treatment (14.7 percent), and lowest in the control (12.8 percent), but these differences were not significant (P = 0.411) (fig. 6).

Tree Mortality

There was no mortality of the 146 trees between 1994 and 1996. However, four trees died between 1997 and 2000. Of these dead trees, all were located in thinned or thinned + burned plots. In thinned plots, 1 of 30 trees died. In thinned + burned plots, 3 of 49 trees died. In control plots, 0 of 67 trees

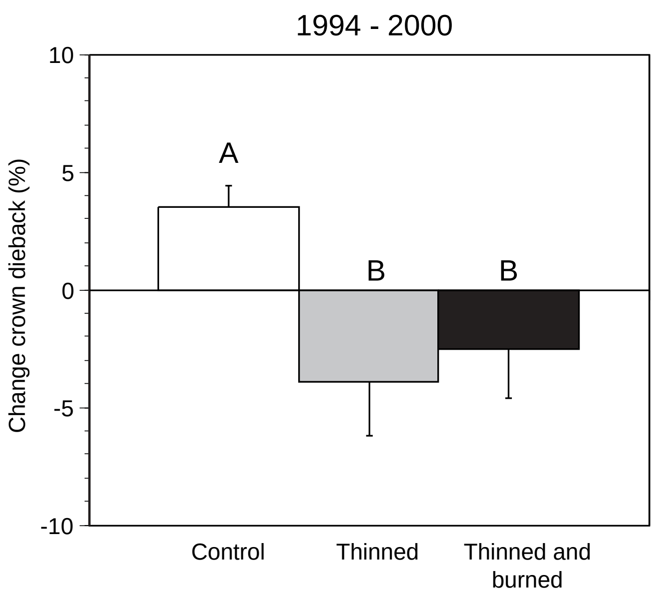

Figure 3—Average change in crown dieback between 1994 and 2000 of presettlement ponderosa pines at the GPNA in three treatments: control, thinned, and thinned + burned. The bars show one standard error of the mean. Means with the same letter do not differ significantly in Fisher's Protected LSD tests (P ≤ 0.01).

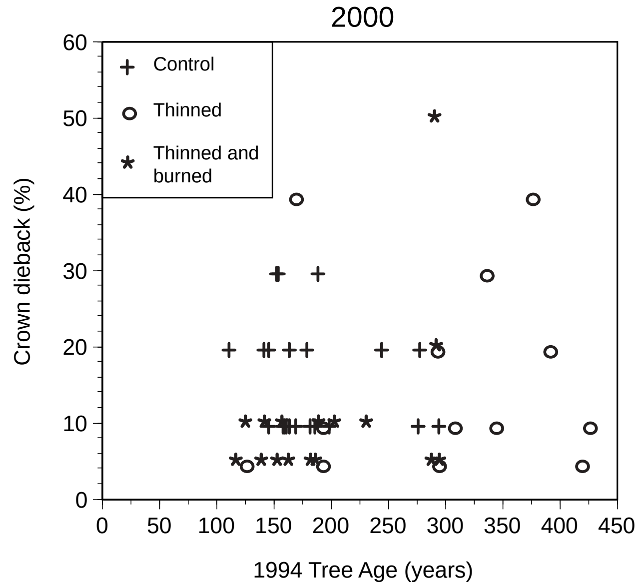

Figure 4—2000 crown dieback of presettlement ponderosa pines at the GPNA versus 1994 tree age in three treatments: control, thinned, and thinned + burned. Correlation coefficient for data pooled over all treatments is R = 0.153 (P = 0.284), control data R = 0.183 (P = 0.427), thinned data R = 0.062 (P = 0.841), and thinned + burned data R = 0.466 (P = 0.059).

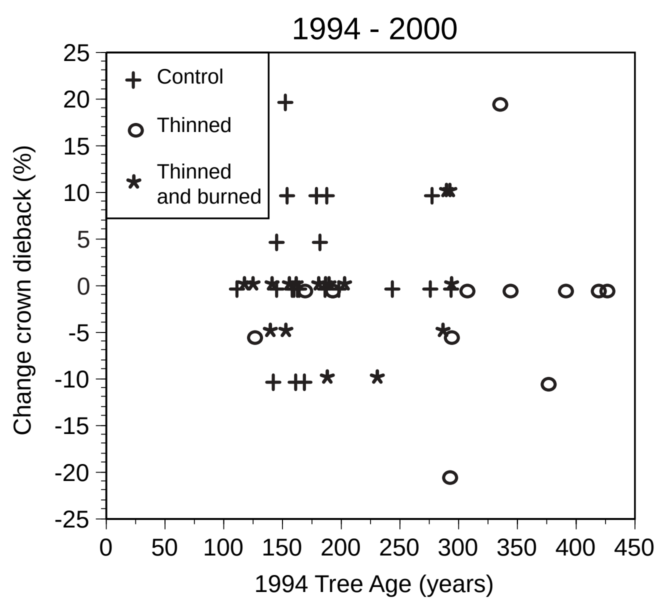

Figure 5—Change in crown dieback between 1994 and 2000 of presettlement ponderosa pines at the GPNA versus 1994 tree age in three treatments: control, thinned, and thinned + burned. Correlation coefficient for data pooled over all treatments is R = 0.103 (P = 0.473), control data R = 0.085 (P = 0.715), thinned data R = 0.091 (P = 0.769), and thinned + burned data R = 0.321 (P = 0.209).

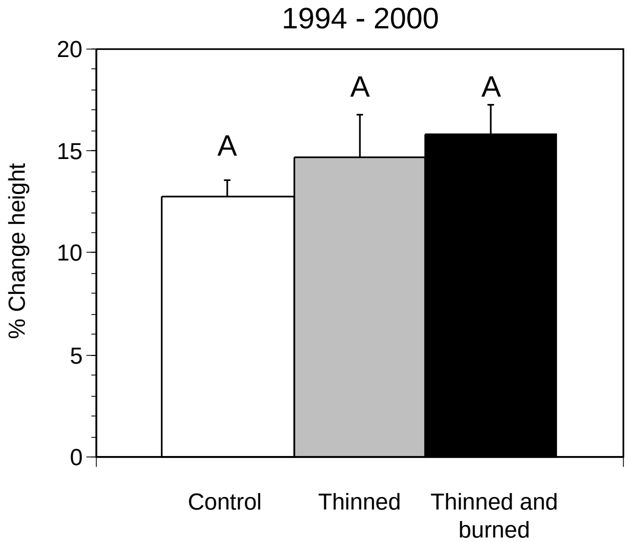

Figure 6—Average percent change in tree height between 1994 and 2000 of presettlement ponderosa pines at the GPNA in three treatments: control, thinned, and thinned + burned. The bars show one standard error of the mean. Means with the same letter do not differ significantly in Fisher's Protected LSD tests ($P \leq 0.01$).

died. Two of the trees that died in the thinned + burned treatment broke or toppled in a severe windstorm following a heavy, wet snow in 1997.

Discussion

Our results on changes in crown dieback of presettlement trees over 6 years since initiation of the restoration treatments are consistent with differences in tree radial growth rate and rates of resource uptake reported by Feeney and others (1998) and Stone and others (1999) for the first 3 years after treatment initiation. In these studies, thinning increased uptake of nitrogen and water, stimulated radial growth, and ameliorated effects of the 1996 drought on tree growth. Further, most growth and physiological characteristics were similar in the thinned and thinned + burned treatments in the first 3 years after the initiation of treatments (Feeney and others 1998). The increase in crown dieback that occurred between 1994 and 2000 in the control, coupled with the decrease in dieback in the thinned treatments, suggest that improved resource uptake in the thinned treatments has influenced carbon allocation to canopy growth and maintenance processes. We note, however, that these changes have been small in magnitude, which is not surprising considering the slow growth rate of old ponderosa pines. Moreover, the increase in canopy dieback in the control between 1994 and 2000 indicates a continued slow decline in the condition of presettlement trees when competition from postsettlement trees is severe.

Interestingly, both canopy dieback in 2000 and changes in canopy dieback between 1994 and 2000 were not related to tree age for ages between 100 and 450 years. This result suggests that mortality of presettlement-aged ponderosa pines at the GPNA is more strongly influenced by tree genotype or local variation in environment and disturbance rather than tree age alone. We lack a clear understanding of these factors, but we speculate that lightning strikes, other disturbances, genetic variation in response to stress, and perhaps local variation in rooting depth may be important factors that contribute to canopy dieback at the GPNA.

Our results on tree mortality should be viewed with caution because of the small number of trees assessed (146 over all treatments) and the small number of trees that died (four over all treatments). With this caveat, we believe that the occurrence of tree death by windthrow or wind breakage exclusively in the thinned treatments is not a coincidence. Our results suggest that windthrow and wind breakage are more common for presettlement trees growing in open stands that result from heavy thinning of postsettlement trees. Given that severe winter weather with high winds is common in ponderosa pines forests in northern Arizona, tree death because of wind breakage and windthrow may have been a common occurrence in open, savannalike forests dominated by old-growth trees prior to Euro-American settlement.

Wind damage was not an obvious causal factor in the death of two of the four presettlement trees that died in the thinned treatments between 1997 and 2000. One explanation is that thinning contributed to their death. However, we note that both of these trees had large amounts of dieback when the treatments were initiated. The tree that died in the thinned + burned plot was classified as "alive but declining" with a different crown classification system (Fulé and others 1997) in 1992, prior to initiation of the treatments. The tree that died in the thinned plot had 40 percent dieback in the 1994 assessment of crown condition. Instead, we speculate that these trees were already declining because of unknown factors at the time of treatment initiation, and were subsequently severely stressed by the regional 1996 drought in the Southwest. Further, bark beetle pitch tubes were evident in 2000 on both trees, suggesting a secondary role of herbivory in mortality. This interpretation is consistent with Manion's (1991) model of tree decline where tree death is often the result of predisposing and contributing factors that occurred many years before mortality and acted to reduce tree carbohydrate levels. Following this sequence of events, tree defense against biotic agents and capacity to recover from severe abiotic stress are diminished, and thus mortality is imminent.

Conclusions

In summary, our results indicate that heavy thinning of postsettlement trees improved the crown condition of presettlement trees at the GPNA over 6 years since treatment, but also may have increased windthrow and wind breakage. The effect of heavy thinning of postsettlement trees on wind damage to presettlement trees should be addressed in other ecological restoration experiments in ponderosa pine forests.

Acknowledgments

We sincerely thank K. Griffis, T. Heinlein, J. Roccaforte, B. Steed, J. Stone, and G. Verkamp for their help with field work. This work was partly supported by grants from the National Science Foundation (DEB-9322706), the NAU Mission Research Program, and the McIntire-Stennis Program.

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Seeding Versus Natural Regeneration: A Comparison of Vegetation Change Following Thinning and Burning in Ponderosa Pine

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Peter Z. Fulé
Margaret M. Moore
W. Wallace Covington

Abstract—The decision whether to seed with native species following restoration treatments should be based on existing vegetation, species present in or absent from the soil seed bank, past management history, microclimate conditions and soils. We installed three permanent monitoring plots in two areas (total 18.6 ha) at Mt. Trumbull, AZ. Trees were thinned and the sites burned in 1996 and 1997. A 5 ha area was seeded with native shrub, grass and forb species; the remaining 13.6 ha were unseeded. Pretreatment species richness ranged from none to five species per plot. We recorded 13 graminoid and eight shrub species in the seeded area, and four graminoid and four shrub species in the unseeded area. The greatest increase in species richness in both seeded and unseeded plots occurred approximately 1.8 years posttreatment. Perennial native species dominated plant cover by 2.8 years, although annual native forbs dominate the soil seed bank. Perennial grasses are nearly absent from the seed bank. The seeded area had the highest diversity, but it also had twice as many nonnative species (14 versus 7 in the unseeded plots). By August 1999, maximum species richness reached 51 species on the seeded plot. Of these species, 80 percent were native. Although seeding increases diversity, it may also have the long-term tradeoff of introducing new genotypes and species, both native and nonnative.

Introduction

Human-induced changes that have impacted herbaceous forbs, grass and shrub species over approximately the last 150 years in Southwestern ponderosa pine forests have been attributed primarily to domestic livestock overgrazing, active fire suppression, increased ponderosa pine density, and climate changes (Arnold 1950; Weaver 1951; Cooper 1960; Covington and Moore 1994; Touchan and others 1995; Covington and others 1997). Attempts to restore the herbaceous and shrub species in Southwestern ponderosa pine forests are hampered by both ecological and social considerations. The scale of ecological restoration projects in the Southwest may be on the order of thousands of hectares. In addition to tree thinning and prescribed burning, restoration of these areas sometimes requires seeding large amounts of native seed. Seeding with native species poses many challenges: seed may be prohibitively expensive, unavailable in the quantities necessary, or collected from an area that is geographically or climatologically dissimilar to the area undergoing restoration. Even if these seeds are available and applied to a site, seed or seedling mortality may occur from competition (by both native and nonnative species), disease, herbivory, lack of mycorrhizal inoculants, inclement weather, or unfulfilled germination requirements or cues. Written historic records and photographs may be unavailable, further hampering efforts to define past plant communities and conditions. Other considerations are a lack of communication between and within government agencies and other organizations, past management practices (such as logging and road building), and a lack of understanding or interest by the general public. Some of these factors may be corrected or improved in order to establish vegetation on restored sites. Others cannot be overcome due to limited resources or permanent landscape changes and must be considered in order to achieve management goals for restoration.

Recent historic herbaceous and shrub species composition and abundance are largely unknown for Southwestern ponderosa pine forests, and thus restoration targets must be determined without the benefit of this information. Knowledge of historic herbaceous plant communities is limited due to a reduced capacity for preservation of nonwoody vegetation in the soil, and limited historical records. A variety of methods, techniques, and tools have been used to determine historical species composition and abundance. These methods and tools include historical records and photographs, packrat middens, palynology, relict sites (Kaufmann and others 1994), diaries, surveys, military expeditions (Dick-Peddie 1993) and opal phytoliths (Fisher and others 1995; Kerns 1999).

The main objective of this study was to compare a seeded and an unseeded area to collect material for further data exploration and generation of hypotheses. This information can then be used to make seeding decisions in areas of ponderosa pine forest undergoing ecological restoration.

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Study Area

The study area lies within the Grand Canyon–Parashant National Monument, in northwestern Arizona, between the Mt. Logan and Mt. Trumbull Wilderness areas and approximately 95 km southwest of Fredonia, AZ. The study area is of primarily the ponderosa pine forest type with intermingled patches of New Mexico locust (*Robinia neomexicana*), Gambel oak (*Quercus gambelii*), and big sagebrush (*Artemisia tridentata*). Precipitation occurs annually in a bimodal distribution pattern and annual precipitation varies between 38–64 cm (USDI BLM 1990). Fire scars collected from individual ponderosa pine trees indicate that fires occurred every 4 to 6 years prior to Euro-American settlement (Fulé 1997 and unpublished data).

The Southwestern Ecological Restoration Project is a cooperative study among the Bureau of Land Management, Northern Arizona University, and the Arizona Game and Fish Department. The area is managed by the BLM and encompasses approximately 18,500 ha. Specifically, the data for this study were collected in two units, one 5.3 ha in size, which was seeded with a mixture of native grasses, forbs, and shrubs. The other unit (13.4 ha) was not artificially seeded and relied on natural regeneration by propagules already onsite or dispersed onto the site by wind, water, animals, or some other mechanism. The objective of the seeding was to increase species richness and diversity and to decrease soil erosion following ecological restoration treatments of tree thinning and prescribed burning.

Methods and Materials

Modified National Park Service fire monitoring protocols (NPS 1992) were used for the plot installation and long-term vegetation monitoring across the entire 18,500 ha project area. The point line-intercept method was used for vegetation monitoring. We installed two 50 m transects per plot, and inventoried these lines every 30 cm for a total of 332 points per plot. Species that were not intercepted by the transects were recorded within a 5 m belt on either side of the transects (hereafter referred to as "belt transects"). Three of these monitoring plots fell within the immediate study area: one in the seeded unit and two in the unseeded unit. Although many more plots were established across the landscape over several years, only data from these three plots will be discussed in this paper due to the longer period over which these plots have been monitored and their use as "demonstration" plots.

We conducted pretreatment inventories in the fall of 1995 in the unit to be seeded and in the spring of 1996 in the unseeded unit. The area was thinned of postsettlement trees in the summer of 1996 and burned in the fall of 1996. The 5.3 ha unit was broadcast seeded in December of 1996 and reseeded the following year using a chain pulled behind an all terrain vehicle (ATV) to cover seed with soil. Both units were fenced to exclude cattle, but deer were able to access the area. The plots were reinventoried in August 1997, 1998 and 1999. Species that were seeded are listed in table 1. The cost of seeding was estimated to be approximately $214 per acre (1996 dollars), with additional money spent on reseeding in 1998 ($26 per acre). Seed was certified "weed free," but germination testing was not conducted by the authors.

Table 1—List of species seeded at Mt. Trumbull in 5 ha area.

Grasses	*Purshia tridentate*
Achnatherum hymenoides	*Rhus trilobata*
Bouteloua gracilis	*Ribes cereum*
Bromus marginatus	*Sambucus cerulea*
Elymus elymoides	*Symphoricarpos oreophilus*
Elymus trachycaulus	**Forbs**
Koeleria macrantha	*Eriogonum umbellatum*[a]
Pascopyrum smithii	*Linum lewisii*[a]
Schizachyrium scoparius	*Penstemon barbatus*
Shrubs	*Penstemon palmeri*
Mahonia repens	

[a]Species seeded in 1997, but not in 1996.

Nomenclature in this paper was adopted from the USDA Plants Database (2000), Utah Flora (Welsh and others 1993), and Intermountain Flora (Cronquist and others 1972, 1977, 1984, 1989, 1994 and 1997).

Results

Percent Ground Cover

Percentages of substrates and plant cover are shown in figure 1. Prior to ecological restoration treatments, 86 percent of the ground cover in the seeded unit consisted of litter (primarily pine needles), and plant cover was less than 1 percent. In the unseeded unit, litter averaged 47 percent prior to treatment and fell to 23 percent the year following ecological restoration treatments. By 1999 (the third year following treatments), litter had increased to 44 percent in this unit, while in the seeded unit, litter increased to 24 percent. Plant cover in the unseeded unit rose from 6 percent (pretreatment) to a high of 50 percent in 1998 due primarily to the occurrence of early successional annuals and short-lived perennials, such as lambsquarters (*Chenopodium* spp.), spreading groundsmoke (*Gayophytum diffusum*) and silver lupine (*Lupinus argenteus*). Plant cover leveled off in 1999 to 25 percent in this unit. However, in the seeded unit, plant cover continued to increase in 1999 to 44 percent. Soil cover dramatically increased in the seeded unit following treatments: 5 percent pretreatment and 58 percent in 1997. It has continued to decrease in both units as other types of cover begin to predominate.

Species Composition

The only two plant families represented in the seeded unit in 1997, the first year following restoration treatments, were the Chenopodiaceae (goosefoot family) and the Poaceae (grass family) (table 2). In 1998, 11 families were represented. The Chenopodiaceae family was most common (23 percent of the ground cover). There were 15 families in 1999. The most common families were the Scrophulariaceae, Asteraceae, and Chenopodiaceae.

The most numerous species in the seeded unit in 1997 were *Chenopodium leptophyllum* (narrowleaf goosefoot), *Agropyron* sp. (wheatgrass), *Bouteloua gracilis* (blue grama), and *Elymus elymoides* (squirreltail). By 1999, perennial

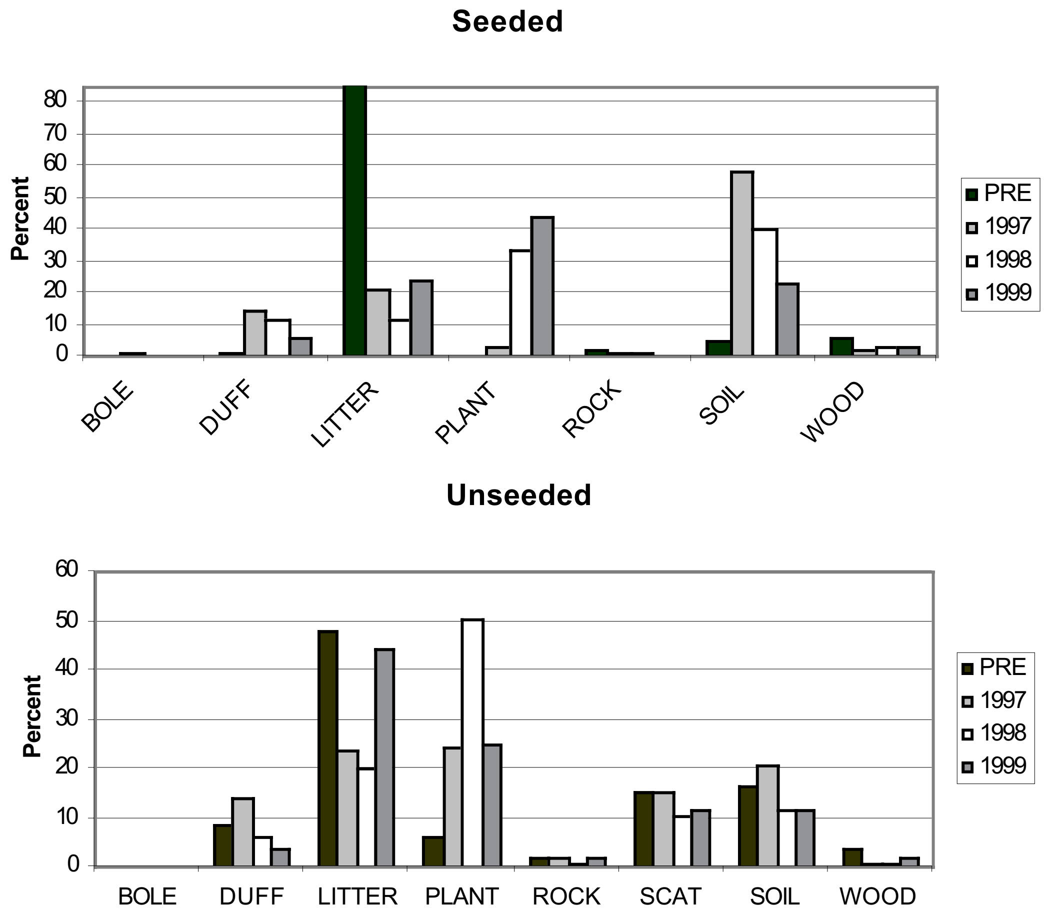

Figure 1—Percent ground cover of plants and substrates in the seeded and unseeded units at Mt. Trumbull.

grasses and forbs were beginning to dominate this site. The five most numerous species were *Agropyron (Pascopyrum) smithii* (western wheatgrass), *Bromus carinatus* (mountain brome), *Conyza canadensis* (Canadian horseweed), *Thinopyrum intermedium* (intermediate wheatgrass) and *Chenopodium album* (lambsquarters).

In the unseeded unit, pretreatment family representation was limited to Poaceae, Asteraceae, Fabaceae, and Scrophulariaceae (table 2). In 1997, there were eight families represented in this unit. The most common families were the Fabaceae, Asteraceae, Polygonaceae, Poaceae, and Scrophulariaceae. The number of families was highest in 1998 (11). The number of families decreased in 1999 to eight as some of the early-successional species began to disappear from the aboveground vegetation. Families included Poaceae, Asteraceae, Fabaceae, Onagraceae, Scrophulariaceae, Polygonaceae, Fagaceae and Chenopodiaceae.

The most numerous species in the unseeded unit in 1997 were *Artemisia tridentata* (big sagebrush), *Lupinus argenteus* (silver lupine), *Polygonum douglasii* (Johnston's knotweed), and *Elymus elymoides* (squirreltail). The most common species in 1999 were the same four with the addition of *Senecio multilobatus* (lobeleaf groundsel) and *Bromus tectorum* (cheatgrass).

In the seeded unit, the percent cover of annuals peaked in 1998, 2 years following restoration treatments, at approximately 29 percent of the ground cover. Perennials continued to increase and composed 31 percent of the ground cover in 1999. In the unseeded unit, both annuals and perennials peaked in 1998. Annuals composed 20 percent and perennials approximately 40 percent of the ground cover. By 1999, annuals had decreased to 7 percent and perennials to 20 percent of the ground cover. Biennials remained at less than 5 percent of the ground cover in all years and in both treatments.

Nonnative Species

The number of nonnative species was higher in the seeded unit. Eight species were intercepted by the transects: *Agropyron cristatum* (crested wheatgrass), *Bromus inermis* (smooth brome), *Bromus tectorum* (cheatgrass), *Lappula occidentalis* var. *occidentalis* (stickseed), *Marrubium vulgare* (horehound), *Salsola tragus* (Russian thistle), *Thinopyrum intermedium* (intermediate wheatgrass) and *Verbascum thapsus* (common mullein). In addition, *Bromus commutatus* (hairy chess), *Bromus japonicus* (Japanese brome), *Convolvulus arvensis* (field bindweed), *Lactuca serriola* (prickly

Table 2—Percent frequency of plant families over time in a seeded and an unseeded unit at Mt. Trumbull, AZ.

Unit and year	Plant families	Percent
Seeded unit		
Pretreatment	No species present	
1997	Chenopodiaceae	2.1
	Poaceae	1.5
1998	Chenopodiaceae	24
	Scrophulariaceae	3.0
	Onagraceae	2.7
	Polygonaceae	1.5
	Poaceae	1.5
	Fumiariaceae	0.9
	Boraginaceae	0.9
	Papaveraceae	0.6
	Lamiaceae	0.3
	Asteraceae	0.3
	Solanaceae	0.3
1999	Poaceae	25
	Scrophulariaceae	7.5
	Asteraceae	5.1
	Chenopodiaceae	4.8
	Boraginaceae	2.4
	Onagraceae	1.8
	Polygonaceae	1.5
	Lamiaceae	1.5
	Amaranthaceae	1.2
	Solanaceae	1.2
	Nyctaginaceae	0.6
	Verbenaceae	0.6
	Linaceae	0.3
	Fabaceae	0.3
	Brassicaceae	0.3
Unseeded unit		
Pretreatment	Poaceae	2.6
	Asteraceae	2.4
	Fabaceae	0.8
	Scrophulariaceae	0.5
1997	Fabaceae	10
	Asteraceae	5.1
	Poaceae	3.0
	Polygonaceae	2.3
	Scrophulariaceae	1.7
	Euphorbiaceae	0.6
	Chenopodiaceae	0.2
	Rhamnaceae	0.2
1998	Fabaceae	22
	Asteraceae	11
	Poaceae	6.9
	Onagraceae	6.9
	Scrophulariaceae	5.6
	Polygonaceae	4.8
	Chenopodiaceae	2.9
	Euphorbiaceae	1.2
	Fagaceae	1.1
	Polemoniaceae	0.5
	Brassicaceae	0.2
1999	Poaceae	9.9
	Asteraceae	8.7
	Fabaceae	5.0
	Onagraceae	1.7
	Polygonaceae	1.2
	Scrophulariaceae	1.1
	Fagaceae	0.5
	Chenopodiaceae	0.3

lettuce), *Poa pratensis* (Kentucky bluegrass) and *Tragopogon dubius* (salsify) were encountered within the 20 m wide belt transect. Percentages of native and nonnative species are given in table 3. There was an increasing trend of nonnatives in the seeded unit, from less than 1 percent of the vegetative cover in 1997 to approximately 24 percent of the vegetative cover in 1999. In the unseeded unit, nonnatives showed an increasing trend as well, from 5 percent in 1996 to 13 percent in 1999. Cheatgrass and field bindweed, both inventoried in this study, are listed in Arizona as noxious weeds

Only three nonnative species were captured by the transects in the unseeded unit: *B. tectorum* (cheatgrass), *L. serriola* (prickly lettuce) and *V. thapsus* (common mullein). Four additional species were inventoried in the belt transects: *Polygonum aviculare* (prostrate knotweed), *Polygonum convolvulus* (black bindweed), *P. pratensis* (Kentucky bluegrass), and *T. dubius* (salsify).

Reasons for the larger number of nonnatives in the seeded unit could be due to: (1) their presence in the soil seed bank; (2) as contaminants in the seed mix; (3) increased foot traffic in this unit because of its use as a demonstration site, however, the unseeded unit is also used for that purpose; or (4) through colonization from offsite by dispersal mechanisms. The seeded unit is in close proximity to a meadow.

Grasses

Ten species of grasses were inventoried in the seeded unit. These species were *Achnatherum hymenoides* (Indian ricegrass), *A. cristatum, Agropyron (Pascopyrum) smithii* (western wheatgrass), *Bouteloua gracilis* (blue grama), *Bromus carinatus* (mountain brome), *B. inermis, B. tectorum, Elymus elymoides* (squirreltail), *Koeleria cristatum* (junegrass), and *T. intermedium*. In addition, the belt transect contained *B. commutatus, B. japonicus,* and *P. pratensis*. *Schizachyrium scoparium* (little bluestem) was seeded, but was not recorded in the monitoring plots prior to 2000.

Only four species of grasses were present in the plots in the unseeded unit and thus most likely regenerated from the soil seed bank or were carried in from the area surrounding the site. These species were *Bouteloua gracilis, Bromus tectorum, Elymus elymoides*, and *Poa fendleriana*. *Poa pratensis* was also captured by the belt transect.

Table 3—Percentages of native and nonnative species in the seeded and unseeded units at Mt. Trumbull.

Plot	Year	Native	Nonnative	Unknown
		percent		
Seeded	1995	—	—	—
	1997	75	—	25
	1998	92	6	2
	1999	74	24	2
Unseeded	1996	86	15	—
	1997	95	5	—
	1998	92	9	—
	1999	87	13	—

Shrubs

Eight species of shrubs were present in the seeded unit. These species were *Amelanchier utahensis* (Utah serviceberry), *Arctostaphylos pungens* (pointleaf manzanita), *Garrya flavescens* (ashy silktassel), *Purshia tridentata* (antelope bitterbrush), *Rhus trilobata* (skunkbush sumac), *Ribes cereum* (wax currant), and *Symphoricarpos oreophilus* (whortleleaf snowberry). Bitterbrush, sumac, currant, and snowberry were seeded, so it is difficult to ascertain whether the species monitored on the plots came from natural regeneration or from seeding. However, seeding with shrub species appeared to be effective with the exception of *Mahonia repens* (Oregon-grape). No specimens of this taxa were found in the seeded unit. *Sambucus cerulea* (blue elderberry) was not captured by the monitoring plots, although it was observed nearby.

In the unseeded unit, four species were inventoried on the plots: *A. pungens, Artemisia tridentata, A. utahensis,* and *R. cereum*. Species that were observed in the unit, but not captured by the monitoring plots, included *G. flavescens, A. pungens* and *Ceanothus fendleri* (buckbrush). These species presumably sprouted after fire. With the exception of big sagebrush (*A. tridentata*), shrubs were not detected in the soil seed bank (or propagule bank) from samples taken at Mt. Trumbull (Springer 1999). However, some species apparently are still present in the form of underground rhizomes or buds and will regenerate following fire. Nearly all shrub species in these units were heavily browsed, most likely by deer or rabbits, since elk are not present at Mt. Trumbull and cattle were excluded from the study area (personal observations).

Legumes

No species of legumes (Fabaceae or pea family) were manually seeded in either unit and no species of legumes were captured by the line transects in the seeded unit. The only occurrence of legumes in this unit was one record of *Lotus utahensis* (Utah birdsfoot trefoil) from the belt transects in 1998.

In the unseeded unit, *Lupinus argenteus* (silver lupine) was recorded on the line transects prior to treatment. Silver lupine was recorded in all years in both plots in this unit. Its frequency was highest in 1998, the year following thinning and burning (an average of 16 percent of the ground cover). Other legumes recorded in the unseeded unit were *Lotus spp*. (*Lotus plebius, Lotus wrightii, Lotus* sp.) in 1997, 1998, and 1999 and *Lupinus kingii* (King's lupine) in 1998 on a belt transect.

Simpson's Index and Species Richness

Pretreatment species richness was very low (less than five species) on the line transects on all plots (fig. 2). In 1998, it peaked in the unseeded unit (17 species), but continued to increase in the seeded unit in 1999 (28 species). The total number of species in the seeded unit, including species captured on the belt transects, was 51. The average number of species per plot in the unseeded unit was 36. The Simpson's diversity index showed a similar trend to species richness in the seeded unit (fig. 3), increasing from 0 in the pretreatment inventory to 14 in 1999. In the unseeded unit, it peaked at 10 in one plot in 1998, but continued to increase slightly in the other plot through 1999 (fig. 3).

Discussion

Seeding is a complex issue, and the positive and negative aspects must be weighed before making any type of management decision relating to ecological restoration in Southwestern ponderosa pine forests. Seeding has many advantages including decreased soil erosion (Beyers and others 1998) and increased vegetative ground cover (Williamson 1984; Tyser and others 1998). It also produces an increase in the amount of available seed stock for colonizing nearby areas (Jacobson and others 1994) and, if successful, it will increase species richness and diversity (Grant and others 1997). However, seeding also has disadvantages, such as high cost and lack of locally-adapted genetic material for most areas (Dunne 1999; Roundy 1999). Also, increased species diversity can be a major disadvantage if this increase is due to nonnative species. This study, though exploratory, seems to indicate that although seeding increased the species richness and diversity, these increases were due in part to increases in nonnative species that were possibly brought onsite by the seed mixture. The percentage of nonnatives in the seeded unit continued to increase in 1999 and will be monitored to determine if this trend continues.

There is presently a dearth of native perennial grasses, legumes, and shrubs in the ponderosa pine forests of the Mt. Trumbull area, based on this study, on previous soil seed bank studies (Springer 1999), and on data from other areas of ponderosa pine forest in northern Arizona. Rasmussen (1941), in one of the few historical publications from the Arizona Strip containing information on species frequency, mentions that by the time of his publication, certain legumes had entirely disappeared from the Kaibab Plateau (about 45 km to the east) due in part to severe overgrazing by wild

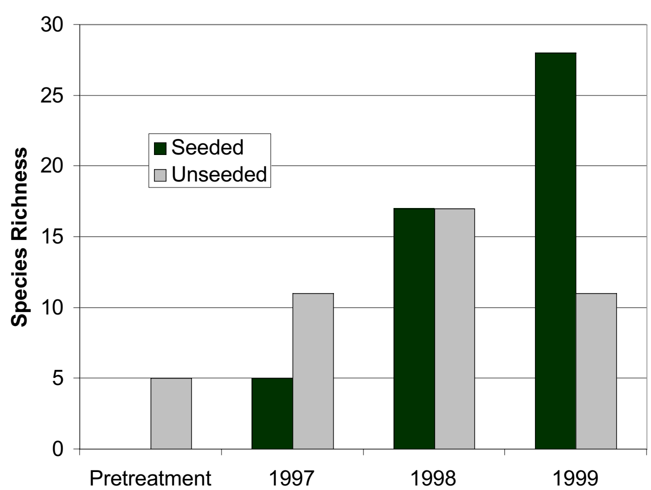

Figure 2—Species richness in the seeded and unseeded units at Mt. Trumbull following ecological restoration treatments.

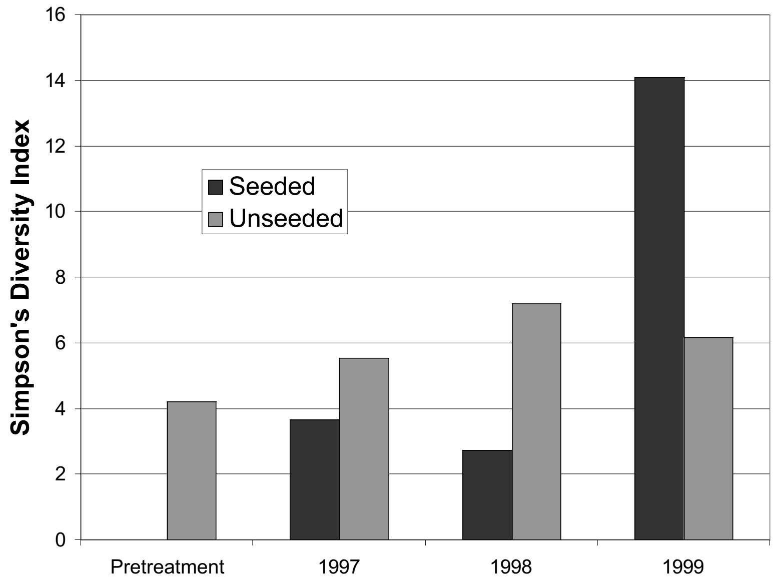

Figure 3—Simpson's diversity index (SI) in the seeded and unseeded units at Mt. Trumbull following ecological restoration treatments.

deer populations and/or domestic livestock in the late 1800s and early 1900s. Native shrub species also were decimated from much of the Plateau, presumably by fluctuations in the deer population. The Mt. Trumbull area has populations of deer but no elk, and it has a history of grazing by domestic cattle (Altschul and Fairley 1989). Seeding can return extirpated species to a site, although information is necessary to ascertain if these species were historically present. If species have been decimated from a site, then the genetic material from species adapted to that site is likely no longer available.

Nonnative species, particularly cheatgrass and common mullein, increased following restoration treatments in the study area at Mt. Trumbull. Common mullein is considered to be an early-successional species and is expected eventually to be replaced by native species in the treated areas, but cheatgrass has been known to change the species composition and fire cycle of areas where it has become dominant (Sheley and Petroff 1999). If the goal of management efforts is to decrease the amount of cheatgrass in the restored areas, then these areas should be heavily seeded with early successional native grass species able to compete with the cheatgrass, such as squirreltail (Jones 1998). Early successional species made up a large percentage of species in the soil seed bank in the study area (Springer 1999). Their decline after the first few years was expected because many of these species are annuals or biennials, rely on disturbance to maintain their populations, and use the soil seed bank to store genotypes for future environmental variability (Springer 1999). However, the low frequency of perennial species in the unseeded unit is of some concern if this pattern is indicative of the entire study area.

Species richness and diversity began to drop off in the unseeded unit in the third year following restoration treatments, and few new species were recorded. This trend could be attributable to weather or to a slowdown in the rate of successional response after treatment. Both units should continue to be monitored to observe trends. As additional units across the entire project area are treated with overstory tree thinning and prescribed burning, more data will be available to assist with management decisions.

This study points to the need to conduct seeding trials to determine which seeds are germinating, and how long it takes for these species to become established. Seeding trials can also be used to pinpoint the origin of nonnative species appearing onsite, whether through dispersal mechanisms or in seed mixtures. There is also a need to conduct studies to determine the effects of prescribed burning on leguminous species in Southwestern ponderosa pine forests, as well as the inputs of nitrogen by these species, for their role in nitrogen cycling is just beginning to be quantified (Hendricks and others 1999).

When considering whether and with what species a site should be seeded, each area needs to be treated individually and blanket treatments should not be applied to huge areas of the landscape. Factors to take into account are existing vegetation, species in the soil seed bank, past management history, microclimate conditions and soils. Soil seed bank samples taken prior to ecological restoration treatments can give an idea of the species that will colonize a site following these treatments (Springer 1999). As the demand for native seed increases across the West, we hope that supply will grow to meet demand and eventually costs will decline; however, there is a tradeoff of cost versus maintaining seed mixtures that are weed-free. Further effort should be made to collect seeds in areas undergoing restoration to maintain the genetic material adapted to that area.

Acknowledgments

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Effect of Restoration Thinning on Mycorrhizal Fungal Propagules in a Northern Arizona Ponderosa Pine Forest: Preliminary Results

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Abstract—The inoculum potential for arbuscular mycorrhizal (AM) and ectomycorrhizal (EM) fungi were investigated in thinned and uncut control stands in a northern Arizona ponderosa pine forest. A corn bioassay was used to determine the relative amount of infective propagules of AM fungi, and a ponderosa pine (*Pinus ponderosa*) bioassay was used to determine the relative amount of infective propagules of EM fungi. Three stands of each treatment were sampled by collecting soil cores along 10 randomly chosen transects within each stand. The relative amount of infective propagules of AM fungi was significantly higher in samples collected from the thinned stands than controls. Conversely, there was a slight decrease in the relative amount of infective propagules of EM fungi in samples collected from thinned stands in comparison to the controls; however, this difference was not significant. These preliminary results indicate that population densities of AM fungi can rapidly increase following restoration thinning in northern Arizona ponderosa pine forests. This may have important implications for restoring the herbaceous understory of these forests because most understory plants depend upon AM associations for normal growth.

Introduction

Low intensity fires carried by grassy understories were prevalent every 2–20 years in southwestern ponderosa pine (*Pinus ponderosa*) ecosystems prior to Euro-American settlement and played a major role in determining the structure, composition, and stability of these ecosystems (Cooper 1960). Current alterations to the structure and function of southwestern ponderosa pine forests are the result of heavy grazing, intensive logging of old-growth trees, and fire suppression by Euro-American settlement around the turn of the 20th century (Covington and others 1997). This has resulted in a large number of small trees with closed canopies and little herbaceous understory production. The current reduction in herbaceous understory diversity and production in ponderosa pine forests has contributed to the alteration of the natural fire regime, loss of habitat for numerous animal species, and overall reduction of species diversity. As a result, a major objective of restoring Southwestern ponderosa pine forests is to increase herbaceous understory diversity and production by reestablishing community structure and function within a range of natural variability.

Mycorrhizae are a major component of soil ecosystems, playing an important role in plant nutrition, nutrient cycling, food webs, and the development of soil structure (Johnson and others 1999). A mycorrhiza is generally a "mutualistic relation between plant and fungus localized in a root or root-like structure in which energy moves primarily from plant to fungus and inorganic resources move from fungus to plant" (Allen 1991). Forest tree species, particularly those within Pinaceae, Fagaceae, Betulaceae, and Dipterocarpaceae form ectomycorrhizal (EM) relationships with basidiomycetes and ascomycetes (Harley and Smith 1983). Over 80 percent of all plants form arbuscular mycorrhizal (AM) relationships with Glomalean fungi, a single order of Zygomycetes. A few exceptional plant families (Cyperaceae, Juncaceae, Caryophyllaceae, Chenopodiaceae, and Brassicaceae) typically do not form any mycorrhizal relationship (nonmycotrophic) (Pendleton and Smith 1983). Numerous researchers have suggested a relationship between the recovery time of disturbed ecosystems and the abundance of infective propagules of fungi (Allen and Allen 1980; Bentwenga and Hetrick 1991; Noyd and others 1995; Reeves and others 1979). Therefore, quantifying the effect of restoration thinning on densities of mycorrhizal fungal propagules may provide insight to the recovery rate of herbaceous understory communities to restoration treatments in ponderosa pine forests.

This is the first known study looking at AM fungal propagule densities in Southwestern ponderosa pine forests. We hypothesized that AM fungal propagule densities would increase and EM fungal propagule densities would decrease in response to restoration thinning in ponderosa pine forests, and that these changes would be correlated to host plant abundance. The specific objectives of this study were to: (1) quantify the effect of restoration thinning on AM and EM fungal propagule densities; and (2) assess the

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relationships between mycorrhizal fungal propagule densities and soil and plant community properties.

Methods

Experimental Design

Three blocks of four restoration treatments were established during the summer of 1998 within approximately 1,700 acres of the Fort Valley Experimental Forest and adjacent areas near Flagstaff, AZ (fig. 1). Treatment units within each block were randomly assigned one of four treatments: (1) no thinning or burning (control); (2) thinning to a low level of replacement trees and burning; (3) thinning to an intermediate level of replacement trees and burning; (4) thinning to a high level of replacement trees and burning. Specific details of thinning treatment guidelines are outlined in Covington and others 1998. Soils for this study were collected before any of the treatment units had been burned, and therefore these results only reflect the effect of restoration thinning on mycorrhizal fungal propagules. Continuation of this study will investigate the effects of full restoration treatments (thinning and prescribed burning) after all treatment units have been burned.

Field Sampling

Soil samples for “bait-plant bioassays” and soil and vegetation analyses were taken in mid-May, 6 months after thinning, along 10 randomly placed transects within each of the three blocks in the controls (1-1, 2-1, 3-1) and low level of replacement trees thinning treatment units (1-2, 2-2, 3-2) (fig. 1). Soil samples were taken to a depth of 15 cm using a hand trowel. Samples were taken to this depth because AM fungal propagule densities are generally highest in the surface 15 cm (Johnson and others 1991). Two samples from each transect were immediately placed into 4 cm diameter x 20 cm diameter deep Conetainers (Stuewe and Sons, Inc., Corvallis, OR) for “bait-plant bioassays.” The other sample for soil analysis was placed in a ziploc bag and stored in a freezer until analysis. A 0.5 x 2 m (1 m^2) plot was located adjacent to each soil sample, and the percent cover of each herbaceous and woody species present along with substrates (litter, soil, and rock) were estimated using cardboard cutouts as visual guides. The soil seed bank, soil disturbance, bulk density, fuel loads, fire behavior, herbaceous biomass production and abundance, overstory tree structure, and understory tree regeneration were also assessed along these 50 m transects.

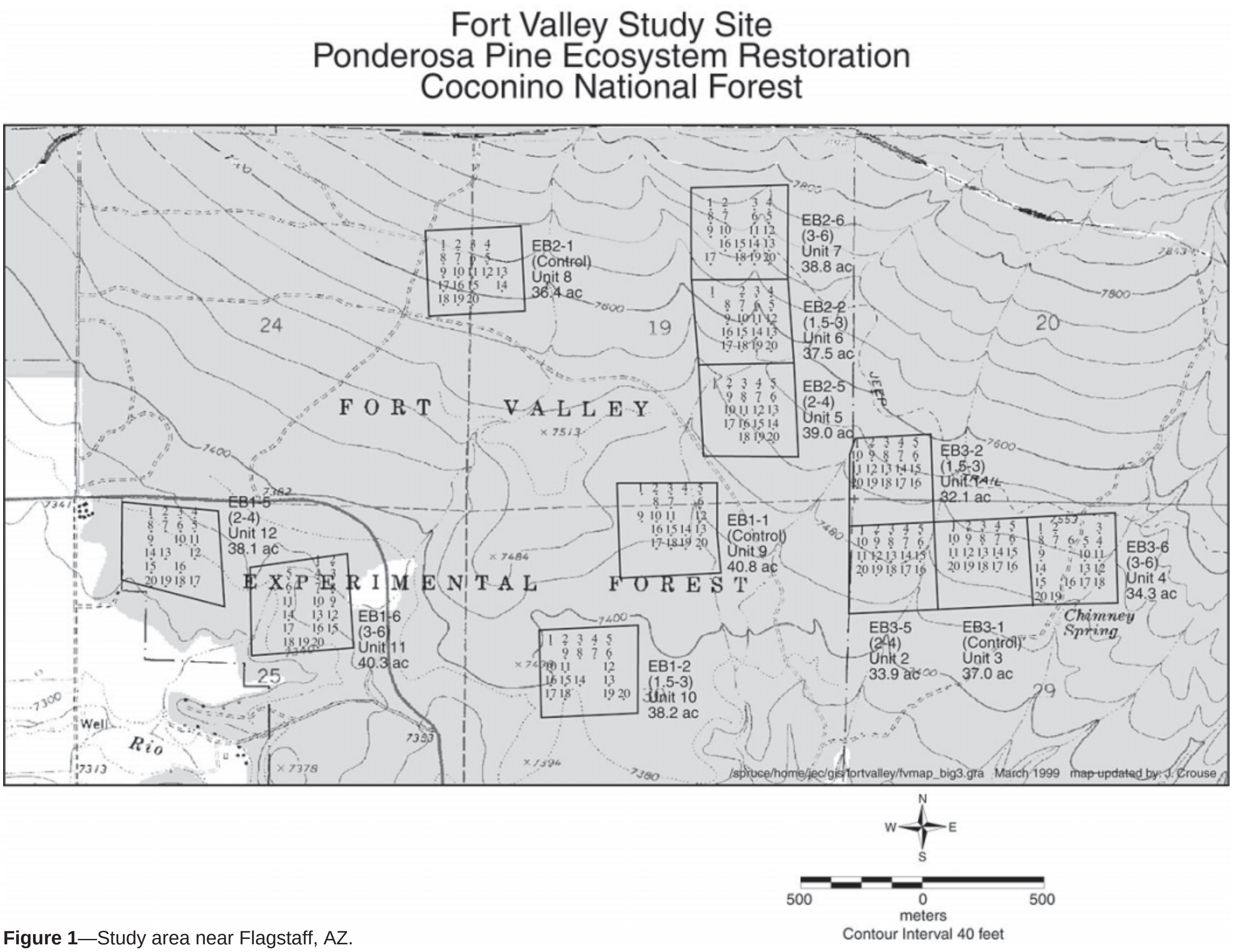

Figure 1—Study area near Flagstaff, AZ.

Lab Analysis

A corn (*Zea mays*) bioassay was used to determine the relative amount of infective propagules of AM fungi, and a ponderosa pine bioassay was used to determine the relative amount of infective propagules of EM fungi. Corn is a strongly mycotrophic plant and grows rapidly and uniformly, and its advantages outweigh the disadvantage of not using native host plants (Johnson and others 1999). "Bait-plant bioassays" are designed to detect all types of viable mycorrhizal fungal propagules including spores, fragments of mycorrhizal roots and extraradical hyphae and therefore may more accurately quantify total mycorrhizal fungi than direct counts of sporocarps, spores, or colonized root lengths (Brundrett and others 1994; Johnson and others 1999). Plants were placed in a greenhouse and watered every 3 days until corn plants were harvested at 6 weeks and pine plants were harvested at 12 weeks. Roots were carefully washed free from the soil and were measured for length and weighed. Following mycorrhizal analysis, roots were dried in an oven at 70 °C for 24 hours and then were weighed again. Shoot lengths and dry weights were also measured. Corn roots were prepared for fungal propagule density measurements by taking a random subsample of roots of a known mass, clearing roots in 5 percent KOH, and then staining roots with trypan blue (Johnson and others 1999). The gridline intersect method using a dissecting microscope was used to measure the proportion of root length containing AM fungal structures: arbuscules, vesicles, coils, internal mycorrhizal hyphae and external mycorrhizal hyphae (Givanetti and Mosse 1980). Pine roots were measured for fungal propagule density through direct examination using a dissecting microscope to quantify the proportion of root tips colonized with EM fungi (Gehring and Whitham 1991). Root tips were also classified as dead or living.

Soil Analysis

Soil samples from each transect were analyzed for pH, total N, total P and organic C at the Bilby Research Soil Analysis Laboratory, Flagstaff, AZ. Soil pH was determined in a 1:1 slurry by pH meter. The general method used for N and P was a Kjeldahl digestion of the soil material followed with the analysis of N and P by automated colorimetry using a Technicon auto-analyzer (Parkinson and others 1975). Organic matter was determined by loss on ignition. Samples were heated in crucibles in a muffle furnace at 450 °C and net weight loss was estimated as organic matter.

Statistical Analysis

Analysis of variance for a randomized complete block design was conducted using SAS JMPIN version 3.2.1 (SAS Institute 1997) to determine the effect of thinning on mycorrhizal propagule densities. Root infection data were log $(x + 1)$ transformed prior to analysis, and bait plant length and weight, herbaceous plant abundance, and soil properties were ln transformed. Simple correlation analyses were conducted to assess the relationship between infectivity, soil and plant community properties, and bait plant characteristics (length and weight) using SAS JMPIN. Significance of correlations and analysis of variances were accepted at alpha ≤ 0.05.

Results

Mycorrhizal fungi colonized all "bait-plant" roots grown in soil from the thinned and control units. Bioassay results indicated that infective propagule densities of AM fungi were significantly higher in the samples collected from the thinned units than controls ($F = 5.4437$, $P < 0.005$) (fig. 2). Also, corn grown in thinned soils had relatively more vesicles and less hyphae than corn grown in control soils (fig. 3). Conversely, there was a slight decrease in the relative amount of infective propagules of EM fungi in samples collected from thinned stands in comparison to the controls ($F = 2.1601$ $P > 0.05$); however, this difference was not significant (fig. 2). Sparsely branched bifurcated tips were the dominant EM morphotype with tuberoid mycorrhiza also dominant.

AM fungal infectivity was not correlated with corn shoot weight, corn root weight or percent herbaceous cover ($r = –0.037$, $r = 0.031$, and $r = 0.035$ respectively, $N = 60$, $P \geq 0.05$). EM fungal infectivity was not correlated with pine shoot weight ($r = 0.11$) but was positively correlated with pine root weight ($r = 0.28$, $N = 60$, $P \geq 0.05$). The only soil property that was correlated with AM fungal infectivity was pH ($r = 0.27$,

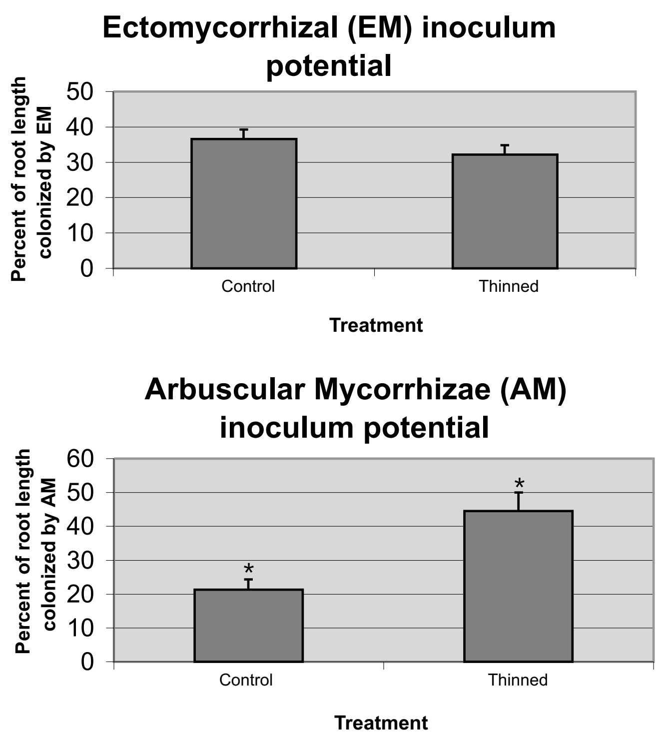

Figure 2—The inoculum potential for EM fungi (top graph) and AM fungi (bottom graph) from "bait-plant bioassays" in the thinned and control units. Differences between the thinned and control units were determined by analysis of variance for a random block design on log $(x + 1)$ transformed data ($F = 2.1601$, $P > 0.05$, $F = 5.4437$, $P < 0.005$, respectively).

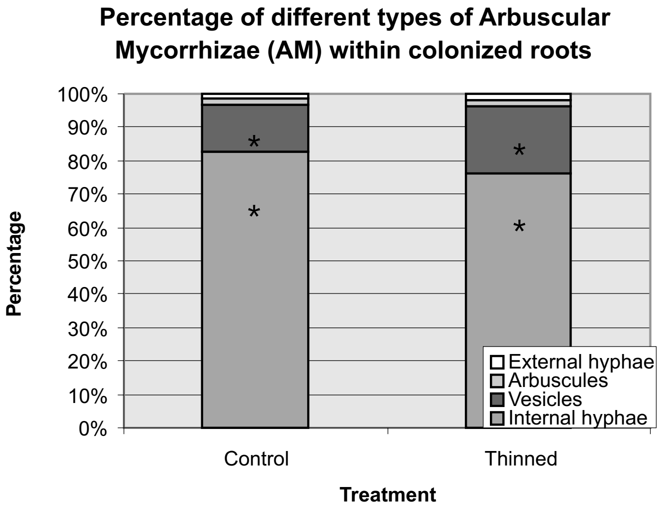

Figure 3—The percentage of external hyphae, arbuscules, vesicles, and internal hyphae within colonized corn roots for each treatment. An asterisk indicates that one type of AM is significantly different ($P < 0.05$) between treatments as determined by Analysis of Variance for a random block design on log (x + 1) transformed data ($F = 8.3001$, $P > 0.005$, $F = 15.4818$, $P > 0.0002$, respectively).

$N = 60$, $P \geq 0.05$). There were no significant differences between soil properties in the thinned and control stands (table 1). However, pH levels were higher in the thinned units than the controls, which is congruent with the findings that thinned units had greater propagule densities of AM fungi than the controls.

Both the thinned and control units were dominated by graminoids followed by nonlegume forbs, legumes, and woody shrubs. Unidentified sedges, *Carex* sp., dominated the herbaceous cover for both units followed by C3 grasses including *Elymus elymoides* (squirreltail), *Poa fendleriana* (muttongrass) and *Festuca arizonica* (Arizona fescue). *Muhlenbergia montana* (mountain muhley) was the only dominant C4 grass in both units; however, its presence was less than the dominant C3 grasses. The dominant forbs in both the thinned and control units were *Cirsium wheeleri* (thistle) and *Solidago* sp. (goldenrod). There were slightly higher species richness, Simpson's diversity, and herbaceous cover in the thinned units in comparison with the control units; however, these differences were not significant (table 2).

Table 1—Soil parameter comparisons between thinned and control ponderosa pine treatments. Average values are presented, N = 30. Standard error of the mean (SEM) for each value is in parentheses.

	Thinned		Control	
pH	6.06	(0.756)	5.85	(0.352)
Total P	0.86	(0.148)	0.95	(0.199)
Total N	1.57	(0.209)	1.65	(0.174)
Organic C	10.29	(1.158)	10.59	(0.994)

Table 2—Plant species richness and Simpson's diversity index (SI) for 1999 herbaceous transects. Minimum, maximum and average richness is by individual transects, not units.

Treatment	Average richness	Min.	Max.	Average SI	Min.	Max.
Control	5.1	1	13	4.34	0.58	12.78
Thin	5.8	1	12	4.71	0	13.75

Discussion

Restoration thinning increased the cover of herbaceous, AM hosts and decreased the cover of ponderosa pine, EM hosts. These aboveground changes were accompanied by the expected belowground changes: propagule densities of AM fungi increased by 23 percent while those of EM fungi decreased by 6 percent. These preliminary results indicate that mycorrhizal fungal population densities respond rapidly to restoration thinning in northern Arizona ponderosa pine forests. Two main processes control population densities of mycorrhizal fungi following disturbance: immigration of new propagules from nearby areas and survival and spread of residual propagules (Warner and others 1987). Rapid colonization of AM fungi has been illustrated in other studies. For example, Johnson and McGraw (1988) found that unreclaimed taconite tailings devoid of AM fungi were colonized within weeks of reclamation. It was hypothesized that spores were transferred to the reclaimed tailings by biotic (animals) and abiotic (wind and water) vectors (Johnson and McGraw 1988). In the present study it is likely that AM fungi spread from preexisting mycorrhizal hyphae in living and dead roots.

Soil disturbance has been reported to generally reduce AM propagule densities due to the destruction of the hyphal network during the break-up of the soil macrostructure (Reeves and others 1979; Fairchild and Miller 1988). Consequently, one may predict that propagule densities should decrease following thinning due to soil disturbance from mechanized logging equipment. This effect was not observed in our study, which may indicate that soil disturbance was not severe enough to have destroyed AM fungal propagules, or that the sites were rapidly colonized by AM fungal spores and residual hyphae. Another study, conducted by Rives and others (1980), showed that there was no reduction in population densities of AM fungi following soil disturbance. It has been suggested that environments with a high proportion of grasses are more tolerant of disturbance because their fiberous root systems generate high densities of infective propagules (Jasper and others 1991). A successional study of AM propagule densities across a grassland to forest chronosequence showed AM inoculum potential increased with increasing grass cover and decreased in later successional sites with EM trees (Johnson and others 1991). Similarly, a study by Benjamin and others (1989) illustrated that dominant herbaceous plants had lower AM colonization as tree density increased, possibly because these plants had insufficient photosynthetic capability to support AM infection. Finally, some research indicates that while the fungal propagule densities do not decrease following disturbance, AM species composition may change (Johnson and Pfleger 1991). Specifically, AM species from the Glomaceae family depend more on hyphae for reproduction than spores. In contrast, AM species from the Gycosperaceae family depend more on spores for reproduction than hyphae (Biermann And Linderman 1983).

Implications for Restoration

Pre Euro-American ponderosa pine understory communities were dominated by warm-season (C4) grasses, which are often obligately mycotrophic (Cooper 1960; Wilson and Hartnett 1998). However, current ponderosa pine forests are dominated by species that form no mycorrhizal associations (*Carex* sp.) or C3 grasses that are often facultatively mycotrophic. This pattern is contrary to other studies that suggest succession proceeds from dominance of nonmycotrophic and facultative mycotrophic species to obligate mycotrophic species in later successional stages (Janos 1980; Allen and Allen 1984). This reversed pattern has been established because the natural fires that historically would minimize competition between ponderosa pine and herbaceous plants have been suppressed, and now shade-tolerant C3 grasses are better adapted to these environments. Therefore, without the reduction of EM tree competition and reintroduction of fire, the understory of Southwestern ponderosa pine forests will likely continue to be dominated by nonmycotrophic *Carex* and facultatively AM species. Noyd and others (1995) demonstrated that obligate C4 grasses, *Andropogon gerardii* (big bluestem) and *Schizachyrium scoparium* (little bluestem), were unable to grow or survive as seedlings in soil where AM fungi were eliminated; however, *Elymus canadensis* (C3 grass) establishment was unaffected by AM fungi availability. Similarly, Hetrick and others (1994) found that C4 grasses were competitively superior to C3 grasses when grown in the presence of AM fungi; but C3 grasses were competitively superior in soils without AM fungi. As a result, one can predict that increased populations of AM fungi in the thinned restoration units may promote increased C4 grass cover. Finally, recent mycorrhizal fungi research in a variety of environments has shown that mycorrhizal interactions may be important determinants of plant diversity, ecosystem variability, and productivity (Hartnett and Wilson 1999; van der Heijden and others 1998a,b). Microcosm experiments simulating European and American grasslands have shown that increasing AM fungal diversity can increase diversity and overall community structure (Klironomos and others 2000; van der Heijden and others 1998b).

Conclusions

Restoration thinning impacts propagule densities of EM and AM fungi. Reducing pine cover and increasing herbaceous cover shifts the system toward a more AM dominated community. Increasing propagule densities of AM fungi may favor the establishment of a C4 grassland community. As a result, knowledge of AM fungal propagule densities and AM fungal species composition will be crucial to understanding the successional response of herbaceous understory communities to Southwestern ponderosa pine forest restoration thinning and prescribed burning treatments.

Acknowledgments

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Plant Community Responses to Livestock Grazing: an Assessment of Alternative Management Practices in a Semiarid Grassland

Matthew R. Loeser
Thomas D. Sisk
Timothy E. Crews

Abstract—One of the most prevalent land-use practices in the American Southwest, and one of the most contentious issues among land-use policymakers, is the grazing of domestic livestock. In an effort to contribute scientific understanding to this debate, we have designed experiments comparing the effects of alternative grazing regimes on plant communities. In a semiarid grassland of northern Arizona, we have implemented a replicated study of four treatments: (1) low-intensity, long-duration grazing rotations; (2) high-intensity, short-duration rotations (Holistic Resource Management-style grazing); (3) very high intensity, short duration grazing (to simulate herd impact); and (4) livestock exclosure. Beginning in 1997, we conducted annual surveys of the plant communities with Modified-Whittaker plots. Preliminary results suggest that interannual variability affecting all study plots is high, and that these alternative management strategies do not have dramatic short-term effects on the plant community. Comparisons of native and exotic species richness, as well as ground cover of grasses and forbs, showed no consistent pattern due to treatment over a 3-year period. Our results suggest that the effects of alternative livestock management styles in the semiarid grasslands studied are modest, at least in the short-term, and that future plant monitoring programs would greatly benefit from a multiscale sampling design.

Introduction

Semiarid grasslands are diverse and productive ecosystems that have been subjected to anthropogenic forces for over a century. Among the many perturbations affecting these systems, the grazing of domestic livestock has been recognized as the most pervasive (Fleischner 1994). Although the effects of grazing have been studied for over a century, little consensus has been reached with regard to grazing's impact on grassland composition and productivity, due in part to the difficulty of separating the effects of livestock from tremendous interannual variation in climatic patterns. While some studies have found grazing to be the primary determinant of plant community composition (for example, Conley and others 1992; Fuhlendorf and Smeins 1997), others have found climatic patterns to outweigh grazing effects (for example, Gibbens and Beck 1988; Herbel and others 1972). Moreover, the interactions of climatic variation and grazing disturbances may exacerbate the changes in grassland plant communities (Fuhlendorf and Smeins 1997).

Within this sphere of unpredictable climatic events, humans endeavor towards sustainable land-use practices, but in many cases we fail due to our ignorance of the complex array of forces, both natural and anthropogenic, operating in all ecosystems. Certainly the last century of fire prevention programs in the forests of the American Southwest are a testament to our efforts to eliminate or control disturbance, but as a consequence, we have placed many forest ecosystems in tenuous states where they are prone to catastrophic fire (Covington and others 1994). Scientists and public alike recognize a need to restore disturbance cycles to many natural systems. However, efforts to restore grasslands have been slow at best, and degradation by land-use activities are so acute that they have become a global conservation priority (Samson and Knopf 1996). Noss and others (1995) concluded that grasslands were among the most endangered ecosystems in the United States, and of those ecotypes that were most severely degraded, grasslands suffered areal losses that were threefold greater than any other ecosystem. Likely explanations of this decline include their widespread use for agriculture and development, and the commonly assigned low aesthetic value that has translated into a low priority for conservation (Samson and Knopf 1996).

The Southwest has nearly one million acres of arid and semiarid grasslands, but unfortunately our understanding of their ecology is fairly superficial, especially of the effect of large mammal herbivory such as domestic livestock. By some accounts the number of cattle on rangelands of the Western United States doubled, from 25.5 to 54.4 million head between 1940 and 1990 (Belsky and others 1999), affecting approximately 70 percent of the land surface in the Western States (Fleischner 1994), and 86 percent of the land in Arizona (Mayes and Archer 1982).

Surprisingly little is known about the effects of varying intensities of livestock grazing on the biological diversity of semiarid grasslands. Although many studies have examined the effects of livestock on the soils, vegetation, and birds

In: Vance, Regina K.; Edminster, Carleton B.; Covington, W. Wallace; Blake, Julie A. comps. 2001. Ponderosa pine ecosystems restoration and conservation: steps toward stewardship; 2000 April 25–27; Flagstaff, AZ. Proceedings RMRS-P-22. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

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of riparian areas (Berendse 1999; Saab and others 1995; Trimble and Mendel 1995), this information cannot be applied to grassland systems because upland grasslands differ substantially, both in structure and function, from riparian ecosystems. In addition, Hastings and Turner (1965) have presented compelling photographic evidence that the last century has seen a transformation over large areas of the Southwest, of grasslands to woody shrublands. While ecological impacts from current livestock grazing practices are widespread and ongoing, it is important to recognize that the greatest damage from overgrazing appears to have taken place in the 1890s (Wildeman and Brock 2000).

Because livestock grazing can be conducted at varying intensities, it is likely that some management techniques alter biological diversity more than others. The standard method for measuring such effects has been to establish exclosures and conduct pairwise comparisons of species richness and ground cover (Hughes 1996). Because of the tremendous variability in responses as seen in the literature, it is exceedingly difficult for land managers to confidently proceed with grazing management plans. In addition to the confounding effects of variable climatic patterns mentioned previously, numerous criticisms have been directed at previous grazing studies, including the small size of exclosures (Fleischner 1994), a lack of standard methods for measuring grazing intensity (Fleischner 1994), changes in the behavior of small herbivores and granivores (McNaughton 1983), and poor experimental design (Stohlgren and others 1998). Stohlgren and others (1998) address many of the sampling design concerns and draw specific attention to inadequate assessment of spatial heterogeneity of landscapes. In grassland ecosystems, choosing the appropriate spatial scale at which to measure a given response variable is not trivial, yet the majority of plant diversity studies have arbitrarily chosen sample plots of 1 m^2 or less (Stohlgren and others 1998). Collins and Glenn (1997) have demonstrated that measures of local species richness generated from a 1 m^2 plot often correlate poorly with landscape-scale species richness. Similarly, Stohlgren and others (1998) compared common rangeland vegetation sampling techniques and found the traditional, fine-scale measurements to greatly underestimate rare or patchily distributed species.

To address many of the problems associated with previous study designs, we initiated an experiment with both spatial and temporal replication. This paper addresses three key issues in the study of livestock grazing impacts:

- Does increasing intensity of livestock grazing have a consistent effect on the ground cover and species richness of both native and exotic grasses and forbs?
- Do grazing treatments cause a predictable shift in species composition of the overall plant community?
- Can our sampling techniques and models of plant species richness be scaled accurately to reflect conditions of larger landscapes?

Methods

Site Description

Located at 2,160 m elevation and approximately 32 km southeast of Flagstaff in north-central Arizona, the Reed Lake field site occupies Upper Great Basin grassland (Brown 1994) at the edge of ponderosa pine (*Pinus ponderosa*) forest. The dominant perennial grasses are *Agropyron smithii* and *Elymus elymoides*. A vertisol soil occurs across the entirety of the 25-ha study site. Annual precipitation averages between 300 and 460 mm and is approximately evenly divided between winter and the July–September monsoons (Brown 1994).

Experimental Design

Meaningful application of science to grazing issues will require comparisons of the effects of actual management practices (“treatments”) as well as experimental treatments designed to elucidate the relationships between grazing and important ecosystem variables. Our experimental design, replicated in time and space, consists of four treatments in three blocks on the landscape (Sisk and others 1999). The four treatments are (abbreviations shown in parentheses): (1) conventional low-intensity, long-duration grazing rotations (CON); (2) high-intensity, short-duration rotations informed by the principles of Holistic Resource Management (HRM, see Savory 1988); (3) very high-intensity, short-duration grazing to simulate herd impact (VHI); and (4) livestock exclosure (EXC).

Stocking rates and rotations for the first two treatments are determined by ranchers on adjoining pastures, while the latter two treatments are implemented on experimental plots by researchers and ranchers. Instead of manipulating small plots in a way thought to mimic range management, we are working on active ranches, where we have carefully selected matched sites in which spatially and temporally extensive treatment effects can be complemented with 1-ha EXC and VHI treatments. Of the four treatments, only the VHI treatment is not a simulation of a current grazing policy; it serves as a critical upper-end treatment representing a ranching practice that could be implemented only with open-range herding. Its primary purpose is to help identify mechanistic relationships and inform interpretation of results from all treatments.

Plant Surveys

To measure plant community patterns, we established a Modified-Whittaker plot in each treatment plot (fig. 1; Stohlgren and others 1995). The Modified-Whittaker plot samples the heterogeneity of plant communities with a higher level of accuracy then either true random sampling or the original Whittaker plot design (Stohlgren and others 1995). This sampling design involves a 1,000 m^2 plot with nested subplots consisting of one 100 m^2, two 10 m^2, and ten 1 m^2 subplots (fig. 1). Our version of the Modified-Whittaker plot differs slightly from the design proposed by Stohlgren and others (1995) in that we have placed four of the 1 m^2 subplots within the central 100 m^2 plot. Each Modified-Whittaker plot has been measured annually since 1997, with surveys conducted between mid-July and early September. The 1 m^2 plots were surveyed with a 50-point frequency frame (Mueller-Dombois and Ellenberg 1974) that provided a more objective measurement of ground cover by species and a more comprehensive species list

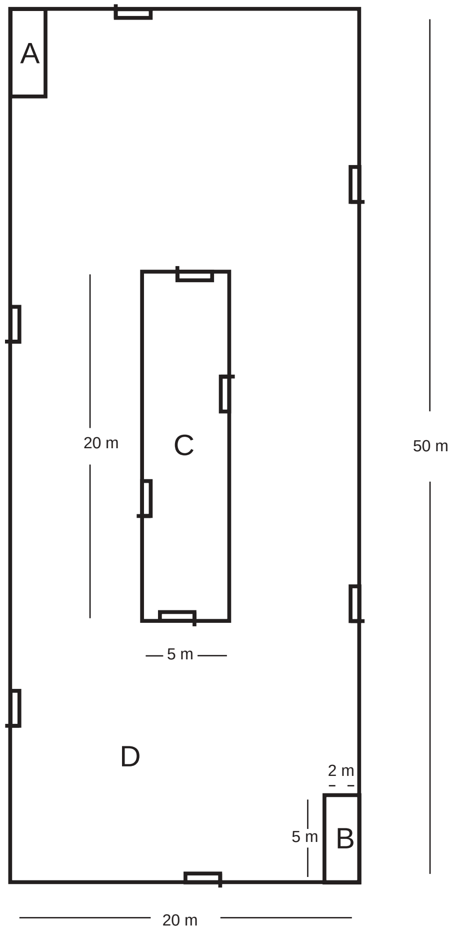

Figure 1—A schematic diagram of the version of Modified-Whittaker plot employed in this study showing multiscale plots in a nested design (see Stohlgren and others 1995).

than could be obtained with visual estimates. Only species lists were assembled for the 10 m^2, 100 m^2, and 1,000 m^2 plots. Plant identifications were referenced with the Deaver Herbarium located at Northern Arizona University in Flagstaff, AZ, and voucher specimens were retained in the Sisk Laboratory. Of the approximately 80 plant species found at the Reed Lake site, 15 could not be identified due to poor specimens but were considered unique species and therefore included in overall estimates of species diversity.

Grass Versus Forb—Three years of detailed plant community data have generated a wealth of information, and here we present only a subset. We calculated trends in ground cover estimates for the two primary functional groups, grasses and forbs, using data collected from the 1 m^2 plots over all 3 years. Data were transformed with an arcsine (square root) function prior to analysis (Sokal and Rohlf 1995), but are presented as percentages. A repeated-measures ANOVA was conducted for each functional group (Sokal and Rohlf 1995).

Native Versus Exotic—To address the issue of whether livestock grazing management techniques affect the establishment of exotic species, we calculated the number of native and exotic species in each treatment for each of the 3 years. In addition, we calculated the total ground cover of native and exotic species by treatment and year. A repeated-measures ANOVA was conducted for species richness and ground cover.

Community-Level Analysis—To determine whether treatment was having a predictable effect on overall plant diversity, we used 3 years of 1-m^2 subplot data to calculate three indices, as described in Magurran (1988). First, we calculated values for the Shannon-Wiener index of diversity based on the proportional abundance of species, with higher values suggesting more species and greater evenness. Second, we estimated evenness by comparing the observed diversity to the maximum potential diversity. Values fall between 0 and 1, with higher values indicating a more evenly distributed community. Third, we calculated the Simpson's index, which is weighted towards the more dominant species and is fairly insensitive to differences in rare species. All calculations were made based on Magurran (1988).

As suggested by Stohlgren and others (1998), we estimated the similarity of plant communities between treatments using Jaccard's Coefficient (signified as *J*, Krebs 1989). Jaccard's Coefficient (or index) is a standard similarity index that indicates the amount of overlap between the communities of any pair of sites. Within each replicate, we compared the CON treatment to each of the other three treatments and calculated the corresponding Jaccard's Coefficient. We chose to use the CON treatment as the "control" because it represents the most common grazing management style for Southwestern rangelands, both historically and currently (Kruse and Jemison 2000), while other treatments are more recent deviations from this management. The *J* value was then calculated for each 1,000 m^2 plot comparison and averaged across all three replicates. A *J* value of 1.0 would represent complete overlap of species between two sites.

Another useful product from the nested design of the Modified-Whittaker plots is the ability to generate species-area curves. Species-area curves have the very practical application of extrapolating total species richness to areas much larger then the plots in which the plant surveys were conducted. One standard variant of the species-area curve is defined as:

$$y = m(\log \text{Area}) + b$$

where y is the number of species, m is the slope of the line, and b is a constant. However, Stohlgren and others (1997) have demonstrated that a corrected species-area curve provides a far more accurate extrapolation of plant species to large areas:

$$y = (m/J)(\log \text{Area}) + b$$

where *J* is the mean Jaccard's Coefficient. We calculated the species-area curve for each treatment based on 1999 data from the 1 m^2, 10 m^2, and 100 m^2 plots. We then calculated the *J* value based on the species lists of each 1,000 m^2 plot within a treatment and used the average J value to correct the original species area curves. The estimated species richness for an area of 3,000 m^2 was calculated using both the standard (uncorrected) species-area curve and the corrected species-area curve and compared to the observed species richness of the three 1,000 m^2 plots summed together.

The final analysis of the plant community data was conducted with a relatively new ordination technique of demonstrated utility, nonmetric multidimensional scaling (NMDS, Minchin 1987). Based upon NMDS, Minchin (1987) developed a simulation procedure that makes it possible to model patterns of variation in community composition along ecological gradients. We used species abundance data from the 1 m^2 plots of all 3 years to compare communities of each plot to one another.

Results

Grass Versus Forb Cover

If cattle graze grasses more commonly than forbs (Stuth 1991), then a moderate intensity of grazing may limit grasses from becoming the dominant ground cover. In this scenario, forbs may have higher germination success in areas where some grazing occurs because grasses aren't able to hoard resources such as light and nutrients (Collins and others 1998). We analyzed ground cover data from our 1-m^2 plots for total grass and forb cover across all treatments and years. Since the initiation of the study in 1997, grass cover appears to have increased roughly 10 percent. However, this change was not statistically significant (fig. 2; df = 3, F = 0.540, p = 0.668). In contrast, forb cover appears to be decreasing by approximately 20 percent, but again this trend is not statistically significant (fig. 2; df = 3, F = 0.697, p = 0.579). Although neither trend demonstrated clear statistical significance, both suggest changes that occurred due to yearly patterns rather than treatment effects.

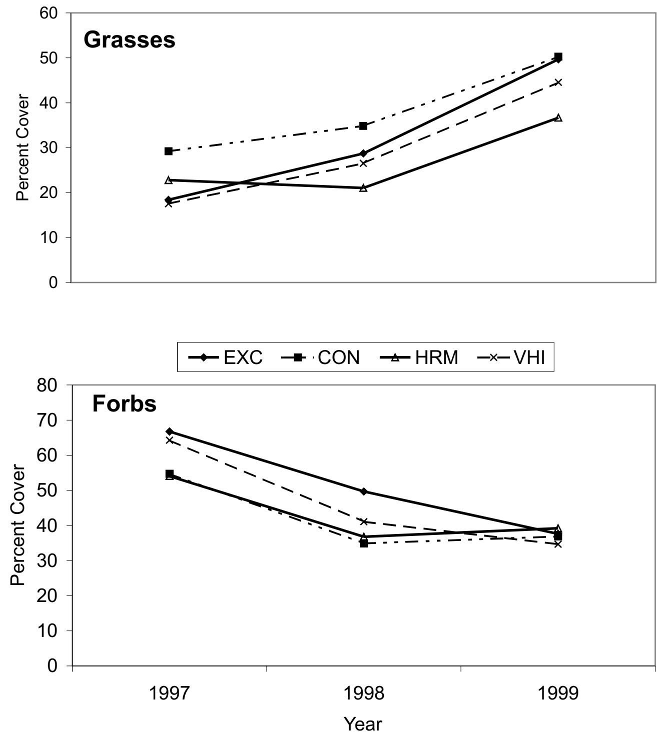

Figure 2—Percent ground cover by year and treatment for grasses and forbs as calculated from 1 m^2 subplot data. Treatment shows no significant effect for either grasses or forbs, but climatic patterns appear to be benefiting grasses to the detriment of forbs. Standard error bars are not shown because they overlap for all treatments within any given year indicating no statistically significant differences (grasses: df = 3, F = 0.540, p = 0.668; forbs: df=3, F = 0.697, p = 0.579).

Native Versus Exotic Richness and Cover

Exotic plants are generally considered to be successful colonizers of disturbed habitats, and high intensities of livestock grazing have been suggested to increase invasibility (Hobbs and Huenneke 1992). To explore this hypothesis, we first computed the total native and exotic species richness from our 1 m^2 plots for each treatment by year. No statistically significant differences were found within years (table 1; 1997: df = 3, F = 1.933, p = 0.165; 1998: df = 3, F = 1.274, p = 0.317; 1999: df = 3, F = 0.917, p = 0.455). Similarly, ground cover by natives and exotics from our 1 m^2 plots did not show significant differences among treatments within years (table 1; 1997: df = 3, F = 1.525, p = 0.213; 1998: df = 3, F = 2.434, p = 0.103; 1999: df = 3, F = 0.254, p = 0.952).

Table 1—A comparison of species richness and ground cover of native and exotic plants from data collected in 1 m^2 subplots over 3 years. Means listed across from year and standard errors shown in parentheses below its respective mean.

		# of native species				# of exotic species				Cover of native species (%)				Cover of exotic species (%)			
Year	n	EXC	CON	HRM	VHI	EXC	CON	HRM	VHI	EXC	CON	HRM	VHI	EXC	CON	HRM	VHI
1997	3	10.0	8.3	9.3	7.0	3.7	3.7	3.0	3.3	54.3	54.2	58.8	65.7	29.7	29.9	17.9	16.1
		(1.2)	(0.3)	(0.7)	(0.6)	(0.7)	(0.9)	(0.6)	(0.3)	(8.4)	(8.6)	(5.6)	(1.9)	(1.6)	(10.6)	(5.2)	(4.3)
1998	3	16.3	13.0	13.3	12.0	4.7	4.3	2.7	4.7	43.3	44.8	50.3	45.0	24.0	33.7	7.6	22.6
		(0.7)	(2.1)	(3.0)	(1.0)	(0.3)	(0.3)	(0.9)	(0.3)	(10.5)	(9.5)	(3.0)	(3.7)	(6.4)	(11.6)	(2.3)	(3.2)
1999	3	24.7	22.3	27.3	22.7	7.7	8.0	7.3	9.0	54.0	55.2	56.2	54.4	32.8	31.8	19.8	24.9
		(0.9)	(0.3)	(2.4)	(2.2)	(1.2)	(0.6)	(0.9)	(0.6)	(9.1)	(11.8)	(5.6)	(2.2)	(11.4)	(12.4)	(7.3)	(3.2)

Table 2—A comparison of measures of community composition (Shannon-Wiener, Evenness, and Simpson's Diversity) for 3 years of plant community data collected from 1 m^2 subplots. Mean values provided with standard errors shown in parentheses.

Year	Treatment	Shannon-Wiener	Evenness	Simpson's Diversity^{-1}
1997	EXC	2.0 (0.2)	0.7 (0.04)	5.9 (1.0)
	CON	1.8 (0.1)	0.7 (0.03)	4.7 (0.7)
	HRM	1.8 (0.2)	0.7 (0.05)	4.8 (0.9)
	VHI	1.6 (0.1)	0.7 (0.05)	4.0 (0.2)
1998	EXC	1.9 (0.2)	0.7 (0.05)	5.1 (1.1)
	CON	1.9 (0.0)	0.7 (0.03)	5.1 (0.3)
	HRM	1.7 (0.2)	0.7 (0.05)	4.4 (0.7)
	VHI	1.8 (0.0)	0.7 (0.03)	5.0 (0.2)
1999	EXC	2.1 (0.2)	0.6 (0.06)	5.1 (1.2)
	CON	2.1 (0.2)	0.6 (0.05)	5.4 (1.0)
	HRM	2.1 (0.2)	0.6 (0.03)	5.5 (0.9)
	VHI	2.2 (0.1)	0.6 (0.02)	5.2 (0.2)

Diversity Indices

Assessing diversity can be done with many different analytical approaches, but perhaps the most common methods involve the calculation of diversity indices. Although ubiquitous in the ecological literature, diversity indices can be difficult to interpret because they are often codependant on species richness and species evenness. We have calculated the means and standard errors for each treatment, by year, for the Shannon-Wiener index, Evenness, and Simpson's Diversity. As displayed in table 2, treatments are fairly similar, regardless of index or year. These data are provided as descriptive information and no statistical tests were conducted.

Furthermore, to determine if alternative management styles, namely EXC, HRM, and VHI treatments, exhibited predictable patterns of unique species we calculated the Jaccard's Coefficient by comparing each alternative treatment plot with its partnered CON treatment plot. There was moderate similarity between plant communities in treatments, with an average of 60 percent overlap of species and less than 6 percent standard deviation (fig. 3).

Community-Level Comparisons

In addition to difficulties in analysis and interpretation, it is difficult to graphically display information from diversity indices. Instead, we display community data through ordination techniques, and we statistically tested the patterns with nonmetric multidimensional scaling. While no treatment effects were found to be significant, a strongly significant pattern was discovered for plant communities when analyzed by year. The data suggest that the plant communities differ significantly among years regardless of treatment type (fig. 4).

Species-Area Curves

It is commonplace in the ecological literature to extrapolate results from small-scale studies to larger landscapes. While most sampling methods fail to provide a framework for extrapolating data, the nested design of the Modified-Whittaker plot is ideally suited to these efforts. Extrapolations employing the species-area equation, corrected with Jaccard's Coefficient, were more accurate than extrapolations with the uncorrected species-area curve. More specifically, the uncorrected species-area curve tended to underestimate the plant community by roughly 20 species, whereas the corrected species-area curve underestimated by as few as three and no more than 13 (table 3).

Discussion

The significance of livestock grazing in determining patterns of community organization is generally believed to be great, but the literature demonstrates a poor understanding

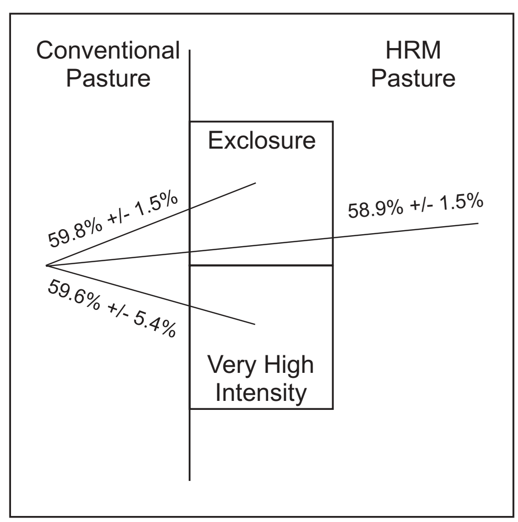

Figure 3—Species overlap between the CON treatment and each of the other three treatments. Jaccard's Coefficient values are averaged across the three replicates.

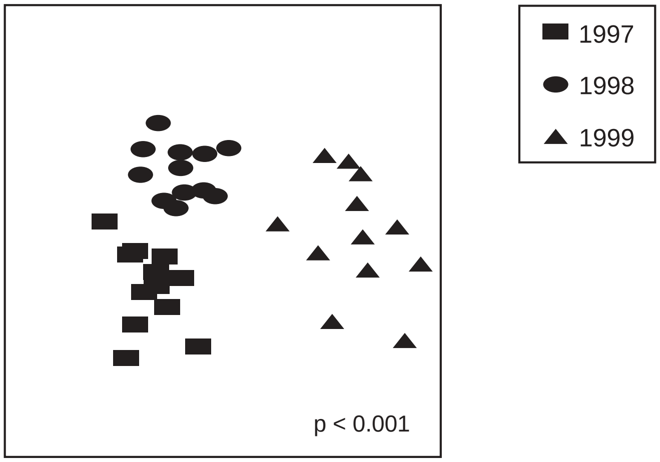

Figure 4—An ordination of plant community data determined with nonmetric multidimensional scaling. Each point represents the plant community of an individual plot from a specific year. Those communities that are more dissimilar will be found farther apart than those that are more similar. Treatments were not statistically significant. However groupings of plant communities by year that ignore treatment are statistically significant.

of specific patterns and processes. Milchunas and Lauenroth (1993) summarized 152 studies in grasslands and found sharply contrasting results with respect to the responses of species composition to large-herbivore grazing. The authors suggested that these conflicting findings may be due to one or more of the following explanations: (1) differing methodologies, including the intensity of grazing and the scale at which the response variable was measured; (2) the productivity of the grassland, which includes its soil resources and moisture; and (3) the evolutionary history of each grassland type (Milchunas and Lauenroth 1993). This paper most directly addresses the first, that is, the appropriate methodology for measuring grassland community responses to grazing treatments.

Grass Versus Forb Cover

Shifts in the proportional ground cover of plant functional groups as a function of grazing intensity have been well documented in highly productive sites. Long-term research conducted in the tallgrass prairies of Kansas demonstrated that perennial grasses created a near monoculture by outcompeting forbs when disturbance events were eliminated (Collins and others 1998). In comparatively less productive ecosystems, such as semiarid grasslands, the interactions of grasses and forbs are poorly understood. In one study of a semiarid grassland, Brady and others (1989) found ground cover increased for both grasses and forbs following the removal of cattle. In contrast, Rambo and Faeth (1999) found grass cover increased by 20 percent in two of three sites when grazing was eliminated for at least 8 years. Our own work suggests no immediate differences in grass or forb cover due to alternative grazing treatments. However, across all treatments grass cover is increasing while forb cover is decreasing. Although it may be too soon to draw conclusions about the fairly recent treatments, EXC and VHI, we have a higher level of confidence in comparing CON and HRM treatments, which have been in place for a minimum of 12 years. With 3 years of ground cover data, the alternative HRM management style does not exhibit a markedly different level of grass or forb cover.

Native Versus Exotic Richness and Cover

Although livestock grazing has often been implicated as a dispersal agent for exotic species (Hobbs and Huenneke 1992; Mack 1981), several studies show no link between grazing and exotic species richness. A multi-State sampling effort in the Northern Rocky Mountains found no consistent differences in native or exotic species richness due to livestock grazing (Stohlgren and others 1999). Similarly, in a semiarid grassland, Rambo and Faeth (1999) found no differences in exotic species richness in either long-term or short-term grazing exclosures. Preliminary findings from our work agree with these latter studies, showing no statistical difference in either species richness or ground cover of native or exotic species due to treatment. Our results do show a consistent increase in the overall number of species

Table 3—Standard species-area curves compared with species-area curves corrected with Jaccard's Coefficient. Equations were determined from 1 m^2, 100 m^2, and 1,000 m^2 plots, and then validated against observed species in the sum of the three 1,000 m^2 plots in each treatment.

Treatment	Standard Equation Y = m (log Area) + b	Estimate of spp. in 3,000 m^2	Corrected Equation Y = m / *J* (log Area) + b	Mean J (S.E.)	Corrected Estimate of spp. in 3,000 m^2	Actual spp. in 3,000 m^2
EXC	y = 6.2581x + 7.6129	29.4	y = 6.2581/0.631 + 7.6129	0.631 (0.066)	42.1	45
CON	y = 4.7742x + 9.8387	26.4	y = 4.7742/0.655 + 9.8387	0.655 (0.075)	35.2	48
HRM	y = 6.5x + 8	30.6	y = 6.5/0.609 + 8	0.609 (0.041)	45.1	49
VHI	y = 5.0806x + 8.129	25.8	y = 5.0806/0.649 + 8.129	0.649 (0.034)	35.3	44

from 1997 to 1999, which is likely to be a function of time since a severe drought year in 1996. Although not statistically significant, it is interesting to note that the HRM treatment had the lowest cover of exotic species in 2 of 3 years of sampling.

Plant Community Responses

By employing a variety of analytical tools we have demonstrated that the plant communities in each treatment do not differ significantly. This may be due to minimal ecological impacts or it may be that the duration of this study is not yet sufficient to detect ongoing, gradual change. Community changes are difficult to document because they involve the combination of species richness and the abundance of each individual species. To address this issue we compared traditional diversity indices and employed more powerful ordination techniques. The standard diversity indices, Shannon-Wiener, Evenness, and Simpson's, suggest no consistent differences among treatments. Furthermore, the species overlap between the conventional treatment and each of the other three treatments is nearly 60 percent for each comparison.

The ordination of community data with NMDS also demonstrated no significant differences due to treatment; however, there were strongly significant differences between each year. This finding suggests that year-to-year variation has had a stronger effect on the plant community than the treatments themselves. Even though the EXC and VHI treatments represent strikingly different livestock treatments, and the CON and HRM treatments represent longer-term, but less extreme treatments, the plant communities have not shown a strong response to grazing intensity over a 3-year period. Based on findings from other studies (for example, Sala and others 1992), we suggest that the plant community of this water-limited ecosystem is often more sensitive to the high variability in precipitation and soil moisture than to the livestock management treatments compared in this study.

Improved Power with Nested Sampling

Stohlgren and others (1995, 1998) have demonstrated a number of benefits from conducting multiscale nested sampling in plant communities. Our results support their conclusions. For our semiarid grassland study site, standard species-area curves consistently underestimated species richness by about 40 percent, and correcting the species-area curve with Jaccard's Coefficient improved estimates to within 15 percent of the known species richness. If the goal of a particular monitoring program is to estimate the total number of species in a given area of grassland, then nested sampling designs such as the Modified-Whittaker plot should provide increased prediction accuracy.

Management Implications

Land managers appear to be increasingly placed in the catch-22 situation of doing more with less. Especially in the stewardship of public lands, the short-term future appears to include further cuts to monitoring budgets, even while pressure grows to make ecologically sound decisions in the face of increasing litigation. Our preliminary results allow us to make two management suggestions, with the caveat that our preliminary results are specific to the Reed Lake study site, and should influence broader interpretations only where they concur with other published works.

First, future monitoring efforts should heed the warning that small sampling plots may provide significantly less accurate information then multiscale, nested sampling plot designs. At our study site, the use of Modified-Whittaker plots and replicated treatments allowed us to improve our species richness prediction accuracy by 45 percent. Efforts are under way to apply this sampling design to other response variables, including the diversity of macro- and micro-arthropods, and the size and fluxes of nutrient pools.

Second, alternative livestock approaches, including exclosures and very high intensity treatments, did not have immediate or dramatic effects on the plant community of this semiarid grassland. In fact, the HRM treatment that has been in place for a dozen years shows no statistically significant differences from the plant community of the adjacent conventionally grazed grasslands. Therefore, if the goal of a grassland management program is to move plant communities in a particular direction with alternative grazing practices, planners and practitioners must recognize the necessity for a long-term commitment to treatments and to an efficient, well targeted monitoring program.

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Butterfly Response and Successional Change Following Ecosystem Restoration

Amy E. M. Waltz
W. Wallace Covington

Abstract—The Lepidoptera (butterflies and moths) can be useful indicators of ecosystem change as a result of a disturbance event. We monitored changes in butterfly abundance in two restoration treatment units paired with adjacent untreated forest at the Mt. Trumbull Resource Conservation Area in northern Arizona. Restoration treatments included thinning trees to density levels comparable to densities at the time of Euro-American settlement, and reintroducing a low to medium intensity fire to the system. One unit was treated in 1996, the second in 1998. Butterfly communities, nectar availability, and herbaceous species richness were compared between treated and adjacent control forests, and between 3-year posttreatment and 1-year posttreatment forests. Butterfly species richness and abundance were two and three times greater, respectively, in restoration treatment units than in adjacent control forests. Nectar plant species richness ranged from two to 10 times greater in restoration treatment units than in adjacent control forests. Comparison of the 3-year posttreatment unit with the 1-year posttreatment unit showed little difference in butterfly species richness and abundance, although no statistical comparisons can be made due to sample size. These restoration treatments offer a unique opportunity to study responses to and recovery from disturbance and restoration at a landscape level.

Introduction

Current studies and methods of ecosystem restoration are often focused on structural components, such as overstory or understory plant composition, and not on functional processes, such as nitrogen cycling, plant pollination, and/or trophic level interactions (but see Kaye 1997; Covington and others 1997). As a result, ecosystem restoration often overlooks invertebrates as important components necessary for ecosystem function and process. This emphasis on ecosystem structure is primarily a result of limited available information. Historical records were often inventories of merchantable resources and did not describe how species interacted with each other, or what processes were important to ecosystem functioning (for example, Dutton 1882). Knowledge of ecosystem function and process can be obtained from studying current-day undisturbed ecosystems (Leopold 1949), although few if any "undisturbed" ecosystems exist. In addition, historical function and process can be inferred by studying function and process in experimentally restored ecosystems (Leopold 1949).

In ponderosa pine forests in the Southwest, forest structure can be reconstructed from presettlement remnants and historical records (Covington and Moore 1994; Fulé and others 1997). This information is limited to overstory species, with some information on herbaceous components also available from early land surveys (Dutton 1882) and phytolith studies (Rovner 1971; Bozarth 1993; Fisher and others 1995; Fredlund and Tieszen 1997). These data show that Southwest forests prior to Euro-American settlement had lower tree densities than current forests (50–150 trees per hectare (tph) in 1870 versus 500–3,000 tph in 1994, Covington and Moore 1994) with grassy openings. Experimental restoration of these forests has been initiated using thinning to reduce current tree densities and reintroducing fire.

Restoration in ponderosa pine is hypothesized to impact all components of the ecosystem, including arthropods. The change from a closed-canopy forest with little or no herbaceous understory community to an open forest with a dominant herbaceous community results in plant diversity and plant production increases (Covington and others 1997; Springer and others, this proceedings). This in turn can result in increases in the abundance and diversity of herbivore arthropods (Erhardt and Thomas 1991). In addition, restored forests show increases in soil moisture and soil temperature when compared to control forests (Covington and others 1997); both factors directly influence the success rate of arthropod pupation (Erhardt and Thomas 1991; Scoble 1992).

The Lepidoptera (butterflies and moths) can be excellent indicators of herbaceous community diversity and composition (Gilbert 1984; Erhardt 1985; Kremen 1994; Sparrow and others 1994). Both butterflies and moths can be host-specific as larvae, but become nectar generalists as adults, encompassing a broad range of ecological niches. Changes in butterfly diversity can indicate changes in the abundance and diversity of a wide variety of invertebrates (Scoble 1992). Presence of a butterfly or moth species indicates presence of the larval host plant, as well as sufficient adult food resources. Day-flying butterflies in particular have a well-known taxonomy, and often can be easily identified in the field (Scoble 1992).

In the ponderosa pine ecosystem, the diurnal Lepidopteran (butterflies) can be used to monitor important changes in ecosystem function as a result of disturbance, rehabilitation, or restoration events. Restoration of ponderosa pine forests

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involves thinning to create openings, thereby initiating an herbaceous successional pattern (Springer and others, this proceedings). While butterflies have shown decreases in abundance after clear-cut logging events (Hill and others 1995), openings in forests or changes created by roadways and paths often show higher butterfly abundances than nearby forests (Pollard and others 1975). It has also been shown that the butterfly community structure changes in response to successional changes from grasslands to forests in Europe (Erhardt and Thomas 1991). These studies suggest the potential for both an immediate and a long-term response to ponderosa pine restoration. For example, butterfly diversity and abundances could initially decrease in response to logging and ecosystem disturbance, then increase in abundance and diversity with the increasing herbaceous community.

The study presented here was initiated to establish the response of butterflies to ecological restoration treatments and potential mechanisms of that response. If changes in butterfly communities are noted, future studies will examine the role of butterflies as bioindicators of invertebrate pollinator groups.

Research Questions

To address the usefulness of butterflies as indicators of restoration treatments, this paper specifically addresses the questions:

1. Does the butterfly community differ between ponderosa pine restoration treatment units and untreated forest?
2. What are potential mechanisms of these differences: (a) Are nectar resources distributed differently? (b) Are host plants distributed differently? (c) Does butterfly habitat preference explain butterfly distribution?
3. Finally, do butterflies show a successional response to restoration treatments?

Methods

Study Site

The study site used for this research is a ponderosa pine (*Pinus ponderosa*) and Gambel oak (*Querqus gambelii*) forest located between Mt. Logan and Mt. Trumbull, about 35 km north of the Grand Canyon on the Arizona Strip. This land is currently managed by the Bureau of Land Management, and falls within the newly designated Grand Canyon-Parashant National Monument. Mt. Logan, Mt. Trumbull and the surrounding highlands form a sky island of ponderosa pine, with desert grassland to the north and the Grand Canyon to the south. The nearest ponderosa pine forest is about 100 km east, on the Kaibab Plateau. The elevation of the sky island ranges from 1,675 m to 2,620 m. The area receives an average of 40–45 cm of rainfall annually, and contains some of the biota of the Great Basin (Utah Flora 1986), in addition to the flora of northern Arizona (Kearney and Peebles 1951). The forest is predominately ponderosa pine, although Gambel oak composes 15 percent of the overstory. Other tree species in the area include aspen (*Populus tremuloides*), pinyon (*Pinus edulis*), juniper (*Juniperus osteospermus*), and New Mexican locust (*Robinia neomexicanus*). Although New Mexican locust is classified as a shrub, it grows to tree stature at this site, and is sampled as a tree. The understory component is dominated by sagebrush (*Artemesia tridentata*), and shows evidence of invasion by nonnative species, such as cheatgrass (*Bromus tectorum*) and wheatgrasses (*Agropyron* spp.). Although 150 herbaceous species have been documented at Mt. Trumbull since the late 1990s (J.D. Springer, personal communication), the forest floor cover prior to restoration treatments was 70 percent litter and duff, with only 15 percent of the cover represented by understory species.

Approximately 1,450 ha of the 5,000-ha forest is targeted for restoration treatment (Covington and others l995), and as of 1999, approximately 200 ha had been thinned and burned. This restoration project is jointly sponsored by the BLM, the Arizona Game and Fish Department, and Northern Arizona University Ecological Restoration Institute. The ponderosa pine ecosystem restoration project takes an adaptive management approach, so that results from initial treatments can be incorporated into later treatments. The actual treatments in place are therefore continuing to evolve. At this site, analysis of fire scars provided a fire exclusion date (in other words, the date of the last widespread fire) of 1870. All treatment sites incorporate thinning trees to densities resembling those at the time of fire exclusion. Trees established at the time of fire exclusion are retained, as well as younger "replacement" trees for presettlement era trees that have died since fire exclusion. Fire is used preliminary to help reduce slash, and then will be returned to the landscape every 4–7 years, depending on weather conditions. For complete details of the treatment, please see the 1996 Annual Report to BLM. Treatment of all 1,450 ha is to be completed by 2002.

Butterfly Sampling

Butterfly monitoring data presented in this paper were taken from two units treated in 1996 (Lava Unit) and in 1998 (Trick Tank Unit). Butterfly monitoring transects (Pollard 1977) were established in the two treatment units, and were paired with monitoring transects in untreated forests (control) adjacent to each unit. Transects were placed 50 m from unit boundaries to minimize edge effects and were at least 50 m apart. Although length of transects varied with total treatment unit size to maintain buffer from edges, lengths of the paired treatment-control transects were the same. Transects in the Lava unit totaled 450 m in each the treatment and the control; transects in the Trick Tank unit totaled 600 m in each the treatment and the control for a total of 2,100 m per survey.

Transects were monitored every week, between May and September of 1999. Diurnal butterflies are very sensitive to cool and windy conditions, often limiting their flights on cloudy, cool days, thereby reducing chance of observation. Therefore, sampling was done between 1,000 and 1,600 hours, on days warmer than 17 °C, with winds less than 10 mph, and mostly sunny skies (Pollard 1977). A total of 5 minutes per 100 m was spent looking for butterflies. Butterfly species encountered on each transect were recorded, along with location along transect, and lateral distance from

Table 1—Butterfly species found at Mt. Trumbull Resource Conservation Area, Summer 1999. * denote most common species.

Hesperiidae	Lycaenidae	Nymphalidae	Papilionidae	Pieridae
*Epargyreus clarus**	*Callophrys gryneus*	*Chlosyne californica*	*Papilio multicaudata*	*Anthocharis sara*
*Erynnis telemachus**	*Strymon melinus*	*Danaus gilippus*		*Colias eurytheme**
Heliopetes ericetorum	*Hypaurotis crysalus*	*Danaus gilippus*		*Nathalis iole*
Pyrgus communis	*Glaucopsyche lygdamus**	*Euphydryas chalcedona*		*Pieris protodice**
Thorybes pylades	*Hemiargus isola*	*Euptoieta claudia*		
	Leptotes marina	*Limenitis bredowii*		
	Plebejus acmon	*Limenitis weidemeyerii*		
	Plebejus icarioides	*Nymphalis antiopa*		
		Nymphalis californica		
		Phycoides campestris		
		Polygonia gracilis		
		Precis coenia		
		Vanessa cardui		
		Vanessa atalanta		
		Vanessa carye		

transect (perpendicular to transect). In addition for each observation, we recorded behavior (in other words, nectaring, basking, flying), and if collected. If the butterfly could not be identified in flight, attempts were made to capture and collect the insect. The timed portion of the survey corresponded only to the observations and did not include time spent in pursuit of a butterfly.

Nectar Resources and Host Plant Distribution and Habitat Preferences

To quantify nectar resources and host plant distribution, vegetation along the butterfly monitoring transects were monitored in 1-m^2 plots every 20 m along the established transect. A total of 30 plots were sampled in both the Trick Tank treatment and the nearby control, and 23 plots were sampled in both the Lava treatment and its adjacent control. Plots were monitored three times during the summer: May, June and August. At each plot, flowering and nonflowering plants were tallied by species. In addition, total number of flowers per 1-m^2 plots was tallied. These data were summarized by unit and treatment to determine differences in flowering plant species richness and host plant abundance.

Habitat preferences for each butterfly were determined from the literature, predominately Scott (1984). All butterfly observations from both units and the entire season were then grouped into habitat preference classes. Total tallies are presented here. No statistical analysis was done because data were lumped, resulting in no replication.

Successional Response to Restoration

To address whether butterflies show a successional response to restoration, we compared the butterfly communities from Lava and Trick Tank units, using the same monitoring data from above. The Lava unit was treated in 1996, and is referred to as a 3-year posttreatment unit. The Trick Tank unit, treated in 1998, is referred to as a 1-year posttreatment unit. Butterfly species richness, abundance, and composition were compared.

Results

Thirty-three butterfly species were collected at the Mt. Trumbull site in 1999 (table 1). The most common of these included the silver-spotted skipper (*Epargyreus clarus*, EPCL), the Gambel oak dusky-wing (*Erynnis telemachus*, ERTE), the silvery blue (*Glaucopsyche lygdamus*, GLLY), the orange sulfur (*Colias eurytheme*, COEU), and the checkered white (*Pieris protodice*, PIPR). The butterflies used a range of hostplants, including legumes, mustards, various shrubs, shrub-trees (New Mexican locust) and trees (oak).

Butterfly Community Response to Restoration Treatments

We found up to three times as many butterfly species in restoration treatments as in the adjacent, untreated control forests (fig. 1, Repeated Measures ANOVA, F = 12.9, p < 0.10). Table 2 lists species found in the Lava and Trick

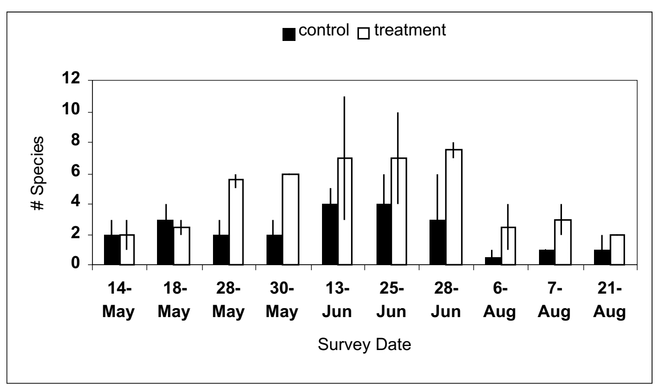

Figure 1—Butterfly species richness encountered in restoration treatment and adjacent control units. Up to three times as many species were observed in restoration treatments than in adjacent controls. Repeated measures ANOVA, F = 12.0, p < 0.10.

Table 2—Butterfly species and total observations found in restoration treatments and adjacent controls of Lava and Trick Tank units. Tally is total observations from 10 surveys, each survey covered 2,100 m (1,050 in control, 1,050 in treatment).

Control		Treatment	
Species	Tally	Species	Tally
Erynnis telemachus	22	*Epargyreus clarus*	70
Epargyreus clarus	16	*Erynnis telemachus*	59
Glaucopsyche lygdamus	12	*Glaucopsyche lygdamus*	46
Limenitis bredowii	10	*Colias eurytheme*	42
Colias eurytheme	8	*Pieris* sp.	35
Plebejus icarioides	3	*Plebejus icarioides*	9
Phycoides campestris	2	*Vanessa cardui*	9
Plebejus acmon	2	*Leptotes marina*	6
Papilio multicaudata	1	*Plebejus acmon*	6
Euphydryas chalcedona	1	*Phycoides campestris*	5
Limenitis weidemeyerii	1	*Strymon melinus*	4
Pieris sp.	1	*Limenitis bredowii*	4
		Papilio multicaudata	2
		Polygonia gracilis	2
		Pyrgus communis	2
		Heliopetes ericetorum	1
		Vanessa carye	1
		Thorybes pylades	1
		Euptoieta claudia	1

Tank treatment and control units. Common species were observed in both control and restoration treatment units but were seen more often in restoration treatments. Rare species (such as *Heliopetes ericetorum*) were observed in restoration treatments, when seen.

Butterfly abundance was also significantly greater in restoration treatment areas as in adjacent control forests (Repeated Measures ANOVA, F = 7.98, p = 0.106). Four to eight times as many butterflies were observed in treatment units as in adjacent controls on any given survey date. As shown in table 2, silver-spotted skipper (*Epargyreus clarus*) was the most common species observed, and was observed five times as frequently in the treatment unit as in the control units. *Pieris* species (whites) where highly abundant in the area, but were observed only once in the control units. Only one butterfly showed higher abundances in the control units than in the treatment units. The Arizona sister (*Limenitis bredowii*) was rare in 1999 but was observed 10 times in control units and only four times in treatment units. This pattern for the Arizona sister is consistent with 1998 data (Waltz and Covington 1999).

Nectar Resource Richness and Abundance

We examined potential mechanisms for increased butterfly species richness and abundance in treated forests. From the 1-m^2 plots surveyed along butterfly transects, we compared the species richness, species abundance, and flower abundance of the plants that were flowering at the time of survey. Number of flowering species (species richness) was significantly greater in treatment vegetation plots than in control vegetation plots (fig. 2, Kruskal-Wallis test (effects = survey and trt), treatment Z = 5.45, p < 0.05). In addition, flowering plant abundance (or number of plants) was also significantly greater (fig. 3, Kruskal-Wallis, treatment Z = 5.50, p < 0.05). Not surprisingly, abundance of flowers per 1-m^2 plot was also significantly greater in treated units, with up to 200 times as many flowers in 1-m^2 plots (Kruskal-Wallis, Z = 5.77, p < 0.05). These results showed that both a higher diversity of plants and a higher total number of plants were flowering in restoration treatment units.

Host Plant Distributions and Habitat Preferences

We examined host plant distributions for the five most common butterfly species (table 1, * species denote most common). Table 3 lists these butterflies, their associated

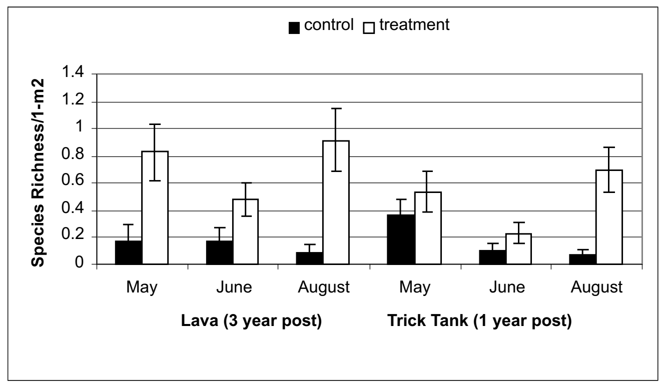

Figure 2—Nectaring plant species richness per 1-m^2 plot was significantly higher in restoration treatment units than in adjacent controls. Kruskal-Wallis for effects survey and treatment. Treatment Z = 5.45, p < 0.05.

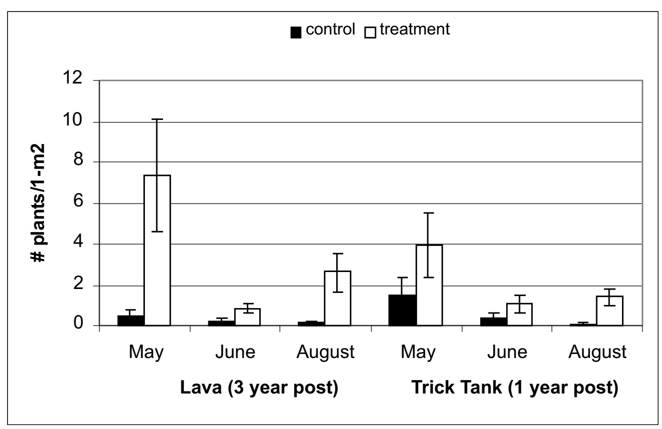

Figure 3—Nectaring plant abundance (number of plants observed) per 1-m^2 plot was significantly higher in restoration treatment units than in adjacent controls. Kruskal-Wallis for effects survey and treatment. Treatment Z = 5.50, p < 0.05.

Table 3—Distribution of host plants for the five most common butterfly species. Distributions showed no significant differences between restoration treatments and adjacent control units. Sampling design does not incorporate the tree species well. Butterfly species codes refer to first two letters of genus and first two letters of species.

		Plants / 1-m^2	
Butterfly	**Host plant**	**Control**	**Treatment**
EPCL	*Robinia neomexicana*	0.02	0.02
ERTE	*Quercus gambelii*	0.21	0.21
GLLY	Fabaceae	1.26	2.47
Pieris sp.	Brassicaceae	0	0.17
COEU	Fabaceae	1.26	2.47

host plants and the host plant abundance per 1-m^2 plot. Two of the species, the silver-spotted skipper (EPCL) and the Gambel oak dusky-wing (ERTE), host on tree species. Only tree seedlings are measured on 1-m^2 plots. However, we included the tally of tree seedlings per 1-m^2 plot in this table, which shows no differences in tree abundances between treatment and control. In fact, host plant species were distributed equally between control and treatment units.

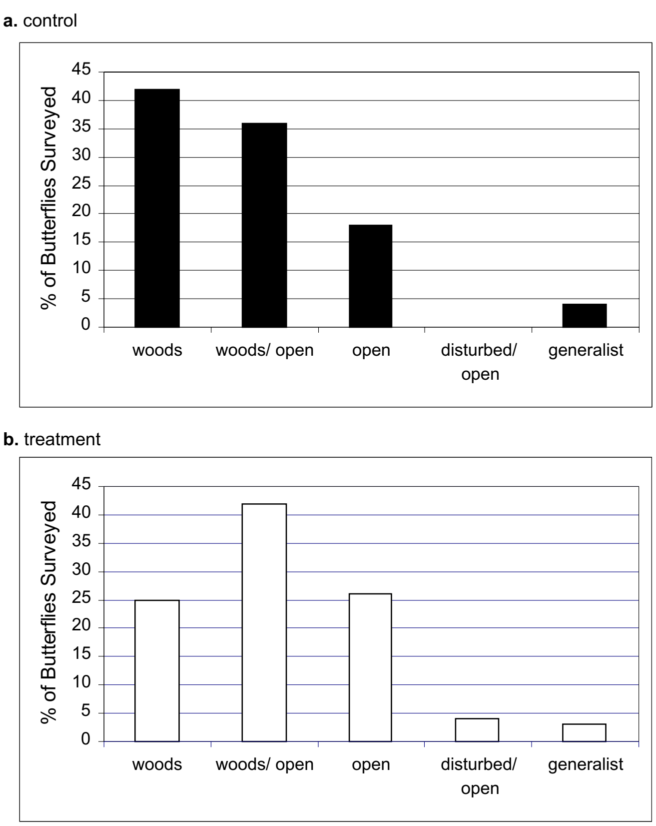

Figure 4—Habitat preferences of observed butterflies in (a) control and (b) restoration treatment units. Graphs represent proportion of observed butterflies only, lumped from 10 surveys from both sites. No statistical analysis performed.

Alternative methods are planned to adequately assess host plant distributions in treatment and control units.

Data on habitat preference and host plant preference are reported only as observed trends, and will be used to generate hypotheses to be tested in future field seasons. The highest proportion of the butterflies observed in control units were species preferring wooded habitat (fig. 4a). The highest proportion of the butterflies observed in treatment units were species preferring woods/open habitat (fig. 4b). Host plant preferences also displayed interesting trends. Butterfly species that hosted on tree species made up the highest proportion of the butterflies observed in the control areas. Alternatively, butterfly species hosting on legumes and forbs made up the highest proportion of the butterflies observed in the treatment areas. Basically, the butterfly species observed in control areas were species that preferred wooded habitat, and most often were species that hosted on tree species. Butterflies observed in treated areas were species that preferred more open habitat, and most often were species that hosted on legumes or forbs. Because of lack of sample size, the variables habitat preference and host plant type were not statistically tested. These results represent proportional trends only.

Butterfly Community Successional Response to Restoration

Figure 5 displays the butterfly species richness data across surveys between the Lava unit (3-years posttreatment) and the Trick Tank unit (1-year posttreatment). Because we observed only one unit for each successional stage, these data are presented as trends. Although some trend exists toward higher numbers of species in the 3-year posttreatment unit, we cannot assess those differences with only one sampling unit per successional stage. Plant communities shift from an annual forb community in 1-year posttreatment units to more perennial forbs and grasses in the 3-year posttreatment unit (Springer, personal communication). We did see increases in diversity of flowering species

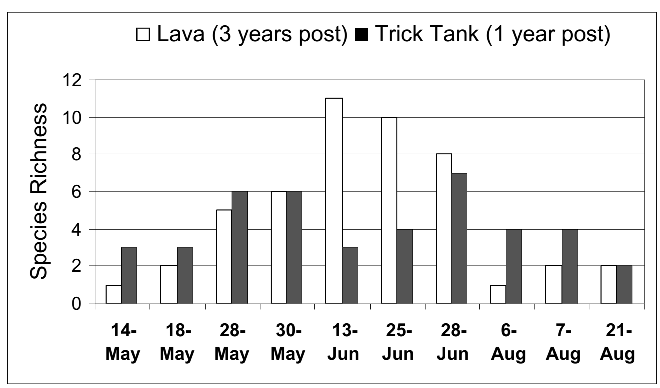

Figure 5—Butterfly species richness in Lava (treated 1996) and Trick Tank (treated 1998) units. Although there are trends toward higher species numbers in the Lava unit (treated in 1996), statistical analysis is not possible with only one sampling unit per successional state.

in the 3-year posttreatment unit compared with the 1-year posttreatment unit. Future studies will be designed to increase sample size and address the successional response of butterflies to ecological restoration.

Discussion

We have shown that the butterfly community had higher species richness and abundance in restoration treatments when compared with adjacent control forests. We also suggest that species with low abundance are more often found in treated units than in control units. Ponderosa pine restoration treatments alter habitat by opening up tree canopies and increasing herbaceous production. The fast responses of butterflies to these changes (within one season after treatment) suggest that arthropods may be one of the first responders to ecological changes. Erhardt and Thomas (1991) also documented butterfly responses to plant successional changes, showing butterfly community changes even before plant community changes could be detected. While some studies show that butterflies decrease after logging events (Hill and others 1995), the logging monitored in those cases was clear-cutting, with little regard for understory establishment. Our results agree with several studies that show gaps created in forest canopies increase butterfly abundances, whether through increased host plant diversity, or changes in microhabitats (abiotic variables) (Pollard 1977; Pollard and others 1975; Holl 1996).

The mechanistic hypotheses we examined to explain butterfly distributions suggest nectar resource availability may contribute greatly to adult butterfly distribution patterns. Our preliminary results showed large differences in available nectar resources in Lava and Trick Tank treatment units, when compared with adjacent control forests. Increased nectar resources can be associated with disturbed areas; many early successional plants are flowering forbs (Springer 1998). Studies have shown that nectar resources are important to adult oviposition selections. Host plants are utilized only when sufficient adult resources (nectar) are also available (Grossmueller and Lederhouse 1987; Murphy 1983). Successful butterfly habitat must therefore include sufficient larval and adult food resources.

Our restoration treatments have much higher plant diversity than control forests (Springer and others, this proceedings), implicating a higher diversity of butterfly host plants in these areas. Our results showed no differences in the host plant distributions of five common butterflies between treatment and control forests. However, the sampling method used here was not designed for tree species, which act as host plants for two of the most common butterflies. In addition, the variability of plant abundance measured in these units was also very high, suggesting plant distributions should be measured on a greater than 1-m^2 scale. Both reasons suggest our current sampling method is not adequate to address questions of host plant distributions.

Habitat preference may also contribute to butterfly community composition shifts. A habitat change from closed-canopy, low plant diversity forests to open canopy grasslands may also see a corresponding shift in butterfly community composition. Our preliminary observations suggest that butterflies found in treated forests were more likely to be species preferring open habitat, requiring legume and forb host plants. Conversely, butterflies observed in control forests were more likely to be species preferring wooded habitat, and hosting on tree species. Although we could not test these patterns statistically, other papers have shown the importance of habitat selection in community composition (Ehrlich and Raven 1964; Ehrlich 1993; Erhardt and Thomas 1991).

Studies of the successional response of butterflies to restoration may give insight to how fine of a scale butterflies can respond to. The successional change in plant communities following thinning and burning can be dramatic. In the Lava unit, plant species richness increased from less than 5 species in 1995 to 7, 16, and 20 species in posttreatment years 1, 2, and 3 respectively. However, it may take longer for these plant species to establish viable, reproducing populations. Steffan-Dewenter and Tscharntke (1997) observed successional responses of butterfly communities in set-aside fields in Germany. Although species richness of the butterfly communities did not change through 4 years of agricultural field succession, the butterfly community composition did show differences. If butterfly communities can show responses at these yearly scales, they may be very useful indicators of successional stage following a disturbance or a restoration treatment.

Problems/Confounding Factors

These results are from a small sample size (two units with paired controls) and should be treated as preliminary, but at the same time suggest more rigorous studies are validated. Currently, a paired block design is set up to monitor butterfly response to restoration treatments and mechanisms of these responses. This increased sample size should help reduce this problem in future studies.

To successfully examine butterfly population responses to restoration treatments, reproductive success and host plant usage should also be documented. Our current design monitors only adult butterfly populations. However, studies have shown positive correlations between adult butterfly densities and larval densities, suggesting monitoring of adult butterflies may provide a close indication of larval densities (Steffan-Dewenter and Tscharntke 1997). Of the 20 species recorded in the Lava and Trick Tank units and associated controls, 12 were classified as locally distributed, not ranging far from their host plants as adults (Scott 1984).

Implications

The response of butterfly communities suggests other arthropod herbivores may respond to restoration treatments in similar ways. Arthropods constitute the largest biomass of any taxon and occupy a large range of functional niches (Kremen and others 1993). Bees and other nectar or pollen feeding arthropods may show increases in diversity and abundance as a response to increased nectar resources, and therefore may parallel the responses of the butterfly community. The importance of pollinators to ecosystem function has recently become the focus of many questions, due to the decreasing abundance of native bees (Buchmann and Nabhan 1996; Kevan 1999). Although butterflies are not as efficient at pollination as the Hymenoptera (bees and wasps) or Diptera (flies) (Scoble 1992), they are easier to monitor and identify. In addition, arthropods decompose organic material, release nutrients back into the ecosystem, and provide the largest food source in almost every ecosystem (Wilson 1987).

Using these restoration experiments, the efficacy of butterflies as biodiversity indicators of these other arthropods can easily be tested, by increased sampling of other taxonomic groups, correlations, and finally testing in other areas undergoing restoration treatments.

The research presented here provides insight into how the butterfly community responds to habitat change, and some of the mechanisms behind that response. Not only do butterfly communities contribute to ecosystem functioning through herbivory, providing food source and pollination events, they also have the potential to be bioindicators of biodiversity in other arthropod guilds.

Acknowledgments

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Habitat Associations of the Sagebrush Lizard (*Sceloporus graciosus*): Potential Responses of an Ectotherm to Ponderosa Pine Forest Restoration Treatments

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Stephen S. Germaine

Abstract—Little is known about the response of ectotherms to ponderosa pine (*Pinus ponderosa*) restoration treatments. The ambient body temperature of an ectotherm affects its physiology, development, and behavior. Microhabitat availability and heterogeneity are critical factors in determining which thermoregulation choices are available to a terrestrial ectotherm (Stevenson 1985). Forest restoration treatments (for example, thinning and burning) will alter herpetofauna microhabitats by decreasing tree canopy cover and allowing more sunlight penetration to the forest floor. This change could, depending on the species, have positive or negative effects on the populations of the area. We sampled microhabitat use by *Sceloporus graciosus* (sagebrush lizards) in northern Arizona at Grand Canyon-Parashant National Monument using standard "pitfall-array" sampling methodology. Univariate analyses were used to relate lizard abundance to ponderosa pine tree density, percent soil cover, percent rock cover, litter depth, and insect density. In a multivariate analysis, ponderosa pine density (negatively correlated) and bare soil cover (positively correlated) were the best predictors of lizard abundance. Restoration treatments will increase small-scale heterogeneity within *S. graciosus* territories by increasing accessibility into and out of sunlight. Based on the thermoregulatory demands of this species, these changes should benefit *S. graciosus*. However, other possible indirect effects of restoration treatments such as increases in predation on lizards (due to greater visibility), as well as changes in food availability, could negatively impact lizard populations. Future research should focus on pre- and postrestoration treatment monitoring of herpetofauna, and on the direct effects of fire on herpetofauna populations within restoration sites.

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Introduction

Ponderosa pine (*Pinus ponderosa*) forest restoration is an experimental management practice currently being applied in the Southwest. This practice varies depending on the management objectives, but generally consists of some combination of forest thinning and prescribed burning. Covington and others (1997) has shown increases in: (1) forest floor sunlight penetration, and (2) herbaceous productivity (grasses, forbs, and shrubs) among restoration-treated ponderosa pine stands. These changes are expected to benefit the fauna of the area, although the direct effects of forest thinning and prescribed fire are poorly represented in scientific literature.

Data from Germaine (1999), in an ongoing restoration study at Grand Canyon-Parashant National Monument, suggest that lizards of the *Sceloporus* genus (*Sceloporus graciosus* and *Sceloporus undulatus*) are more abundant in areas with lower ponderosa pine density. This suggests that restoration treatments could result in higher densities of these *Sceloporus* lizards. However, studies of herpetofauna responses to ponderosa pine restoration before and after treatment do not currently exist.

The direct effects of fire on herpetofauna populations are also poorly understood. Cunningham and others (2000) pitfall-trapped lizards in burned and unburned chaparral and Madrean evergreen forests the year following a high-intensity wildfire, and captured primarily immature lizards. Subsequent trapping over the following two seasons resulted in higher species richness, diversity, and capture rates, indicating rapid settlement of the burned area by adjacent residents. Lizard genera represented in this study included *Sceloporus undulatus* and numerous *Cnemidophorus* species.

A few studies have shown that certain lizard species of fire-dependent ecosystems, such as long-leaf pine (Mushinsky 1985), chaparral (Lillywhite 1977a,b), and sandpine scrub (Greenberg 1993), increase in diversity or density following fire. Both Mushinsky (1985) and Greenberg (1993) found that some lizards of the genus Eumeces remained more abundant in control areas (unburned and unharvested for at least 20 years in both cases). It is apparent that different species have different thermoregulatory demands and should be addressed on a species-specific basis.

In this study we identified current forest condition microhabitat characteristics of the sagebrush lizard (*Sceloporus graciosus*) in a Southwestern ponderosa pine forest.

Microhabitat preferences for this species were determined, based on habitat variation within the study area. Potential responses of *S. graciosus* to restoration treatments were determined by extrapolating habitat relationships from current condition forests to conditions expected in treated forest areas. In the future, these data will be further applied to a long-term study measuring the posttreatment effects of restoration on the age structure, composition, and abundance of this lizard population.

Methods

The study site was in northern Arizona at Grand Canyon-Parashant National Monument. The elevation ranges from approximately 2,080 m to 2,290 m, and the area is comprised primarily of ponderosa pine forest, with abundant sagebrush meadows. During May and June of 1999, prerestoration treatment microhabitat data were collected from 56 pitfall arrays. Arrays consisted of four 5-gallon buckets buried in a "peace-sign" configuration (Jones 1986). Each array included three wire mesh drift fences radiating out 7 m from the center bucket, connected to the other three buckets (fig. 1). Arrays were opened and checked every third day, for a total of 168 trap days. At each array, percent soil cover, percent rock cover, litter depth, and insect density were sampled within a 1 m circular radius surrounding each bucket. Insect density was sampled along four 1 m transects per bucket. The data were collected at timed intervals of 10 seconds per transect, scanning the entire transect length. All insects within the top 2.54 cm of litter were tallied during sampling.

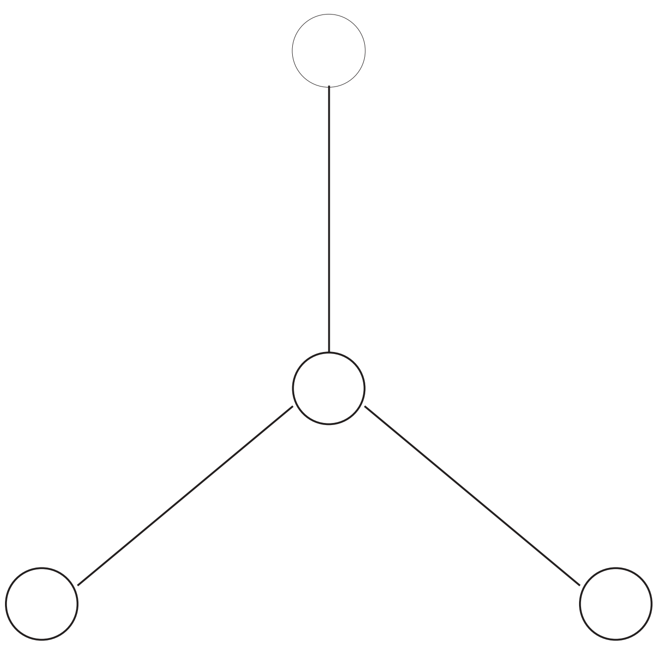

Figure 1—Aerial view of a pitfall array used to sample sagebrush lizard abundance at Grand Canyon-Parashant National Monument in June 1999, assembled in a peace-sign configuration with drift fence delineated by lines radiating out from the center bucket at 0°, 120°, and 240°, respectively.

Germaine (1999) gathered percent coniferous canopy closure and ponderosa pine density at each array. Coniferous canopy closure was measured at four transects as 40 "hits" or "misses", out to a 25 m radius, originating from the center of each array. Ponderosa pine stem density was measured in a 6 m belt running N-S and E-W across each array.

Values for each microhabitat variable collected were averaged for each array. We used stepwise linear regression analyses, with entry and exit levels of alpha = 0.15, to identify those habitat variables that, in combination, were associated with lizard abundance (Sokal and Rohlf 1981; SAS Institute Inc. 1985). Spearman correlations were used to determine univariate relationships between lizard abundance and microhabitat variables that were selected in multivariate models (SAS Institute Inc. 1985). Those microhabitat variables that were not normally distributed, or with unequal variance, were transformed by taking the square root of each variable (Sabin and Stafford 1990).

We calculated the Mallow's Cp statistic to select the best fitting model. This statistic is a measure of total squared error for each model and should be approximately equal to the number of parameters (including the intercept) in the model to choose a parsimonious model (Younger 1979: 493–495).

Results

Three species of lizards were captured during the trapping period. Captures consisted of *Sceloporus graciosus* (n = 81), *Eumeces skiltonianus* (n = 4), and *Sceloporus undulatus* (n = 1). *S. graciosus* abundance ranged from 0 to 9 (± 2.06) captures per array. Ponderosa pine density ranged from 0 to 1,968 (± 466.84) trees per hectare. Bare soil cover ranged from 0 to 91.25 (± 18.84) percent per array. Results from the stepwise linear regression analyses are presented for *Sceloporus graciosus* (table 1). The best model for explaining the variation in *S. graciosus* abundance contained the variables of ponderosa pine density (trees/ha) and percent bare soil cover:

$$\textit{S. graciosus} \text{ abundance} = 1.78 - 0.06 \text{ (ponderosa pine density)} + 0.24 \text{ (percent bare soil cover)}$$
$$C(p) = 2.16$$

This model explained 34 percent of the variance. Ponderosa pine density was negatively correlated with *S. graciosus* abundance (fig. 2). *S. graciosus* abundance peaked in areas that contained 50–200 ponderosa pine trees/ha. Bare soil cover was positively correlated with lizard abundance (fig. 3).

Table 1—Summary of the stepwise linear regression analysis of the sagebrush lizard with habitat variables at Grand Canyon-Parashant National Monument in June 1999.

Parameter	Partial R^2	Model R^2	C(p)[a]
Pine density per hectare	0.28	0.28	3.43
Percent bare soil cover	0.06	0.34	2.16

[a]Mallow's Cp statistic is a measure of the total squared error for each model and should be approximately equal to the number of parameters (including the intercept) in the model.

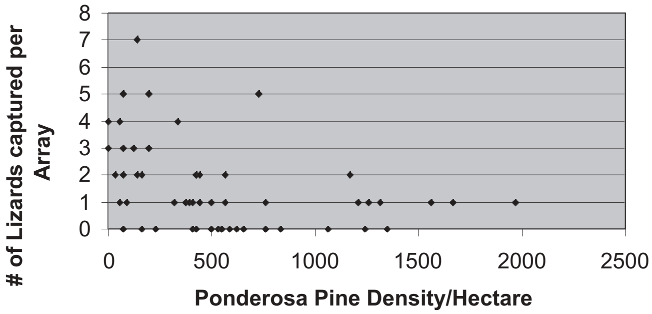

Figure 2—Sagebrush lizard abundance at Grand Canyon-Parashant National Monument in June 1999 was negatively associated with ponderosa pine density in a univariate analysis.

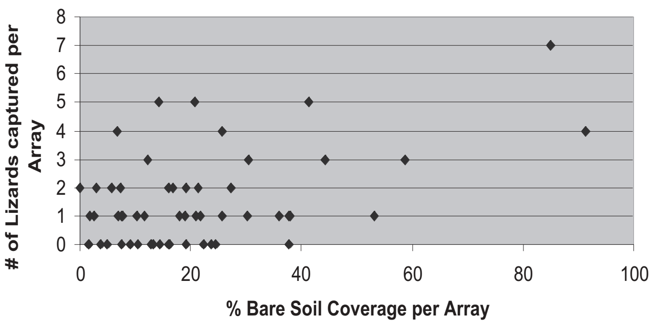

Figure 3—Sagebrush lizard abundance at Grand Canyon-Parashant National Monument in June 1999 was positively associated with bare soil coverage in a univariate analysis.

Discussion

This study indicates the need for small-scale heterogeneity within *Sceloporus graciosus* territories. Contiguous, homogeneous, fire-suppressed ponderosa pine forests inhibit adequate sunlight penetration to the forest floor. This can reduce sunlight accessibility for ectothermic animals with small home ranges, such as most lizards and snakes, which rely upon solar heat for metabolic processes and mobility.

Previous research has established that current Southwestern ponderosa pine forests contain much higher densities of ponderosa pine than were present in these same forests before Euro-American settlement (Cooper 1960; Covington and Moore 1994b). This suggests that sunlight penetration to the forest floor has also decreased during this period due to large increases in ponderosa pine densities. Covington and others (1997) have shown that forest restoration treatments, through reduction in forest tree density and prescribed fire, do in fact open up the canopy and create a more open, parklike landscape. Restoration, carried out through these means, will enhance the thermoregulation options available to *S. graciosus* by lowering litter accumulations (which will provide easier access to burrows), and ultimately increasing accessibility into and out of sunlight.

Ponderosa pine forest restoration treatments, implemented on an adaptive management basis, have potential to increase the habitat suitability of some terrestrial ectotherms by increasing small-scale heterogeneity. Thinning dense stands of ponderosa pine will not only allow more sunlight penetration, but will also create a more patchy distribution of sunlight at the ground layer level. Results of this study suggest that optimal *S. graciosus* habitat for thermoregulation would entail leaving between 50–400 trees/ha after forest thinning treatments.

It is important to recognize the many other factors that could change as a result of restoration treatments, which may inadvertently affect *S. graciosus*. Lawrence (1966) found increases in predatory birds and mammals after a chaparral fire in the Sierra Nevada foothills. Increases in predation and competition, as well as changes in food availability, may also play significant roles in the postrestoration habitat of terrestrial ectotherms.

We were able to explain one-third of the variance in sagebrush lizard abundance. Possible explanations for the lack of a model that explains a considerable amount of the variance can be attributed to several factors: (1) It is possible that the collective contribution of several microhabitat variables determines the success of the animal. (2) There is a high degree of difficulty associated with accurately sampling small-scale heterogeneity within the small territories of Sceloporus lizards. (3) The sample size could have been too small. (4) The length of the sampling period could have been too short. (5) Thorough seasonal population representation of an animal that spends two-thirds of its life below ground is questionable.

Suggestions for future research include the need for detailed behavioral studies that capture common behavior patterns of these lizards. Other recommendations include the need to obtain larger sample sizes, as well as long-term (>5 years) studies. Studies on the direct effects of fire on lizard fitness are essential to correctly understand which scale and intensity of prescribed fires to use in restoration treatments.

Although no studies exist that measure lizard responses directly to forest restoration before and after treatment, we hope that this study will encourage additional research. In those ecosystems where restoration treatments increase sunlight availability to the forest floor, terrestrial ectotherms can be significant indicators of thermal change. Data from this and related studies need to be incorporated into future restoration treatment prescriptions to ensure that habitat needs are met for all native wildlife species during forest restoration activities.

Acknowledgments

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Can We Create and Sustain Late Successional Attributes in Interior Ponderosa Pine Stands? Large-Scale Ecological Research Studies in Northeastern California

William W. Oliver

Abstract—Conflicts over changing demands on our increasingly scarce stands of late successional ponderosa pine could be abated by increasing the proportion of stands with late successional attributes in the forest land base. However, we don't know whether these attributes can be developed through the management of younger stands. Nor do we know whether late successional stands can be managed to perpetuate these values through time. To answer these questions, two long-term large-scale studies were begun to study ecosystem responses to a series of silvicultural treatments that include timber harvest and prescribed fire. In one study, treatments are designed to test several pathways toward late successional forest attributes in a young, even-aged stand. In the other study a treatment is aimed at sustaining an existing late successional stand and contrasting the response of that ecosystem with that of a young, even-aged stand. Although not yet completely installed, the stand structure in one treatment resembles two late successional stands with periodic fire.

Conflicts over changing demands on our increasingly scarce stands with late seral structures often leave forest managers faced with difficult decisions. One obvious way to abate these conflicts would be to increase the number of stands in the forest land base with late seral attributes, such as large trees, snags, large down woody debris, multiple canopy layers, associated shrub, herb, and grass components and canopy gaps. However, it is not known which active management strategy could speed development of these attributes in younger stands, nor is it known whether we can manage late seral structures to perpetuate their values through time. In addition to concerns about esthetics, biodiversity, and sustainable productivity, stands with late seral attributes often contain the most valuable timber and critical wildlife habitat.

To address these questions, we began two long-term large-scale studies in the interior ponderosa pine (*Pinus ponderosa*) forests of northeastern California. The primary objective of the study on the Goosenest Adaptive Management Area (GAMA) is to accelerate late successional attributes in a young-growth stand (Ritchie and Harcksen 1999). One of the objectives of the study located on the Blacks Mountain Experimental Forest (BMEF) is to sustain late successional attributes in stands threatened by the absence of fire (Oliver and Powers 1998). The inception of effective fire control about 1930 at both BMEF and GAMA has resulted in dense stands with a higher proportion of white fir. Before 1930, the median fire return interval at BMEF was about 7 years (personal communication with C. N. Skinner, September 19, 2000). Although data are not yet available for GAMA, we expect the interval to be similar. Neither site has experienced wildfire since 1930. Many stands have suffered severe mortality from competition exacerbated by drought. At BMEF, intense competition has accelerated the demise of the large old trees. Similar shifts in species composition and stand structure resulting from fire suppression have been reported in the San Bernardino Mountains of southern California (Minnich and others 1995). At GAMA, the resulting buildup of fuels has caused fire hazard to become so extreme that long-term protection of the forests seems virtually impossible.

Both sites are ideally suited for a large-scale, ecological research project. Both sites provide enough land for operational-scale studies on areas dedicated to research. Plots can be large enough to monitor treatment effects on small mammals and passerine birds. But the large size of each unit, a minimum of 100 acres, made spatial control on the ground necessary to integrate data collected at different scales across disciplines. It was accomplished by a permanently monumented grid on 328-ft (100-m) centers throughout each unit to which all measurements and other activities are referenced.

We struggled as a team with treatment descriptions and how to translate our idea of what an interior ponderosa pine forest looked like when fire was part of the ecosystem. Although we knew of the aggregated stand structure existing in late seral ponderosa pine stands from silvicultural studies at BMEF (Hallin 1959) and from studies in northern Arizona (Cooper 1961; White 1985), we did not attempt to reconstruct them as did Harrod and others (1999) in central Washington. At BMEF, the method was too time consuming for the large area to be treated. At GAMA, no clues to the original structure existed in the young-growth stand. The 50-year records from a Methods-of-Cutting study (Dolph and others 1995) that provided data on stands about 20

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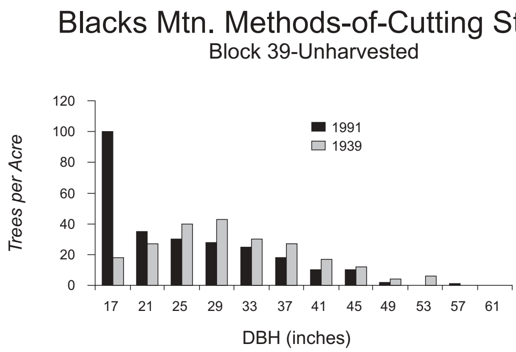

Figure 1—Distribution of tree diameters in 1939 and 1990 in the unharvested plot in Block 39 of the Methods-of-Cutting study on Blacks Mountain Experimental Forest.

years after initiation of effective fire control (fig. 1) were most relevant for diameter distributions, as were data from the Beaver Creek Pinery (fig. 2). This stand of ponderosa pine and California black oak (*Quercus kellogii*) on the west-side of the Sierra Nevada never has had effective fire protection. Although the Beaver Creek Pinery is in an environment very different from BMEF, we believed that the stand structure resulting from three wildfires in the 1990s was instructive.

Goosenest Adaptive Management Area Research Project

The Goosenest Adaptive Management Area, within the Klamath National Forest, lies on the north side of the Medicine Lake Highlands—an area of recent volcanics running east from Mt. Shasta. The entire Adaptive Management Area is clothed in 60- to 80-year-old forest, composed primarily of ponderosa pine and white fir in dense stands containing about 230 trees and 170 ft^2 of basal area per acre. The U.S. Forest Service acquired the area in the 1950s from the Long Bell Lumber Company, the successor to the Weed Lumber Company of Weed, California. Long Bell logged these stands in the 1920s and 1930s via railroad. Very few merchantable trees were left standing. After logging, pine and fir regenerated abundantly. Also at about that time the area was brought under effective wildfire control.

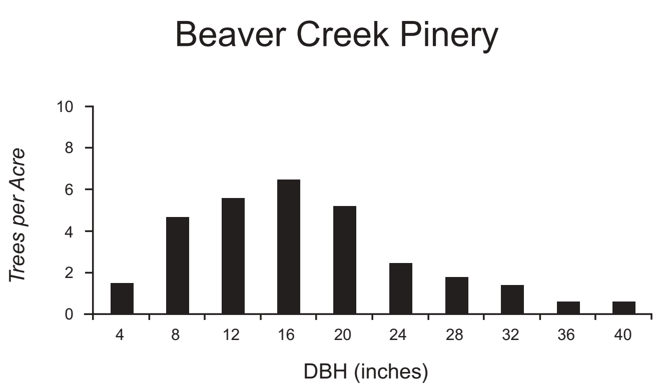

Figure 2—Distribution of tree diameters in the Beaver Creek Pinery.

Project Description

The overall research objective is to test silvicultural practices that may accelerate development of late successional forest characteristics in young growth stands, utilizing partial cutting and prescribed fire. Active management has the potential to reduce the hazard of wildfire and to accelerate development of late successional attributes, but a precise pathway for this development is unknown. Therefore, treatments were designed to place the forest on various stand development trajectories toward what are commonly conceived as late successional attributes (table 1). The pathways to late successional attributes are:

- **No treatment**—In reality this treatment is minimal management because fire prevention and fire suppression activities will continue as before. This provides an inactive management strategy for comparison with the other three pathways.
- **Accelerate late successional large trees without fire**—A major late successional attribute, large trees, should be created most rapidly by this treatment. Trees left following treatment are spaced 18 to 25 ft apart and are the largest, regardless of species. Fuel reduction is by mechanical methods not by prescribed fire, because of the danger of killing the white fir.
- **Accelerate late successional pine without fire**—We recognized that late successional stands in the area had a much greater proportion of ponderosa pine than exists today and that tree spacing was clumpy. To achieve these attributes we left all dominant and co-dominant ponderosa pines regardless of spacing, saving only white firs larger than 30 inches in diameter at breast height (d.b.h.) and those where no pines were present. We further increased the proportion of pine in this initial entry by removing clumps of decadent white fir and regenerating with pine. Our goal is to have ponderosa pine comprise 80 percent of the stems. Prescribed fire is withheld to investigate whether mechanical methods will be an effective surrogate.
- **Accelerate late successional pine with fire**—The same treatment as the previous one plus the restoration of the function of frequent low intensity fires by periodically igniting controlled fires.

Table 1—Treatment contrasts in the Goosenest Adaptive Management Area Research Project in northeastern California.

Treatment contrasts	
Species Emphasis	**Large Tree Emphasis**
Ponderosa pine retention is the most important criterion.	Tree diameter is the most important criterion.
High horizontal diversity—15 percent of area in created openings.	Minimal horizontal diversity—no openings created.
Tree spatial distribution is clumpy.	Tree spatial distribution is homogeneous.

Each strategy is replicated five times on 100 acres in each unit of a well-buffered, fully randomized field design. We anticipate a reentry in 10 years at which time we will evaluate these pathways and may adjust the treatments.

Blacks Mountain Ecological Research Project

Blacks Mountain Experimental Forest is located within the Lassen National Forest in northeastern California. Typical of vast areas of the interior ponderosa pine forest type, BMEF has two major age classes—a scattered overstory of 300- to 500-year-old pines and a dense understory of pines and white fir that originated about when intensive livestock grazing ended and when effective wild fire suppression began. An intermediate age class of 200-year-old trees is largely absent. Portions of BMEF had been harvested by a Methods-of-Cutting study between 1938 and 1947 (Dolph and others 1995), but much surrounding forest remained unlogged. At present an average acre contains 331 trees with a basal area of 144 ft^2.

Project Description

A major objective of the Blacks Mountain project is to determine if existing late seral attributes can be sustained and possibly enhanced by active management. Two forest structures, which we termed High Structural Diversity and Low Structural Diversity, were created with and without cattle grazing and prescribed fire on 12 units of 250 acres each. High Structural Diversity is characterized by the presence of many large, old trees, abundant snags, multiple canopy layers with dense clumps of smaller trees, and many small gaps in the canopy. Low Structural Diversity, created to provide an extreme contrast, is characterized by a single canopy layer of well-spaced pole- and small sawtimber-sized trees and few, large gaps in the canopy. Our objective was not to test the classic silvicultural systems of even-aged and uneven-aged management. Rather, the High Structural Diversity treatment was designed to improve the health and longevity of the large, old trees and create a multilayered forest, but not necessarily three or more age classes of trees. The Low Structural Diversity treatment was designed to provide the extreme contrast needed to determine if forest structure influenced biodiversity and sustainable productivity (table 2).

The entire forest is included in grazing allotments. Thus, six of the 12 units are fenced to exclude livestock. In each unit prescribed fire is introduced to one half and excluded from the other. We anticipate that a series of three burns, closely spaced, will be needed to bring fuels to a maintenance level. Each treatment (structural diversity x grazing) is replicated three times in a randomized block design with fire applied in split plots. Blocking is by proportion of white fir because the proportion varies among units. Fir becomes more common with increasing elevation and is a surrogate for subtle site differences. Although the design does not include untreated controls per se, four Research Natural Areas well distributed within the BMEF will be studied, also, to provide quantitative and qualitative information on undisturbed systems.

Table 2—Treatment contrasts in the Blacks Mountain Ecological Research Project in northeastern California.

Treatment contrasts	
High Structural Diversity	**Low Structural Diversity**
High vertical diversity—Retain/sustain all large old trees by removing competing trees.	Low vertical diversity—Remove all large old trees.
High horizontal diversity—Retain 10 percent to 20 percent in dense clumps. Only plant if openings >2 acres comprise >15 percent of unit.	Low horizontal diversity—Uniformly space at about 20 feet all saplings and poles. Plant all openings >2 acres.
Retain all snags and down logs.	Retain all snags and down logs.

Stand Structure Changes at BMEF

Because both projects required the removal of large volumes of wood over an extensive area—2,600 acres at GAMA and 3,000 acres at BMEF—timber harvest was spread over 3 years with prescribed fire applied the fall following timber harvest. All treatments have not been installed in either project but we do have some posttreatment results from one replication at BMEF.

The diameter distribution of the units in Block I at BMEF before treatment demonstrate the exaggerated reverse "J" shape—typical of interior pine stands developing without fire (fig. 3). Seedlings and saplings make up 61 percent of the stems. When the pole size classes are added, 96 percent of all trees in the stands have originated since fire exclusion. Although not shown in figure 3, much of the seedling, sapling and pole-size components are white fir. The pretreatment inventory at GAMA shows a similar but less exaggerated reverse "J" shape (fig. 4). At GAMA, a cooler and moister site, white fir is more abundant.

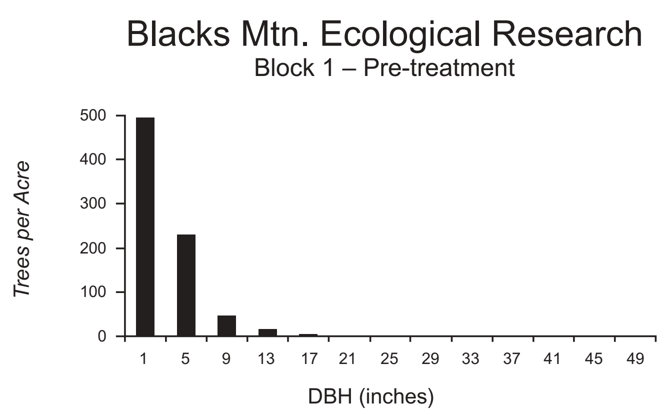

Figure 3—Distribution of tree diameters before installing treatments in Block I of the Blacks Mountain Ecological Research Project. Numbers of trees with diameters greater than 17 inches are too few to show on graph.

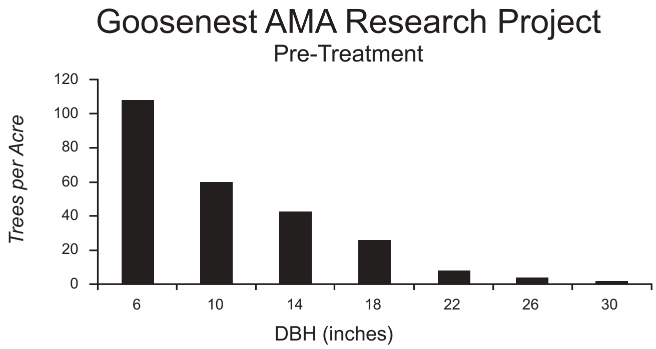

Figure 4—Distribution of tree diameters before installing treatments in the Goosenest Adaptive Management Area Research Project.

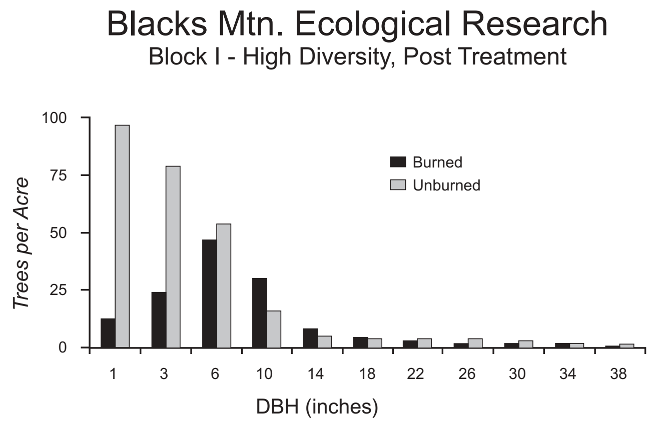

Figure 6—Distribution of tree diameters after installing the high structural diversity treatment in Block I of the Blacks Mountain Ecological Research Project.

In 1999, we measured the trees in BMEF Block I that had all treatments (tree harvest and prescribed fire) installed. The diameter distribution was altered dramatically by both the low structural diversity treatment and prescribed fire. Most of the seedlings, saplings, and small poles are gone, leaving a shape best described as a normal distribution skewed toward the larger diameters (fig. 5). Stand density as measured by basal area was reduced from 122 ft^2 per acre to 40 ft^2 per acre. In sharp contrast, the high structural diversity treatment without fire maintains the reverse "J" shape diameter distribution because more of the smaller size classes remain (fig. 6). Basal area reduction was much less in the high structural diversity treatment—from 122 ft^2 per acre to 90 ft^2 per acre. When prescribed fire was added to the high structural diversity treatment, a skewed normal distribution was formed because fire mortality was restricted to the smaller trees. Basal area per acre was little affected. A similar distribution was reported from reconstruction of historic stands in central Washington (Harrod and others 1999).

We are pleased to discover that the diameter distribution in this first block of the high structural diversity treatment with fire at BMEF has begun to resemble the shape of the diameter distribution of the stand in the Beaver Creek Pinery (fig. 2). It also is beginning to resemble the distribution in 1939 of the unharvested plot in the Methods-of-Cutting study (fig. 1). The major difference is that the modal values for the distributions at Beaver Creek Pinery and the 1939 unharvested plot are higher—16 and 29 inches d.b.h., respectively. The modal value for the diameter distribution of the high structural diversity with fire treatment is only 6 inches d.b.h.—a legacy of the dense understory originating after fire exclusion and the demise of many of the large old trees. With the program of planned periodic reentry of prescribed fire, this modal value should rise.

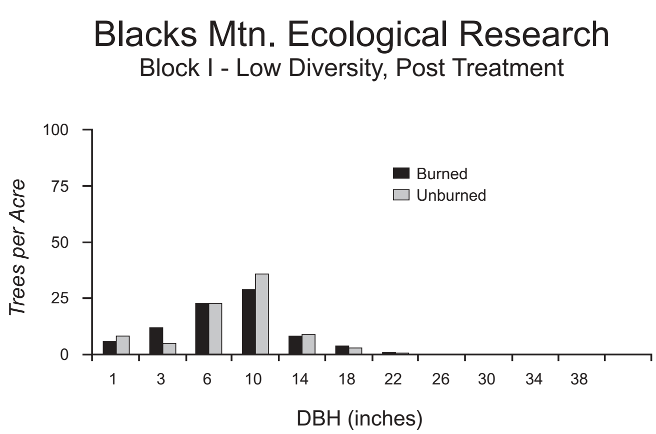

Figure 5—Distribution of tree diameters after installing the low structural diversity treatment in Block I of the Blacks Mountain Ecological Research Project.

Conclusions

At BMEF, because some large old trees are still present, we seem to be recreating a stand structure that resembles interior ponderosa pine stands of a century ago in which fire was an integral component of the ecosystem. Only one block of the replicated experiment has been treated so far, therefore, firm conclusions must remain tentative. At GAMA no large old trees are present. The present stand originated in the absence of fire after the late seral stand was logged off. Here we are testing pathways to achieving late seral attributes. Some pathways may be more effective in achieving late seral attributes and some may achieve them more rapidly than others. Early results from BMEF suggest that prescribed fire is a critical component. Maybe mechanical treatments can achieve similar effects as prescribed fire. We anxiously await results from the fire/fire surrogate study described elsewhere in this conference proceedings.

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Alternative Ponderosa Pine Restoration Treatments in the Western United States

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Phillip Weatherspoon
Carl Edminster

Abstract—Compared to presettlement times, many ponderosa pine forests of the United States are now more dense and have greater quantities of fuels. Widespread treatments are needed in these forests to restore ecological integrity and to reduce the risk of uncharacteristically severe fires. Among possible restorative treatments, however, the appropriate balance among cuttings, mechanical fuel treatments, and prescribed fire is often unclear. Resource managers need better information on the effects of alternative practices such as fire and mechanical/manual "fire surrogates." A group of scientists and land managers has designed an integrated national network of long-term research sites to address this need, with support from the U.S. Joint Fire Science Program. Seven of the 11 sites in the network are in ponderosa pine-dominated Western coniferous forests with low-severity natural fire regimes. The study will assess a wide range of ecological and economic consequences of four alternative restoration treatments: (1) cuttings and mechanical fuel treatments alone; (2) prescribed fire alone; (3) a combination of cuttings, mechanical fuel treatments, and prescribed fire; and (4) untreated controls. The study is long term, with treatments repeated over time. Each site will have at least three replications of the four treatments, applied to treatment units of at least 14 ha in size (including buffer). Where feasible, the replicated units will be supplemented by unreplicated large areas treated similarly to study larger scale ecological and operational issues. A comprehensive set of core variables will be measured at each site, including aspects of fire behavior and fuels, vegetation, wildlife, entomology, pathology, soils, and economics. The core design will allow interdisciplinary analysis at both the site and multisite scales. Investigators at each site will also have the freedom to add treatments and/or response variables to the core design as dictated by local interests, available resources, and expertise.

Introduction

Restoration has become necessary in many ponderosa pine forests of the Western United States. Current forests are denser, more spatially uniform, have more small trees and fewer large trees, and have greater quantities of forest fuels than did their presettlement counterparts (Bonnicksen and Stone 1982; Chang 1996; Parker 1984; Parsons and DeBenedetti 1979). Causes of these changes include fire suppression, past livestock grazing and timber harvests, and changes in climate (Arno and others 1997; Parsons and DeBenedetti 1979; Skinner and Chang 1996). These changes have caused a deterioration in forest ecosystem integrity, and an increased probability of large, high-severity wildfires (Dahms and Geils 1997; Patton-Mallory 1997; Stephens 1998; Weatherspoon and Skinner 1996). Reports from the Blue Mountains of Oregon and Washington (Everett 1993), the Columbia River Basin (Quigley and Cole 1997), and the Sierra Nevada Ecosystem Project (SNEP 1996; Weatherspoon and Skinner 1996) have highlighted these problems and have explained the need for large-scale and strategically located thinning (especially of small trees), fuel treatment, and use of prescribed fire. A recent speech by Interior Secretary Babbitt (2000) pointed out that similar problems and the need for similar solutions are now being acknowledged by national policymakers.

The need for widespread use of restorative management practices is clear (for example, Hardy and Arno 1996). Less clear, however, is the appropriate balance among cuttings, mechanical fuel treatments, and prescribed fire (SNEP 1996; Stephens 1998; van Wagtendonk 1996; Weatherspoon 1996). Economic and technical feasibility of various treatments, as well as social and political acceptability, are important considerations in managers' decisions about tools to use. To achieve goals for ecosystem integrity and sustainability, however, we also need better information about the ecological consequences and tradeoffs of alternative restoration practices. The frequent, low- to moderate-severity fires that characterized presettlement disturbance regimes in many of our ponderosa pine forests influenced not only forest structure, composition, and fuels, but also a wide range of other ecosystem components and processes (Agee 1993; Chang 1996). What components or processes are changed or lost, and with what effects, if "fire surrogates" such as cuttings and mechanical fuel treatments are used instead of fire, or in combination with fire? While there is considerable information on the costs and ecological effects of both prescribed fire and thinning treatments in Western forest ecosystems (for example, see Walsted and others 1990), no studies have directly compared these two methods in the same place and at the same time.

Long-term, interdisciplinary research is needed to quantify and compare the consequences and tradeoffs of alternative fire and fire surrogate treatments. Ecological and economic aspects must be included as integral components.

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The research should be experimental, rather than retrospective or correlative, to permit stronger inferences about cause-and-effect relationships. Through this research it will be possible to determine which ecosystem functions of fire can be emulated by other means, which may be irreplaceable, and how much restoration will cost society. Such an effort must be collaborative, involving land managers, researchers, and interested public.

A team of scientists and land managers has designed an integrated national network of long-term research sites to address this need, with support from the USDI/USDA Joint Fire Science Program (http://www.nifc.gov/joint_fire_sci/index.html). The steering committee (see Acknowledgments) and other participants in this national Fire-Fire Surrogate (FFS) study represent a number of Federal and state agencies, universities, and private entities, as well as a wide range of disciplines and geographic regions. The 5-year study, now funded by the Joint Fire Science Program, applies a common experimental design over 13 sites nationally, with each site representing a forest that is at risk of uncharacteristically severe wildfire (Weatherspoon 2000). This paper focuses on the work as applied to eight of the sites in the Western United States, all dominated by ponderosa pine (fig. 1).

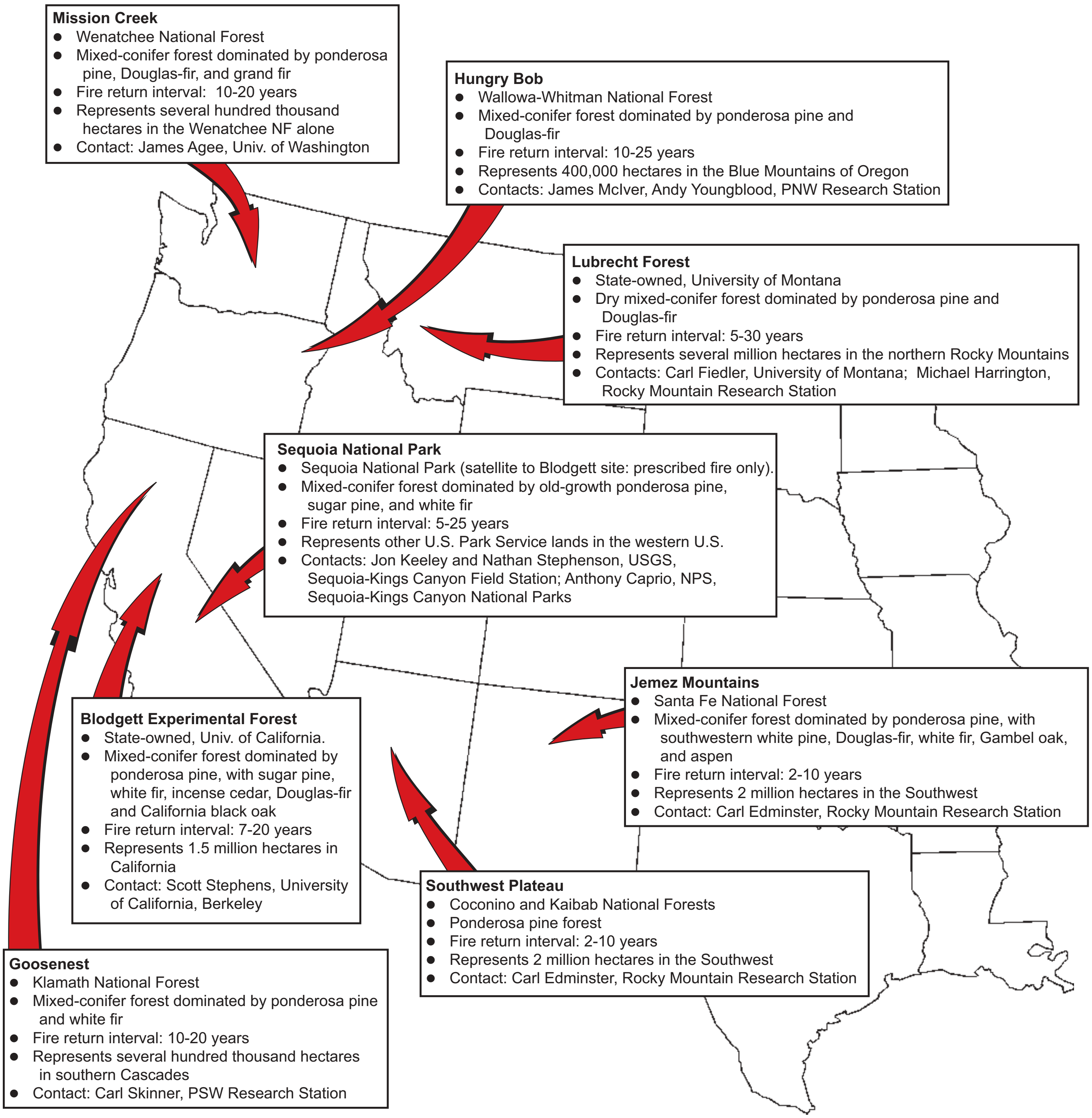

Figure 1—Ponderosa pine-dominated sites of the Fire-Fire Surrogate study.

Objectives

The goal of the study, as it applies to the Western sites, is to quantify the ecological and economic consequences of fire and fire surrogate treatments in ponderosa pine-dominated forests of the Western United States. The primary audience for the study is land managers, the people who make the decisions about which tools are most appropriate to use under different circumstances. Objectives include:

1. **Effects**: Quantify the effects of fire and fire surrogate treatments on a number of critical response variables including (a) fuel and fire behavior, (b) vegetation, (c) soils and forest floor, (d) wildlife, (e) entomology, (f) pathology, and (g) treatment costs and utilization/economics.
2. **Design**: Provide a research design that (a) establishes and maintains the study as an integrated national network of long-term interdisciplinary research sites using a common "core" design, (b) allows each site to be independent for purposes of statistical analysis and modeling, as well as being a component of the national network, and (c) provides flexibility for investigators of each research site to augment—without compromising—the core design to address locally important issues and to exploit expertise and other resources available to local sites.
3. **Models**: Develop and validate models of ecosystem structure and function, and refine recommendations for ecosystem management.
4. **Relationships**: Within the first 5 years of the study, establish cooperative relationships, identify and establish network research sites, collect baseline data, implement initial treatments, document treatment costs, report results, and designate FFS research sites as demonstration areas.
5. **Database**: Develop and maintain an integrated and spatially referenced database to be used to archive data for all network sites, and to allow interdisciplinary and meta-analyses.
6. **Monitoring**: Identify and field test a suite of response variables that are sensitive to the fire and fire surrogate treatments and are technically feasible for use in management contexts.

Research Approach

Experimental Design

The FFS project can best be described as an operational experiment, in which rigorous control is applied to a design meaningful to managers. Thus while the experiment has sufficient replication and control for each site to stand alone statistically, the treatments, variables, and scales have been chosen with the manager in mind. The treatments closely match the options available for managers, the variables chosen for study reflect those of greatest concern to managers, and the scale of the experimental units matches for the most part the sizes of management units typically designed by managers. In addition, the manner in which variables are measured at each site facilitates an integrated analysis of response for the range of variables, thereby providing the kind of information managers need to assess tradeoffs among the treatment options.

Treatments—Study treatments represent various combinations of the most common restoration activities used in forested ecosystems: cutting trees or other vegetation, using prescribed fire, and mechanically treating residues or scarifying the soil. Four study treatments include those that address widely shared concerns about forest health and wildfire hazard, those that deal with environmental concerns, and those most practical from an operational standpoint:

1. Untreated control
2. Prescribed fire only, with periodic reburns
3. Initial and periodic cutting, each time followed by mechanical fuel treatment and/or physical removal of residue
4. Initial and periodic cutting, each time followed by prescribed fire; fire alone also could be used one or more times between cutting intervals

These four treatments also span the range of restoration activities advocated by proponents of "structure restoration" (treatment 3), "process restoration" (treatment 2), or both (treatment 4) (Stephenson 1999).

Cuttings in treatments 3 and 4 will be repeated at intervals appropriate to the forest type and site conditions—for example, every 20 years. Periodic prescribed burns in treatments 2 and 4 will be based on available information about presettlement fire intervals for each research site. Irregular rather than fixed burn intervals are preferable where supported by fire history evidence, as it seems likely that important elements of ecosystem diversity were promoted historically by natural variability in fire intervals (Agee 1993; Skinner and Chang 1996).

We recognize that treatment specifications can encompass considerable variability in both cutting/mechanical and fire treatments that may differentially affect ecological responses of interest. While more precise specifications would reduce treatment variability among sites, such precision is neither feasible or desirable across so diverse an array of sites. The real world of forest ecosystems and resource management would not be well served by such a prescriptive approach. Flexibility in treatment specification does, however, increase the need for: (1) local replication to allow each research site to stand on its own statistically; (2) a specified desired future condition (DFC) for each site to help guide the application of treatments; and (3) careful documentation of treatments actually applied at each research site. We have defined a network-wide minimum standard short-term DFC for the study, based on stand resistance to wildfire:

> Each noncontrol treatment shall be designed to achieve stand and fuel conditions such that, if impacted by a head fire under 80th percentile weather conditions, at least 80 percent of the basal area of overstory (dominant and codominant) trees will survive. The definition of 80^{th} percentile weather conditions will be based on an analysis of fire season conditions, calculated for mid-afternoon, over a period of 10 to 20 years at the closest fire weather station. The prescription to implement the treatment will be developed based on fire behavior modeling (for example, FARSITE; Finney 1998) and predicted fire effects. Effects will be predicted using techniques such as FOFEM (First Order Fire Effects Model; Reinhardt and others 1997) and/or other modeling efforts that may include expert opinion.

This standard presumes the retention of a viable residual stand following treatment (clearcutting would not be an acceptable treatment option). The DFC will be well defined and implemented using a specific prescription to ensure consistency among treatment units. Each site DFC will consider management goals appropriate to that site, to stand conditions, and to the expectations of resource managers and other stakeholders. While early treatments may focus on thinning from below, or the equivalent using a series of burns, long-term restoration of historic stand structure will require provisions for recruitment of tree regeneration and development of a sustainable age-class distribution. Although fire hazard reduction will be a continuing emphasis for treatments, in the long run it is expected that stand structure will be increasingly able to accommodate wildfires that occur under the 80^{th} percentile weather conditions.

Assuming the same starting point of stand and fuel conditions, moving toward a given DFC using the fire-only treatment will clearly be much less precise than using cutting treatments. For example, some desired changes in stand structure—for example thinning relatively large trees with fire without doing damage to the overall stand—may not be feasible. However, use of innovative prescriptions, firing techniques, and other methods such as stage burning may, over successive burns, permit considerable progress toward most DFCs using prescribed fire alone.

Replication and Plot Size—So that each site can be analyzed independently, each treatment will be replicated at least three times per site, using either a completely randomized or randomized block design. The core set of four treatments will thus be represented in 12 treatment units at each of the eight ponderosa pine research sites.

Each of the 12 core treatment units at a research site will consist of a 10-ha measurement unit, within which core variables will be measured, surrounded by a 4-ha treated buffer. The 10-ha unit size is a compromise between advantages of smaller units (for example, reduced costs, reduced within-unit variability) and those of larger units (for example, the need to represent natural variability at an operational scale, and the need to accommodate some larger scale ecological responses). The buffer, treated in the same way as the measurement unit it surrounds, will have a width at least equal to the height of a best-site potential tree. A 30-m treated buffer, for example, would bring the total size of the treatment unit to about 14 ha. Site participants will need to determine appropriate separation of treatment units and the nature of treatment in the matrix between units.

We recognize that many aspects of wider ranging wildlife species, bark beetles, and some economic questions can be studied at the 10-ha scale only indirectly—for example, via habitat attributes and modeling methods. Where feasible at a given research site, two additional approaches may help to address larger scale issues: (1) Larger replicated treatment units (for example, larger buffers) can be used, provided that the core 10-ha units are embedded within them and are used for measurement of core response variables. Additional, larger scale variables could then be measured on the larger treatment units. (2) The core 10-ha replicated units can be augmented with much larger (200 to 400 ha or more), generally unreplicated areas nearby treated to the same specifications. These large treatment areas could provide useful information concerning operational-scale economics and practicability, as well as larger scale ecological responses, especially if linked to the smaller replicated units via appropriate models.

Response Variables—Ecosystem management requires an understanding of three interacting components: societal expectations and desires, management costs and revenues, and how management activities affect the ecology of whole systems. The social component will be linked to the study through other efforts funded by the Joint Fire Science Program and others. The FFS study is focused on economics and ecology and, because the study is directed toward management, is designed to provide information on how the whole system responds to treatment, such that managers can assess tradeoffs. Because core response variables will be measured at all network sites in a consistent way, we will be able to provide a package of information on how forest ecosystems of this kind respond to management. This is critical in a world where a number of issues are debated simultaneously for every parcel of land. For example, while fuel reduction may lower fire hazard and risk, removing down woody material will also reduce foraging habitat for birds and macroinvertebrate species. Measuring both the extent of fuel reduction and its effect on biodiversity may help identify thresholds that would be useful for fine tuning management to achieve more holistic objectives. In addition, measuring the costs and revenues of fuel reduction provides the kind of information that allows the manager to assess tradeoffs on the application of alternative management tools. Finally, applying this design to eight different sites will provide more robust information to guide management decisions on restoration of ponderosa pine forests.

Several members of our FFS steering committee (see Acknowledgments) have been serving as disciplinary group leaders with responsibility for developing major sets of response variables (table 1). Each group leader has worked with a team of people with appropriate expertise to identify a core set of response variables and measurement protocols to use at all research sites. Their activities also have included cross-group coordination to ensure consistency, compatibility, and nonduplication of data collection efforts. As

Table 1—Disciplinary groups and group leaders.

Fire and fuels	
	Sally Haase, PSW Station, and Bob Vihnanek, PNW Station
Vegetation	
	Jon Keeley, USGS, Sequoia-Kings Canyon National Parks
Soils and forest floor/hydrology	
	Ralph Boerner, Ohio State University
Wildlife	
	Steve Zack, Wildlife Conservation Society
Entomology (primarily bark beetles)	
	Patrick Shea, PSW Station
Tree pathology	
	Bill Otrosina, SO Station
Treatment costs and utilization/economics	
	Jamie Barbour, PNW Station

project implementation proceeds, they will work to ensure that data collection protocols are followed consistently at all the sites. This may include training, oversight of field crews, or other measures as appropriate.

Within-unit sampling of all variables will be keyed to a 50-m square grid of permanent sample points to be established and maintained within each measurement unit. Any number of grid points in a measurement unit may be used for a given variable depending on the nature and appropriate intensity of sampling for that variable. Referencing of all data to the grid, coupled with digital orthophotography, will facilitate spatial, interdisciplinary analysis.

Research Site Locations

Criteria for Site Selection—A network of research sites using a common experimental design has the potential for synergistic output exceeding what could be accomplished by a series of separate, uncoordinated studies. In selecting research sites we have developed and used the set of criteria given in table 2.

Proposed Initial Sites—The proposed initial network comprises 13 sites, each representing a forest with a historically short-interval, low- to moderate-severity fire regime. Eight sites are in Western coniferous forests, ranging from the Pacific Northwest to the Southwest (fig. 1). These sites all share ponderosa pine as an important tree component, but sites vary in composition of other conifers and differ substantially in topography and soil. We recognize that this network of pine sites does not represent all of the geographic localities of Western pine forests that are in need of restoration. However, its composition is a reasonable compromise considering the widespread need for the information, anticipated availability of funding, and available expertise and commitment. Furthermore, depending on the level of interest and support available, future sites at other localities may be added to the network.

Table 2—Criteria used in site selection.

1. Site is representative of forests with a historically short-interval, low- to moderate-severity fire regime and a currently high risk of uncharacteristically severe fire.
2. Site is representative of widespread forest conditions (site characteristics, forest type and structure, treatment history) that are in need of, and likely to benefit from, fire or fire surrogate treatments, and in which such treatments are feasible.
3. Site contributes significantly to balancing the overall network in terms of regional representation and/or land ownership type.
4. Partners and cooperators are committed to and capable of participating in the program. This involves several factors, including active support and interest in involvement on the part of partners/cooperators; available land base for the study; ability and willingness of land managers to implement the full suite of experimental treatments successfully within required timeframe, repeat treatments over time as appropriate, commit selected sites for long-term research uses, and document these commitments in amendments to long-term land management plans.
5. On Federal lands, treatment costs are borne by lead agency or partner.
6. Partnerships exist across agencies and with universities, and between researchers and managers.

Acknowledgments

We would like to acknowledge the Steering Committee, the group of scientists and managers who jointly conceived and developed the “Fire-Fire Surrogate” study. In addition to the three authors of the current paper, the Steering Committee consisted of the following individuals: James Agee (University of Washington), Phil Aune, Jim Baldwin, Matt Busse, Gary Fiddler, Sally Haase, Kathy Harksen, Bill Laudenslayer, Martin Ritchie, Patrick Shea, and Carl Skinner (PSW Station, USDA Forest Service), Sue Husari and Mike Landram (USDA Forest Service, Region 5), Frank Beall and Scott Stephens (University of California), Ron Hodgson (California State University), Jon Keeley and Nate Stephenson (Sequoia-Kings Canyon National Park), Jamie Barbour, John Lehmkuhl, Roger Ottmar, Bob Vihnanek, and Andy Youngblood (PNW Station, USDA Forest Service), Steve Zack (Wildlife Conservation Society), Carl Fiedler (University of Montana), Julio Betancourt (USGS), Mick Harrington, Kevin Ryan, Elaine Sutherland, and Dan Neary (Rocky Mountain Station, USDA Forest Service), Ralph Boerner (Ohio State University), Dan Yaussy (NE Station, USDA Forest Service), Bill Otrosina, Dale Wade, and Tom Waldrop (Southern Station, USDA Forest Service). We also wish to thank the USDI/USDA Joint Fire Science Program for its support throughout the life of this project.

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Upper South Platte Watershed Protection and Restoration Project

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Abstract—The Upper South Platte Basin is a critical watershed in Colorado. Nearly 80 percent of the water used by the 1.5 million Denver metropolitan residents comes from or is transmitted through this river drainage. The Colorado Unified Watershed Assessment identified the Upper South Platte River as a Category 1 watershed in need of restoration. Most of the river basin is located within the Pike National Forest southwest of the city of Denver. The South Platte River is also a major recreation area in Colorado and is highly regarded for its trout fishery.

The Upper South Platte Watershed Protection and Restoration Project was proposed in 1998 by Denver Water, the Colorado State Forest Service, Colorado State University, the Environmental Protection Agency, and the USDA Forest Service to respond to concerns about catastrophic disturbances in this watershed. The project is addressing these concerns by focusing on landscape vegetation patterns, soil erosion, and water quality within the Upper South Platte River Basin.

Introduction

The Upper South Platte Watershed Project was initiated in August 1998 to restore and protect the watershed. The project involves an interagency partnership between the USDA Forest Service, Colorado State Forest Service, Denver Water Board, and Environmental Protection Agency. These partners are concerned with continued soil and water problems from the Buffalo Creek Fire of 1996 and the potential for future fires to cause problems in other parts of the watershed. The partners intend to use watershed restoration as a guideline for management and project planning.

The Upper South Platte Protection and Restoration Project will reduce the potential for adverse effects to water quality, human life, and property by achieving the following goals:

- Reduce sediment, crown fires, and risks to property
- Create sustainable forest conditions in the Upper South Platte River Basin

The partners intend to improve water quality by reducing road and trail sediment, stabilizing stream channels, and reducing noxious weeds. Using prescribed fires, natural fires, mechanical vegetation treatments, and creating sustainable forest conditions will also reduce the risk of high intensity crown fires. Urban interface hazards will be reduced through educational programs and vegetation treatment on private lands. Sustainable forest conditions will be created by thinning stands, establishing openings, and maintaining snags and down logs. Research from the Cheesman historic forest landscape will guide the forest restoration activities.

The project will begin in three sixth-level watersheds. Restoration actions on public lands will focus in the Waterton/Deckers and Horse Creek watersheds. The USDA Forest Service, the Colorado Forest Service, and Denver Water will coordinate with other Federal and State agencies, local governments, and interested parties to plan, implement, and monitor restoration projects in the Upper South Platte River Basin. The partners involved in the Upper South Platte Project will implement new methods of doing business to protect watersheds that cross ownership or jurisdictional boundaries.

Land, Water, and Ecosystems

Characterization

The Upper South Platte Basin is a critical watershed in Colorado that encompasses about 1,000 square miles. Nearly 80 percent of the water used by the 2.5 million Denver metropolitan residents comes from or is transmitted through this river drainage. The Colorado Unified Watershed Assessment, as required in the Federal Clean Water Action Plan, identified the Upper South Platte as a Category 1 watershed in need of restoration. Most of the basin is within the Pike National Forest southwest of Denver, CO. The South Platte River is also a major recreation area in Colorado and is a highly regarded trout fishery.

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The Pike National Forest portion of the protected area comprises about 500,000 acres. Private land in holdings comprise about 100,000 acres; the State of Colorado owns about 2,000 acres within the project area and manages 15,725 acres of Denver Water lands. The Bureau of Land Management and the city of Aurora also manage lands within the project area.

The Upper South Platte Project can be divided into three major vegetation zones based on elevation. The montane zone ranges from 6,500 to 10,000 feet in elevation and is made up primarily of ponderosa pine, Douglas-fir, and, in the upper portions, lodgepole pine. About 450,000 acres are in the montane zone. The subalpine zone ranges from 10,000 to 12,000 feet in elevation and comprises primarily lodgepole pine, aspen, Engelmann spruce, and subalpine fir on about 150,000 acres. The alpine zone includes the areas above tree line and primarily comprises alpine meadows, shrub land, rock and pockets of bristlecone pine on about 50,000 acres. The montane forests were intensively harvested in the late 1800s and early 1900s to supply mining needs, railroad ties, and building materials. However, a 12-square-mile area of montane forest was not logged and provides valuable information for restoration guidelines.

Timber management today primarily involves the harvest of dead wood for firewood. Most stands are mature, with 80 percent of the forested area in densely stocked, late seral conditions. The principal wildlife species within the project area are mule deer, elk, Merriam's turkey, Abert's squirrel and Wilson's warbler. The principal fish species are rainbow trout, brown trout, cutthroat trout, brook trout, white sucker, long nose sucker, and long nose dace. Wildlife viewing, photography, hunting, and fishing are an important part of many recreation activities.

Recreation in the Upper South Platte Project area includes sight-seeing, picnicking, camping, hiking, mountain biking, motorcycle and ATV riding, cross-country skiing, boating, fishing, and hunting. There are two designated Wildernesses within the watershed, Lost Creek and Mount Evans. Developed recreation facilities along the river corridors include 20 campgrounds. In addition to the campgrounds on the South Platte River, there are several developed picnic areas and numerous trailheads and parking sites. Use of the South Platte River and the adjacent uplands has increased annually over the past decade. Recreation use was estimated at 1,650,000 visitor days in 1995. This increase is largely attributable to the rapid population growth in the Denver metropolitan area and surrounding counties. The population has been increasing about 2.5 percent annually or 40,000 people per year. In-migrating residents tend to be active and affluent, giving rise to an increase in demand for dispersed recreation activities.

Landscape Assessment

A recent Colorado Front Range Assessment identified large areas along the Colorado Front Range where current forest conditions and urban interface are not conducive with the natural disturbance processes. The current forested landscape does not reflect the historic disturbance regime and is not sustainable. Fire suppression in the 20th century has allowed smaller, thin-barked trees to reproduce. The Upper South Platte forests of today are denser than historic conditions, containing more small trees compared to the forests prior to 1900. These small trees serve as fire ladders, allowing ground fires to climb into the tree canopy and become crown fires. Wildfire severity and frequency have increased in recent years. These forest conditions, when combined with greater human encroachment into the forestlands, have dramatically increased the risk for loss of life and property from wildfires.

The Buffalo Creek Fire burned about 12,000 acres in 1996, destroying several homes and removing essential forest cover on highly erosive soils. Heavy rainfall and floods following the fire resulted in two fatalities and substantial erosion and sedimentation. Downstream reservoirs that supply water for the Denver metropolitan area were adversely affected. Denver Water spent nearly $1 million on water quality cleanup and dredging operations in their reservoirs. They estimate they will spend an additional $25 million on future cleanup, dredging, and water treatment modifications as a result of the Buffalo Creek Fire.

A landscape assessment was completed for the 645,000-acre Upper South Platte Watershed in September 1999. Key issues were identified and management recommendations were made to address catastrophic disturbances. The landscape assessment identified the dominant ecological processes and developed recommendations to restore and maintain the health of the Upper South Platte watershed.

Landscape Pattern of Vegetation—The structure, composition, and landscape pattern of vegetation have been altered from conditions found prior to the arrival of Euro-American people by the cumulative effect of human activities. Lowering the density of timber stands and creating more openings in ponderosa pine and Douglas-fir forests should reduce the risk of large-scale catastrophic fires, such as the Buffalo Creek Fire. Maintaining healthy forest stand conditions should also reduce the severity of other disturbances, including insect epidemics. Reducing existing fuel buildups by the use of prescribed fire and other treatments will reduce the threat of high-intensity wildfires and the associated risks of flooding, erosion, and downstream sedimentation.

Soil Development and Movement—Soil development and movement in the Upper South Platte Basin may be changed significantly by human influences on disturbance processes. Soil erosion hazard is correlated to road and trail density, vegetation, and drainage patterns, an especially severe problem in the highly erosive soils of the project area. Paved and unpaved roads and trails with inadequate maintenance, inadequate drainage, or improper engineering can lead to considerable erosion and increased sedimentation. Realigning or improving drainage and maintenance of existing roads and trails will reduce soil erosion and sedimentation, and improve road and trail safety. Closing and restoring unnecessary roads and trails will also reduce soil erosion and sedimentation.

Water Quality, Quantity, and Aquatic Habitats—Recent catastrophic fire and flood events have resulted in the movement of large amounts of sediment into the streams, harming water quality, aquatic habitat, and valuable municipal water systems. Water treatment plants had to be shut down, and the tap water had a bad odor and taste. Restoring the landscape vegetation to more sustainable conditions will reduce the potential for catastrophic events. Abandoned mine reclamation and drainage control will also have a positive impact on aquatic habitat.

Customers and the Public Benefit

The Upper South Platte River Basin is southwest of Denver, CO. The area includes portions of Park, Jefferson, Douglas, Teller, and Clear Creek Counties. Residential land use in the Upper South Platte Basin is primarily rural; recreation, mining, and agriculture form the economic base. The watershed is sparsely populated, with several small towns located near historic mining and recreation areas. Many of the small communities have a mixture of permanent and seasonal residents. The communities of Bailey (population 9,100) and Woodland Park (population 9,000) are the largest urban areas within the watershed. Other small communities include Shawnee, Pine, Trumbull, Oxyoke, Nighthawk, and Deckers. Many homes are located in unincorporated areas adjacent to the South Platte River and its tributaries.

Stakeholders

The stakeholders include local and county governments, fire departments, landowners, and the business and environmental communities. The Upper South Platte Watershed Protection Association is a stakeholder group that shares similar interests with the Upper South Platte Watershed Protection and Restoration Project. The project will seek to develop a partnership with the association.

The Upper South Platte Project will benefit watershed stakeholders in several ways. Reducing wildfire severity will also reduce the risk of large, disastrous fires and the resulting home and property damage in the urban interface. Many of the residents' livelihoods are dependent on the surrounding natural resources. Sustainable forest conditions would support continued recreation and employment opportunities in the natural resource-related jobs.

Customers

Customers include horseback riders, hikers, mountain bikers, motorcycle, ATV and four-wheel drive enthusiasts, campers, anglers, hunters, and guides, Denver Water consumers, and downstream irrigators. The Upper South Platte Basin supplies about 80 percent of the water needs of Denver and the surrounding communities (50 percent from the South Platte and 30 percent from the North Fork). The current demand on the Denver Water system averages 265,000 acre-feet per year. The supply is about 345,000 acre-feet per year. Water resource development interests have identified the Upper South Platte Basin as the most efficient supply with the least costly storage sites for the Denver metropolitan area's future water.

The Denver metropolitan area residents also benefit from the Upper South Platte Project in several ways. Reducing sediment transport will minimize impacts on water quality. Denver Water will save money on maintaining reservoir capacity and water treatment. Water bills will remain low for Denver Water customers, and they will continue to have quality drinking water. The Denver metropolitan area residents comprise the majority of the recreation users in the Upper South Platte River Basin. Sustainable forest conditions will permit continued high quality forest recreation opportunities.

The Public

As a result of the Upper South Platte Restoration Project, the forest will be less prone to catastrophic wildfire. This will save suppression costs on large wildfires and create more sustainable landscape conditions for soil, water, fish, wildlife, and recreation.

Controversies

Fire Risks

A report prepared by the Federal General Accounting Office in 1999 describes the seriousness and problems that now exist from the threat of catastrophic wildfires to forest resources and communities in the following statement:

> Uncontrollable wild fires should be seen as a failure of land management and public policy, not as an unpredictable act of nature. The size, intensity, destructiveness and cost of wildfires are no accident. It is an outcome of our attitudes and priorities. The fire situation will become worse rather than better unless there are changes in land management priority at all levels.

The early logging from 1870 to 1900, grazing of domestic livestock, and fire suppression effects on the Upper South Platte Watershed have resulted in conditions that differ markedly from pre-Euro-American settlement conditions, especially in the ponderosa pine, Douglas-fir forests. These forests have changed from a mosaic of patches with different aged trees and tree densities to a more uniform, dense forest. The fire regimes have changed over time from mixed severity fires to crown fires.

In the last 150 years, no extensive fires have occurred in the drainage. A low-intensity fire that occurred in the 1850s was extensive. A similar fire should have reoccurred in 60 years. These "natural" fires and tree recruitment periods following the fires resulted in considerable spatial and temporal heterogeneity. There are four components to this type of forest. The first component is forested patches with a distinct age cap. The second component includes patches of old-growth ponderosa pine with no evidence of past stand-replacing fires. In these patches, ponderosa pine and, often, Douglas-fir trees range widely in ages and states of health and decline. The third component is nonforested openings created by fire. The fourth component is the riparian system. These components proportions have changed significantly in the Upper South Platte Watershed, which increases the likelihood of a catastrophic fire.

Air Quality

Wildland fire is a major source of air pollutants that have the potential to create high concentrations of fine particulates. The Environmental Protection Agency 24-hour standard for these particulates with a diameter of less than 10 microns is 150 micrograms per cubic meter. Concentrations of 5,000 micrograms per cubic meter have been measured on some wildland fires.

The emissions vary significantly between flaming and smoldering combustion. Six to 10 times more particulates are produced by smoldering combustion compared to flaming combustion. Most small fuels are consumed by flaming combustion and have a relatively small emission factor. About 40 percent of the larger fuels, those materials 6 inches in diameter and larger, burn by smoldering combustion, which results in large emissions. By removing larger materials before the ignition, the potential for large amounts of smoke and larger fuels to smolder for long periods of time is reduced. This smoldering often occurs after ideal meteorological conditions have passed and an inversion has set in. We are working to develop markets for small diameter materials so they can be removed by methods other than prescribed fire.

Prescribed fires are scheduled during periods when meteorological conditions will prevent violating air quality standards. Of course, it is not possible to schedule a wild fire. Prescribed fire can be an excellent technique to prevent extreme emission from being generated from a wild fire. Agencies in Colorado are working hard to improve monitoring and predicting the impacts of fire emissions to the ambient air quality and visibility.

Water Quality

The Upper South Platte watershed is a valuable drainage system to Denver and surrounding communities. Of particular concern is the quality and quantity of water produced by this watershed. Shortly after the Buffalo Creek Fire, a strong thunderstorm occurred over the burned area. Flooding from this storm moved large amounts of sediment, destroyed homes and bridges, and decreased soil stability. Following the fire there were efforts to rehabilitate the burned area. Although some of the rehabilitation was successful, another flood caused two human fatalities and moved tremendous quantities of sediment. Much of the sediment settled in the Strontia Springs Reservoir, which supplies municipal water. Aquatic habitat was also damaged as a result of the fire and floods.

This project is now making a concentrated effort to protect and restore the landscape to a more sustainable condition to reduce the potential for future catastrophic events that would dramatically affect water quality and aquatic habitat. The intent of these efforts is to move watershed conditions and functions towards more sustainable conditions.

Erosion

Human activities have affected soil development and movement during the past 100 years, primarily by influencing disturbance processes. Alterations to natural disturbance processes may increase catastrophic events, such as a fire. Soil development and movement depend on several factors: climate, parent material, time, vegetation, and disturbance. Granite is the parent material of most of this watershed. Granite weathers to gruss, which is a coarse gravel to fine sand crystalline regolith. Gruss and the soils that develop on it are highly erodible when exposed to direct impacts of rain, overland flow, or removal of vegetation by fire. Consequently, soil development and soil movement may be much different spatially and temporally during the past 100 years than the 1,000 years before Euro-American influence. An imbalance of soil development processes that may have potential long-term detrimental effects to the health and vigor of the watershed has likely occurred.

Land uses, such as suburban and rural development, are activities that can increase the surface erosion and soil loss. For a period, these activities may expose detrimentally compacted, displaced or fragmented surface organic and mineral layers to erosion. The net effect of these conditions may leave the surface soil layers in an unstable or unprotected state that can erode and deposit in streams and reservoirs.

Road construction, home construction and recreation use are on the rise in the Upper South Platte watershed, fueled by population growth in Denver. The Buffalo Creek Fire burned homes and forest cover on about 12,000 acres of highly erosive soils. Fortunately, this fire did not burn in a heavily populated area. While the homes were quickly rebuilt, ecological recovery has been slow. Hillsides are still barren of woody vegetation, and soil erosion continues. The soils are being transported to a storage reservoir for Denver Water. Denver Water estimates it will take $25 million to dredge this reservoir.

Marketing

Marketing that involves the public and organizations has been designed to help develop a desired image of the watershed and the project. The objectives include:

- Providing timely and accurate information about the Upper South Platte Project to interested parties, media, public officials and others.
- Preparing and issuing news releases from the partners pertaining to the Upper South Platte Project in coordination with the Pike National Forest, Rocky Mountain Regional Office, Colorado State Forest Service and Denver Water.
- Informing the public and affected interests of the importance of healthy watersheds and the disturbances that can affect them.
- Emphasizing partner's commitment to implementing the Upper South Platte Project and completing activities on the ground.
- Offering opportunities for individuals and affected interests to contribute to the project—support, comments, ideas, assistance.
- Building and strengthening relationships with community leaders as well as interested individuals and organizations.
- Gaining recognition of and support for the project.

Partnerships

Partners in the Upper South Platte Watershed are those agencies and organizations contributing funds or services to the Restoration Project. The Upper South Platte Restoration Project is coordinating with and seeking the involvement of stakeholders, customers, and the public. The project is also developing additional stakeholder partners. The partners are concerned with water quality issues and fire risks within the Upper South Platte River Basin.

The partners agree to use watershed restoration as a guide for management and project planning within the Upper South Platte River Basin. The partners agree to the following collaboration principles: no one is the center of a network; keep commitments; communicate in a candid and tactful manner; honor each others' interests and contributions; and keep shared work products visible.

Rocky Mountain Region and Pike National Forest

The USDA Forest Service manages about 500,000 acres of the Pike National Forest within the Upper South Platte Basin. National forest management occurs within a framework set by Federal laws and regulations. The agency's ultimate responsibility is to manage National Forest lands for multiple benefits on a sustainable basis. The USDA Forest Service operates within the annual budgets appropriated by Congress. Individual projects are planned with public input using the National Environmental Policy Act (NEPA) process. The Federal budget, acquisition, and planning processes result in the USDA Forest Service moving slower than the other partners.

The Rocky Mountain Region and the Pike National Forest entered into the partnership to facilitate meeting landscape objectives. Although the agency manages nearly 80 percent of the lands within the Upper South Platte River Basin, it cannot hope to achieve the landscape objectives without partners. The partners will provide resources to improve conditions adjacent to the largest streams and in the urban interface where private lands predominate. The partnerships provide a means to increase efficiencies in planning and implementing projects on a landscape basis. The partners provide a collaboration to leverage funds to achieve shared objectives.

The management branch of the USDA Forest Service has a three-person team assigned full time to the Upper South Platte Project. The team is involved in all aspects of the project, including public involvement, partner recruitment, restoration projects, and coordination with other Forest Service projects and programs. The team has identified numerous specific restoration projects to improve terrestrial and aquatic conditions. The planning for these projects began in 2000. The majority of the projects described in the Business Plan will occur on the Pike National Forest.

Rocky Mountain Research Station

The research branch of the USDA Forest Service focuses on academic issues in forest management. Peer-reviewed research provides tools and context for National Forest management. The research branch is relatively independent of the land management branch to minimize any scientific bias. The Rocky Mountain Research Station entered the partnership to assure the relevance of research to land management by formalizing the feedback loop to the knowledge base. Monitoring by the research community assures research conclusions can be tested on a landscape basis, and management practices can be adapted accordingly.

The Upper South Platte Project is based on science. It is not a pilot or test. It will rely heavily on research being conducted at Cheesman Lake, an intact historical landscape that can serve as a model for restoration activities in the lower montane zone for the Colorado Front Range.

A great deal is known about the natural disturbance history of this historical landscape and about the structure of the landscape components. Management actions will be based on this knowledge without compromising options. Research during the first years of the project will improve knowledge about the overall landscape and the structure and processes regulating it. New information can be incorporated into project planning and implementation. However, technical limitations may prevent new research information being implemented in a timely way.

The Forest Service research branch will conduct the research for the restoration activities, at an annual cost to the project of $175,000. We are using the Forest Vegetation Simulator and Stand Visualization System with existing data. A GIS layer of forest structure at the landscape scale has been developed for the historical Cheesman Lake landscape. Fire behavior is being evaluated for several landscape scenarios using the FARSITE model. Additional research for fiscal year 2000 included: (1) developing an integrated landscape Historical Range of Variability for the historical landscape; (2) testing this Historic Range of Variability elsewhere in South Platte Basin; and (3) preparing three-dimensional visualizations of various landscape scenarios.

After the first year, subsequent research will focus on tightening the description of the historical landscape and natural variation in the processes affecting landscape patterns, with the overall goal of extending results to the larger montane zone of the Front Range. This will include refining restoration scenarios for the project landscape and evaluating crown fire potential and water balance, comparing the project area and historical landscape. Subsequent research also will assess pretreatment and posttreatment under story plant diversity in the project area, including noxious weeds.

Colorado State Forest Service

The mission of the Colorado State Forest Service is to achieve stewardship of Colorado's environment through forestry outreach and service. The mission includes protecting natural resources from damaging elements and increasing public understanding of forestry's role and value in a healthy environment. The State owns about 2,000 acres within the Upper South Platte River Basin. The Colorado State Forest Service has a contract to manage Denver Water lands in addition to the State lands. The State will work closely with private landowners to reduce the fire risk in the urban/forest interface. The partnership provides the State with demonstration areas for other landowners on Colorado's Front Range. The partnership also provides a mechanism to leverage funds and improve communication with the public.

Denver Water

Denver Water owns 15,725 acres within the Upper South Platte River Basin. The forest management is under contract to the Colorado State Forest Service. They manage dams and reservoirs within the basin, which provide 80 percent of the water used by Denver metropolitan residents. Strontia Springs Reservoir was adversely affected by sediment following the Buffalo Creek Fire. Denver Water wants to reduce the risk of future events like the Buffalo Creek Fire by proactively managing its lands and the public lands within the basin. Denver Water can communicate the partners' objectives to nearly one million residential water customers.

USDI Geological Survey

The U.S. Geological Survey maintains stream gauges and monitors water quality across the United States. The U.S. Geological Survey collected water quality and soil erosion data in the Upper South Platte River Basin following the Buffalo Creek Fire. The U.S. Geological Survey has GIS data available for the landscape. Their monitoring experience has resulted in well-established monitoring protocols for soil and water parameters. They will help develop and implement the monitoring plan.

USDA Natural Resource Conservation Service

The Natural Resource Conservation Service provides soil and conservation technical assistance to private landowners. They provide an additional avenue of public outreach and have a close working relationship with the local Soil Conservation Districts (local officials appointed by county commissioners). The Natural Resource Conservation Service may be a source of potential cost-share funding for private landowners. They have soil inventories for the area and can provide water quality testing.

Trout Unlimited

The Trout Unlimited Cutthroat Chapter is concerned about road- and trail-related sediment that is adversely affecting fish habitat in the South Platte River. They are interested in identifying potential restoration projects to reduce sediment and can provide volunteers to help complete the work. Trout Unlimited has expressed an interest in reconstructing the Gill Trail.

Elk Creek Fire Protection District

The Elk Creek Fire Protection District provides fire protection in the urban/forest interface. The district is interested in creating defensible space to fight forest fires before homes become engulfed in flames. The district will work with the Colorado State Forest Service to raise public awareness and educate homeowners on how to create defensible space.

Landscape Restoration

Mechanical Vegetation Treatment

The Landscape Assessment identified the Cheesman, Trout Creek, and Waterton/Deckers and Horse Creek watersheds as high priority for forest vegetation and wildlife habitat restoration. The ponderosa pine/Douglas-fir forests are at high risk of catastrophic fire because of dense, even-aged, closed-crown forest conditions. These forests have very little down wood to permit low intensity ground fires. Under extreme fire conditions (hot, dry, and windy), fire will carry as a high intensity crown fire. Mechanical treatment is needed to reduce the canopy density and create openings. The objective is to reduce canopy density to 30 percent or less on up to 80 percent of the ponderosa pine/Douglas-fir landscape. Openings of 1–40 acres will be created on up to 25 percent of this landscape. The mechanical vegetation treatments will include commercial timber sales, stewardship/service contracts, noncommercial thinning, and chipping or shredding to masticate the trees on site. Prescribed fire will be used in conjunction with mechanical vegetation treatments.

About 2,000 acres will have mechanical vegetation treatment on an annual basis. The operational costs vary considerably based on the method used. Monitoring plan that includes vegetation plots and landscape mosaics will measure accomplishments.

Generally, lands with existing road access and slopes less than 35 percent may use commercial timber sales to meet the vegetation objectives. The timber value in the basin is relatively low; therefore, some flat areas with roads may require stewardship/service contracts to meet the vegetation objectives. Noncommercial hand felling, shredding, or chipping will be used in steep or limited access areas.

Most of the mechanical vegetation treatments will occur on National Forest lands managed by the USDA Forest Service. The Colorado Forest Service will manage State and Denver Water lands and provide assistance to private landowners. Costs for mechanical vegetation treatment vary from commercial value for some timber sales to several hundred dollars per acre for hand falling. An average cost of $137.50 per acre is planned for mechanical treatment. The actual cost could be considerably different if the treatment mix differs significantly from the assumed 50–75 percent in commercial removal.

Reforestation

A portion of the Buffalo Creek Fire area requires reforestation to provide vegetation diversity. The USDA Forest Service will plant about 1,000 acres with ponderosa pine widely spaced. Standard reforestation survival and growth protocol will be used to measure accomplishments. The seed inventory is currently insufficient to provide an adequate number of seedlings. Seeds will be sowed in 2000 to plant 100 acres of container seedlings in 2001. Cones will be collected in 2000–2002 to replenish the seed inventory. Additional seed will be sowed in 2001–2003 to plant 300 acres per year the following years. Reforestation costs are expected to be $500 per acre.

Noxious Weeds

Leafy spurge, diffuse knapweed, yellow and Dalmatian toadflax, and Canada and musk thistles are noxious weeds along 25 miles of the South Platte River. These noxious weeds are less palatable to wildlife, are less effective in stabilizing soil, and often out compete native vegetation. The goal is to reduce the infested acres. About 200 acres will be treated annually using chemical, biological, mechanical, and manual methods. Accomplishments will be measured by surveying the river corridor annually to determine if the infestation zone is shrinking, remaining constant, or growing.

The USDA Forest Service, Colorado Division of Wildlife, Colorado Department of Agriculture, and county weed boards are working with private landowners and volunteer groups to manage the noxious weed problem. County weed management departments will assist in developing integrated weed management plans for all land ownerships. The National Fish and Wildlife Foundation and the USDA Forest Service have provided grants for noxious weed treatments in previous years. A contribution of $20,000 annually will be used to leverage an additional $40,000 in grants from partners.

A noxious weed prevention strategy will require treating an additional 200 acres annually prior to mechanical and prescribed fire treatments. The additional 200 acres will require $40,000, for a total of $60,000 funds annually.

Roads

Many roads are poorly located and poorly maintained. Roads are the major source of anthropogenic erosion and sedimentation. The Pike National Forest plans to inventory its roads and update the information during the next 3 years. The project will accelerate the inventory within the basin and supplement the collected information to include site-specific erosion and sedimentation concerns. The updated information will be used to assess and prioritize roads for maintenance, closure, and obliteration. The cost in fiscal year 2000 cost will be $55,000.

Road maintenance in the basin costs about $100,000 annually. Currently, several roads not normally maintained are in obvious need of maintenance or need more effective closure devices installed. In fiscal year 2000, $45,000 will be used to place effective water bars or closure devices on 100 miles of priority roads known to be contributing high amounts of sediment. The road assessment is anticipated to identify $120,000 of road maintenance, $73,000 of road reconstruction improvements from funds, and $57,000 of road obliteration annually in fiscal years 2002–2005. Accomplishments will be measured by visual inspection to assure best management practices are implemented and effective. The monitoring plan identifies the protocols to evaluate if roads are affecting water quality.

Trails

The goal of the following trails projects is to create a safe, sustainable trail system in the South Platte watershed to meet the needs of hikers and other trail users while minimizing environmental impacts.

Cheesman Canyon is one of Colorado's treasures. The Gill Trail travels through the Cheesman Canyon and accesses a nationally known fishery along the Platte River. The trail is used by many hikers interested in seeing one of Colorado's major canyons absent roads, railroads, and residential development. Views of the historic Cheesman Dam and rugged canyon scenery also attract trail users. The Gill Trail was constructed about 40 years ago, stopping short of Cheesman Reservoir. No major trail improvements have occurred since then. An estimated 25,000 visitors per year use the trail and their effects are clearly evident. Crumbling side-slope trails have caused numerous braided routes and excessive erosion. Many social trails have been created to try to access the South Platte River, and some sections of this route are unsafe. The excessive and braided trails also cut through habitat used by the Federally listed Pawnee montane skipper, killing the plants on which they depend.

The USDA Forest Service, the National Park Service, consultants, and partners have prepared preliminary trail plans that address the issues described above. Key partners in this project include the Cutthroat Chapter of Trout Unlimited, Denver Water, and the USDA Forest Service. The final trail plan will be refined and evaluated under the National Environmental Policy Act planning process by the end of fiscal year 2000. Trail and restoration work will include safe rerouting and repair of existing trail, building a new safe sustainable trail from the original alignment to Cheesman Reservoir, expanding parking areas, eliminating braided trails, rehabilitating damaged side slopes, and restoring native vegetation that can be used by the skipper. All work will be completed by fiscal year 2004. Expected project benefits include improved safety, hiking experience, and visual quality along the trail, restored skipper habitat, and reduced sediment input from eroding trails. Local economic benefits may result from expenditures for goods and services by anglers and hikers attracted to the improved South Platte River access.

The estimated total cost for this project is $400,000 over the next 5 years. The trail design, construction, and maintenance portion is estimated at about $355,000, restoration about $30,000, and monitoring at $15,000. Besides the USDA Forest Service's contribution, Trout Unlimited and Denver Water will contribute about $126,000 (32 percent of the project total cost). Trout Unlimited and the USDA Forest Service have also applied for grants totaling $145,000 (36 percent of the project total) from the Colorado State Trails Program and Fishing is Fun. This project will make extensive use of volunteers from Trout Unlimited and Volunteers for Colorado Outdoors to perform trail work.

Prescribed Fire

Fire has been suppressed in the Waterton-Deckers and Horse Creek watersheds for more than 100 years. Therefore, the natural fuels have been building up over time and have the potential for large catastrophic stand- replacing fires. These stands are generally dense, even-aged, with closed crown conditions. Extreme fire conditions (low humidity, low fuel moistures, high temperatures and wind) allow fire ignitions to result in high-intensity crown fire. The objective of this project is to treat 2,000 acres annually with prescribed fire to reduce natural and activity fuels and, where possible, create openings in the ponderosa pine-Douglas-fir stands. The areas to be treated will be companion areas to those being treated by mechanical methods.

The operational costs should not vary from costs experienced over the past 5 years in the prescribed fire program. Accomplishments will be measured based on the monitoring plan. Prescribed fire can be used to treat lands with little or no access and slopes greater than 35 percent. The slopes greater than 35 percent are on the upper limit for mechanical treatment. There may be a need to return to the stands treated by prescribed fire to supplement the prescription by hand felling trees not killed by fire to enlarge openings. Mechanical treatments will generally be followed by prescribed fire on lands managed by the Colorado State Forest Service and the USDA Forest Service.

Monitoring

Monitoring will serve as the quality control aspect of this restoration project. Data collection and observations of activities will provide a basis for evaluation of the overall restoration project. Data collection and analysis will follow established scientific procedures. Data collected will be analyzed and evaluated. Monitoring will be designed to determine if activities are being carried out in compliance with the project plan and existing Forest Plan. Objectives, long-term relationships, and the ability of the project to adapt to the research findings from the Cheesman research project will be evaluated. Also, the effectiveness of management activities in moving the vegetation and water quality toward desired conditions and in reducing the threat of catastrophic wild fires and their associated effects on soil, water and the human environment will be evaluated.

Monitoring will be completed annually and it will be based on the *Upper South Platte Watershed Protection and Restoration Project Monitoring Strategy* developed by a subcommittee of the Steering Committee. The monitoring will determine whether project activities are meeting the goals of reducing fuel hazards and associated catastrophic fire risk while maintaining soil productivity, minimizing erosion, and improving water quality. The monitoring strategy is designed to be dynamic and will be changed as new information becomes available. Monitoring stations will be established in the upper portion of the watershed, as well as in the lower portion of the watershed. Activity-level monitoring will be determined during project planning. Where possible, the Denver Water lab will be used for water analysis. The USDA Forest Service will provide data storage, and information will be shared with all partners on a regular basis.

The USDA Forest Service will take the lead in funding the monitoring activities and will work with partners to secure additional funding. The USDA Forest Service will work with partners to assign personnel to conduct the monitoring activities. A monitoring report will be done annually and presented to the partners and Steering Committee. The Middle East Regional Cooperation Program will contribute some data to the monitoring process. This project may also contribute some funds toward the overall project but this remains to be seen.

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Social and Cultural

Problem Solving or Social Change? The Applegate and Grand Canyon Forest Partnerships

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Abstract—Natural resource conflicts have resulted in attempts at better collaboration between public and private sectors. The resulting partnerships approach collaboration either by problem solving through better information and management, or by requiring substantial social change. The Applegate Partnership in Oregon and the Grand Canyon Forest Partnership in Arizona illustrate each approach. These approaches show the formative influences that shape the evolution and activities of a partnership, and show the need for multistakeholder participation.

Introduction

Over the past decade, numerous communities, agencies, and interest groups in the American West have turned to collaboration to manage natural resource conflict and develop and implement plans for managing, preserving, and restoring the landscape. These collaborations take many different forms because they emerge in response to local political, economic, and ecological circumstances. Yet all must respond to a common challenges: complex problems, diverse and often conflicting interests, fluid and often shifting relationships of power and authority, and the need to develop local capacity and resources.

These collaborative initiatives, often referred to as "partnerships," have attracted increasing attention as communities, agencies, and interest groups create, join, or actively resist these multistakeholder initiatives. A growing body of literature has attempted to comprehend and characterize these groups (Brick and others 2000; Cestero 1999; Conley 2000; Kenny 1999, 2000; Moore and Koontz 2000; Moseley 1999; Sturtevant and Lange 1995; Weber and Herzog 2000). Our assertion is that many of the efforts to explain partnerships and collaboratives overlook a fundamental distinction among groups that explains variations in group character, configuration, focus, and success.

We argue that a partnership chooses between at least two fundamentally different ways of seeing and defining its problem. One approach views the challenge as a quasi-technical problem requiring better information, technical expertise, organizational efficiency, and public education. The other views the situation as requiring substantial social change—the reorganization and redistribution of the decisionmaking authority, responsibility, and resources, and the allocation of costs and benefits associated with land management/stewardship.

We draw on two partnership initiatives—the Applegate Partnership of southwestern Oregon, and the Grand Canyon Forest Partnership of northern Arizona—to illustrate this distinction. Using these two groups, we further suggest that a group's choice between these two orientations depends on the characteristics of the founding group—those who initiate the partnership, define the problem to be addressed, and recruit the rest of the participants. We assert that the disposition of this founder group will largely determine a group's choice of strategies. Therefore, it is essential to understand what factors determine who assumes the founding role, shaping both the partnership, its perception of the problem, and the strategies it will create to address that problem.

Method

In this paper, we compare the Applegate Partnership in southwestern Oregon and the Grand Canyon Forest Partnership in northern Arizona. The Applegate Partnership came together in 1992 to address forest conflict and create ecosystem-based management in the Applegate Watershed. Founders created the Grand Canyon Forest Partnership in 1996 to reach agreement about how to reduce fire hazard and restore the forests surrounding the City of Flagstaff. The Grand Canyon Forest Partnership and the Applegate Partnership offer fertile grounds for comparison, in part, because they share similar ecological challenges and yet developed markedly different forms to address them. Both the Applegate Valley and northern Arizona are naturally fire-dominated ecosystems with the federal land management agencies controlling a large percentage of the land. In both regions, extensive timber extraction, grazing, and fire suppression had led to dense stands of small trees. Many participants in both groups were worried that these forest conditions would lead to large-scale, disruptive change especially wildfire, species loss, and habitat degradation.

The material for this paper is drawn from the authors' participant observation in these two groups. Brett KenCairn was a founding Applegate Board member and was active with the Partnership from 1992 to 1997. KenCairn was involved in the Grand Canyon Forest Partnership in 1998 and 1999. Cassandra Moseley was a board member of the Applegate Partnership and also conducted extensive interviews and participant observation in the Applegate Valley between 1995 and 1998.

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Formative Influence of Local Context

Kingdon (1984) argues that political change occurs when a “window of opportunity” opens that creates the political space for a new policy or idea to be introduced into politics (see also Moseley 1999). These windows open for a variety of reasons, ranging from regular events such as elections to extraordinary political crises. In both of our case studies, political crises created the opportunity to realign institutional arrangements. In the Applegate, the injunction that halted timber harvest in the territory of the northern spotted owl created the window of opportunity. The anger and administrative chaos that the injunction created allowed a community leader to pull together people to talk about new forms of land management.

In northern Arizona, disputes over federal forest management coupled with dramatic wildfires that threatened Flagstaff created a sense of crisis. This crisis pushed natural resource agencies, elected officials, and other community leaders to search for common ground that would allow federal managers to implement new forms of forest management.

Despite similarities of political crises, the partnerships that emerged were markedly different. In Flagstaff, the institutional dynamics of participant agencies dominated the Partnership. In the Applegate, a hybrid community/institution-based group emerged that eventually evolved into a community-based group.

Participant Worldview and Perceptions of “the Problem”

The Grand Canyon Forest Partnership (GCFP) was founded as a collaboration of government agencies and nongovernmental organizations to reduce fire hazard and improve forest health on the 100,000 acres of public lands surrounding Flagstaff. GCFP founders believed that legal and political conflict and disagreement over federal forest management were preventing the formulation and support for a new approach to active forest management. Participant groups were selected based on their technical expertise or their role as opinion leaders in the region. Partnership founders believed that they could get the public to trust their efforts if they recruited a diverse collection of leading organizations to support the work of forest restoration specialists. The group focused on convening experts to develop a solution and creating public support for the proposed solution.

When faced with a similar if more extreme political crisis, Applegate leaders founded a very different partnership. The Applegate Partnership founders wanted to develop a comprehensive management strategy at a watershed (500,000 acre) scale. New management strategies would be “ecologically sound, economically viable, and socially acceptable.” (Unattributed quotes come from anonymous interviews of Applegate Partnership participants.) At the outset, Applegate leaders choose to address the ecological problems they faced at a larger scale—ecologically and socially—than-the effort in Flagstaff. But more importantly, Applegate leaders viewed the central causes of the problem they faced—and how they could be best addressed—in a much different way. The Applegate founders believed that the model in which government agencies developed plans and then put them out for public review was not working. This process left out too many people until it was too late in the process, which created a narrow problem definition, and offered too few sources of ideas for solving complex problems. Applegate leaders saw problems as being so complex that no one person or set of people with particular perspectives or interests could hope to solve them.

Why were these groups so different and what have been the effects? We argue that the difference in group form and problem definition was caused by the different political context, especially institutional arrangements. These different visions affected partnership organization, accomplishments, and how the groups responded to outside challenges.

Political Context and Locus of Authority

Although Kingdon (1984) points us toward political crisis as an opportunity for change, he offers us little guidance for understanding which ideas and changes will flow through the window and who will push them. Historical institutionalists encourage us to look at institutional form and “thickness” to understand what shapes change (*sensu* Skowronek 1997). Skocpol (1979), Orren and Skowronek (1998 to 1999), and Skowronek (1982) argue that past institutional development will shape the form of change in the present and future. Following their lead, we argue that, although there is no guarantee that partnership groups will choose social change over pragmatism, this is more likely to occur when private citizens are able to play a major role in the group’s founding and problem definition than when leadership is primarily representatives of pre-existing institutions, especially government agencies.

The Grand Canyon Forest Partnership came together in an institutionally thick environment. Flagstaff and its surrounding forests were at the center of the jurisdictional interests of numerous elected officials, local, and state government agencies. Flagstaff is the county seat of Coconino County and hosts most of the county government and services. It is also the location of the Coconino National Forest Supervisor’s Office and two of the four Ranger District offices. Flagstaff maintains a full-time wildfire specialist and coordinates multiagency wildfire responses. Flagstaff is also an administrative center for a variety of other organizations that were to become important members of the Grand Canyon Forest Partnership including Arizona Game and Fish, the Arizona State Land Department, the U.S. Fish and Wildlife Service, and the Nature Conservancy. In addition, several regional environmental groups and chapters of national groups are headquartered in Flagstaff including the Grand Canyon Trust, the Sierra Club, and the Southwest Forest Alliance (a consortium of more than 90 environmental advocacy groups).

The consequence of this institutional thickness was the dominance of institutions in the formation of this collaborative. The thickness privileged institutional representatives over private citizens. This difference between citizens and institutional representatives matters because of the ways that these representatives are tied to their institutional mandates and cultures. Our argument is not that citizen innovation was impossible but rather that it is difficult and unlikely in these circumstances without extraordinary energy on the part of citizens.

In Flagstaff, leadership from government agencies did not divorce itself from their organizational perspective. Consequently, rather than seeking social change, the leadership sought to solve technical problems. What happened in Flagstaff reflects more generally what we might expect in an environment thick with bureaucratic institutions. People working in particular organizations bring with them the mandates of these agencies and are seeking to solve the problems they face.

This tendency is heightened because many natural resource agencies were formed with Progressive Era ideology in which problems were seen as primarily technical and experts were the best qualified to solve those problems (Hays 1959, 1991; Hirt 1994; Kaufman 1960; Moseley 1999; Thomas 1999; Luker 1984; Ross 1991). Technically trained experts could manage resources efficiently for the greater common good (Pinchot 1990). Through the application of scientific principles, nature could be simplified and ordered to bring it under human control (Scott 1998). By implication, citizens were either too self-interested or too ignorant to participate. Even when people in natural resource agencies take a broader view, formal and informal institutional processes have already been shaped using a Progressive Era model. Thus when faced with a crisis, or a failure of the old order, these agencies seek change but still find themselves tied to older institutional processes (Moseley 1999; Orren and Skowronek 1998–1999; Skocpol 1979; Skowronek 1982). Exceptional leadership, complete breakdown of normal institutional politics, or intensive outside pressure, we argue, would be required to shift these politics.

The jurisdictional conditions and social development could not have been more different in the Applegate Valley. There are no incorporated communities in the Valley and, thus, no city government. A county line divides the Valley in half, and thus no county-level elected officials see the Valley as their territory. No state or local agencies locate their offices in the Valley. The major exceptions to this are the federal land management agencies. The Forest Service has a Ranger Station in the Valley and major portions of two BLM resource areas are located in the Applegate Valley. With the exception of the Applegate Ranger District, all government offices are 30 to 45 minutes away.

In addition to the relatively weak organizational bases of local and state government, few staffed nongovernmental organizations focused on the Applegate. Headwaters, the longest standing regional environmental organization in southern Oregon, was active in the Applegate but was housed in Ashland, a town outside the Applegate. Similarly, the Sierra Club and other national organizations had active members living in the Applegate but local chapters were located in towns outside the Valley and were servicing an area several times the size of the Applegate.

Instead of thick formal bureaucratic and political institutions, the Applegate Valley had a long history of small-scale social and political organizations (Moseley 1999; Preister 1993; Sturtevant and Lange 1995). Particularly important for natural resources were a number of small-scale, volunteer environmental groups that had monitored federal timber sales during the 1970s and 1980s. They opposed federal actions by mobilizing networks of residents to attend public hearings, write letters, and submit appeals. These small, volunteer groups developed a cadre of community residents who understood the basics of forest management and Forest Service and BLM administration.

These community residents, moreover, were not stereotypical rural community residents but rather people who had moved to rural areas in the 1970s and 1980s to escape traditional suburban life. (For a description of "back-to-the-landers" see Jacob [1997].) Social change had long been a part of their personal ideology. Several of the core group of Applegate Partnership founders had long been acting upon their personal social change agendas by protesting federal timber sales. Through these protests they came to see the limits of old political processes and developed ideas for alternative approaches to citizen involvement.

Instead of numerous government agencies and sophisticated nongovernmental organizations facing the limits of old management strategies first, it was community residents and the federal land management agencies. Instead of placing leadership from formal organizations at the center of change, the jurisdictional paucity in the Applegate put community residents at the center. With the Forest Service and the BLM in crisis, it was community residents who stepped forward to create change.

Core Group Formation and Participant Recruitment

In these different institutional climates, the initial leaders drew upon different networks to identify and recruit initial participants. In the institutionally rich environment of northern Arizona, organizers used formal networks of existing institutions and agencies to identify representatives. In the Applegate, organizers frequently turned to the informal network of community residents to recruit individuals known.

In the Applegate, Jack Shipley, the organizer of the initiative, was a private citizen. Shipley had a long history as an environmental activist. He was part of a network of citizen activists and knew many government officials and other professional and technical specialists. Shipley and his early allies broadened these networks driven by the belief that only with broad participation could a collaboration find solutions. Long before Jack Shipley decided to hold a founding meeting, he undertook several months of shuttle diplomacy between various interest groups, neighborhood groups, and people in the Applegate and larger Rogue Basin. Shipley's strategy was an organizing campaign—similar to traditional union organizing—not an information campaign. He put forward provisional ideas and sought suggestions. He was not offering a solution but an opportunity for people to participate in figuring out a solution. Driven by an idea of social change that pushed experts out of the center of decisionmaking and created broad participation, Shipley believed that he had to recruit people from a wide variety of perspectives. This choice was motivated not simply by political considerations, but by the core belief that only with this wide variety of perspectives participating could innovative solutions be found.

The founding leaders understood, however implicitly, that their central problem was to move people out of their traditional boxes as "industry," "enviros," or agency." If people could open up to other perspectives then, leaders hoped, the group could find solutions that no one could develop independently.

In contrast to the Applegate, where a private citizen working largely independent of any organizational affiliation created the partnership, a frustrated Forest Service official stymied by public opposition to traditional management approaches founded the Grand Canyon Forest Partnership. In the early 1990s, Fred Trevey, then Supervisor of the Coconino National Forest, watched litigation and public opinion prevent the Forest Service from implementing familiar forest management. Recognizing that implementing forest management required a new base of public support, Trevey began convening informal meetings with a small circle of local environmental leaders, academics, and city and county government personnel. The group decided to form an institutional partnership to develop a consensus for restoration-based forest management.

This initial group identified organizations that would be viewed as stakeholders in public forest land management. In assessing who should participate, the core group considered institutions that needed to be represented to ensure the support for any strategy that the partnership developed. Stakeholders were brought to the table if they had technical expertise or veto power in other arenas. Founders asked these groups to designate representatives to sit on a board. In contrast to the Applegate Partnership's emphasis on recruiting participants that could be representative of the core views and issues of their constituencies, the Grand Canyon Forest Partnership sought participants to represent their organizations in drafting and endorsing specific proposals.

Common Challenges, Contrasting Responses

The founding process of each of these two groups and institutional character of their settings shaped how the partnerships formed. With differing local contexts, founding dynamics, and perceptions of the central problem (and associated response), these two partnerships developed markedly different organizational forms and habits. To contrast these differences, we identify three major challenges common to both partnerships and contrast their responses to these challenges.

Balancing Participation and Decisionmaking Authority

The relationships of the people who actively participate and nonactivists who have some interest in the work of a partnership are important because everyone cannot be at the table at once. Active participants are only a small fraction of the people who have a stake in the work of a partnership. Yet, active participants make management decisions for the larger society. To succeed a partnership needs social "permission" to do their work. How did the participants represent the larger society and what effect did it have on the form and possibilities of these groups? There are, really, three ways that people "represent" in these sorts of groups. Although they are readily distinguished theoretically, in fact, people practically shift from one mode to the other.

One way that people come to the table is as representatives of particular groups. For example, in the Grand Canyon Forest Partnership, the 17 initial partner organizations each selected a formal representative that had the authority to represent his or her institution in any negotiation or agreement. In this form, the representative tends to be tied to the institutional concerns of his or her agency. These concerns could include fulfilling legal mandates, avoiding political conflict, taking leadership in reform efforts, and defending turf. The personality, skills, and ideology of the representative and the freedom that his or her home agency gives them mediate the extent of the institutionalist mode of participation of particular participants.

The second form is representation of the interests of a constituency. In this case, someone comes to the partnership with constituent interests firmly in mind. For example, a timber industry lobbyist might come to the table to protect and promote the industry's economic interests. Similarly, an environmentalist may come to the table committed to defending the values and interests of other environmentalists. We can imagine, in these instances, that these representatives will tend to act self-interestedly on behalf of their constituents. They may tend to be strategic in their position taking. In addition, they might be inclined to take extreme positions and then bargain on behalf of their constituents. Again, these tendencies will be moderated by the personality of the representative. Divisions or lack of clear opinions or positions within the representative's constituency may also create flexibility and opportunity for independent behavior.

A third form of representation is that of perspective. In this instance, participants are viewed as bringing to the table their knowledge, experience as someone who has particular life experiences and values that lend insights to the conundrums that the group faces. For example, a logger might bring values about the importance of continuing to harvest and knowledge and skills about the opportunities and limits of particular harvest plans, and the views of her peer group. An agency scientist might bring values of species protection and information about ecosystem dynamics in the region under discussion, and an understanding of his agency's organizational limitations.

The Grand Canyon Forest Partnership viewed representation primarily in the first two ways. The Applegate sought to create the third form of representation but the first two forms were important at particular times as well.

At its outset, the founders of the Grand Canyon Forest Partnership believed that the solution to public skepticism over public forest management was to establish a partnership of respected organizations and institutions that constituted a diverse set of positions and views on forest management. If this group of powerful institutions could then reach consensus, public support would follow.

With their focus on influential organizations, institutional representation dominated the Grand Canyon Forest Partnership. By and large, participants represented the agencies and organizations from which they came. Founders assumed that the people at the table represented the key views of major constituencies and all the important perspectives on forest management.

In contrast to the Grand Canyon group's emphasis on institutional representation, the Applegate Partnership encouraged its participants to represent perspectives rather than institutional positions. In this way, the group believed the key to innovation was bringing people and ideas together in new ways. Participants talked about "leaving positions at the door and bringing only values and interests." This approach was also reflected in the group's culture of negotiation. Rather than supporting a process in which representatives started negotiating from extreme positions and then negotiated towards the middle, the group tried to focus on problem solving and collectively identifying alternatives. However, despite this explicit focus on perspective over positions, all three forms of representation existed in the Applegate, and individuals would shift from one form of representation to another as circumstances and mood dictated.

Group Structure and Participant Authority

The second challenge facing both collaboratives was how to structure the roles of participants. The Grand Canyon group quickly sought to formalize and institutionalize the partnership structure but the Applegate Partnership resisted formalization.

In the institutionally rich environment of northern Arizona, the member organizations of the Grand Canyon Forest Partnership, the majority of whom are public or quasi-public institutions, sought to establish a clear organizational structure and legal body capable of managing the coordination process. The Grand Canyon group created a new not-for-profit organization to assume the responsibility for the forest management projects. It hired a staff person who met weekly with a management team composed of representatives of five of the 15 partners to handle the day-to-day decisionmaking.

The full partnership met monthly to review partnership priorities and progress in the implementation. The group gave informal responsibility for meeting facilitating to the representative of one of the most active partnership groups. The meetings were open to the public and a number of interest group representatives and other agency personnel frequently attended. However, formal decisionmaking authority rested only with the designated partner groups. The group's bylaws stipulated the necessity of consensus among a majority of participating partners. Voice votes were typically used to make decisions on priorities.

In contrast, the Applegate Partnership actively resisted formalizing an implementing body separate from the collaborative for its first 4 years. It sought to avoid the emergence of separate organizational structures. In its first years, governmental and nongovernmental participants accomplished the majority of the Applegate Partnership's work. These groups were responsible for identifying the resources necessary to provide support for the partnership's projects and functions.

Eventually the Applegate Partnership did establish a not-for-profit entity and a watershed council that acted as subsidiaries to the larger partnership. These entities became an important part of the Applegate Valley's capacity to implement a wide range of planning and restoration projects. However, the long period of distributive leadership had created a culture of decentralized decisionmaking and responsibility.

Also reflecting this caution against consolidating leadership, the facilitation of the Applegate's meetings constantly rotated. This informal culture of rotating leadership has helped the group avoid the perception that any particular person has a disproportionate influence on group decisionmaking.

We can see, then, an institution-focused participation and representation led the Grand Canyon Forest Partnership to create an early organization and bureaucratic structure for itself. By contract, the informal, nonorganizational Applegate Partnership and its notion of perspective-based representation led to distributed leadership and project implementation.

Communicating with Nonparticipants

The final challenge that both groups faced was developing systems to communicate with people beyond core partnership activists. Because not everyone can logistically or practically come to the table, one central task of partnerships is to communicate with nonparticipants. The Grand Canyon Forest Partnership followed two distinct strategies for communication. One was with the agencies and close constituents of participants and the second was with the general public. By contrast, the Applegate Partnership used similar communication strategies with close allies and with more distant ones. As distance from the partnership increased the intensity of communication diminished.

The communication among Grand Canyon participants was direct and involved iterative policy development. But even within the Grand Canyon Partnership, participation in communication depended on the level of organizational involvement in the partnership's activities. A management team represented five of the most active partnership organizations. This group met weekly to develop the specific implementation plans. Most of the decisionmaking took place within this core group of active partner organizations. The larger group of partners was involved primarily through the monthly partnership meetings, special meetings, and electronic correspondence.

When the Grand Canyon group sought to communicate with people beyond constituent agencies and organizations, its focus shifted from deliberation and iterative proposal development to information and education. The group used a series of public meetings during its initial phases to solicit public comment on proposed actions. When some of the public opposed the Grand Canyon's proposal to limit certain types of recreation use, the partnership dropped those proposals. Subsequently, the partnership relied almost entirely on press releases, media coverage, and the formal government review processes to announce to the public what they planned to do and why.

In the Applegate, the distinction between "public" and "constituent" was less pronounced because participants were rarely formally representing people but rather were representing perspectives. Clearly, people working for organizations and for interest-based constituencies had responsibilities to communicate proposals back to their organizations to get feedback. But partnership participants knew this institutional sign-off would not be sufficient to garner broad community support. Because people sitting at the table were often not formal representatives they had no explicit permission to enter into agreements on anyone's behalf. Consequently, participants needed an interactive relationship with a large number of stakeholders beyond the partnership itself.

At meetings, decisions were often preliminary agreements that participants would take to their social networks to test out. Subsequently, at or between meetings, on the phone or in person, participants would bring back reactions of the various people in participants' networks and modify decisions. Of central importance to the Applegate group were informal social networks. Rarely did the Applegate Partnership use the media to get their message out. Instead, participants each turned to their social networks, be these connections with environmentalists, timber workers, people in the Regional Office of the Forest Service, or community residents that one saw at other community meetings or social events. Some connections were professional, others were social. Some ties were based on strong ties of friendship, others were associational ties from repeated meetings on natural resource or community business (Granovetter 1973).

Rarely was the primary task of the Applegate Partnership educational. Because the Applegate group did not represent people formally, much of their communication was about gaining permission from the larger community and relevant organizations to move forward with their ideas. As partnership participants talked to people, they were looking for feedback, suggested modifications, support, and permission to move forward. With their social change agenda, communication was really an "organizing" task, not an information strategy to explain a decision.

Responses to Challenge and Conflict

The socially-based approach of the Applegate Partnership and the institutional, problem-solving approach of the Grand Canyon group affected each group's notions of appropriate representation, internal structural development, and communication. One can most clearly see effects of these differences in their responses to external opposition and conflict. Often partnership groups spend months or years developing a delicate balance of views, positions, and actions. Once under pressure, these agreements can become unstable and stretch the collaboration to its limits.

Going Public, Building Expectations

For both the partnerships, the founding periods were spent largely out of public view. This period involved dozens of meetings in which participants tried to find common ground among diverse interests and agendas. In both groups, this period ended by "going public" with events in which the partnerships described the newly found common ground and celebrated the great potential for common action.

Having raised public expectations, these groups had to demonstrate the efficacy of their claims. Because both groups were working largely on public land issues, this necessitated a federal lands planning process. Because they were federal land projects, the public had to review the plans. In both cases, the initial projects were on Forest Service lands. In each case, local agency officials, both District and Forest level, worked closely with the partners to develop a project and then the environmental assessment. Also common to both cases, developing the proposed action was lengthy, and diverse interests, particularly environmental and industry, conflicted frequently. At various points in both partnerships, negotiations nearly broke off and threatened the continued existence of the partnerships. In the Grand Canyon Forest Partnership, some of these negotiations involved groups that were not formally partners but were capable of obstructing the process if not satisfied. Ultimately, in both efforts, participants made compromises to craft an action all of the major interests would support, usually by reducing the scope and intensity of proposed actions.

Once these concessions were made, most members of each partnership believed that they had appeased the important stakeholders. Both groups recognized that there were outlier groups that had not been satisfied but dismissed their power to obstruct the process because of the agreement among participants and apparent public support.

In both cases, environmental groups that had not actively participated in project formation, and opposed the concept of collaborative management more generally, sought to stop these first projects through the administrative appeals process. In both cases, an environmental organization pursued administrative appeals using the National Environmental Policy Act (NEPA) that led the Regional Offices to require that the respective National Forests (Rogue River National Forest in southern Oregon, Coconino National Forest of northern Arizona) modify the environmental assessments to meet the demands of appellants. In the case of the Grand Canyon project, some environmentalists remained unsatisfied and pursued legal settlement in the courts.

Contrasting Responses

These successful appeals tremendously impacted the morale and internal cohesion of both partnerships. The tenuous and often strained accommodations these unlikely partners had created now began to fray. Internal dissension over strategy began to grow. It is at this point that the real differences between these two partnerships begin to become more apparent.

The Applegate Partnership—As the older of the two partnership groups, more time and history are available to consider the effects of external opposition on the Applegate's orientation. The Applegate's focus on social change and informal structure allowed the group to shift substantive focus temporarily and continue to work on institutional change and citizen involvement. When the Forest Service project stalled, the group shifted its attention to the development of similar management changes with the Bureau of Land Management and private landowners.

For the Applegate, the initial challenges created by the appeal of its premier project were only the beginning of its tribulations. The success of the appeal of the Applegate

Partnership's first project, called Partnership One, substantially shifted the balance of opinion within the environmental community. (It should be noted, however, that Partnership One was not simply challenged from the environmental side. The timber industry also opposed the sale by not bidding for the timber once the Forest Service gained approval to proceed.) The lead environmental organization in the Partnership, Headwaters, eventually bowed to pressure from the rest of the environmental community and withdrew from the Partnership.

Although the Headwater's withdrawal was a public blow for the partnership and emotionally anguishing for many participants, ironically, it did not substantially change the makeup of the partnership. Two of the three the individuals representing Headwaters continued to participate in the partnership. These were people who were both representatives of Headwaters and residents of the local Applegate community. When Headwaters withdrew its formal organizational support, these people continued to participate, now as community residents active in local watershed groups.

As it tried to recover from what many assumed would be a fatal blow—Headwaters's departure—the partnership took yet another major hit. Paradoxically, this time the source of the damage was the timber industry. In an effort to invalidate President Clinton's Northwest Forest Plan, representatives of regional and national timber organizations sued the federal government, arguing that the Plan's creation violated the Federal Advisory Committee Act (FACA). Passed in the early 1970s, Congress designed FACA to reduce the influence of special interests on federal agency policy making, especially defense contractors on Pentagon planning and decisionmaking. FACA outlined a series of provisions that federal agencies must follow if it participates in any joint decisionmaking processes with nonagency groups.

Afraid that it might jeopardize the huge investments it had made in the development of the Northwest Forest Plan, the Clinton Administration ordered all public agencies to withdraw from partnerships or other collaborative efforts that might violate FACA. Agency officials who had been formal partners in the Applegate process could no longer be Applegate Partnership board members. Despite their pledges to continue to participate as nondecisionmaking members, the basic foundation on which that the partnership had been built was now shattered. In the span of just 6 months, the Applegate Partnership lost two of its organizational constituents.

Facing these difficult events, the Applegate Partnership went through an important internal reevaluation and metamorphosis. Throughout its early period, the Applegate had attempted to balance two sometimes-opposing roles. The Applegate group viewed itself as an effort to make peace between two opposing forces and the federal agencies caught in the crossfire. Much of its early notoriety and influence was derived from the perception that it was a model for resolving otherwise intractable conflict in federal land management. At the same time, the partnership was, from its outset, a solidly community-based, community-driven initiative. It was as likely to have a potluck as a conference. Its weekly meetings were designed to be accessible to community people. Meeting participation was open to all and locals frequently outnumbered their agency and interest group counterparts who were paid to attend.

With its first major project stalled indefinitely and two of is major power brokers formally withdrawn from the leadership of the group, the Applegate reconsidered its focus and constituency. It broadened its focus to include a wider range of community issues. Aware that community support was now essential, the Applegate initiated a major community outreach and assessment process to identify the range of community concerns and ideas. From this and other forms of informal outreach, partnership activists launched a broad set of local initiatives. For example, one subset of participants was interested in private land restoration that would improve wild salmon fisheries.

At the same time, the partnership began working more actively with the BLM to evaluate restoration projects on its land. Over time, with considerable perseverance, the partnership developed ways to work with the federal agencies that did not involve formal joint decisionmaking. Within 6 months, the struggles over its first Forest Service project and the consequences of FACA were largely forgotten as a broader set of activities began to occupy the partnership's attention.

Three years after its founding, the consequences of this shift from interest-based partnership to community-based partnership could be clearly seen in the projects and accomplishments for which the partnership was either directly or indirectly responsible. Beyond shifting of BLM management in one resource area and private land restoration projects, the Applegate Partnership began to publish a bimonthly newspaper distributed to all 10,000 residences in the watershed. It spun off a small-scale economic development group looking at land-oriented small enterprise opportunities. Later, the partnership brokered a deal with one county to have the community develop a management plan and land use regulations related to gravel extraction. Partnership participants helped bring a county park under community management after the county shut it down because of a lack of funding. Repeatedly the Applegate Partnership took on the task of mediating conflict and brokering agreements between community residents and government agencies, be they federal, state, or county. At every turn, partnership activists argued for different processes that included community residents earlier in planning processes, bringing together conflicting parties to find common ground.

As we see, then, when faced with blockages such as the administrative appeal of the Partnership One project and then FACA, Applegate Partnership participants did not give up. Rather, they fell back on their larger goals of social change and ecosystem management in the watershed. These broad goals allowed the group to conceive a wide variety of particular projects to achieve their ends. This broadening was possible because, from the beginning, a substantial proportion of its active participants saw themselves as community residents first and organizational representatives second.

The Grand Canyon Forest Partnership—When faced with opposition the Applegate Partnership broadened its variety of strategies and shifted its attention from stalled to viable projects. In contrast, the Grand Canyon group's narrow problem definition limited its room for maneuver when confronted by effective opposition.

Founded in the summer of 1996, when fires burned over 40,000 acres of northern Arizona forests, the Grand Canyon Forest Partnership's proposals to develop restoration and wildfire reduction programs had enjoyed wide public support. After 2 years of working out its proposal internally, the partnership held a series of public meetings to communicate its proposal to the public. The partnership outlined thinning, prescribed fire, and also a series of recreation management proposals, including the closure of some popular trails and remote roads.

To the surprise of many organizers, the public seemed to broadly support extensive forest thinning and prescribed fire. People could not yet visualize the posttreatment appearance of the restored areas, but were supportive of reducing the potential for wildfires that could threaten human life and property. This bolstered the partnership's perception that the community was behind its efforts to develop a restoration/wildfire hazard reduction strategy.

In contrast, the meeting that focused on recreation use generated substantial concern and some controversy. Rumors began circulating that the Grand Canyon group was attempting to exclude one or another recreation user group. A nascent opposition to the partnership's efforts began to organize, creating unlikely alliances between mountain bike users, ATV enthusiasts, equestrian advocates, and others. It quickly became apparent that recreation issues could become a flash point for opposition to the overall restoration initiative.

In response, the group decided to narrow its focus and exclude the controversial elements of the recreation plan from the proposed action. The group discussed the possibility of developing a restoration task group to convene a collaborative process with recreation interests. After a series of tentative steps in this direction, the effort was dropped and Grand Canyon participants focused on developing and implementing the thinning and fire treatments.

Confident that it now had a broad public mandate to proceed, the Grand Canyon Forest Partnership and the Forest Service drafted the environmental assessment and prepared for formal public review. However, throughout the preparation of the final proposal, the partnership experienced agonizing disagreements with prominent local and regional environmentalists over a variety of provisions. For example, the Forest Service began a series of test plots using a range of treatments. The graphic reality of the actual treatment impacts nearly destroyed the fragile agreements that had been crafted with these environmental groups. Yet despite these setbacks, partnership leaders continued to make concessions and deferrals that maintained the truce and allowed the project to proceed. Having conceded to most of the demands made by these groups, the partnership and the Forest Service were confident its proposed action would withstand any outside challenges.

It was no surprise when an environmental group from outside the area filed an appeal. But the partnership was shocked when the Regional Office of the Forest Service upheld the appeal. A group that most had dismissed as an irrelevant outsider was now a champion trumpeting its victory around the country. A partnership regarded as a national model for transcending the gridlock of federal forest management found itself buried in political mud.

The defeat created a period of internal disorder within the partnership. The timelines that had been established for implementing treatments were now irrelevant. Efforts to develop markets for the restoration byproducts, a step essential to making the restoration treatments economically viable, were destroyed as entrepreneurs canceled plans to invest in new tools and technology. Other financial resources the partnership was pursuing also evaporated as word spread that the partnership was stalled.

These setbacks further aggravated nascent tensions within the partnership over strategy and priorities. One faction advocated for the partnership taking a leading role in regional and national dialogues on the underlying issues of the opponents of cross-interest collaborations. Others favored abandoning the conciliatory approach in favor of aggressive counterattack using legal challenges and high profile efforts to question the credibility of opposing groups.

In response to these challenges, the partnership chose again to narrow its focus. The primary focus of project development shifted to fire hazard reduction projects in proximity to human developments. At the same time, efforts to secure final approval for the partnership's first stalled project began to focus on preparations for a direct legal battle with project opponents. As the local agency officials made the mandated changes to the document that the Forest Service's Regional Office had ordered, leading groups in the partnership began formulating a legal strategy for intervening in the event of a lawsuit.

Conclusion

Faced with complex and difficult natural resource management challenges, institutions and communities are increasingly looking to multistakeholder collaborations as an approach to developing widely supported solutions. These collaborations face many similar challenges: how to integrate diverse interests and issues, how to share decisionmaking authority, and how to organize resources and expertise necessary to develop solutions.

The prevailing perception within these groups about the nature of the problem they face and how it can be most effectively addressed largely shaped the choices these groups made in addressing these common challenges. With two examples, we suggested that the perception of the problem can shape the strategy and conduct of the resulting collaborations.

Given the central importance of this initial perception of the problem, we have gone further to try and explain why two partnership facing relatively similar circumstances developed different perceptions of the problem and strategies for addressing it. We have asserted that the institutional context in which a partnership takes place determines much of this initial orientation. In settings rich with preexisting institutions—Federal agencies, local governments, and nongovernmental organizations—it appears that collaborations are more likely to have a technocratic problem solving orientation. Circumstances without this density of institutions leave more of the influence over the character of the collaboration in the hands of the private citizens who create the impetus for formation.

Given the prior history and experience in social change work—environmental, labor, and peace activism—among many of its founding organizers, it is perhaps predictable that the Applegate initiative viewed its challenge as less a technical problem and more of a social issue. As a consequence, the strategies it selected and the ways it adapted to changes and challenges are markedly different than those of the Grand Canyon Forest Partnership.

Our intention in this paper is not to make explicit or implicit judgments of the value or effectiveness of one strategy over another. Few collaborative efforts have existed long enough to reasonably evaluate their performance history. In drawing distinctions between these two partnerships our intention has been to underscore the formative influences that shape the evolution and activities of a partnership. In so doing we hope to contribute to the set distinctions that will enable both practitioners and researchers to more clearly understand the similarities and differences between initiatives.

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Ecological Wilderness Restoration: Attitudes Toward Restoring the Mount Logan Wilderness

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Abstract—By law, wilderness areas are intended to be unmarred landscapes where evidence of modern civilization is generally absent. This presents a problem, since ecological wilderness conditions have been impaired by human activities. For example, some forest wilderness ecosystems have been altered by livestock grazing, logging, fire exclusion, and through other environmental manipulations. Additionally, there are socio-political factors that must be considered prior to discussing wilderness restoration methods. This paper focuses on the need for ecological wilderness restoration and presents options for managers to consider while discussing wilderness restoration. To determine the most preferred wilderness restoration method among communities located near the Bureau of Land Management Arizona Strip Field Office, a social survey was conducted that focused on attitudes of community residents living in proximity to the Mount Logan Wilderness, which is in northern Arizona within the Grand Canyon-Parashant National Monument. The study focused on this wilderness because an ecological restoration project was occurring outside the wilderness boundary. The survey was designed to determine the local acceptance of a mechanical, nonmechanical, or prescribed fire wilderness restoration method. Mechanical methods are often the most controversial, but survey respondents held the most positive attitude toward this method.

Introduction

According to the Wilderness Act of 1964 (P.L. 88-577), management of wilderness includes "retaining its primeval character…which is protected and managed so as to preserve its natural conditions…with the imprint of man's work substantially unnoticeable." This statement implies that wildernesses have been protected and preserved, and that they are in a natural state. According to Cole (2000), managing wildernesses for natural conditions, which includes the absence of recent human influences, may be a desired management outcome. It is debatable if wildernesses are in a natural state, and we assert that many wildernesses have not retained their natural conditions, as human evidence in regards to past and present land manipulations are apparent in many wilderness ecosystems. Additionally, prior to the passage of the Wilderness Act of 1964, many wildernesses did not have special management provisions, therefore, they were not treated differently than nonwildernesses. Wildernesses may not be as pristine as policymakers believed 35 years ago when the Wilderness Act was passed (Brunson 1995; Cole 1996; Murry 1996). Therefore, managers are now faced with the dilemma of determining if wilderness restoration is necessary to return areas to more natural conditions, and determining ways to restore wilderness conditions through the least damaging and most appropriate wilderness restoration methods.

Proposed methods usually include mechanically reducing fuels, the use of prescribed fire, or a combination of these methods to restore the natural fire regime (Covington and Moore 1994a; Landres and others 2000). For wildernesses, these types of restoration methods may not be a popular option and would have to consider NEPA (National Environmental Policy Act of 1970) in assessing environmental impacts. Also, it may not be possible to develop ecologically, economically, and politically acceptable restoration management regimes, especially for wilderness and other natural areas (Covington and Moore 1994a). To determine the acceptability for restoration of the Mount Logan Wilderness, Northern Arizona University, with input from the Bureau of Land Management (BLM), designed a survey to determine which types of wilderness restoration methods may be considered a wilderness management option. The survey of local communities did not ask respondents if they would like to see a combination of methods. Instead, it focused on which restoration methods the local communities preferred. A local sample was selected because managers wanted to determine attitudes of residents within proximity to the wilderness and the amount of support they held toward wilderness restoration methods. Managers will have to decide, from an ecological standpoint, how wilderness restoration methods can be combined to meet restoration goals, while considering socio-political factors.

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Wilderness Changes

Western wildernesses have changed drastically since the time of Euro-American settlement in the 1800s. To determine the ecological changes in the Mount Logan Wilderness, a forest reconstruction was completed prior to conducting a social survey. In reconstructing the wilderness, a pre Euro-American reference date of 1870 was determined and selected as the reference condition. This reference condition (date) was selected because it represented a measurable and replicable point in time, and a time previous to a large human population influx that created noticeable human impacts. For example, in southwestern ponderosa pine (*Pinus ponderosa*), Euro-American settlers introduced livestock grazing, logging, and fire exclusion, which contributed to changes in the forest structure (Covington and others 1994, 1997; Covington and Moore 1994b; Fielder and others 1996; Swetnam and Baisan 1996). The disruption of the natural fire regime created the most pervasive source of degradation in ponderosa pine ecosystems (DeMillion and Covington 2000), increased the susceptibility of insect and disease outbreaks (Kolb and others 1994), and has allowed tree densities to increase. The result is dense forests, rather than the open and parklike forests that were described by early Euro-American settlers (Cooper 1960; Covington and others 1994; Covington and Moore 1994a; Fulé and others 1997).

In the Mount Logan Wilderness, the forest reconstruction determined that the tree density in 1870 was 36 trees per acre, versus today's forest density of 571 trees per acre (DeMillion 1999). Research has shown that an increase in forest densities and fuels often leads to high intensity crown fires (Arno and others 1995; Covington and Moore 1994a,b; Covington and others 1994; Swetnam and Baisan 1996). Research results from the Mount Logan Wilderness study determined that wilderness forest conditions have been dramatically altered, which is similar to results found outside this wilderness area (NAU 1996). If wilderness managers intend to restore this wilderness to conditions that existed prior to Euro-American settlement, they must consider restoration methods in tandem with socio-political factors to decide when and how to restore the wilderness.

Ecological Wilderness Restoration

The Wilderness Act of 1964 defines wilderness as "an area of undeveloped Federal land." Unfortunately, undeveloped areas do not always equate to unchanged or unmanipulated landscapes. The majority of western wildernesses have been manipulated by Euro-American settlers and other external influences. The question that wilderness managers need to answer is, should restoration methods be used to replicate conditions previous to modern human impacts? If wilderness restorative actions are taken, the sense of wildness and the belief that an area is free of human manipulation is compromised (Landres and others 2000). If actions are not taken, high intensity fires will start in wildernesses and spread beyond their boundaries to rural and urban communities.

Two examples of fires in 2000 that began in wildernesses are the Cerro Grande (Bandelier National Park) fire that interfaced with Los Alamos, New Mexico, and the Clear Creek (Salmon-Challis National Forest) fire that began in the Frank Church Wilderness of No Return and spread to Panther Creek along the Salmon River in Idaho. Both fires started in wilderness and spread from wildland areas to urban or rural communities destroying structures and in some instances livelihoods. Fuel loadings within and outside of the wilderness boundaries in both locations were similar and the fires did not stop at the administrative boundaries. Due to the heightened awareness of fire and other factors, some land managers are beginning to explore the socio-political concerns that the public holds toward restoring wildernesses. Managers of the BLM Arizona Strip Field Office wanted to conduct a survey to determine the range of alternatives and the types of wilderness restoration methods that might be acceptable to local community residents.

Mount Logan Ecological Wilderness Restoration Methods

Community residents within proximity to the Mount Logan Wilderness were surveyed to determine their attitude toward mechanical, nonmechanical, and prescribed fire wilderness restoration methods. Within the survey, a mechanical method was defined as any activity where machines are used to cut and/or transport personnel or material. An example of this would be the use of chainsaws to reduce small diameter trees, decrease ladder fuels around old growth trees and snags, and create fuel breaks. Mechanical methods could also include the use of motorized equipment and vehicles with wheels or tracks. A nonmechanical method was defined as any activity that involved a living power source, for example, the use of a handsaw, hand rake, or horses. A prescribed fire method was defined as intentionally igniting and controlling fires. This definition is different from the national definition that described prescribed fire as "any fire ignited by management actions under certain, predetermined conditions to meet specific objectives related to hazardous fuels or habitat improvement. A written, approved prescribed fire plan must exist, and NEPA requirements must be met prior to ignition." This definition was not used because the terminology may have been confusing, it may have been too lengthy, and it does not refer to prescribed fire as a restoration method.

Attitudes Toward Wilderness Restoration Methods

The Arizona Strip Field Office, BLM and Northern Arizona University surveyed communities within proximity to the Mount Logan Wilderness about their attitude toward wilderness restoration methods. The wilderness is within the Grand Canyon-Parashant National Monument in Arizona and was selected because an ecological restoration project was under way in areas outside the wilderness boundary. The closest communities to the wilderness include St. George, Kanab, and Hurricane in Utah, and

Fredonia and Colorado City in Arizona. Residents within these communities were selected to respond to a survey about the use of mechanical, nonmechanical, and prescribed fire wilderness restoration methods. Managers wanted to determine the attitudes that local community residents held toward these specific wilderness restoration methods.

To conduct the survey, a mail-back questionnaire was designed, pre-tested, and sent to a random sample of 1,000 residents. Questionnaires were sent to 500 residents of St. George, Utah (urban sample), with another 500 sent to residents in two rural communities in Utah (Kanab and Hurricane) and two rural communities in Arizona (Fredonia and Colorado City). The overall survey response rate was 55 percent. There were no significant differences in attitudes toward ecological restoration treatments between respondents in the urban and rural samples.

Attitudes Toward a Mechanical Treatment—The most favorable response was for the mechanical method, with 74 percent of the respondents holding a positive attitude (table 1). We suggest that positive attitudes toward a mechanical method are likely to be higher among local communities than among residents from a regional or national sample. Research on attitudes has shown that the more familiar people are with an attitude object (subject), the more supportive they are toward that object (Bruvold 1973; Chaiken and Stangor 1987; Fazio 1982). In other words, residents of the communities we surveyed may be more familiar with the use of mechanical methods, and therefore may have a more positive attitude toward them.

Table 1—Attitudes toward the use of three restoration methods in the Mount Logan Wilderness.

Type of wilderness restoration treatment	Positive	Negative
	------ *Percent* -----	
Mechanical	74	15
Nonmechanical	54	32
Prescribed fire	55	33

Attitudes Toward a Nonmechanical and a Prescribed Fire Treatment—Slightly over 50 percent of the respondents held positive attitudes toward using a nonmechanical and a prescribed fire wilderness restoration method to restore the Mount Logan Wilderness. Respondents with positive attitudes toward these methods agreed that restoring the wilderness in a natural way is good. In comparing our results with other studies that focused on prescribed fire, we found similarities. Results from a regional sample of Montana and Wyoming residents showed that 55 percent of the residents supported prescribed fire, compared to 48 percent of a national sample (Manfredo and others 1990). Between our results and other studies, there appears to be support for using prescribed fire as a management tool to restore the Mount Logan Wilderness, as well as other wilderness areas.

Options for Wilderness Ecological Restoration

Every wilderness restoration method is controversial. Mechanical treatments have potential wilderness impacts, nonmechanical treatments (the use of hand raking around all old-growth trees) may not be a viable option on a landscape scale, and the use of prescribed fire creates the potential for stand-replacing fires, especially in proposed or designated wildernesses, where fuels are rarely reduced prior to burning. Often prescribed fires become high intensity and uncontrollable wildfires, due to years of fuel accumulations. One example of this is the Yellowstone National Park prescribed natural fire ignitions of 1988, which burned under a fire prescription without a prior reduction in fuels and became uncontrollable. Firefighters frantically reduced fuels with mechanical methods by cutting trees in a matter of days instead of over a number of years (Wakimoto 1990). This illustrates the need in some wildernesses to reduce fuels prior to reintroducing fire, or prior to allowing naturally ignited fires to burn under certain fire prescriptions.

Some managers may be limited to only the use of prescribed fire as a wilderness restoration method, because of socio-political concerns with the use mechanical methods. Some people believe that the use of mechanical restoration methods is an unacceptable way to enhance the naturalness of these areas (Parsons and Landres 1998), but others believe that manual removal of fuels should precede the use of prescribed fire (Noss 1991).

Regardless of the restoration methods selected to reduce forest fuels, managers are aware that wildfires will continue to occur across legislative and administrative boundaries. The wilderness ignitions that led to the high intensity Cerros Grande and Clear Creek fires of 2000 are forcing managers to evaluate the range of wilderness restoration alternatives, especially where rural and urban communities interface with wildernesses.

Conclusions

Unhealthy forest conditions result from altered wilderness conditions that are inhibiting natural processes. Years of grazing, logging, and fire exclusion have altered forest wilderness ecosystems. These altered landscapes are conducive to unnaturally intense and often stand-replacing fires. Managers need to decide if wildernesses should be left unmanipulated or be restored to more natural conditions (Cole 1996).

Restoration outside wildernesses usually entails the use of mechanical methods to reduce fuels, along with the use of prescribed fire to restore the natural fire regime. Using mechanical methods prior to the use of prescribed fire may not seem to be a popular option among the general public, but the social research presented above concluded that mechanical methods were the most acceptable and preferred wilderness restoration method. These results were taken into consideration by the Arizona Strip Field Office, BLM, but management decisions are not based upon the results of this one study. The study did create the impetus for the BLM to begin a dialog about wilderness restoration among diverse regional public groups. Managers are encouraged to discuss wilderness restoration methods if they want to restore wilderness conditions to a more natural state and decrease the potential for wilderness fires to interface with rural and urban communities. Wilderness managers considering socio-political parameters are struggling to determine when wilderness restoration is appropriate and the range of restoration methods that can be employed.

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Incorporating Ecological and Nonecological Concerns in the Restoration of a Rare, High-Elevation Bebb Willow Riparian Community

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Abstract—Activities were initiated by The Nature Conservancy, the USDA Forest Service, and the Northern Arizona University School of Forestry and Department of Geology in 1996 to restore hydrologic and ecological function to a high-elevation Bebb willow (*Salix bebbiana*) and mixed grass riparian community in Hart Prairie, near Flagstaff, AZ. Initial restoration removed small water diversions above the willow community to restore hydrologic flow to the downslope community. Because of the small scale of this effort, the viewshed, recreation, and wildlife in the area were not affected. Subsequent monitoring indicated that although restoration increased water availability and improved Bebb willow water status, soil moisture conditions were still below those needed for willow seed germination and seedling growth. Therefore, the long-term sustainability of the Bebb willow community is still at risk. Current restoration plans are to manipulate the upslope watershed to provide additional water needed for willow regeneration. This restoration will include removing stock tanks, thinning trees encroaching into the meadow above the willows, and burning to restore the upslope area to its historic nonforested, prairie condition. These plans could significantly impact aesthetics, wildlife, and recreation in the area. In addition, Indian Tribes who consider the San Francisco Peaks to be sacred and the general public may have concerns with these restoration activities. This paper discusses our efforts to address these issues while still providing sufficient ecosystem restoration for long-term sustainability of the Bebb willow-mixed grass community.

Introduction

In Arizona, upland riparian communities located at elevations above 2,500 m consist of small plant stringers occupying channels of first- and second-order streams (Debano and Schmidt 1990). They are largely supported by shallow ground water, and are typically in meadows at the base of watersheds dominated by mixed conifers and ponderosa pine (*Pinus ponderosa* Laws.). The upland riparian community at Hart Prairie near Flagstaff, Arizona, has a Bebb willow (*Salix bebbiana* Sarg.) and trembling aspen (*Populus tremuloides* Michx.) overstory with a grass understory. This type of upland riparian community is rare in Arizona and supports high levels of wildlife and plant community diversity. Both aspen and Bebb willow are listed by the USDA Forest Service (USFS) as species of concern due to their biodiversity, aesthetic, and wildlife habitat values. Upland riparian communities such as Hart Prairie make important contributions to the biodiversity necessary for resilient and healthy ecosystems at the regional/landscape level (Noss 1990). Unfortunately, many of these communities, including Hart Prairie, do not have adequate regeneration to sustain themselves and are thus considered unhealthy (Briggs and others 1994).

Abundant shallow ground water supplied through perched aquifers is the critical factor for maintaining healthy upland riparian communities such as at Hart Prairie. These perched aquifers occur discontinuously across the Colorado Plateau. They have limited storage and require annual recharge through runoff and infiltration from heavy snowpacks and consistent precipitation. In high elevation areas in the Western United States, approximately 70 to 90 percent of annual streamflow originates from these types of events (Troendle 1983). Drought, surface water diversions such as stock tanks and dams, and water loss through increased tree transpiration modify hydrologic function by changing the quantity and timing of the water recharge to these perched aquifers and the subsequent water delivery to riparian ecosystems at the base of watersheds. These factors are likely causing a decline in the health of the Bebb willow riparian community at Hart Prairie (Avery 1991; Church 2000; Gavin 1998).

Initial Restoration Efforts

Restoration of hydrologic and ecological function to the Hart Prairie Bebb willow riparian community began in 1996–1997 through a cooperative project involving The Nature Conservancy (TNC), USFS and the Northern Arizona University (NAU) School of Forestry and Department of Geology. This restoration project was funded by the Arizona Water Protection Fund (grants 95-006WPF and 96-0019WPF) and NAU School of Forestry. The restoration involved digging out two small earthen dams that were diverting water out of its original channel away from the

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downslope Bebb willow riparian area. The original channel was reconstructed by removing debris that had accumulated in it, and rock weirs were constructed in the channel to slow high-flow water velocities. Drainages that had been diverting water from the original channel were blocked using sediment and rocks removed from the original channel. In addition to the channel work, restoration activities included constructing wire fences around some of the Bebb willow populations to protect them from overgrazing by deer and elk. The fencing and channel restoration activities were all completed using hand tools.

An extensive monitoring network was established to measure hydrologic and Bebb willow responses to the restoration. This monitoring network included a drilling series of shallow wells, setting up soil moisture monitoring points, and inserting a metal "H"-flume at the base of the watershed to measure surface and subsurface water movement within and out of the base of the watershed. A permanent weather station was set up in a nonforested area of the meadow near the top of the Bebb willow riparian community to collect a variety of weather data, and an electric fence surrounding the weather tower was constructed to protect the electronics from deer and elk. Finally, Bebb willow water status and regeneration were also measured. Details of these restoration and monitoring activities are described in Gavin (1998) and Church (2000).

Because restoration of the channel primarily involved redistribution of soil and rocks already present, and it did not change the visual appearance of the area, water sources for wildlife were not decreased and there were no impacts to hikers, bicyclists, and skiers in the area. However, there have been concerns expressed by the public regarding the monitoring equipment. The "H"-flume measuring streamflow at the base of the watershed has been tampered with, and the 7-foot tall, silver-colored weather tower with its electric fence is an eyesore. In response to the public's concerns, the exterior of the weather tower and the fence posts around it were painted to blend better into the environment to improve aesthetics of this monitoring station. In addition, alternative fencing is being considered.

Results of the monitoring activities indicated that the initial restoration only partially restored hydrologic function (Church 2000; Gavin 1998). Moisture conditions in the riparian system *have* improved, but they were still below those needed in the late spring for successful Bebb willow seed germination and early seedling growth. Church (2000) reported that the Bebb willow trees were producing abundant viable seed (85 percent germination under laboratory conditions), but soils were too dry during seed dissemination to support Bebb willow regeneration. Therefore, although the restoration activities improved conditions for the adult trees, they did not provide for the long-term sustainability of this riparian community.

Past riparian restoration efforts including those at Hart Prairie illustrate the importance of evaluating the condition of a degraded riparian area from a watershed perspective. Because of the close connection between upslope and downstream processes, riparian restoration efforts must avoid the approach that was taken at Hart Prairie where recovery strategies were based solely on an evaluation of the immediate degraded riparian site (Briggs 1996). Addressing isolated components of a watershed such as was done at Hart Prairie is ecologically incomplete; to be complete, restoration must address conditions in the upslope watershed.

In the case of the Hart Prairie area, a thick cover of grass and high density of ponderosa pine and white pine (*Pinus strobiformis* Engelm.) now dominate the meadow above the Bebb willow riparian community. These conditions are in contrast to historic photographs of Hart Prairie showing a meadow with light grass cover and only a few scattered trees. The increased grass cover and encroachment of woody tree species from the upper watershed into the meadow are the result of fire suppression and grazing. This encroachment of pines is consistent with increases in the number of trees per acre on National Forest lands in Arizona and New Mexico over the past 30 years (Covington and Moore 1994). In high-elevation meadows such as Hart Prairie, the encroaching pine forest competes with the downslope riparian communities for ground water stored in the perched aquifers. Restoration of the upslope area back to historic conditions would involve removing the pine trees where they have encroached into the previously wet meadow, and prescribed burning to reduce grass cover. Experiments in the Beaver Creek watershed in Arizona show that streamflow can be increased for 6–10 years following removal of trees in upslope communities (Baker 1986; Brown and others 1974).

Ecological Concerns

Complete restoration of the prairie meadow would remove up to 90 percent of the ponderosa and white pines in the area directly upslope from the Bebb willow community (fig. 1), and prescribed burning would reduce grass density and prevent future tree encroachment. In addition, a complete restoration would remove stock tanks in the watershed above the Bebb willow community. In 1998, NAU received funding from the Arizona Water Protection Fund (grant 98-050WPF) for a collaborative effort by NAU, TNC, and the USFS to restore the watershed at Hart Prairie.

The first step in the restoration effort was to map and measure all of the trees encroaching into the meadow. The total amount of water lost to the downslope riparian community from the encroaching trees will be estimated by NAU by combining the tree size data with sapflow data collected during the summer of 2000. This information will be used to compare the amount of water released downslope following different thinning treatments, which will help determine the actual number and which trees to remove. Water lost due to herbaceous vegetation transpiration is being measured by NAU to determine the amount of water that would be released downslope following prescribed burning. This information will also help determine the ecologically best time(s) of the year to burn. For example, if the grasses are using a lot of water during Bebb willow seed dissemination, then a prescribed burn prior to willow seed dissemination would be the best ecological time to burn.

There are five stock tanks currently being evaluated by NAU to determine their effects on the downslope Bebb willow community. This information will be used to direct activities such as deconstruction of tanks, reduction in tank size, and bank and channel reshaping. Tank removal and mitigation will be based on field measurements, historical photographs and maps, and other supporting information.

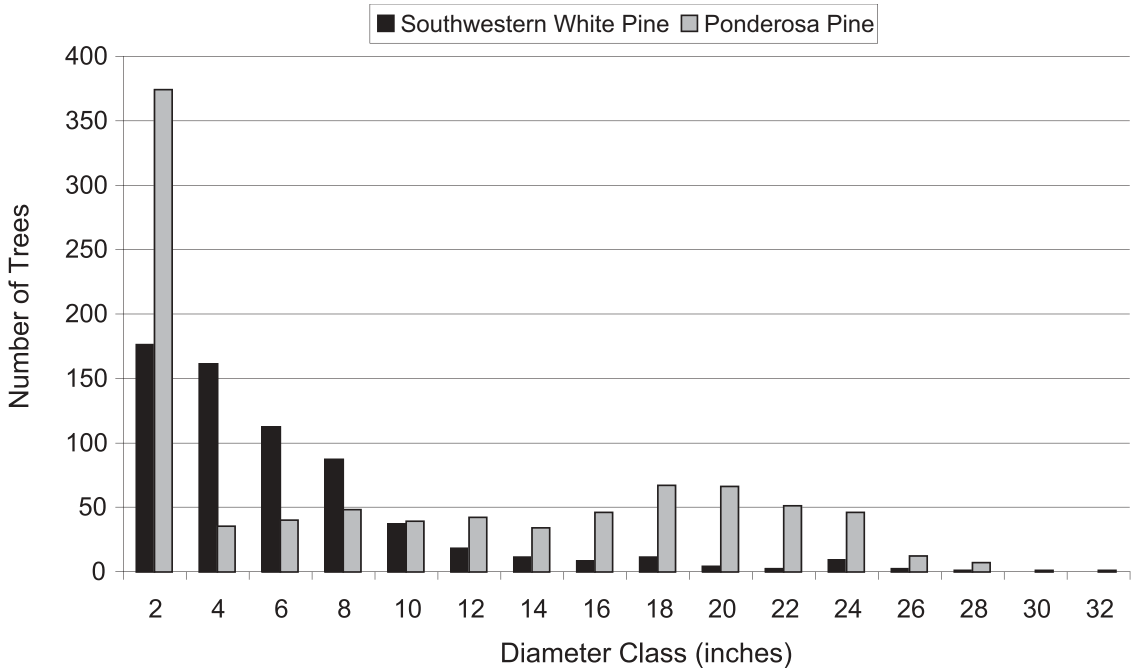

Figure 1—Trees encroaching into the meadow at Hart Prairie, Arizona

The tree and tank removal, and prescribed burning will increase water available to the riparian community, but in contrast to the initial restoration efforts, these watershed activities could also generate considerable public concern due to their potentially significant aesthetic, wildlife, and recreation impacts.

Nonecological Concerns

NAU, TNC, and the USFS conducted planning meetings during 1999–2000 where impacts to the viewshed and recreation from a full watershed restoration were discussed, and modifications to the full restoration were developed to address these concerns. In addition, public comments were collected in response to the USFS Proposed Action describing the restoration, and concerns were collected from the neighbors of TNC. The concerns raised during the planning meetings and by the public are described below, and modifications to the full restoration as a result of these concerns are described in the next section.

The Hart Prairie area is popular with hikers, bikers, and cross-country skiers, and the restoration treatment area is within the Snowbowl ski area and proposed Arizona Trail viewsheds. The removal of the large numbers and sizes of trees involved in the full restoration is expected to have negative aesthetic impacts. In addition to the aesthetics, there are hazards to recreationists associated with downed woody debris and snags that would be created by the tree removal. Smoke from the proposed prescribed burning can create health and aesthetic problems, and 2–3 foot firebreak lines and road construction may also have negative aesthetic effects. In addition, fears were expressed that the prescribed burn would get out of control and destroy private property.

The San Francisco Peaks are considered a sacred place for many of the Native American Tribes in the Southwest, so a consultation process was set up with 13 Tribes to ensure that potential cultural impacts of the proposed restoration were considered. Tribes were also invited to Hart Prairie for a tour and further discussion of the proposed watershed restoration. During this consultation process, it was revealed that the removal of the larger trees and the total number of trees involved in the restoration have raised concerns. In addition, concerns were expressed regarding the time of year when restoration activities would occur; the timing of such activities should be compatible with Native American religious beliefs.

Many concerns were raised that the amount of open water available for the diverse species of wildlife at Hart Prairie will be reduced with the removal of stock tanks. In addition, there was a great deal of opposition to the removal of Snowbowl tank because this tank is used for education and is a favorite spot for recreationists.

Incorporating Ecological and Nonecological Concerns

In order to address concerns of the large numbers and sizes of trees to be removed, the prairie meadow encroached by the pines was divided into subwatersheds that were prioritized in terms of their potential contribution to water loss downslope. A subwatershed directly upslope from the riparian community was identified as the first priority to be thinned. This area represents less than one-third of the total prairie meadow encroached by the pines, resulting in restoration that will remove significantly fewer trees and will therefore result in fewer aesthetic and safety impacts. Some of the larger trees will be girdled to create snags rather than removed to provide wildlife habitat. In addition, computer-generated, visual simulations are being developed by NAU from the mapping data to compare the current viewshed to viewsheds following different thinning treatments. Locations, numbers, and sizes of trees that would be removed will be included in the visualization of the different thinning treatments. These simulations can be used to help design the tree removal to minimize negative aesthetics and to inform the public about what is being removed and what the viewshed will look like. Finally, "fuzzy" boundaries will be created between the restored area and the areas of the meadow not being restored to further minimize visual impacts. Some of the tanks will be reduced in size rather than removed in order to maintain the wildlife benefits they currently provide, and a "wildlife drinker" will be installed in the area to provide access to water lost due to the stock tank removal. Despite the fact that Snowbowl tank prevents water from entering the channel directly above the Bebb willow community, it will most likely not be modified due to public sentiment not wanting this tank to be removed. Finally, the timing of restoration activities will be compatible with Native American religious beliefs, and attempts will be made to burn when recreational and aesthetic impacts would be minimized.

As a result of addressing nonecological concerns associated with the full ecological restoration, the proposed activities will not fully restore the prairie meadow. However, incorporation of nonecological concerns will still permit restoration of one-third of the meadow and this should provide significant additional water to the downslope Bebb willow riparian community. Prerestoration data are still being collected, and the Tribal consultation and public comment processes are ongoing. Once these processes and the restoration activities are completed, the postrestoration monitoring will tell us whether further restoration is needed to provide additional water flow to the Bebb willow riparian community.

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Economics and Utilization

Financial Results of Ponderosa Pine Forest Restoration in Southwestern Colorado

Dennis L. Lynch

Abstract—From 1996 to 1998, the Ponderosa Pine Partnership conducted an experimental forest restoration project on 493 acres of small diameter ponderosa pine in the San Juan National Forest, Montezuma County, Colorado. The ecological basis and the financial analysis for this project are discussed. Specific financial results of the project including products sold, revenues collected, harvesting costs incurred, and profits or losses realized are reported. Restoration costs are also compared with fire suppression costs experienced both in Colorado and nationwide. Using data collected since the conclusion of the project, the future potential for financing forest restoration in southwestern Colorado is explored.

Introduction

This paper presents specific financial results from a forest restoration project conducted by the Ponderosa Pine Partnership in southwestern Colorado. This Partnership was created by a group of organizations to explore potential benefits of forest restoration and includes Montezuma County, San Juan National Forest, Ft Lewis College, Colorado Timber Industry Association (CTIA), Colorado State Forest Service, College of Natural Resources, and Cooperative Extension at Colorado State University (CSU). The Partnership was formed because the County and the San Juan National Forest were concerned about several problems present in the 183,000 acre ponderosa pine forest northeast of Cortez, CO. These problems included:

- Potential for insect and disease outbreaks
- Risk of catastrophic forest fires
- Decline of small forest product businesses
- Recognition that ponderosa pine forests in this area are not within their range of natural variability and are probably not sustainable in their current condition (Romme 1999).

Dr. William Romme, professor in the Department of Biology at Ft Lewis College, Durango, CO, had principal responsibility for ecological data collection, development of an ecological prescription for this experimental forest restoration project, and subsequent monitoring. In his previous research in southwestern Colorado (Montezuma, La Plata, and Archuleta Counties) Romme (1999) reached generally similar conclusions as research from Arizona and New Mexico (Cooper 1960; Covington and others 1997; Dahms and Geils 1997; Fulé and others 1997). He found that in this forest area there were seven times more trees per acre today than in 1900. Specifically, there are 280 to 390 trees per acre today versus 40 to 50 trees per acre in 1900. Stands were characterized in 1900 by clumps of trees. These clumps were typically one-tenth to one-quarter acre in size, while today the clumped pattern of stands has largely been lost. Stump diameters of trees living in 1900 averaged 27 inches and ages were likely to reach 300 years. Today 95 percent of the trees are 16 inches in diameter at breast height or smaller and 110 years or less in age. Additionally, less than 20 percent of today's stands have pine regeneration. Mortality is also disproportionate, occurring in the very young and oldest trees.

He also discovered that fire frequencies have changed. For example, in the forest area, of which this study is a part, he found that during the period 1685–1872, the interval between fires ranged from 5 to 20 years, with a median of 12 years. In another part of the area he found that between 1729 to 1879 the range in fire frequency was 2 to 31 years with a median of 10 years. Since 1879, there has not been a fire that created a fire scar in most of the area.

On the basis of his research, Dr. Romme suggested that the current forest could be modified and restored to conditions similar to pre-1870 structure and processes. He designed an ecological prescription that retained large trees, developed the clumped nature of tree groups within the forest, developed glades and parks between tree clumps, reduced shrub understory while increasing the grass-forb component, and reintroduced low intensity fire as a process.

A dominant concern of this study was to ensure that the ecological prescription controlled implementation. It was also obvious that forest restoration could not proceed unless costs were paid. As previously stated, no government money was available to implement this study. Further it was highly unlikely that tax dollars would be appropriated to pay for forest restoration in any situation except, perhaps, those most severely threatening to life and property. Even then, the material to be removed would have to be disposed of in some way. Leaving thinning slash in the forest would only increase fuel loading and the probability of intense fires.

Sackett (2000) commented that prescribed burning alone has been unable to remove the volume of existing material in the Chimney Springs, AZ, area forest to a healthy state, let alone presettlement conditions. Recent catastrophic fires in the Southwest also attest to the danger associated with the use of prescribed fire alone. Therefore, this financial

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study was conducted to determine harvesting costs and potential revenues available from products removed in this experimental attempt at forest restoration.

It is also important to recognize that forest restoration projects are very different from typical timber sales. In forest restoration, the material removed is the least desirable from a product standpoint and may contain a predominance of small diameter trees that cost more to remove than they are worth. A large body of timber harvesting research, from the United States and internationally, clearly documents that manual or mechanical removal of small diameter trees is expensive. Thus the basic premise of forest restoration, which requires removing small trees and leaving the largest, presents a difficult financial problem. This problem is further complicated in Montezuma County because there are only a few local industries available to process small diameter material. Therefore, long haul distances are necessary to take small material to processing points. Forest Service procedures that required treating forest restoration activities as timber sales and an appraisal process that did not accurately estimate values of the material removed made the problem even more complex.

Carla Garrison Harper, representing Montezuma County, served as project coordinator for the Ponderosa Pine Partnership. The Partnership completed forest restoration on six representative sites in the Mancos-Dolores District of the San Juan National Forest. However, because financial data were incomplete and inconclusive on one unit, this report discusses financial results for five of the areas completed. These five study units totaled 492.6 acres. An overview of the ecological prescription, restoration work, costs, and subsequent monitoring is included in a paper by Lynch and others (2000). It does not, however, provide the cost detail presented here nor a comparison with wildfire costs.

Three key points should be emphasized:

- Dr. Romme's ecological prescription was formed on the basis of his previous research. The project objective was to restore the forest toward the ecological balance he envisioned.
- The Partnership did not have any Federally appropriated money to use to subsidize the removal of small diameter trees.
- After the largest trees had been retained in clumps, as specified by the ecological prescription, the excess material was removed and sold. The financial results presented here are for that undesirable and excess material. The project was not designed to make money, but to determine if forest restoration work based on good ecology could pay its way.

Description of Project Implementation

The five units selected for this study are representative of forest conditions in this area and are quite similar in regard to soil, slope, terrain, and access to haul roads. To implement forest restoration it was necessary, under existing Forest Service procedures, to transfer Dr. Romme's ecological guidelines to a timber sale format. Phil Kemp, forester with the Mancos-Dolores District, had primary responsibility for applying Dr. Romme's research and guidelines to develop and implement ecological restoration prescriptions. The retention of larger trees, and the clumped nature and spacing of tree groups were key considerations. Phil developed on-the-ground silvicutural prescriptions, and trained and supervised crews to mark the trees to be retained in the forest. He coordinated with Dr. Romme to develop a marking guide that would achieve the desired forest condition. Trees to be retained on the site were marked with blue paint. Following marking of each unit in this study, a cruise was conducted to determine the amount and character of the material to be removed. The Forest Service, using this information, prepared administrative timber sale contracts.

Montezuma County purchased each timber sale and arranged to pay for the wood removed and associated costs such as road rock replacement and slash disposal. Initially, costs to conduct the ecological and economic studies were also to be charged to the timber sale. However, research funds from other sources were secured to pay for the studies. Timber volumes and subsequent payments were initially based on cruise data that ultimately proved to be inaccurate. Cruise estimates tended to overestimate the amount of sawtimber available for removal and underestimate the volume of POL material to be removed. Such faulty estimates are of critical concern to both the Forest Service and small businesses. Cruise estimates of timber are not only costly to perform, but in forest restoration sales have the potential to be very erroneous because there is such great variation in the material to be removed. Therefore, weight samples were taken from logs and truckloads of logs were scaled after weighing to determine weight-volume relationships. Subsequent payments were based on weight, which proved to be more accurate and fair to both seller and purchaser.

CTIA coordinated with local industry and the San Juan National Forest to determine which industry firms were to be involved in the project. Ragland and Sons Logging of Dolores, CO, purchased the five timber sale units from Montezuma County and had primary responsibility for harvesting and marketing material removed from the forest. The material removed was used to produce a variety of wood products, some of which were experimental attempts to use small diameter trees. Every effort was made to market logs to processors in or near Montezuma County, but some small diameter material simply could not be processed locally. The bulk of this small material was taken as waferwood logs to the oriented strand board plant in Olathe, CO. At the plant, logs are debarked and processed through a waferizer, also known as a flaker, to create wafers or strands for oriented strand board production. Sawlogs from the sale were processed at the Ragland's Stonertop sawmill near Dolores into timbers and lumber. Material brought to the mill was appraised to all other possible mill sites and the value of the logs estimated on that basis.

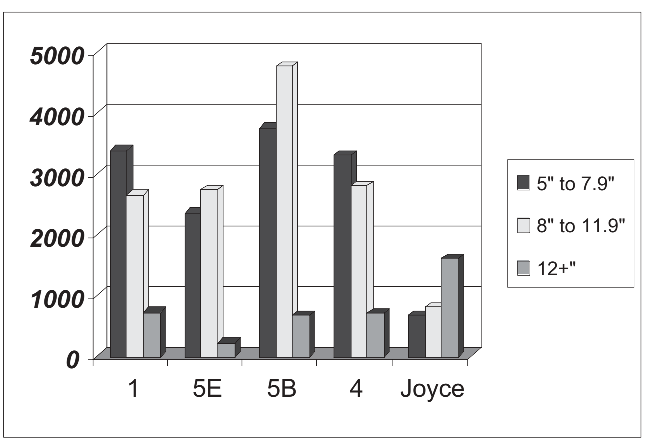
Ragland and Sons Logging utilized both manual (chainsaws) and mechanical (JD 743 Harvester) equipment for felling, limbing, and bucking (FLB) in this project. Slash was lopped and scattered in place for prescribed burning. Skidding was done with rubber-tired grapple skidders (CAT 518 and JD 540). Logs were loaded with a knuckle boom loader on conventional log trucks for transportation by product type to various mills. All trucks were weighed on scales to determine the amount of material removed.

We followed a program of adaptive management. As each unit was completed we reviewed the results to ensure that appropriate changes were made in the next unit to improve ecological accomplishment (Lynch and others 2000). Following logging, broadcast burning of slash by the Forest Service was used to reintroduce fire to the forest. Prescribed burning of these areas in later years will be done at intervals that are historically characteristic of these forests. This should result in low intensity fires in the future. Ecological monitoring of these areas is continuing.

Financial Studies and Results

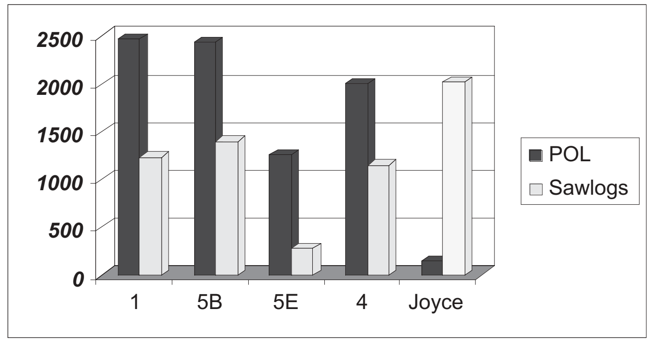
Study Methodology

In this financial study, we recorded every tree cut by diameter class. Distributions of trees removed by unit and diameter class are shown in figure 1. Every log skidded and every log loaded onto a truck were also counted. All trucks were weighed to determine the tons hauled by product type. The relationship of products other than logs (POL) to sawlog material by unit is shown in figure 2. Samples were taken from logs and sample truckloads were scaled by a Forest Service scaler to determine weight to volume relationships. Complete records of time and costs for all aspects of the operation were collected using established logging cost collection techniques. These included all costs associated with labor, equipment, oil, gas, diesel, supplies, field and office administration, and transportation involved in felling, limbing, bucking, skidding, loading, slash treatment (except prescribed burning), roads, and hauling logs. These costs are specific to the company participating in this study, so itemized cost information is not presented in order to respect the privacy of the firm. Revenues for each product were recorded



LEGEND
Left Axis — Number of Trees
Bottom Axis — Units
Bars- Diameter (DBH) Classes

Figure 1—Numbers of trees by unit and diameter class.



LEGEND
Left axis — Tons
Bottom axis — Units
Bars- POL or Sawlogs

Figure 2—Relationship of products by unit (tons of POL and tons of sawlogs per unit).

and profit or loss calculated. The efficiency of the logger was compared to USDA Forest Service regional average harvesting costs and harvesting cost studies in New Mexico, Arizona, Oregon, and Canada. Correlation analysis was used to compare the characteristics of units 1, 5B, 5E, and Joyce with unit profit. Initial analysis indicated that factors associated with the cold temperatures and deep snow in Unit 4 inflated costs and would bias results. Only correlations above 0.8 (+ or –) were considered strong enough to merit consideration.

Unit Results

Unit 1—Also known as the Smoothing Iron unit, Unit 1 is located on Haycamp Mesa and is 125 acres. A total of 6,754 trees were removed from the unit during August 25 to October 3, 1995. Figure 1 shows tree distribution by diameter class. In this unit, 1 cubic foot of ponderosa pine log at the landing weighed 71.78 lb with a standard deviation of 5.11 lb

The removal of trees resulted in the delivery of 1,232.76 tons of sawlogs to Stonertop sawmill in Dolores, 2,322.75 tons of waferwood logs to the Louisiana-Pacific oriented strand board mill in Olathe, and 145 tons of pine excelsior logs to Western Excelsior in Mancos. This was a relationship of 33.3 percent sawlogs to 66.7 percent products other than logs (POL) by weight. It amounted to a removal of 29.6 tons per acre. Any tree 8 inches d.b.h. or larger that would make a sawlog was taken to the sawmill. Other trees were taken for waferwood logs or pine excelsior logs. Table 1 lists unit revenue, and cost to the logger as well as profit or loss data per ton.

There was a total profit (total revenue minus actual costs) for the unit of $440.48. Note that this is total profit before taxes. No previous allowances for profit and risk or income taxes were made during cost collection or analysis.

The effort to develop pine excelsior was experimental. It was first believed that pine might be added as a supplement to aspen excelsior for certain applications and was, therefore, worth trying as a potential local outlet for small, poor quality logs. However, pine contains terpenes, a compound

Table 1— Unit summary of revenue, costs, and profit (loss) per ton.

Product	Revenue	Cost[a]	Stumpage		Profit (loss)
Unit 1—Smoothing Iron					
Waferwood	$31	$31.06	+ $1.51	=	(–$1.57)
Excelsior	$17	$23.35	+ $1.51	=	(–$7.86)
Sawlogs	$30.52	$22.64	+ $3.64	=	$4.24
Unit 5B—Found Park					
Waferwood	$31	$30.84	+ $1.32	=	(–$1.16)
Pulpwood	$18.82[b]	$16.87[b]	+ $1.32	=	$0.63
Posts-poles	$24.00	$22.81	+ $1.32	=	(–$0.13)
Sawlogs	$30.52	$20.86	+ $3.21	=	$6.45
Unit 5E—Found Park					
Pulpwood	$24.04[b]	$21.77[b]	+ $1.14	=	$1.13
Posts-poles	$24.00	$27.55	+ $1.14	=	(–$4.69)
Waferwood	$32.00	$35.37	+ $1.14	=	(–$4.87)
Sawlogs	$30.52	$30.52	+ $8.92	=	(–$3.76)
Unit 4—West Trail —“Snow Unit”					
Waferwood	$31	$36.20	+ $1.48	=	(–$6.68)
Sawlogs	$30.52	$26.42	+ $4.63	=	(–$0.53)
Joyce Unit					
Waferwood	$32	$27.10	+ $ 2.32	=	$2.58
Sawlogs	$30.52	$17.22	+ $ 5.21	=	$8.09

[a]Includes all costs described in text associated with logging and hauling logs to mill.
[b]Note: pulpwood revenue and costs are at the landing, not to the mill.

that has a characteristic pine odor and, when it occurs in sufficient concentrations, can be used to produce turpentine. Terpenes can also attract certain insects, such as termites, and in sufficient concentrations can interfere with the viability of seeds. Further, pine excelsior proved to be brittle and difficult to ship. These characteristics reduced the marketability of pine excelsior and, at this time, pine excelsior does not appear to be a viable product.

The effort to use ponderosa pine for waferwood was also experimental to a degree. Usually, wood materials of lighter density, such as aspen, are sought for waferwood used in the production of oriented stand board (OSB). However, Louisiana-Pacific at Olathe, CO, was willing to mix ponderosa pine from this area with other species as a test. This provided an excellent outlet for poor quality, small diameter material that would have been impossible to market elsewhere. The long haul distance of 160 miles to the plant contributed to losses for this product.

Unit 5B—This unit is also known as one of the Found Park units, and is located off Forest Road 521 on Ormiston Point. It is 108 acres. A total of 9,177 trees were removed from the unit during June 26 to November 11, 1996. Figure 1 shows the distribution of trees cut by diameter class. Ponderosa pine sawlogs at the landing weighed 72.81 lb per cubic foot with a standard deviation of 2.0 lb. The POL logs weighed 76.56 lb per cubic foot at the landing with a standard deviation of 4.03 lb. The removal resulted in the delivery of 1,397.44 tons of sawlogs to the Stonertop mill, 1,899.86 tons of waferwood logs to the Louisiana-Pacific mill, 353.07 tons of wood for posts and poles to Cannon Forest Products in Cortez, and 177.54 tons of pulpwood sold at the landing and ultimately delivered to Stone Container in Snowflake, AZ. This was a relationship of 36.5 percent sawlogs to 63.5 percent POL by weight and a removal of 35.4 tons per acre. Table 1 summarizes the per ton financial results by product for this unit. The total profit for the unit was $6,875.60.

Shipment of pulpwood to Snowflake, AZ, was another experimental attempt to evaluate potential markets. In this case, a trucker from Cortez was hauling house logs from an area near Snowflake to a location in Colorado. He needed a back haul to Snowflake and this provided an opportunity to explore this market. However, he estimated his costs resulted in a loss to him of approximately $8.00 per ton for the haul. Thus, this pulpwood market proved to be uneconomical to the trucker, although the revenue at the landing to the logger was near the break-even point.

Post and pole logs were sorted at the landing and trucked separately to Cannon Forest Products. It takes a reasonably straight, defect free, and uniform tree to yield a satisfactory post or pole. Forest restoration material, however, often contains large volumes of deformed, poor quality, small diameter material that does not fit post and pole specifications. Sorting out higher quality material can introduce additional costs. Also, the quantity of post and pole logs ultimately overwhelmed the small local mill capacity to buy and process the material.

Unit 5E—The second of the Found Park units is bisected by Forest Road 521 on Ormiston Point. It is 65 acres and almost equally distributed on either side of the road. A total of 5,307 trees were removed from this unit during May 29 to June 27, 1997. Figure 1 shows the distribution of trees cut by diameter class. In this unit, ponderosa pine logs at the landing weighed 71.68 lb per cubic foot with a standard deviation of 6.04 lb. This removal resulted in a delivery of 274.85 tons of sawlogs to Stonertop mill, 881.07 tons of waferwood logs to Louisiana-Pacific, 269.64 tons of wood for posts and poles to Cannon Forest Products, and 101.9 tons of pulpwood to Stone Container in Snowflake, AZ. This is a relationship of 18 percent sawlogs to 82 percent POL by weight and was a removal of 23.5 tons per acre. Table 1 presents the per ton financial data for this unit.

The loss for the unit was $6,473.71. This loss does not include any allowances for profit, risk, or income taxes. Factors contributing to the loss were a high stumpage rate for sawlogs coupled with the small amount of sawtimber removed from the unit. The high stumpage price resulted from cruise data that indicated there were more sawlogs in the unit than actually existed. Thus, the logger paid for material that did not, in fact, exist. This was a key reason for using a weight basis for subsequent sales of this low value material.

While pulpwood again appears to be profitable in this unit, the trucker had delayed in calculating his costs only to discover losses similar to those discussed in the narrative for Unit 5B. It is likely that the large amount of small material of poor quality created additional sorting and handling for post and pole logs, thus increasing costs.

This unit illustrates the financial losses that occur when there is a small percentage of sawlogs to POL material. It simply costs more to harvest small diameter material and when sawlogs are not present, overall costs increase. In addition, the stumpage cost for sawlogs was too high and that added to losses.

Unit 4—Also known as the West Trail unit, this unit is located off Forest Road 524. We refer to it as the "snow unit" because it was logged from November 11, 1996, to January 31, 1997, when the area was besieged with very cold weather and deep snows.

The unit is 95 acres. A total of 6,824 trees were removed from the unit. Figure1 shows the distribution of trees cut by diameter class. Ponderosa pine logs in this unit weighed 72.43 lb per cubic foot with a standard deviation of 4.6 lb. The removal resulted in the delivery of 1,152.51 tons of sawlogs to Stonertop Mill and 2,008.57 tons of waferwood to Louisiana-Pacific. This is a relationship of 36.5 percent sawlogs to 63.5 percent POL by weight. This was a removal of 33.27 tons per acre. Table 1 presents per ton financial results for this unit.

This resulted in a loss of $14,028.08. Factors contributing to this loss were increased costs from logging in deep snow and somewhat higher sawlog stumpage prices. The POL stumpage price was also near that of unit 1 and higher than units 5B and 5E. It is important to note that all of the other characteristics of this unit indicate it would have been a profitable unit, like unit 5B, if it had not been logged in deep snow. While it is often desirable to encourage winter logging of areas, this sale illustrates the importance of scheduling forest restoration sales during periods when costs will not be increased by weather conditions. It also underlines the need to provide flexibility for contract extensions when such conditions arise.

Joyce Unit—This unit is located in the Boggy Draw area, which is a favorable site for tree growth. It is 99.6 acres. A total of 3,101 trees were removed from this unit. Figure 1 shows the distribution of trees cut by diameter class. In this unit, ponderosa pine sawlogs weighed an average of 78.82 lb per cubic foot (the mean of two scaled loads). A total of 2,021.24 tons of sawlogs were delivered to the Stonertop mill and 142.46 tons of waferwood logs to Louisiana-Pacific. This is a relationship of 93.4 percent sawlogs to 6.6 percent POL by weight. A total of 21.72 tons per acre were removed from the unit. Table 1 presents the per ton financial data for this unit.

The total profit for this unit was $16,719.38. This unit is, from an economic standpoint, at the upper end of profitability for a forest restoration sale. It is at the opposite end of the spectrum from Unit 5E where the material removed was predominately POL. The quantity of larger trees on the site when the ecological prescription was applied resulted in the removal of more sawlog material than in the other units. The Joyce unit restoration would resemble a traditional timber sale if it had removed the largest trees from the site. However, the largest trees were left on the site in adequate densities and trees in excess of the prescription were removed.

Harvesting costs declined overall as a result of the removal of a larger proportion of sawlogs in this unit. The higher POL costs, which had been typical up to this point, were offset by the lower harvesting costs of sawlogs. In addition, the loggers had learned from our reports that POL was resulting in serious losses. In this unit they did everything possible to salvage material for sawlog use.

This unit illustrates that opportunities can arise to balance costly units (like 5E that have high proportions of POL), against units like this one to achieve break-even or profitable forest restoration projects. This may require combining scattered units at a variety of locations into one sale or adopting landscape scale restoration in order to modify costs.

Summary of Key Results

This forest restoration project incorporated five sale units covering a total of 492.6 acres and the removal of 31,163 trees (63.3 trees per acre) for use as sawlogs (6,075.8 tons), waferwood (7,254.71 tons), and other products such as posts and poles, pulpwood, and pine excelsior (1,047.15 tons). The loggers' costs in this study, for each step in the harvesting process except hauling, were less than USFS Regional averages and very comparable to cost studies from other areas. Forest Service and independent observers rated the restoration logging conducted by Ragland and Sons as excellent work.

Table 2 records a summary of profit and loss by unit. Total profit to the logger of $3,533.67 was 0.81 percent (less than

Table 2—Unit profit or loss.

Unit	Profit or (loss)	Per acre profit (loss)
1	$ 440.48	$ 3.52
5B	$ 6,875.60	$ 63.66
5E	$(6,473.71)	$(99.60)
4	$(14,028.08)	$(147.66)
Joyce	$ 16,719.38	$ 167.86
Total	$ 3,533.67	

1 percent) on gross revenues of $434,645.54. This approximates a break-even situation that is better than suffering a loss, but is hardly a model for a sustainable business venture. Usually, it is appropriate to expect profit and risk allowances of 10 to 15 percent for logging in this region.

Sale characteristics with the strongest correlation to unit profit were percent of sawtimber removed by unit (0.93356), number of trees per acre 12 inches d.b.h. or larger (0.92516), and cubic foot volume per tree (0.81464). Percent of POL was negatively correlated with unit profit (–0.93536).

Harvesting costs were also compared to unit profit. Individual harvesting cost elements that were most negatively correlated included felling, limbing, and bucking (–0.99989), loading of POL (–0.91372), and hauling of POL (–0.83338). Total POL costs were also negatively correlated (–0.88099).

The market conditions that existed at the time of this study were ponderosa pine sawlogs selling at $240 per 1,000 board feet, waferwood logs selling for $31 to $32 per ton, and post and pole logs selling for $24 per ton. Under those conditions, the analyses conducted in this study suggest that future restoration projects *in this area of Colorado* can be near the financial break-even point when material removed meets the following criteria:

1. Sawtimber should be at or above 40 percent of the total weight removed and POL should be at or below 60 percent of the total weight of material removed. Note the relationship of sawlogs to POL in figure 2 and then relate that to profit and loss in table 2. Utilizing trees in the 8 inch to 11.9 inch diameter class for the highest value product possible is critical to profitability. Payments for material removed should be based on weight scaling and not cruise data, given the variability of this material.
2. Six or more sawtimber trees 12 inches d.b.h. or larger should be available for removal per acre with an average cubic foot volume of 12 cubic feet per sawtimber tree removed. Given that forest areas are being reduced from 200 to 350 stems per acre down to half that number or less, this should not be viewed as a difficult problem in landscape scale projects. However, when units do not meet these criteria, they should be combined with other units to create break-even projects.
3. POL harvesting costs must be carefully managed. Note the summary cost distribution for waferwood logs in figure 3. Attention to cost control and record keeping is important when dealing with low value, small diameter material.
4. Felling, limbing, and bucking (FLB) costs must be kept as low as possible. Note the proportion of FLB costs to all POL costs in figure 3. In an analysis of the FLB cost per cubic foot of wood removed, mechanical felling was more expensive in all diameter classes in this study than manual felling. This is due both to the type of harvester used in this study and the scattered nature of the trees being cut. Also it is clear that forest restoration logging in adverse weather conditions should be avoided.
5. Loading and hauling costs for POL must be reduced. Note that the proportion of costs attributable to loading and hauling in figure 3 were 50 percent of all costs. It also suggests the need to strengthen and/or develop local manufacturing businesses that could use small diameter material and decrease hauling distances.
6. Stumpage prices for POL should be zero or less in some cases. Note in table 1 that the loss per ton for POL is often nearly equal to the stumpage paid for it. Also, as a general rule, the revenue from the lowest quality material, such as waferwood logs, must not be below $31 per ton to break even.

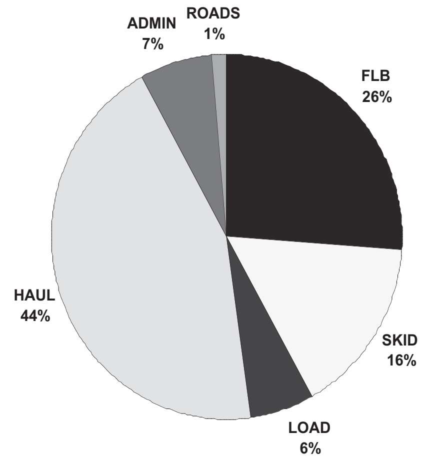


Figure 3—Cost distribution for waferwood harvesting.

Potential to Improve Profitability

Information developed from this study and other market studies by the author was used to persuade the USFS to revise its appraisal process for forest restoration projects in Region 2. Stumpage prices for ponderosa pine POL in Region 2 restoration projects may now be appraised at zero or even a negative value, because the basis for the project is ecological improvement of forest stands and POL material typically costs more to harvest than it is worth. In other words, this is recognition that POL is a liability, not a valuable product. This change would, for example, reduce the loss in Unit 5E from $6,473.71 to $1,204.28 and increase the profit in Unit 5B from $6,875.60 to $9,476.36. Thus, future restoration units with low value POL might actually be financially feasible if new appraisal policies are used.

In addition, tremendous demand for OSB in housing and commercial construction improved the market for waferwood logs immediately following this study. During the study, waferwood log prices paid at the mill were either $31 or $32 per ton. March 2000 waferwood log prices being paid ranged between $35 and $37 per ton. An average price of $36 per ton would have improved the profitability of this project to 7.3 percent. Similarly, a down turn in the OSB market or the ponderosa pine lumber market could adversely affect restoration projects. Thus, market fluctuations are important to consider when restoration projects are being planned.

We gained some insight into ways to reduce harvesting costs. Figure 3 shows the summarized cost distribution for one product, waferwood logs. Note that hauling and loading costs comprise a significant cost center as does felling, limbing, and bucking. Hauling costs could be reduced using more efficient trailers that better accommodate small diameter material. Research should be directed at the potential use of "hay rack" trailers or short log trucks with pup trailers commonly used for hauling pulpwood. Felling and bucking costs could probably be improved with mechanized equipment designed for harvesting small material. However, this requires further experimentation. If FLB could be mechanized efficiently there might also be some improvement in skidding costs. Logging should be planned to avoid adverse weather that increases costs.

We also have learned to avoid some products, such as pulpwood, that did not return sufficient revenue to cover costs or excelsior logs that are not technically feasible. Studies to improve utilization of wood wastes at the mills and the development of new products are also important to future profitability and sustainability of restoration projects.

Comparison of Forest Restoration Costs to Forest Fire Costs

In the previous analysis, the principal focus was on the identification of factors that could produce break-even or profitable forest restoration projects. This focus resulted from concerns that appropriated tax money would not be available for service contracts to accomplish restoration work.

However, there are important nonmarket benefits from forest restoration that might merit investment of public funds. One of these benefits is the reduction of catastrophic forest fire hazard. To analyze the economic benefit of fire hazard reduction, the costs or profit per acre from the five units were calculated and are presented in table 2. These costs were subsequently compared with local, State, and national fire suppression costs per acre.

The nearest large-scale fire to the study area was the 1996 Disappointment Fire that occurred in the pinyon-juniper forest type north of Dolores, CO. San Juan National Forest records show that this fire burned 3,840 acres and cost $992,000 to suppress. This is a per acre suppression cost of $258.33. That cost is almost double the worst-case restoration cost that occurred in Unit 4 (which had extraordinary snow costs and high stumpage costs). It is almost three times more expensive than Unit 5E, which had extensive amounts of POL and the highest stumpage costs in our study. Recall that the other restored units recorded profits. Also, all five restored units still have forests with large trees, pleasing esthetics, desirable wildlife habitat components, and ecologically functioning processes. The Disappointment Fire area, on the other hand, lost tree cover, esthetics, and wildlife habitat, while also requiring additional rehabilitation costs to reduce erosion and reestablish forest vegetation. Unfortunately the amount of those additional rehabilitation costs were not recorded.

From 1991 to 1996, U.S. Forest Service wildfires in Colorado cost an average of $482.78 per acre to suppress (USFS 1996). Property damage and rehabilitation costs for these burned areas were not recorded. During the period 1977 to 1992, national large fire costs averaged $570.98 per acre (USFS 1995). Again, property damage costs and rehabilitation costs for these fires were not recorded.

Suppression, property damage, rehabilitation costs, and subsequent flooding costs were studied for the 1996 Buffalo Creek fire that occurred in ponderosa pine near Denver. All costs after fire and flooding amounted to $679.47 per acre and are expected to climb to $2,000 per acre because of continuing flooding and water quality problems (Dennis 2000).

It can be argued that it may not be fair to compare fire suppression costs with forest restoration costs because no one can predict when and where a catastrophic fire will occur. However, because research in ponderosa pine forests suggests that fuel conditions for catastrophic fire are widespread and that fire can be expected to occur at some time, the comparison is not far fetched.

An estimate of expected fire suppression costs in restored forests versus suppression costs under current forest conditions was also developed as an example. Dr. Romme studied fire frequencies in the Five Pine Canyon area west of Units 5B and 5E. He found that fire frequencies during the past 30 years in the Five Pine Canyon area were similar to fire frequencies that occurred prior to Euro-American settlement. The Doe Canyon Fire occurred recently in this vicinity, just north and west of Units 5B and 5E, and serves as an interesting cost comparison. It covered 600 acres and fire observers noted that the fire was of low intensity and burned like a prescribed fire. Some of the area had been previously thinned, there are some open areas mixed with ponderosa pine forest areas, the terrain is fairly level, and it is likely that fire frequencies there have been similar to those identified by Dr. Romme in the adjacent Five Pine Canyon area. Fire suppression, in this case, consisted mostly of burning out from the roads. San Juan National Forest records show that the cost to suppress this fire totaled $7,500 or $12.50 per acre. Some small ponderosa pines were destroyed, but overall the forest remained intact. This better resembles the lower intensity type of fire expected in restored forests and may well suggest that the costs of future fire suppression in such areas will be much lower.

Commentary

Given the results of this project, I think two questions are pertinent to the future implementation of forest restoration projects in the Southwest. These are:

- Who is going to do the work and why should they do it?
- Why should appropriated tax money be used to finance forest restoration if well designed projects could pay for themselves?

It is clear that an ecological prescription must control project implementation to achieve the goals of forest restoration. Forest restoration projects are not traditional timber sales. Restoration of the forest to its inherent characteristics and processes is, after all, the basic purpose for these projects.

It is also just as clear that forest restoration is a risky, financially expensive proposition. In Colorado all of the forest industry businesses, except two, meet the qualifications for small businesses. Nearly all are family owned and operated, similar to the Ragland and Sons Logging Company studied in this report. It is presumptuous, therefore, to expect small businesses that characterize the forest industry of Colorado and the Southwest to make a silk purse out of a sow's ear, or low quality, small diameter material into a valuable product. Break-even projects or government subsidized programs may accomplish ecological objectives in the short term, but they will never result in the investments necessary to improve efficiencies, develop new products, and provide sustainable forest stewardship. Therefore, it is important to approach these projects with long-term financial sustainability in mind, providing loggers with incentives to function as forest stewards, providing a variety of restoration services such as watershed rehabilitation and wildlife improvement projects. By paying attention to factors that influence profitability, such as the use of new appraisal guidelines that recognize the low quality of this material and alertness to market fluctuations, agencies can improve the playing field for small businesses. A consistent supply of wood from well-designed projects with long-term contracts could encourage investments to improve efficiencies associated with harvesting and product manufacturing. To further aid in this, government could provide incentives in the form of tax advantages, low cost loans to create and strengthen small businesses, and applied research to improve harvesting efficiency and product development. For example, forest product studies currently under way through the Four Corners Initiative (Reader 1998) are intended to support profitability by increasing local utilization of material removed, improving manufacturing efficiencies, and creating new value-added products.

The lessons learned from our experimental forest restoration project are encouraging enough to recommend further projects, I hope, at the landscape level. I also recognize that other forest areas are so beset with worthless material that subsidies will have to support restoration.

Finally, restoration projects cost much less than forest fire suppression. Building a forest restoration infrastructure makes more financial sense than spending tax money fighting catastrophic forest fires.

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Cost / Effectiveness Analysis of Ponderosa Pine Ecosystem Restoration in Flagstaff Arizona's Wildland-Urban Interface

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Abstract—Ponderosa pine ecosystem restoration in Fort Valley (located east of Flagstaff, Arizona) has been proposed as a method of restoring ecosystem health and lowering the risk of catastrophic wildfire in Flagstaff's wildland-urban interface. Three methods of harvest are being used to carry out restoration treatments: hand harvesting, cut-to-length harvesting, and whole-tree mechanized harvesting. This paper presents a theoretical application of a cost / effectiveness analysis to aid in recommendation of an optimum method of harvest for restoration treatments. Harvest methods can be compared on the basis of ratios of harvest cost / effectiveness. Effectiveness in this approach is defined as a harvest method's ability to carry out restoration treatment with the least negative impact on residual stand damage, soil impacts, and fuel loading.

Introduction

Cost / effectiveness analysis is an economic tool that can help decisionmakers with limited financial resources select among alternatives to meet a predetermined objective. In the Fort Valley restoration project, the objective is an ecological restoration / fuel reduction treatment intended to reduce the number of small-diameter ponderosa pine (*Pinus ponderosa*) trees in Flagstaff, Arizona's wildland / urban interface. These small diameter trees inhibit growth of larger diameter trees and can act as fuel ladders in catastrophic stand replacing fires. Three treatment alternatives have been used to reduce current tree densities: hand-harvesting, cut-to-length harvesting, and whole-tree-mechanized harvesting. Each of these harvesting alternatives uses different mixes of equipment to obtain the given objective of restoration. Each mix of equipment has different costs for treating the stand and different environmental impacts on the stand after treatment. This paper reviews economic tools to aid in environmentally related decisions, and argues that cost / effectiveness analysis is superior to benefit cost ratio and impact analysis as an environmental decisionmaking approach for harvest method selection in the restoration projects. The paper introduces the Fort Valley restoration treatments as a case study, and covers the theoretical application of cost / effectiveness analysis for harvest method selection in the Fort Valley restoration treatments.

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Efficiency Analysis

Cost effectiveness analysis is one of many techniques that have been used to analyze the efficiency of environmental improvement or, conversely, to analyze the efficiency in avoiding the damage done by environmental degradation. The three techniques we review in this paper (benefit / cost ratios, impact analysis, and cost / effectiveness analysis) are all types of a benefit / cost analysis. Benefit / cost analyses are used to help policy and decisionmakers assess the desirability of specific government projects for environmental improvement, and to assess the desirability of new regulations to protect certain aspects of the environment from further degradation (Tietenberg 1996). The traditional economic efficiency analysis techniques are difficult to apply to environmental resources because of the lack of economic valuation for these goods and services (Tietenberg 1996). While demand curves for normal goods such as pizza or automobiles can be estimated from readily available market data, there are no such market data for environmental resources, because they do not pass through normal market transactions.

Benefit / Cost Ratio Analysis

Benefit / cost ratios represent an approach to quantifying the social profitability of a project. In isolated situations where the end product of a project has a quantifiable market value, a benefit / cost ratio analysis can help decisionmaking by ranking alternate activities designed to complete the project (Weaver and others 1982). Alternative activities are ranked by relative efficiency, and the analysis is a formalized attempt to obtain the maximum efficiency from a given level of funding. The benefit / cost method evaluates each alternative on a comparison of an earning rate index based on dollar benefits and per project costs. When used to select a set of alternatives under a budget constraint (in other words, capital budgeting problem), benefit / cost ratios ensure that the alternatives maximize benefits from a given level of funding. Benefit / cost ratios can also guide decisions between mutually exclusive alternatives. In this case, the benefit / cost ratios are compared and the alternative with the highest ratio is selected. To provide valid comparison of alternatives, benefit / cost ratio analysis requires that all benefits and costs of a project are quantified in monetary terms and included in the ratio.

Therefore, the major difficulty in using benefit / cost ratio analysis for comparing environmental projects is that not all benefits and costs are easily quantified in monetary terms. Even if market prices exist for inputs and outputs, these market prices do not always reflect the full social value of goods produced, or costs of resources used (Clifton and Fyffe 1977). Even greater difficulties arise when the resources in question do not have market values or are not handled by normal markets. For example, what is the dollar value of forest soil compaction caused by various harvesting alternatives?

The direct financial costs of carrying out a project are generally the most straightforward part of the analysis to quantify. These costs can be measured in terms of equipment, fuel, salaries, and so forth Indirect costs and intangible benefits estimation often pose many difficult problems. Indirect costs and intangible benefits are inputs and outputs not covered by normal markets, such as soil compaction, habitat loss, forest health, and aesthetics. In environmental projects, the majority of cost and benefits often are indirect or intangible. Aside from the sheer number of inputs and outputs that require nonmarket valuation, attempts to value each benefit can encounter problems concerning differing individual values of natural resources (Stevens and Sherwood 1982). Additional difficulties arise when the analysis is applied in the context of benefits and costs associated with avoided damages, for example, pollution control (Tietenberg 1996).

Many of the benefits associated with restoration treatments are avoided damages. For example, some of the benefits derived from forest restoration in Fort Valley are the avoided cost of fire suppression and avoided damages from wildfire. In theory, the benefits gained from various levels of restoration could be quantified in terms of damages avoided. Some of the avoided damage could be quantified since market goods such as fire suppression or property values are affected. This type of approach could be used in Fort Valley, but the valuation of such goods is a project within itself, and no such valuation has been completed. The Fort Valley treatments also affect many nonmarket benefits such as wildlife habitat and recreational opportunity that are not easily valued for use in benefit / cost ratios.

Impact Analysis

Impact analysis is an approach to quantifying the social, economic, and environmental consequences of various actions. Impact analysis is often used as a preliminary tool in developing environmental assessments. Unlike benefit / cost and cost effectiveness analysis, impact analysis makes no attempt to convert impacts into a one-dimensional measure to ensure comparability (Tietenberg 1996). Impact analysis also does not attempt to find an optimum of an economically efficient set of alternatives for accomplishing objectives. Rather it attempts to describe all impacts, quantitatively and qualitatively, as a way of evaluating proposed actions.

In the context of the Fort Valley restoration project, USDA Forest Service personnel completed an environmental impact analysis to evaluate the impacts of different treatments (in other words, silvicultural restoration prescriptions) on the landscape and surrounding community (USDA 1998). The focus of the analysis was to compare effects of different thinning intensities (in other words, different residual densities of trees per hectare) proposed in the restoration treatments. The analysis did not attempt to quantify all costs and benefits in terms of monetary units, and made no comparison of different harvesting procedures and equipment mixes to be used to carry out the treatments. While the impact analysis provides necessary information to determine if the overall project was worth undertaking, it did not provide information that would facilitate the selection of the optimal harvesting alternative.

Cost / Effectiveness Analysis

Cost / effectiveness analysis is a technique designed to assist a decisionmaker in identifying a preferred choice among possible alternatives. The analysis involves a comparison of alternative courses of action in terms of costs and effectiveness in attaining some specified objective (Quade 1967). Cost estimation is identical to direct cost estimation used in a benefit / cost ratio analysis. Direct costs are measured in terms of equipment costs, fuel costs, and salaries. Cost / effectiveness analysis avoids some of the difficulty in quantifying economic benefits of various alternatives by replacing dollar measurements of benefits and indirect costs with the concept of "effectiveness," where effectiveness is a comparative measure of an alternative's ability to meet project objectives.

In its simplest form, a cost effectiveness analysis is equivalent to a minimized cost approach. If all alternatives are equally effective in accomplishing project objectives the cost effectiveness ratio becomes (cost / 1). In this case the lowest cost alternative is optimal. Alternatively, in the case that different alternatives (for example, different harvesting techniques) have different impacts on the site, then effectiveness measures are selected to calculate these impacts.

A cost effectiveness analysis requires the analyst and decisionmaker to choose (1) a specific set of objectives, (2) a complete listing of the alternative solutions to be considered, and (3) acceptable measures of effectiveness in meeting the objectives. The first two decisions are common to all project analysis techniques. The third decision, selection of effectiveness measures, is unique to cost effectiveness analysis and makes this analysis subjective because of analyst assumptions. The analyst and decisionmakers must select attributes, which quantify the ability of an alternative to meet project objectives. In environmental decisions, these attributes are most often measures of damages avoided. To be useful the expected value (or effectiveness score) of attributes must vary across alternatives.

The virtue of the cost / effectiveness analysis lies in a more systematic and transparent use of judgment than any of its alternatives (Quade 1967). While effectiveness analysis may appear subjective in terms of analyst judgment of effectiveness measures, the analysis is presented so the decisionmakers can follow the analytic assumptions. Decisionmakers can choose which assumptions they agree with and arrive at their own conclusions. Other efficiency analysis techniques often mask assumptions in the assignment of dollar values to nonmarket benefits.

The selection of effectiveness measures also poses limitations that must be considered when using cost / effectiveness results. Results of an analysis (in other words, alternative rankings in terms of cost and environmental damage) must be interpreted by qualified and informed decisionmakers. Decisionmakers must understand the assumptions made by the analyst to arrive at the results, such as selecting attributes of effectiveness and assigning importance weights to attributes. The measurement of effectiveness is limited by selected attributes, and these attributes are often constrained by the cost of measurement and other factors that the analyst may not be able to take into account (Livingston and Gunn 1974). Thus, the results of effectiveness analysis are not intended to provide understanding and prediction, as are the results of science. Rather, the results are intended to serve primarily as recommendations or suggestions for selecting among courses of action (Quade 1967).

A cost / effectiveness analysis is most appropriate when used to choose a preferred alternative among possible alternatives that have the same project objectives. This analysis can rank possible alternatives that have a nonmarket nature to their benefits and costs, a situation that fits many restoration activities. The following description of the Fort Valley restoration project sets a base for a theoretical application of a cost / effectiveness analysis, which can be used as an aid in recommendation of an optimum harvest method.

Application of Cost / Effectiveness Analysis to Fort Valley Restoration Treatments

The Fort Valley restoration project is a series of ecological restoration / fuel reduction treatments, guided by pre-Euro-American settlement conditions (prior to 1870). The restoration treatments cover approximately 1,376 ha (3,400 acres), and incorporate various mixes of equipment to complete harvests. The goals and objectives of this ecological restoration and fuel reduction project as stated in the Fort Valley Environmental Assessment (USDA 1998) are to create a mosaic of open, parklike forests that approximate conditions present before Euro-American settlement. The project will also reduce the hazard of catastrophic crown fires.

Three harvesting methods were used to meet the objectives of the Fort Valley restoration project: (1) hand harvesting, (2) cut-to-length harvesting, and (3) whole-tree-mechanized harvesting. The hand harvesting treatment method consisted of a classical sawyer operation. Trees of 12.7 cm (5 inches) diameter at breast height (d.b.h.) and greater were cut, limbed, and bucked into 4.9 m (16 ft) log lengths by hand. Processed logs were transported to waiting trucks at the log landing using a tractor with a log grapple. The merchantable activity was simultaneously accompanied by the cutting, scattering, and lopping of the very small, nonmerchantable trees.

The cut-to-length mechanized harvesting method consisted of a single grip harvester, a forwarder, and hand sawyers to cut and lop all nonmerchantable trees. A tractor with a log grapple forwarded all logs cut and processed by the harvester. The logs coming from trees 12.7 cm (5 inches) and greater in diameter were forwarded to waiting trucks at the log landing.

The whole-tree-mechanized harvesting method consisted of one feller-buncher, whole-tree skidders, a delimber, and a loader to process merchantable trees 12.7 cm (5 inches) and greater. Whole trees were cut and piled using a tree-to-tree feller-buncher. Whole trees were skidded to the landing using a rubber tired grapple skidder. Trees were processed into logs at the landing using a delimber and loaded onto waiting trucks. Hand cutting, scattering, and lopping of the less than 12.7 cm (5 inches) diameter trees followed the merchantable activity (Larson and Mirth 1999).

The effectiveness of these harvesting alternatives depends on the degree to which the stated goals have been obtained. All three of these harvesting techniques performed the same ecological restoration treatments, leaving the same residual stand density, species composition, and age class distribution. The different alternatives also achieved similar reductions in risk of catastrophic crown fire, which was primarily dependent on the treatment prescription rather than the harvesting method. The benefit side of the project objectives does not vary significantly between the harvesting alternatives, and therefore are not useful in selecting the optimum-harvesting alternative. In addition, many of the benefits from restoration currently do not have quantified market value, or established nonmarket value. The absence of economic quantification of returns from restoration precludes the use of benefit / cost analysis (Weaver and others 1982).

These three harvesting alternatives are expected to have had different impacts on the residual forest condition. The key differences between the alternatives are how the trees were felled and processed, piled, and transported to the log landing. These differences will have caused variation in residual stand damage, amount and distribution of logging slash, and degree of soil impacts. While these impacts (indirect costs) are also not easily quantified in terms of dollar values, they are amenable to effectiveness analysis.

Because the alternatives are similar in primary goal accomplishment, the primary benefits are nonmarket in nature, and the alternatives differ in direct and indirect costs, and cost / effectiveness analysis is the most appropriate technique for selecting the optimum alternative. The direct cost of each method can be quantified in terms of dollars. In the absence of differences in indirect costs, the cost effectiveness ratio would reduce to cost, and the minimum cost alternative would be preferred. If indirect costs of residual stand damage, fuel loading, and soil impacts were easily quantifiable in dollar value, then any of the efficiency measures (for example, benefit cost ratios) could be used. Cost / effectiveness analysis avoids the problem of the nonmarket nature of these indirect costs by using them as effectiveness measures. Cost / effectiveness analysis can evaluate the alternatives in terms of their direct cost and relative effectiveness in avoiding these adverse impacts. The primary criteria for attributes to quantify these impacts are data availability, and the attribute's correlation to the impact costs.

The following is a short description of each of three proposed effectiveness attributes for residual stand damage, soil compaction, and fuel loading and a possible method for measuring each. Residual stand damage may be quantifiable as percent of residual trees undamaged, with damage defined as visible wounds, broken tops, and broken branches. Stand damage is important because bole damage (for example, cat faces caused by equipment) is a primary point of entry for diseases such as decay fungi to enter the tree (Storer and others 1997). Broken tops and branches can also leave the tree in a state of low vigor, increasing the possibility of insect and disease mortality. This attribute can easily be measured by inventory plot procedures using a fixed radius plot or strip cruises.

Soil impacts measure damage to soil structure. This attribute was selected because compacted and disturbed soils from machines and humans can impede surface water entry into the lower layers of the soil (Seixas and McDonald 1997). Disturbed soil can also provide a growing habitat for invasive exotic weeds that can out-compete native vegetation, reducing habitat for native wildlife and insect populations. Compaction can be measured with an infiltrometer, and be converted to percent change in compaction. Disturbance can be visually estimated as a percentage of disturbed soil, using a plot or strip inventory procedure.

Fuel loading measures the downed woody material left on the forest floor after treatment. In a study on biomass and nutrient removal, Giles (1979) found that removing whole trees, as opposed to stems only, could increase biomass removal by 28 percent in coniferous forests. Increased removal of biomass and nutrients may cause a decline in soil nutrients and forest productivity. Branches cut from trees in the limbing process contain many of the tree's resource of nutrients. Removing this material from the harvest site and accumulating large amounts at landings can cause effective nutrient mining of the site, can be a breeding center for decay fungi and insects that can spread into the residual stand, and can be a fire hazard. Fuel loading can be measured along transects, giving a measure of average diameter of fuels deposited throughout the site (Brown 1974).

The attribute scores should be measured so that cost / effectiveness ratios for each treatment are the highest when they have the highest cost, and hence least effective. This convention requires that each attribute be measured so those alternatives that cause greater impacts have lower effectiveness scores. Attribute scores may be weighted to reflect relative importance by multiplying the percentage amount of each attribute by an importance weight (Macmillan and others 1998).

The overall effectiveness of a harvest alternative can be quantified by summing attribute scores (percent measurements of each of the above attributes) for each treatment. Along with these effectiveness measurements, each harvesting method must be quantified in terms of the financial cost required to implement each harvesting treatment. Direct costs for the Fort Valley restoration project were estimated by the use of a cost analysis completed by Larson and Mirth (1999). This analysis estimated direct logging costs using the method of implementation costs, modeling equipment sets, and operational procedures for hand, whole-tree, and mechanized harvesting methods. Table 1 presents the direct cost of the harvesting alternatives. The whole tree harvesting method was the least cost approach in all forest conditions calculated. Cut-to-length was the second lowest cost in all conditions, and it equaled the whole-tree cost in the yellow pine stand condition. The hand-harvesting method had the highest cost in all conditions. Unless the effectiveness of the methods varies, a rational decisionmaker would select the whole-tree method as the optimal alternative.

The cost / effectiveness ratio of each harvesting alternative can be computed by dividing the alternative's direct cost by its restoration effectiveness score. Table 2 presents a hypothetical example calculation of cost effectiveness ratios for the black jack stand condition, using only residual stand damage as the attribute of effectiveness. In the example, we assume that the percent of stand damaged is higher for the lower cost methods (30, 20, and 5 percent for whole-tree, cut-to-length, and hand harvesting, respectively). If these values match actual stand damage, then the optimal decision would switch to hand harvesting because it has the lowest cost / effectiveness ratio. In the example, stand damage values reverse the optimal ranking from the least cost approach.

The sensitivity of a decision to effectiveness score depends on the ratio of direct cost of the alternatives. Alternatives with similar direct costs (ratio near one) are more sensitive to effectiveness measures. Table 3 demonstrates this sensitivity of cost / effectiveness ratios to direct cost differences. In the black jack stand condition, whole-tree direct costs are 73 percent of hand harvesting cost. For the decision of optimum harvest method to switch from whole-tree to hand harvesting, whole-tree effectiveness would have to be less than 73 percent of the effectiveness of hand harvesting. In the yellow pine condition direct cost differences are similar, and whole tree effectiveness must be at least 90 percent of hand harvesting effectiveness in order to remain the optimal choice.

Table 1—Direct cost ($/ha) of restoration harvest alternatives by stand condition.

Harvesting method	Stand condition: Black jack	Black jack / yellow pine	Yellow pine
Whole tree	1,098	931	818
Hand harvesting	1,497	1,397	909
Cut-to-length	1,297	1,210	818

Table 2— Hypothetical example of cost / effectiveness ratios for black jack stand condition and residual stand undamaged attribute.

Harvesting method	Harvest cost ($/ha)	Cost/(percent undamaged) ($/ percent)	Cost effectiveness
Whole tree	1,098	(1,098)/65	16.9
Hand harvesting	1,497	(1,497)/95	15.8
Cut-to-length	1,297	(1,297)/80	16.2

Table 3—Sensitivity of cost / effectiveness ratios to direct cost differences, or percent effectiveness required to change decision from minimum cost approach. Percent effectiveness is the level of effectiveness that the lower cost alternative must achieve relative to higher cost alternative in order to remain the optimal decision.

	Stand condition	
	Black jack	Yellow pine
Whole tree direct cost ($/ha)	1,098	818
Hand harvesting direct cost ($/ha)	1,497	909
Required percent effectiveness (percent)	73	90

Conclusions

This paper demonstrates cost effectiveness analysis as a useful tool for optimum harvest recommendation for ecological restoration treatments such as those in Fort Valley. Harvest methods may be compared on the basis of ratios of harvest cost / effectiveness. Effectiveness measures can be defined as a harvest method's ability to carry out restoration treatment with the least negative impacts in terms of residual stand damage, soil impacts, and fuel loading. Cost / effectiveness analysis avoids the valuation difficulties with the nonmarket nature of indirect costs by using them as effectiveness measures. Thus, it avoids problems associated with benefits estimation encountered with traditional benefit / cost approaches.

Important issues must be addressed in the use of a cost / effectiveness analysis, such as the proper selection of effectiveness measures and the weights given to these measures. Weights given to effectiveness measures must appropriately reflect not only their importance with respect to other measures, but also their ability to effect optimum harvest choices.

Possibilities for future use of this approach may incorporate a more objective presentation of results. Analyst assumptions in effectiveness weighting could be partially eliminated from this analysis by presenting cost / effectiveness ratios in a table showing effectiveness weights calculated a variety of ways. For example, if the weighting of each effectiveness attribute is removed (all attributes are assumed to be equally important) the cost / effectiveness ratio for each harvesting treatment may change. Attributes could be weighted or not weighted in order of importance, effectiveness could be calculated in each instance, and ratios could also be reported in a table. The table format would give more interpretive decisionmaking power to the reader and make the entire analysis less subjective. Aside from objective presentation techniques, further investigation into the sensitivity of effectiveness measures is also an important consideration for a meaningful analysis.

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Projected Economic Impacts of a 16-Inch Tree Cutting Cap for Ponderosa Pine Forests Within the Greater Flagstaff Urban-Wildlands

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Abstract—The Grand Canyon Forest Partnership (GCFP), located in Flagstaff, AZ, has implemented a 16-inch diameter breast height cutting cap in the Fort Valley Restoration (Phase One) Project to secure the support of environmental organizations for urban interface forest restoration and fuels reduction projects. This paper provides insights into the economic impacts of this limitation by applying a simulated cap to realistic inventory, logging, and revenue models developed from an earlier representative project—the GCFP's 332-acre Fort Valley Research and Demonstration (R&D) project. The simulation was possible on only four of the nine R&D units, as these were the only units that had trees greater than or equal to 16-inch d.b.h. available for cutting. The simulated cutting cap resulted in implementation cost increases of 5 to 19.4 percent, harvested fiber decreases of 10 to 39 percent on a volume basis, and reductions in operator net returns ranging from 22.3 to 176 percent. The primary market for harvested material, at the time of this analysis, was low-value firewood and pallet stock that was supplemented by occasional sales to high-value users of large diameter logs. The 16-inch cap limited the operators' ability to broker logs to these large diameter users (for example, small volume viga manufacturers located in the Phoenix, AZ, metropolitan area) who would pay upward of $200 per ccf. Projections showed, however, that under more favorable market conditions, such as that of a regional pulp mill or oriented strand board plant, the operators (and consequently the GCFP) could better sustain, economically, the 16-inch cutting cap.

Introduction

The Grand Canyon Forests Partnership (GCFP), in Flagstaff, AZ, is a collaborative effort between the Coconino National Forest, Grand Canyon Forests Foundation, Northern Arizona University, and a number of other governmental and nongovernmental organizations. The GCFP seeks to reduce the risk of catastrophic fire and restore forest ecosystem health through practices that are ecologically sound, economically viable, and socially acceptable. Its implementation strategy includes thinning of forest stands, introducing low-intensity fire, restoring meadows, and thoughtful minimization of trails and roads.

The GCFP finished its first treatment project, known as the Fort Valley Research and Demonstration (R&D) Project, during 1999. This 332-acre project is an adaptive management experiment for researching presettlement reference conditions, contemporary pretreatment conditions, and the impact of four thinning alternatives on three forest stand types. Thinning was limited to trees less than 22 inches in diameter measured at breast height (d.b.h.). This project serves as a demonstration site, providing on-the-ground evidence and information so that the public can effectively contribute, in a learned manner, to the process of treating the urban-wildland forest interface in the greater Flagstaff region.

The GCFP has been planning and working toward its next treatment activity, the 1,700-acre Fort Valley Restoration (Phase One) Project. This project is an interesting one in that the GCFP implemented a cutting cap to limit thinning to only those trees smaller than 16 inches d.b.h. By implementing this cap, the GCFP responded to the concerns and forest management desires of regional environmental organizations (SWFA 1996, 1998; Suckling 2000) and demonstrated that GCFP's intent is on forest fuels reduction and restoration, and not commercial logging.

GCFP's ability to treat forest stands over the 100,000-acre urban wildland interface, however, is highly dependent upon economics. Implementation is expensive, and this project must eventually become financially sustainable; independent, to some degree, of Federal dollars. The GCFP realizes that the same material that is thinned from the forests as part of the restorative stand treatment can be recycled as a marketable resource. Wood fiber, in a favorable market environment, has the potential to fund and sustain forest treatment programs. A cutting limitation like the 16-inch cap, however, can have negative financial repercussions.

This paper provides insights into these economic impacts by simulating the effects of a cap on stand conditions, thinning models, and fiber markets that are representative of the conditions of the greater Flagstaff urban-wildland. The process and results of this simulation included:

- Collecting and characterizing representative stand parameters, which are summarized below in the section titled "Fort Valley Research and Demonstration Project."

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- Amassing realistic thinning information and building representative mathematical models; tasks that are summarized in the section titled “The Cost to Treat.”
- Quantifying fiber markets representative of real regional conditions and incorporating this information into the models. The results of these activities are presented in the section titled “Revenue Expectations.”

Fort Valley Research and Demonstration Project

The R&D project is north of Flagstaff near Highway 180 and Snow Bowl Road at an elevation of 7,400 to 7,600 ft above sea level. This project consists of three areas that are representative of different stand configurations—a yellow pine area with more than five yellow pine trees per acre, a mixed yellow pine and blackjack area with less than two yellow pine trees per acre, and a blackjack area. (Yellow pines are ponderosa pine trees characterized by yellow bark and are larger in size and older than 150 years. Blackjacks are younger and smaller ponderosa pine trees with black bark.) The two yellow pine areas lie within the Fort Valley Experimental Forest, while the blackjack area lies in the Coconino National Forest along the eastern boundary of the experimental forest. Figure 1 is a map of this project.

Each treatment area is subdivided into four units that range from 32 to 41 acres. The four units are differentiated by the type of the proposed treatment, designated as: 1.5-3, 2-4, 3-6, and control. Although these prescriptions provide for different levels of thinning, they are anchored to the presettlement condition as their template. Common to each treatment is that all living presettlement trees, standing snags, and trees greater than 22 inches d.b.h. are retained (Flagstaff Urban/Wildland Interface Treatment Guidelines, 1998). In addition, all treatments called for the removal of all trees 4 inches d.b.h. and smaller. The reader can find additional information about these types of treatments and their effects in the articles by W. Covington and others (1997, 1998).

The Treatments

The 1.5-3 treatment is known as a full restoration prescription. For every direct evidence of a dead presettlement tree (stumps, snags, downed trees, and stump holes), 1.5 replacement trees are left whenever large (>16 inches d.b.h.) and vigorous replacement trees are available within a 30 ft radius of the evidence. If the only available good quality replacement trees are smaller than 16 inches, then three trees are marked for retention. When the available trees within the 30 ft search radius are not acceptable due to

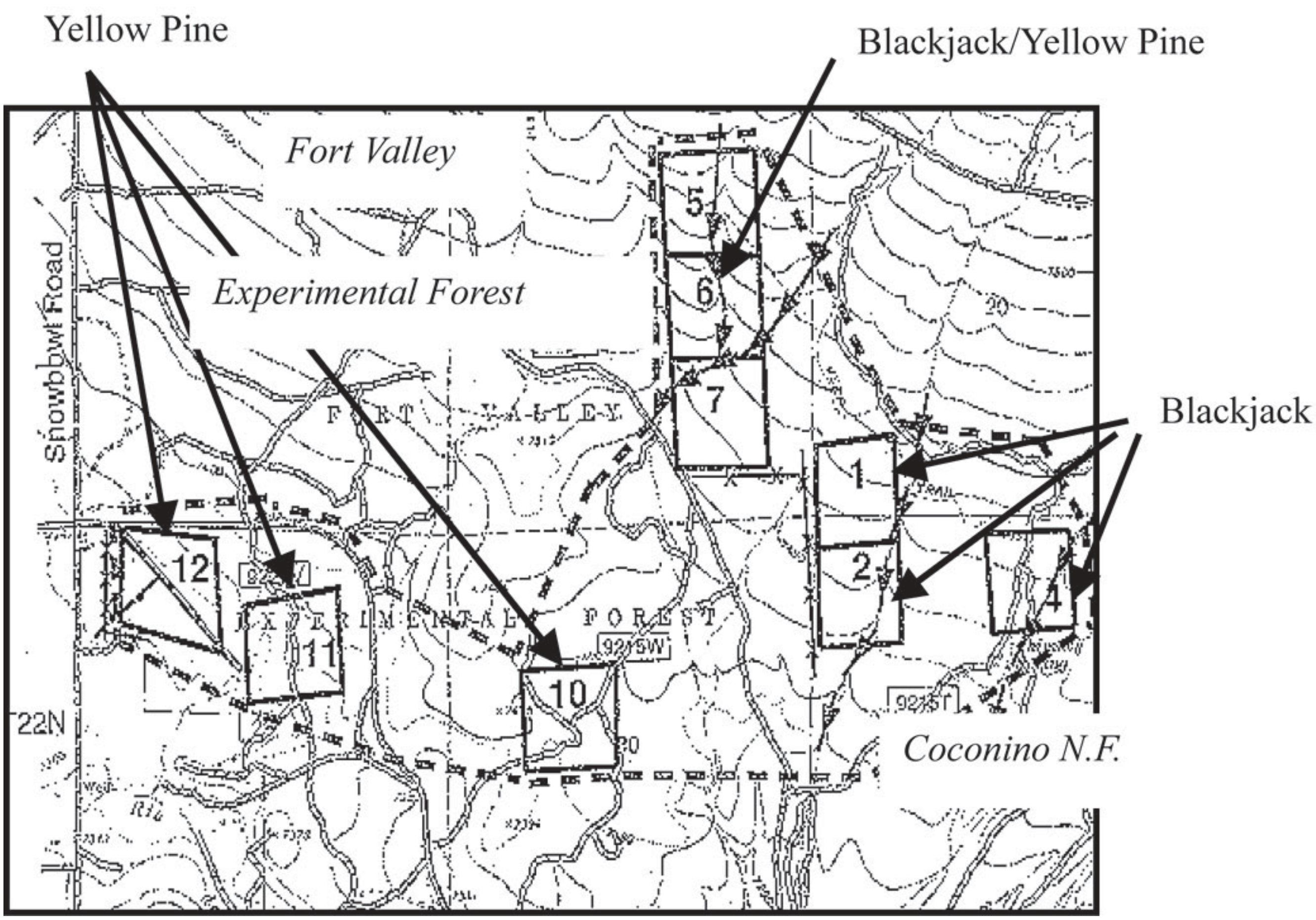

Figure 1—Map of the Fort Valley Research and Demonstration Project.

quality or mistletoe infection, the search radius is extended to 60 ft.

The 2-4 treatment is known as an intermediate level of thinning where more trees per presettlement evidence are retained. In practice, two large or four small dominant and/or vigorous trees are left for every evidence.

The 3-6 treatment is a minimal thinning plan that results in an even greater density of replacement trees where three large or six small trees are left per evidence.

The control units are untreated and will be used for comparative purposes.

Existing Stand Character

An inventory of the marked Fort Valley Research and Demonstration Units was started on September 14, 1998, and completed on November 11, 1998. This work formed the starting point for the analysis of the 16-inch cutting cap economic impacts by providing a reliable projection on the total amount of wood thinned in 1-inch size classes. Only a summary of the inventory is provided here. Details and comprehensive analysis can be found in the report by Larson and Mirth (1999).

The tree stand condition prior to thinning is given in table 1. The sampling technique to determine this condition consisted of counting trees and measuring tree diameters over 33–1/100 acre samples per unit. This provided an 80 percent probability that the estimate of the average total number of trees per acre per unit will fall within the limits shown on the table.

The blackjack units, located outside the Fort Valley Experimental Forest, had been previously harvested for large trees and had been thinned from below (Fulé and others 1999). Consequently, the inventory reflects a small percentage (ranging from 8.3 percent to 23 percent of the total stem count) of existing trees in the 4-inch and under classes. (The trees within each d.b.h class have a possible breast height diameter that range from 0 to 0.9 inches. In example, a 5-inch d.b.h. class could include 5-inch trees as well as 5.9 inches and any tree in between.) The majority of existing stems are found in the 5 to 15-inch classes; ranging from 70 to 92 percent of the total unit tree counts. The number of existing 16-inch and larger trees is small; accounting for only 3.7 percent of the overall tree population of all three units.

The blackjack/yellow pine units are denser than the blackjack units due to large numbers of less than 5-inch trees, reflecting a lack of precommercial thinning in the Experimental Forest. The less than 5-inch trees represent 59 to 75 percent of the total number of standing trees. These blackjack/yellow pine units have a smaller percentage (16.6 to 31.2 percent) of 5-inch to 15-inch trees, even though the absolute numbers are similar to that found in the blackjack units. The blackjack/yellow pine units contain greater numbers of larger (16 inches and greater) trees, averaging 28.0 trees per acre (tpa).

In terms of the numbers of very small and very large trees, the yellow pine units were similar in character to the blackjack/yellow pine units. The population of existing trees that was less than 5 inches accounted for 49 percent of the trees in unit 10 to 73 percent in unit 12. The number of 16 inches and greater trees averaged 23.1 tpa.

Table 1—Fort Valley Research and Demonstration Project standing tree estimates prior to thinning.

Unit	Type	Standing stems per acre: average
1	BJ	330 ± 48
2	BJ	321 ± 52
4	BJ	257 ± 49
5	BJ/YP	1,076 ± 256
6	BJ/YP	618 ± 152
7	BJ/YP	563 ± 114
10	YP	376 ± 97
11	YP	445 ± 128
12	YP	1,182 ± 218

Treatment Effects

Tables 2, 3, and 4 summarize the effects of thinning as part of the GCFP restoration plan. The projected cutting rate is given on a total number of trees per acre basis and on a fiber volume basis of 100 cubic foot per acre (ccf/a). Currently, the less than 5-inch trees cannot be economically

Table 2—Merchantable wood fiber volumes for the blackjack units of the Fort Valley Research and Demonstration Project.

	Unit 1	Unit 2	Unit 4
Treatment	1.5–3	2–4	3–6
Number cut (tpa)	293.3	271.9	175.3
Merchantable fiber (ccf/a)			
0–4 inch d.b.h. class	0	0	0
5–15 inch d.b.h. class	17.724	11.650	8.241
16–21 inch d.b.h. class	0	4.275	.912

Table 3—Merchantable wood fiber volumes for the blackjack/yellow pine units of the Fort Valley Research and Demonstration Project.

	Unit 5	Unit 6	Unit 7
Treatment	3-6	1.5-3	2-4
Number cut (tpa)	999.9	563.3	505.1
Merchantable fiber (ccf/a)			
0–4 inch d.b.h. class	0	0	0
5–15 inch d.b.h. class	9.247	7.902	12.738
16–21 inch d.b.h. class	0	1.853	8.292

Table 4—Merchantable wood fiber volumes for the yellow pine units of the Fort Valley Research and Demonstration Project.

	Unit 10	Unit 11	Unit 12
Treatment	1.5-3	3-6	2-4
Number cut (tpa)	324.0	358.4	1,098.4
Merchantable fiber (ccf/a)			
0–4 inch d.b.h. class	0	0	0
5–15 inch d.b.h. class	9.796	1.705	8.196
16–21 inch d.b.h. class	0	0	0

used as a marketable resource and, although they are part of the tpa cut projections, they are not represented in the merchantable fiber volumes. This very small material was cut, scattered on the ground throughout the units, and lopped to less than 2 ft in height.

The potential fiber harvested from these units is calculated on a diameter class basis that permits analysis of the 16-inch cutting limitation. As shown in tables 2 through 4, had the cutting limitation been imposed on this R&D project, it would have impacted the fiber returns from only four (units 2, 4, 6, and 7) of the nine cutting units. On a 100 cubic foot per acre (ccf/a) basis, the available 16 to 21.9-inch trees represented—respectively over units 2, 4, 6, and 7—26.8, 10.0, 19.0, and 39.4 percent of the total merchantable fiber potential.

The Cost to Treat

Thinning Models

Two thinning strategies that closely replicated the processes and equipment setups of two of the actual three operators that performed the thinning of the Fort Valley R&D project were modeled for this 16-inch cap analysis. These models included:

1. Whole tree mechanized harvesting (WT):
   - (a) This scenario uses a mechanized system consisting of a tracked feller-buncher, whole-tree skidders, a delimber, and loader to process the merchantable 5-inch and greater trees.
   - (b) The submerchantable trees less than 5 inches d.b.h. were hand felled, scattered, and lopped. (In this context, lopping refers to the cutting of downed trees and limbs that project higher than 2 ft above the ground level.) We presumed that this precommercial activity was subcontracted out to a local sawyer operator.
2. Hand felling of all trees (HD):
   - (a) This scenario considers the hand cutting, limbing, and bucking of the 5-inch and greater trees that are then forwarded to the landing using an articulated rubber-tired skidder with a log grapple.
   - (b) The merchantable activity is simultaneously accompanied by the cutting, scattering, and lopping of the very small, nonmerchantable trees. This model assumes that a subcontractor completes all cutting and related processing, regardless of tree size.

The HD operation modeled here is fundamentally different than the WT one. For comparative purposes, the HD operation was modeled as a direct cost only operation; neglecting overhead, profit, ancillary logging-related expenses, equipment depreciation, and so forth. This is in contrast with the WT model that incorporated all ideal business expenses including 10 percent for profit and 11 percent for administrative overhead. Model details are found in the paper by Larson and others (2000), or the unpublished report by Larson and Mirth (1999).

Like the actual contracted R&D work, the different operational models were applied to the different stand types. Stand data from units 2 and 4 (the blackjack units) were used as input to the WT model. Units 6 and 7 data (the blackjack/yellow pine units) were coupled to the HD model. Because our focus is to gain insight into the 16-inch cutting cap effects, the work presented here is limited to only those four units (1, 4, 6, and 7) with available trees in the 16 to < 22-inch d.b.h. classes.

Table 5—WT implementation costs for merchantable trees projected for units 2 and 4 of the Fort Valley Research and Demonstration Project.

Units	2	4
Description	BJ	BJ
Treatment	2-4	3-6
WT scenario 1: 5 inches to <22 inches d.b.h. trees		
Merchantable ccf/acre	15.92	9.15
$/Unit	$44,743.36	$21,793.20
$/Acre	$1,272.46	$665.22
$/ccf	$79.91	$72.68
WT scenario 2: 5 inches to <16 inches d.b.h. trees		
Merchantable ccf/acre	11.65	8.24
$/Unit	$39,078.56	$20,603.82
$/Acre	$1,111.35	$628.91
$/ccf	$95.40	$76.32

Results

Through the application of stand data to the appropriate thinning models, operational cost projections are made and summarized in tables 5 and 6. Two scenarios are presented, allowing ready analysis of the impacts of the 16-inch cutting cap relative to implementation costs. Scenario 1 considers the cost implications of cutting all unmarked trees in the merchantable d.b.h. size classes from 5 inches to less than 22 inches. The contrasting scenario 2 considers that only trees less than 16 inches are cut.

Table 6—HD implementation costs for merchantable trees projected for units 6 and 7 of the Fort Valley Research and Demonstration Project.

Units	6	7
Description	BJ/YP	BJ/YP
Treatment	1.5-3	2-4
Hand scenario 1: 5 inches to <22 inches d.b.h. trees		
Merchantable ccf/acre	9.76	21.03
$/Unit	$26,670.60	$47,034.82
$/Acre	$730.80	$1,291.60
$/ccf	$74.91	$61.42
Hand scenario 2: 5 inches to <16–inches d.b.h. trees		
Merchantable ccf/acre	7.90	12.74
$/Unit	$22,491.91	$29,338.80
$/Acre	$616.30	$805.64
$/ccf	$77.99	$63.25

The tabulated cost summaries are presented in three ways:

- $/unit—the total cost to truck all merchantable fiber to processing mills. Different mill distances were used for the different operators, because each operator historically served different fiber users. A haul distance of 118 miles was used for the WT operator. This mileage represents a weighted travel average, moving wood to a once active pulp mill and to a saw log buyer. (Neither of these purchasers exists today. The sawmill closed in December of 1998. The paper mill converted to a 100 percent recycled product, eliminating the need for pulpwood.) Similarly, the HD weighted average haul distance varied between 60 to 100 miles, depending upon the distribution of log sizes. The main purchaser of the HD fiber was a firewood and pallet stock manufacturer located 60 miles away, but occasionally, the HD operator would haul larger logs to a viga producer 140 miles away.
- $/acre—an expression of the total costs including trucking on a per acre basis.
- $/ccf—an expression of the total costs including trucking on a per 100 cubic foot of extracted merchantable fiber.

An examination of $/unit or $/acre data from tables 5 and 6 erroneously suggests that the imposition of a 16-inch cap reduces cutting costs. A better context, however, for assessing the cap impact is the cost per extracted fiber basis. It is a number that can be directly compared to its revenue potential that thereby offsets implementation costs. For example, in unit 2, 26.8 percent of the projected extracted fiber (on a ccf basis) comes from trees 16 inches and larger. Upon imposition of the cutting cap, implementation costs are shown to increase by $15.49/ccf (or 19.4 percent). Similarly, unit 4 realizes an implementation cost gain of 5.0 percent associated with a 10 percent loss of potential harvested fiber. Unit 6 sees a 7.3 percent increase in cost due to a 19 percent reduction in available fiber. Unit 7 realizes a 10 percent cost increase due to a 39 percent fiber reduction. These results are consistent with the known relationship that "logging costs per cubic foot are higher for smaller removal volumes per acre" (Hartsough and others 1998).

The net revenue loss, however, will be larger than that suggested by implementation cost increases as the larger than 16-inch trees command a better selling price than the smaller diameter trees. This revenue impact due to the 16-inch cap is discussed further in "Revenue Expectations."

Revenue Expectations

GCFP's ability to treat the urban-wildland forests is highly dependent upon the local operators' ability to profitably market the harvested trees. A true picture of the implementation economics and the impact of the 16-inch cap is not complete without a revenue analysis that is grounded in either historical or current market conditions.

Prior to December 1998, the WT operator sold his small diameter logs as pulpwood (smaller diameter logs that are suitable for use in making pulp—the main component in paper production) and the larger logs as saw logs for cutting into boards and lumber. As of winter 1999 when this revenue analysis was completed, the WT operator did not have a regular buyer of harvested wood fiber. Except for a few truckloads of larger diameter logs sold as viga (high-quality poles peeled from large diameter logs that are used in Southwestern roof and ceiling architectural systems) stock or saw logs, most of the logs from units 1, 2, and 4 were eventually sold in late spring 1999 at a discount to Northern Arizona Wood Products for firewood, posts, or poles.

The HD operator historically sold wood to a broad range of markets. During winter 1999, however, the harvested logs were sold into the firewood, pallet stock, and viga markets.

For this revenue analysis on the impact of the 16-inch cutting cap, two market conditions were projected—a favorable one of pulpwood and saw logs, and a subsistence one of firewood and pallet stock that is bolstered by a low volume regional viga market. This analysis, however, neglects the cost of operator down time that does occur due to oversupply in these limited volume markets. A summary of the total net return projections that includes revenue from the sale of fiber, stumpage fees, thinning, and precommercial costs is provided in tables 7, 8, 9, and 10 for the WT and HD operators. These tables provide both a generalized sense of the impact and also a detail analysis on a per unit basis. The general impact trends are summarized here:

- There is a decrease in stumpage fees paid out by the operator as there is less wood fiber harvested.
- Harvesting costs change, decreasing if presented on a per area basis, but increasing if on a per merchantable volume basis.
- The revenue opportunity does not change as a function of cutting restrictions within the small diameter product categories, but decreases significantly in the larger diameter products. As a consequence, the net returns to the operator(s) are likewise severely impacted.
- The large diameter products subsidize the lower value small diameter products. This bolstering is particularly important to the operators selling fiber within a subsistence market where the main opportunity for small diameter logs is a very low value product.

Table 7 summarizes the financial impacts of the 16-inch cap for the WT operator under a market condition that is considered favorable (albeit currently nonexistent for operators in the Flagstaff area) as the small diameter wood is sold at a relatively high price as pulpwood. The projected impact of a cutting cap to the operator is net return reductions of 96.8 percent over unit 2 and 22.3 percent over unit 4. Recall that the WT thinning model included profit, overhead, and other reasonable business expenses. Because of this, and because net returns were always positive, the modeled WT operator could tolerate the 16-inch cap in a market that pays a reasonable price for small diameter wood.

This is not the case, however, for the WT operator in a marginal market of firewood, pallet stock, and vigas as shown in table 8. The projected impact of a cutting cap is quantified in terms of net return reductions of 176.2 percent over unit 2 and 58.8 percent over unit 4. Table 8 suggests that the modeled WT operator could not afford the 16-inch cap in a low-value small-diameter market with a negative net return that completely cancels any profit opportunities.

Table 7—Comparing WT revenue with and without a cutting cap under favorable market conditions.

	Unit 2	Unit 2	Unit 4	Unit 4
Prescription	2-4	2-4	3-6	3-6
With or without cutting cap	Without	With	Without	With
Mill price				
Pulpwood (5 inches to <12 inches d.b.h. @ $81/ccf)	$24,392.57	$24,392.57	$11,568.49	$11,568.49
Saw logs (≥12 inches d.b.h. @ $140/ccf)	$36,233.85	$15,189.91	$21,982.39	$17,801.55
Total mill revenue	$60,626.42	$39,582.48	$33,550.88	$29,370.04
Stumpage	–$4,423.65	–$3,236.17	–$2,368.72	–$2,132.80
Mechanized[a] + trucking costs	–$44,743.36	–$39,078.56	–$21,793.20	–$20,603.82
Precommercial service contract[b]	$3,516.30	$3,516.30	$3,276.10	$3,276.10
Precommercial costs	–$311.50	–$311.60	–$321.59	–$321.69
Net return	$14,664.20	$472.44	$12,343.48	$9,587.83

[a]Includes a 10 percent profit margin and 11 percent overhead on the mechanized portion of the cutting activity.
[b]The Foundation pays this operator $100/acre to complete the precommercial activity.

Table 8—Comparing WT revenue with and without a cutting cap under subsistence market conditions.

	Unit 2	Unit 2	Unit 4	Unit 4
Prescription	2-4	2-4	3-6	3-6
With or without cutting cap	Without	With	Without	With
Mill price				
Firewood[a] (5 inches to <14 inches d.b.h. @ $62/ccf)	$23,748.52	$23,748.52	$13,942.35	$13,942.35
Vigas (≥14 inches d.b.h. @ $200/ccf)	$35,383.08	$5,320.30	$14,992.29	$9,019.65
Total mill revenue	$59,131.60	$29,068.82	$28,934.64	$22,962.00
Stumpage	–$4,423.65	–$3,236.17	–$2,368.72	–$2,132.80
Mechanized[b] + trucking costs	–$44,743.36	–$39,078.56	–$21,793.20	–$20,603.82
Precommercial service contract[c]	$3,516.30	$3,516.30	$3,276.10	$3,276.10
Precommercial costs	–$311.50	–$311.60	–$321.59	–$321.69
Net return	$13,169.38	–$10,041.21	$7,727.23	$3,179.79

[a]This product category also includes pallet stock.
[b]Includes a 10 percent profit margin and 11 percent overhead on the mechanized portion of the cutting activity.
[c]The Foundation pays this operator $100/acre to complete the precommercial activity.

Table 9—Comparing HD revenue with and without a cutting cap under favorable market conditions.

	Unit 6	Unit 6	Unit 7	Unit 7
Prescription	1.5-3	1.5-3	2-4	2-4
With or without cutting cap	Without	With	Without	With
Mill price				
Pulpwood (5 inches to <12 inches d.b.h. @ $81/ccf)	$19,642.49	$19,642.49	$20,360.12	$20,360.12
Saw logs (≥12 inches d.b.h. @ $140/ccf)	$15,894.95	$6,424.97	$72,020.68	$29,748.58
Total mill revenue	$35,537.43	$26,067.45	$92,380.79	$50,108.70
Stumpage	–$697.83	–$565.25	–$1,500.95	–$909.14
Merchantable material costs	–$26,670.60	–$22,491.91	–$47,034.82	–$29,338.33
Precommercial costs	–$2,178.75	–$2,178.75	–$1,905.93	–$1,905.93
Net return	$5,990.25	$831.55	$41,939.09	$17,955.29

Tables 9 and 10 show a similar impact trend for the direct cost HD model as that seen for the full cost WT model. In a favorable market, the cutting cap reduces projected returns by 86.1 percent over unit 6 and 57.2 percent over unit 7. In the subsistence market, the reductions are, respectively, 162.5 percent and 83.8 percent over units 6 and 7. These reductions, however, are particularly severe in the subsistence market for this direct cost model excludes overhead, profit, depreciation, insurance, opportunity loss, mobilization, and roadwork. The cutting cap moves the operator from an adequate financial situation to a losing one where he cannot finance the indirect operational costs.

Table 10—Comparing HD revenue with and without a cutting cap under subsistence market conditions.

	Unit 6	Unit 6	Unit 7	Unit 7
Prescription	1.5-3	1.5-3	2-4	2-4
With or without cutting cap	Without	With	Without	With
Mill price				
Firewood[a] (5 inches to <14 inches d.b.h. @ $62/ccf)	$16,168.56	$16,168.56	$23,577.53	$23,577.53
Vigas (≥14 inches d.b.h. @ $200/ccf)	$19,050.38	$5,521.84	$77,102.02	$16,713.32
Total mill revenue	$35,218.94	$21,690.40	$100,679.55	$40,290.85
Stumpage	–$697.83	–$565.25	–$1,500.95	–$909.14
Merchantable material costs	–$26,670.60	–$22,491.91	–$47,034.82	–$29,338.33
Precommercial costs	–$2,178.75	–$2,178.75	–$1,905.93	–$1,905.93
Net return	$5,671.76	–$3,545.51	$50,237.85	$8,137.44

[a]This product category also includes pallet stock.

The quality of the fiber market is an important variable on the overall impact of a cutting cap. A favorable regional market that can pay a reasonable price for small diameter logs—distinguished by higher value products such as pulp for paper products, oriented strand board, or medium or high density fiber board—is one that might support a cutting cap. A subsistence market—distinguished by low value products such as firewood, pallets, or arts and crafts—cannot with current thinning technologies and administrative procedures support the cap. A comparison of "with cap mill revenues" between tables 7 and 8 and between tables 8 and 9 readily demonstrates this market quality factor. The subsistence market yields gross revenues that are 73.4 to 83.2 percent of what the favorable market is projected to provide. This is roughly comparable to the difference in small diameter fiber price between the low-value use versus the higher value use.

Conclusions

The results presented in this paper were developed by simulating a cutting cap over various models built from a representative forest restoration project, the Fort Valley R&D Project, that involved the thinning of trees that were less than 22 inches d.b.h. This simulation work suggests the following:

- The cost to conduct a forest thinning program to reduce the risk of catastrophic fire and restore forest ecosystem health is substantial.
- Establishing a cap that prohibits the cutting of 16 inches d.b.h. and greater has a negative effect on the economics of a forest thinning project.
- The number of 16- to 21.9-inch trees available for cutting in the original Fort Valley R&D Project represents only a small percentage of the standing large trees. However, these trees represent a disproportionately large percent of the total volume to be cut with profound effects on project economics.
- The economic effect of the 16-inch cap is related to the health of the regional wood fiber market. A healthy market with several users that pay fair prices can support a forest restoration program even with a cap. A weak, limited market for wood fiber probably cannot support the operators, if a cap is imposed.
- This study suggests that a healthy market is one that can pay, on a weighted average, between $70/ccf to $95/ccf within a 100 mile haul radius.

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Lumber Recovery From Small-Diameter Ponderosa Pine From Flagstaff, Arizona

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Abstract—Thousands of acres of densely stocked ponderosa pine forests surround Flagstaff, AZ. These stands are at high risk of fire, insect, and disease outbreak. Stand density management activity can be expensive, but product recovery from the thinned material could help defray removal costs. This project evaluated the yield and economic return of lumber recovered from small-diameter, suppressed ponderosa pine. A sample of 150 trees ranging from 6 to 16 inches in diameter at breast height was selected. Half of the sample was sawn into dimension lumber and the other half into appearance grade lumber. This study yielded about 25 percent No. 2 and Better Common for appearance lumber with the majority of lumber (66 percent) graded No. 3 Common. About 50 percent of the dimension lumber was No. 2 and Better, with very little of the highest grade, Select Structural, produced. Volume recovery was slightly higher for dimension lumber, yet the lumber from the logs sawn for appearance grade was worth significantly more than that from the logs sawn for dimension lumber.

Introduction

In 1998 it was estimated that 40 million acres of National Forest were at high risk of fire, disease, and insect outbreak because of increased stand density (Dombeck 1998). This acreage does not include forest land under other ownership. Some researchers believe that past forest management practices, such as fire suppression and livestock grazing, have altered forest conditions in ways that may increase these hazards (Sackett and others 1996). This is of special concern at the urban-wildland interface because of the risk of property damage and threats to public safety. A representative case study in northern Arizona (Mast and others 1999) found a rise in forest density from 60 trees/ha in 1876 to 3,000 trees/ha in 1992. This case study prescribed a forest restoration plan to return the forests to presettlement conditions. The plan involves thinning the dense understory of stands to allocate more resources to overstory trees and thereby restore stand vigor and health. Costs for this type of management activity are high, and information regarding revenue from use of the extracted resource is minimal.

Historically, ponderosa pine (*Pinus ponderosa* Dougl. Ex Laws.) was a commercially important species in the Southwestern United States. Past research (Ernst and Pong 1985; Fahey and others 1986; Fahey and Sachet 1993) to evaluate product recovery from this resource generally is not specific to suppressed stands and is limited to a few grading options not suitable for evaluating use in engineered structural products such as metal plate wood trusses, I-joists, and glulam beams. Erickson and others (2000) recently completed a grade yield study on small-diameter trees sampled from northern and central Idaho that provides some information on structural framing and machine stress rated (MSR) lumber yields for ponderosa pine sawn into 2 by 4s. The ponderosa pine trees sampled for this study were removals from densely stocked stands that were about 45 years old and thinned from below. The trees removed were smaller, 9-inches diameter at breast height (d.b.h.) or less, that were not dominant or codominant in the forest canopy. The lumber produced from these trees had poor yields in the higher visual grades and would not be suitable for production of MSR lumber. It should be noted, however, that grade yields from these thinnings might not be typical of small-diameter ponderosa pine that were dominant or codominant trees in a stand. Previous studies also have documented that drying degrade is a major cause of grade reduction with younger trees (Arganbright and others 1978; Blake and Voorhies 1980; Mackay and Rumball 1972; Markstrom and others 1984). Even though these past studies have shown that lumber cut from young-growth, small-diameter, ponderosa pine trees is prone to warp during drying, the extent of the problem in lumber from suppressed stands is not entirely clear.

To begin evaluating products from this resource, some basic information on lumber recovery, grade yield, and mechanical properties is needed. The Ecologically Sustainable Production of Forest Resources (ESP) team at the Pacific Northwest Research Station has been conducting product recovery studies for over 40 years. Since the early 1990s, the focus of the team's work in this area has been to quantify the types of wood products that might result from treatments designed to alter structural conditions and development trajectories in densely stocked, small diameter stands. The objective of these studies was to determine the value of these wood products and to incorporate this information into planning tools used to design treatments for these management activities (Fight 1997). Although many people tend to think of small trees as being young growth,

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it became evident that many trees from densely stocked stands were older (usually at least 60 years old), slow growing, and had small branches or knot indicators. Small diameter is defined differently depending on land management objectives. The d.b.h. range of trees (Douglas-fir, *Pseudotsuga menziesii* [Mirb.] Franco; western larch, *Larix occidentalis* Nutt.; and lodgepole pine, *Pinus contorta* Dougl. Ex. Loud.) from a previous study in northern Idaho was 6 to 14 inches. Results from this work (Lowell and others 2000; Willits and others 1997) showed that the quality (volume and grade yield) of this resource was similar to that found in larger diameter trees. Economic gains could be realized by using the lumber as a raw material for a variety of end uses, including engineered and value-added products.

This study was part of a larger project designed to evaluate product potential of the ponderosa pine resource and to determine if certain kiln drying techniques reduce drying defect. The objective of the portion of the study reported here was to determine the yield of structural and nonstructural lumber grades from small-diameter (in this case 6 to 16 inches d.b.h.), suppressed ponderosa pine and to compare the volume and value recovery of dimension versus appearance grade lumber from this resource.

Procedures

Tree Selection

Trees were selected from the Fort Valley demonstration project, Flagstaff, AZ. The demonstration project was set up in three experimental blocks with four treatment plots each. The experimental blocks represented different initial stand conditions; black jack (young-growth) pine, yellow pine (old growth) and a mixture of the two age groups. The treatments within the blocks were different thinning prescriptions designed to return stands to presettlement conditions. This involved thinning from below where the larger, older trees are retained. Trees to be left had been marked but no treatments had been applied prior to sample selection for this study. Sample trees came from three of the four treatment plots in the mixed age block.

A sample of trees ranging from 6 to16 inches d.b.h. was selected by using a matrix consisting of six 2-inch diameter classes. Trees selected were those that would have been removed under the silvicultural prescription. The sample was randomly divided into two subsamples, one to be sawn for dimension lumber and the other for appearance grade lumber.

Logging

A crew from the ESP team was present during tree falling to record measurements from both the standing tree and bucked logs. Woods-length logs had a preferred length of 36 and 40 feet. Measurements included d.b.h., total height, age at stump height, length and diameter of all woods-length logs, length of breaks, and unutilized tops. Woods-length logs were tagged with a number that identified the tree and position in the tree from which it came. Logs were hauled to the cooperating mill for both long-log and short-log scaling.

Log Measurements

After all woods-length logs were delivered to the mill, they were rolled out and scaled as presented. Logs were scaled by a Forest Service check scaler according to rules in the National Forest log scaling handbook (U.S. Department of Agriculture 1985) and by the Forest Service cubic scaling rules (U.S. Department of Agriculture 1991). Scaling deductions were itemized to show cause, location, and amount of each defect. Logs also were graded by using an appropriate log grading rule. The official scales for the study were cubic and Scribner. After scaling, woods-length logs were bucked into sawmill lengths (short logs) and scaled again, recording bucked lengths and diameters (to the nearest 0.1 inch). An ESP team member participated in scaling to ensure that all necessary data were collected.

Log Processing

As the study logs were brought into the mill they were assigned a consecutive sawing number that was matched to the original tree number and log within that tree. One-half the sample was sawn into 2-inch dimension lumber (nominal sizes 2 by 4 and 2 by 6) with some 1-inch jacket boards sawn from the outer portion of the logs. The other half of the sample was sawn for 1-inch boards. Green lumber measurements were taken to enable calculation of the rough-green cubic volume of lumber, sawdust, and chips. Scientists from the Forest Products Laboratory (FPL) also measured warp on the green lumber.

Board Data

After kiln drying, the lumber was surfaced to standard dressed dry dimensions. Each board grade, length, and width was tallied after the grading station. A Western Wood Products Association inspector graded the dimension lumber under the structural light framing grades; Select Structural, No. 1, No. 2, No. 3, and Economy (Western Wood Products Association 1998) and the type of grade-limiting defect was recorded. The appearance boards were graded under the board grades; #1 Common, #2 Common, #3 Common, #4 Common and factory lumber grades; Moulding, #3 Clear, 1 Shop, and 2 Shop (Western Wood Products Association 1998).

The 2 by 4 lumber was shipped to the University of Idaho for inclusion in an ongoing cooperative study between the FPL and the University of Idaho on mechanically grading lumber from small-diameter trees of Inland West species (Green and others1997). An associate inspector of the Western Wood Products Association graded this subsample of lumber using the light framing (Construction, Standard, Utility), structural light framing (Select Structural, No.1, No.2, No.3), lamstock, and machine stress rated systems. Final grade assignment for the lamstock and MSR grades required determination of physical and mechanical properties. Results from this portion of the study will be reported separately.

The FPL determined the warp characteristics of lumber kiln dried in a conventional way and then determined the effect of presteaming and top loading, separately and in combination, on warp. These results also will be reported separately.

Analysis

Regression analysis was used to determine volume and value recovery of ponderosa pine by log small-end diameter. The best fitting model was selected on the basis of coefficient of determination (R^2) and standard deviation about the regression (s_{yx}). Models were developed by using a transformation of log small-end diameter as the independent variable. The dependent variable for volume was either percentage of cubic recovery or cubic volume of lumber per gross cubic log volume expressed as a percentage. Dollars per 100 cubic feet of log scale ($/CCF) was the variable used for lumber value because it is a preferred measure of tree value. It is calculated by dividing the total value of lumber produced by the cubic scale (gross or net) of the log multiplied by 100. The results are dependent on the pricing structure in place at the time. Dollars per 100 cubic feet can be highly correlated with log diameter because log quality tends to increase with log size.

Analysis of covariance was used to determine if significant differences existed in volume and value recovery between logs sawn for appearance and dimension lumber.

Results and Discussion

Tree and log data for each sample are given in table 1. The average age of the trees was 90 years old (range of 50 years to 120 years). Both samples of trees had very little scaled defect, less than 1.5 percent of the total cubic log volume.

Volume Recovery

The percentage of cubic volume recovered as lumber was similar for logs sawn for dimension and appearance grade products (fig. 1). Table 2 contains the analysis of covariance (ANCOVA) statistics and table 3 presents the models and statistics for the volume recovery curves. The logs sawn for dimension lumber had a slightly higher volume recovery. This can be partially explained by the fact that fewer pieces of lumber were produced from each log. Fewer saw cuts were made so there was less loss to saw kerf, fewer boards means less overall loss to shrinkage during kiln drying, and there would be correspondingly less loss in planer shavings. About 10 percent of the volume of lumber produced from the dimension logs were 1-inch boards sawn from the outside portion of the log (table 4). The mean cubic recovery from logs sawn for dimension products was 42.7 percent while logs sawn for appearance lumber recovered 40.4 percent.

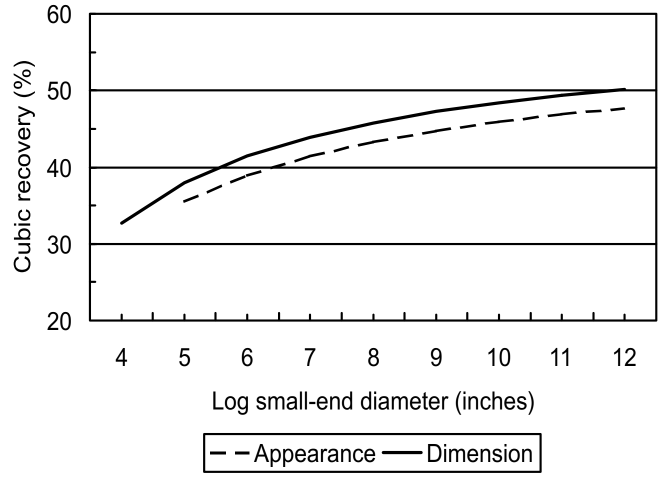

Figure 1—Percentage of cubic volume recovery (based on gross cubic log volume) by log small-end diameter from logs sawn for appearance and dimension products.

Value Recovery

As illustrated in figure 2 (tables 3 and 5 show the models and ANCOVA, respectively), value of lumber differed significantly between the two product types, with appearance lumber worth more than the dimension lumber in the log diameter range sampled. The mean value of the appearance grade lumber was $268 per 100 cubic feet (gross log scale), while the dimension lumber was only $215 per 100 cubic feet. Prices used were from the Western Wood Products Association (1999) report summary through November 1999. No manufacturing costs are included in these values; so the real difference between the products thus would be less because manufacturing and handling costs for appearance lumber are greater owing to the larger number of pieces being produced.

Value recovery depends on both the volume and the lumber grade recovered from a log. Lumber grade recovery is important, with higher grades of lumber commanding higher prices. This study yielded about 25 percent No. 2 and Better Common for appearance lumber with the majority of lumber (66 percent) graded No. 3 Common (table 6). About 50 percent of the dimension lumber was No. 2 and Better (table 4), with very little of that in the highest grade.

Table 1—Sample statistics by the type of lumber product produced in this study.

Item	Appearance	Dimension
Number of trees	76	76
D.B.H. range (inches)	6–16	6–16
Avg. small diameter of log (inches)	7.0	7.0

Table 2—Analysis of covariance for percent cubic volume recovery.

Source	df	MSE	F-value	P-value
1/log SD[a]	1	6172.67	73.45	0.0001
Lumber product[b]	1	513.52	6.11	0.0139
Error	350	84.04		

[a]SD = Log small-end diameter.
[b]Appearance and dimension lumber.

Table 3—Models for volume and value recovery curves shown in figures 1 and 2.

Dependent variable	Product	Model	R2
Cubic volume recovery	Appearance	56.44 – (104.59*(1/SD))[a] (2.02)[b] (12.2)	0.18
	Dimension	58.86 – (104.59*(1/SD)) (1.99)	
Dollars/CCF	Appearance	342.92 – (2953.22*(1/SDSQ))[c] (28.25) (–6.80)	0.25
	Dimension	244.40 – (1161.41*(1/SDSQ)) (22.95) (–3.09)	

[a]SD = Log small-end diameter.
[b]Numbers in parentheses are standard errors of estimates.
[c]SDSQ = Log small-end diameter squared.

Table 4—Lumber grade recovery from logs sawn for dimension grade products.

Board grade	Lumber volume
	Percent
#1 Common	<1
#2 Common	2
#3 Common	5
#4 Common	2
Moulding	<1
3 Clear	<1
1 Shop	<1
2 Shop	<1
Select Structural	<1
#1	4
#2	46
#3	31
Economy	9

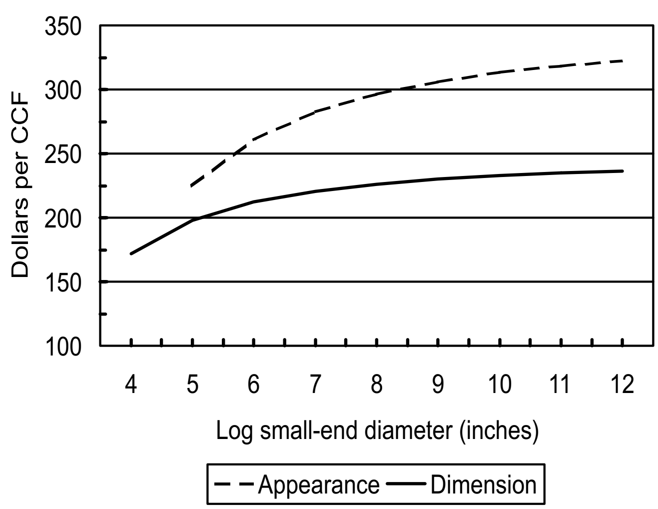

Figure 2—Value in dollars per 100 cubic feet (CCF) by log small-end diameter from logs sawn for appearance and dimension products.

Table 5—Analysis of covariance for value recovery (dollars per 100 cubic feet).

Source	df	MSE	F-value	P-value
1/log SDSQ[a]	1	219487.43	5.14	0.0001
Lumber product[b]	1	158961.34	37.22	0.0001
Product*SDSQ	1	41622.89	9.75	0.0019
Error	349	4270.43		

[a]SDSQ = Log small-end diameter squared .
[b]Appearance and dimension lumber.

Lumber grades are dependent on a variety of board characteristics. Appearance grades (Factory lumber and Boards) are based on characteristics such as stain, pitch pockets, unsound wood, slope of grain, knot size, and wane. Structural lumber is graded on characteristics that affect its strength. Knot type and placement becomes a factor under grading systems for structural lumber. The amount of juvenile wood can also affect the mechanical properties of lumber graded for structural applications. Dimension lumber that can be graded for structural applications (either visually or mechanically) commands a higher price. The research conducted by Erikson and others (2000) indicated

Table 6—Lumber grade recovery from logs sawn for appearance grade products.

Board grade	Lumber volume
	Percent
#1 Common	3
#2 Common	22
#3 Common	66
#4 Common	7
Moulding	<1
3 Clear	<1
1 Shop	1
2 Shop	1

that the lumber manufactured from their ponderosa pine sample yielded a very small proportion of lumber suitable for structural applications. The lumber sawn for this study indicated that just by producing a different primary product (appearance lumber), a gain in value was realized.

Previous work (Lowell and others 2000) shows that there are opportunities to increase the value of appearance lumber through further processing into cut-stock material. The study found boards graded No. 3 Common had the largest increase in value when comparing value as a single piece of lumber versus value as cut-stock material. Board width also influenced yield and value of clear cuttings with 6-inch wide boards being higher in value as cut-stock than lumber. The resource from Flagstaff, AZ, used in this study had a high yield of appearance lumber graded #3 Common and about 60 percent of the boards were 6 inches or greater in width. While the Flagstaff study did not involve evaluation of cut-stock production, there may be an opportunity to recover more value from this material with further processing into secondary products.

Conclusion

Old-growth ponderosa pine traditionally has been used in appearance applications. The results of this study indicated that lumber from small-diameter ponderosa pine growing in densely stocked stands in the Flagstaff, AZ, area has characteristics that make it more valuable as appearance grade lumber than dimension lumber. Results from this study indicated that the wood characteristics of small-diameter ponderosa pine growing in densely stocked stands have sufficient value to warrant its processing for primary products and also may have the potential for secondary processing as a resource for the moulding and millwork industries. This information can be used with new technology and emerging processes to direct this resource to an appropriate end use, where as much value as possible may be captured from the resource to offset restoration costs.

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Explorations of Roundwood Technology in Buildings

Jeffrey Cook

Abstract—A report and critical commentary is presented on the use of small diameter roundwood in building construction in the United States and England. Examples are discussed of roundwood joinery being evaluated at the USDA Forest Service's Forest Products Laboratory, and joinery developed by the British engineering consulting firm Buro Happold, working over 15 years in the development of roundwood use in Europe. Three buildings constructed at Hooke Park in Dorset, England, using small diameter roundwood thinnings as the structural members are profiled, demonstrating the range of applications from small domestic scale to a very large enclosed workshop space. These precedents will inform the construction of traditional Navajo hogan shaped buildings in northern Arizona in the development of local roundwood industries.

Introduction

Roundwood is the original forest product for building construction from the most ancient of human times to today. Whether heavy logs or thin scantlings, the natural form of timbers and saplings has been the traditional geometry of construction. But since the industrialization of forestry and lumbering, especially in the milling of lumber, a prismatic mentality has dominated thinking and practice. Flat surfaces are easiest to join to flat surfaces. But with milled material there is both material and structural loss in trimming a round log into a rectangular cross section.

This exploration paper is part of a comprehensive program to develop a use for the forest thinnings of the northern Arizona ponderosa pine (*Pinus ponderosa*) forest. The project involves the design of a modernized traditional Navajo home, a single room octagonal "hogan," using precut and fitted roundwood parts. Perhaps small industries could be developed locally that would also meet a strong housing need not currently addressed.

Obviously a house is more than a roof. Roundwood and thin logs are undoubtedly best suited to structural frames, trusses, and skeletons. But these need continuous panels or infill if used for shelter. When used as solid walling, or as panel materials, roundwood seems less appropriate. Thus the challenge is to find the most obvious economic uses of these small timbers in small buildings.

Forest Products Laboratory, Madison, Wisconsin

Within the U.S.A., the Forest Products Laboratory at Madison, WI, run by the Department of Agriculture Forest Service, is a national center for research in timber applications to building construction. Experimental, testing, and computational activities are conducted both at the laboratory and by contracted researchers. A personal visit there February 3–5, 2000, allowed intensive discussion with Ron Wolfe, the investigator, about the current technical status of roundwood applications, as well as a review of recent experiments in joinery and fastening. Because wood is strong in tension, but weak in shear, considerable attention has been focused on methods to allow the joint to transfer the potential full strength of the member.

The most thorough recent experiment has been with a "dowel-nut connection" at the end of a piece of roundwood that was conducted by Ron Wolfe. A cylindrical 44.5-mm (1.75-inch) diameter dowel is fitted across the member or strut. A distance of some 7 diameters from the end is recommended. This dowel is tapped and becomes the nut for a threaded rod that extends from the center of the roundwood to connect to a hub, or other transfer point or joint. A tight steel strap or sleeve around the roundwood holds the fibers together to control splitting. Although the experiments were made on 127 mm (5 inch) diameter Douglas-fir (*Pseudotsuga menziesii*) peeler cores, by adjusting structural data the technique could be applied to other wood types, other diameters, and to debarked but untrimmed roundwood, according to the researcher.

A building project of special interest is the design and construction of a full sized circular pavilion of roundwood 7.5 m (25 feet) in diameter, a dimension similar to a hogan, a traditional Navajo dwelling. A 12.5 mm ($\frac{1}{2}$-inch) steel cable around the perimeter makes a tension ring for the truss ends. The roundwood pieces 150 to 200 mm (6 to 8 inches) in diameter are joined by "timber rivets," a steel nail with a wedged head used with a flitch plate. While apparently over designed, the pavilion is both a confidence builder and the subject of full sized monitoring. In addition, at the Forest Products Laboratory a number of other building products are at various stages of development and commercialization that have special interest to those interested in innovative forest products. Trade names such as "fast wall," "tectum," "bison pressed board," and "compreg" are among the many of these upcoming materials.

In: Vance, Regina K.; Edminster, Carleton B.; Covington, W. Wallace; Blake, Julie A. comps. 2001. Ponderosa pine ecosystems restoration and conservation: steps toward stewardship; 2000 April 25–27; Flagstaff, AZ. Proceedings RMRS-P-22. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

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Hooke Park, Dorset, England

The Parnham Trust, Beaminster, Dorset, UK, was founded in 1977 as a nonprofit educational charity by John Makepeace, perhaps the best known furniture designer and maker in Britain. It has been primarily concerned with the teaching and development of the design, production, and business skills needed for the establishment of new enterprises using timber as a primary material. Both forestry and furniture-making skills are taught in apprentice-like settings. Thus Hooke Park was founded as a demonstration forestry tract and test site of 132 ha (330 acres) of Dorset woodland, which contains at its center, a workshop/training facility in the use of roundwood, as well as student residences—a miniature college in the woods.

The educational facilities at Hooke Park consist of a series of custom designed experimental buildings that explore uses of roundwood, a sustainable crop, together with test forests that provide the outdoor classroom. Roundwood has been recognized as a viable construction material worldwide and throughout history. However, it is the use of immature thinnings of little commercial value that is innovative at Hooke, and of most interest elsewhere.

Leading British architects have been involved in the design of each building at Hooke Park. However, the common technical consultant for all structures has been Buro Happold, Bath, with Ted Happold personally involved with carefully engineered high-tech ideas of joinery. Since his death, the home office staff from Bath have continued technical support of structural and services engineering for this program. All buildings demonstrate ideas of what can be achieved with round poles, particularly the first two where details use epoxy plugs, wrappings, and adhesives.

The Prototype House

Richard Burton, of Arends Burton, Koralik, (ABK) Architects, of London designed this single story domestic building (figs. 1 and 2) in 1983, which was completed in 1986. The internationally known partnership of ABK founded in 1961 has been the source of major architectural designs for four decades. The house is used as a seminar room and for administrative offices. It has a size of 11.2 by 8.5-m (37.3 by 28.7 feet) and the material used was Norway spruce (*Picea abies*). A series of A-frames support a doubly curved roof structure using 5.5 m (18.3 feet) long, 60 to 90 mm (2.4 by 3.6 inch) diameter hanging rafter poles with an initial sag of 20 mm (0.8 inch). Thus the roof load is carried in tension and taken out of the poles by epoxy threaded 12 mm steel rods, connected to tension eyes.

The decision to use the thinnings structurally in tension led to the resin joint. To use roundwood only 50 mm (2 inches) in diameter, a very efficient joint was required (fig. 3). This is achieved by drilling a conical hole in the end of the timber and

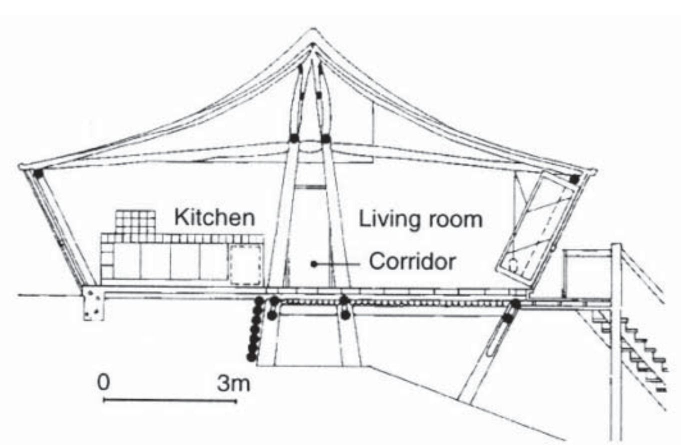

Figure 1—Prototype house, Hooke Park, UK: section.

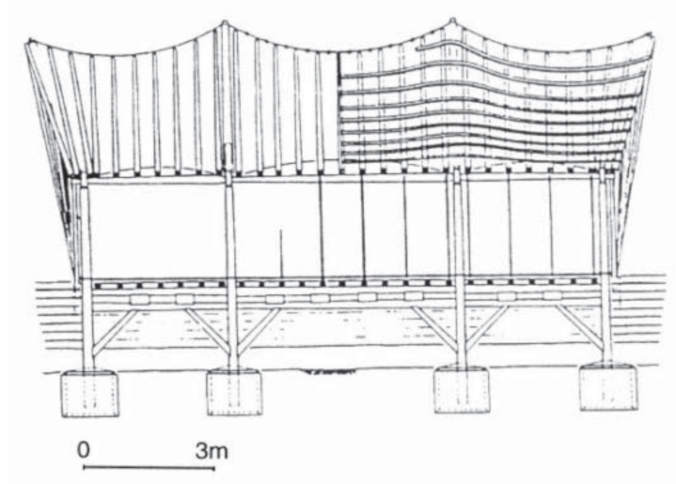

Figure 2—Prototype house, Hooke Park, UK: longitudinal section.

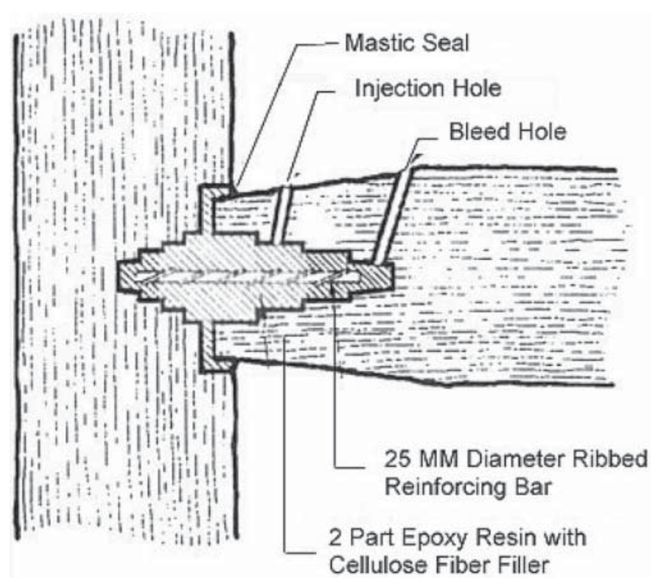

Figure 3—Prototype house, Hooke Park, UK: roundwood design joint.

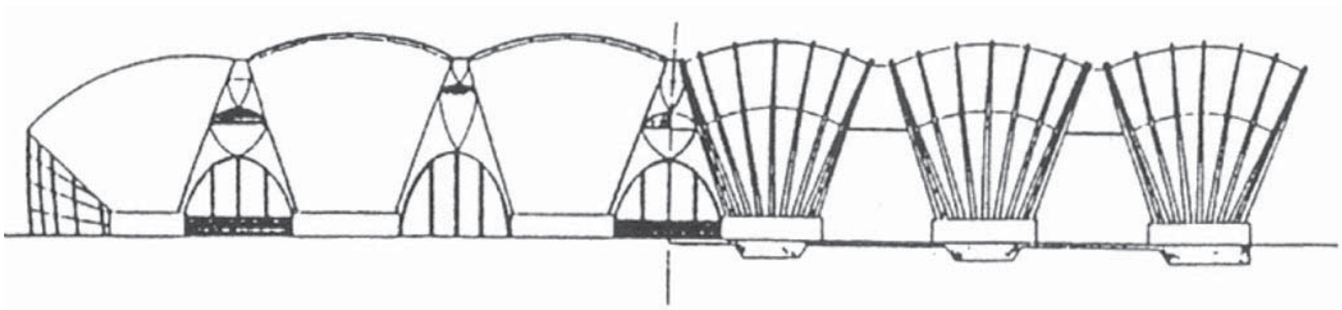

Figure 4—Wood workshop, Hooke Park, UK: elevation/section.

filling it with epoxy resin. The conical hole exposes all the fibers, which are oriented longitudinally, and maximizes the efficiency of the joint. A threaded steel rod is bedded in the resin to enable a connection to be made. In the case of this house, roof timbers hang from the ridge cable. An additional benefit is to avoid metal-to-wood contact, a common area of building material failure. Similarly, the same joint is used in the compression joints as the resin plug helps to bind the center fibers of the roundwood and prevent them splitting apart under pressure.

With the initial sag set at 200 mm (8 inches), the design sag for maximum load is 300 mm (12 inches), which gives a tension force of 7.2 kN. This tension is transferred into the building structure at the ridgeline through a tension joint attached to a wire cable hung between the heads of four A-frames. The tension at the eaves is transferred into a cable spanning between inclined side posts, resisted in turn by a roundwood header beam joining the tops of these posts as a "burnished compression spar." The symmetry of the cross section avoids structural eccentricity.

The outside walls lean outward to help resolve structural, thermal, and rainwater problems, and to provide solar control. An overhanging eave was considered but rejected on the grounds that it is undesirable to have rafters projecting outside the insulated skin and be subjected to different temperature and moisture conditions. This solution also avoided having to cut the vapor barrier around rafters and also avoided wind penetration.

The Wood Workshop Building

The training Wood Workshop (figs. 4 and 5) was also designed by Richard Burton of ABK, and was completed in 1990. It is a 15 m (50 foot) span vault roof using wet Norwegian spruce (*Picea abies*) two bent poles 10.5 m (35 feet) long tapering from 180 mm to 70 mm (less than 8 inches to less than 3 inches) in diameter. A crown section of two poles was used to connect the bent poles through lapping bolted joints. At the base, the poles were fixed to concrete walls using grouted-in-place bolts and plates.

The final building has three shells spanning 15 m (50 feet) wide, forming a total structure 42.5 m (142 feet) long and 7 m (23 feet) high. The shells are formed using thinnings of

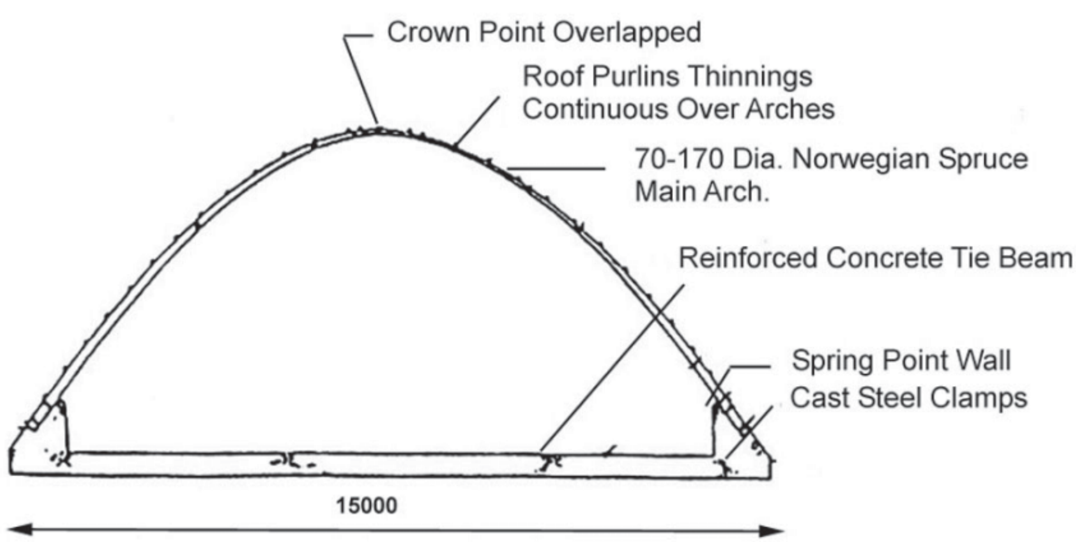

Figure 5—Wood workshop, Hooke Park, UK: structural section.

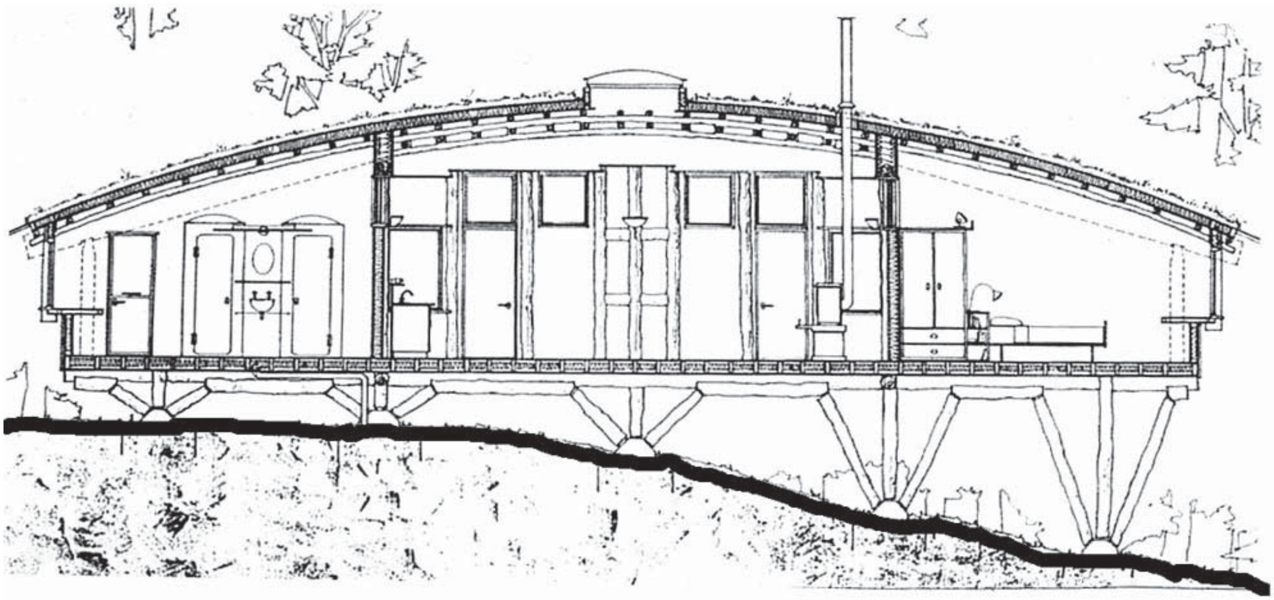

Figure 6—Westminster Lodge, Hooke Park, UK: construction section.

nominal diameter 155 mm (6 inches) at the base to 65 mm ($2\frac{1}{2}$ inches), approximately 9 m (30 feet) long, joined at the crown by a laminate crown arch member. Access for visitors is by a bridge that enters the building at mezzanine level between the workshop and teaching areas. Two of the shells form the workshop area. The third contains the library, seminar rooms, teaching area, and offices, all arranged both at ground floor and mezzanine levels.

The most labor-intensive part of the work involved fitting the plywood edging strips for the membrane and fitting the windows and doors to the complex shapes of the frame on the end walls. This work required several skilled carpenters and took several weeks.

Westminster Lodge

The Westminster Lodge (fig. 6) by architect Edward Cullinan was opened in April 1996. It provides student accommodation with four pairs of private rooms around a central common space. The floor is raised above sloping ground and the load-bearing walls are made from clad roundwood. The main feature of the Lodge is the two-way spanning Vierendeel truss roof (fig. 7), which required eight 3 m (10 foot) long poles to be spliced together to form the top chord. The poles were approximately 100 mm (4 inches) in diameter and were used in four layers to span the 8.4 m (28 feet) in two principal directions. Blocking pieces provided shear connections between the top and bottom layers adding stiffness in the central part of the span.

Named after the Duke of Westminster, who provided substantial funding, the most interesting roundwood element again is the roof. The two-way spanning roof has a large number of alternative load paths to allow for problems with isolated timbers. Using this strategy the variable properties of the roundwood can be exploited without the need to downgrade the stresses to the level that the statistical variation in strength might imply.

The solution was to develop a joint, which could easily be made in the forest, with simple, readily available tools. This joint was a combination of a scarf joint and finger joint. The

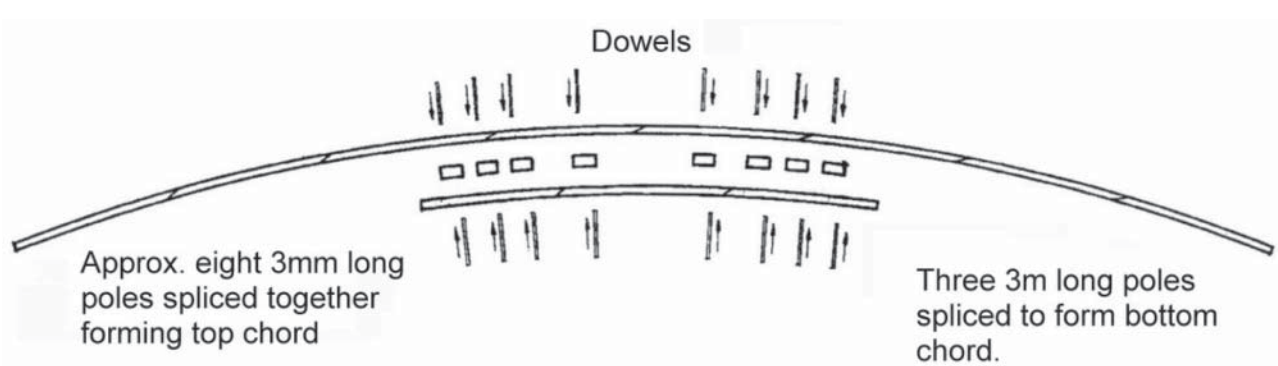

Figure 7—Westminster Lodge, Hooke Park, UK: structural assembly.

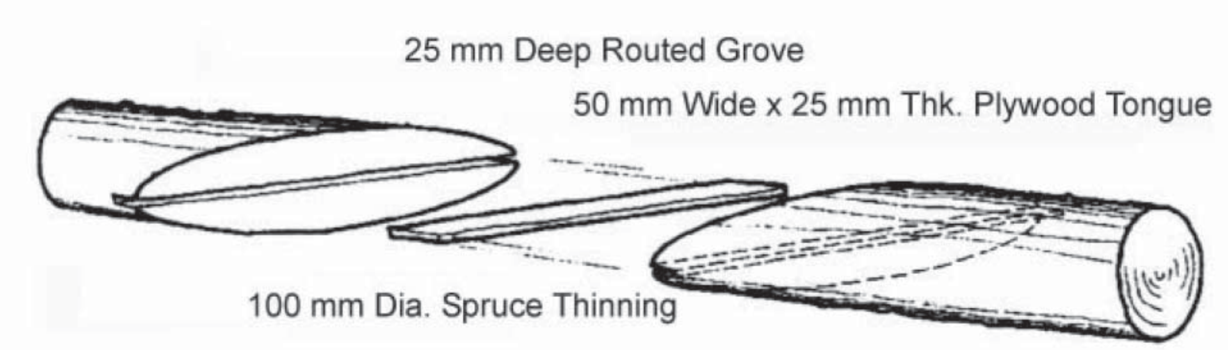

Figure 8—Westminster Lodge, Hooke Park, UK: Roundwood end joint detail.

normal scarf joint is weak and unreliable; the normal finger joint is difficult to make in the forest. By cutting a plywood tongue or key into the roundwood scarf joint diagonally, a simply made, strong, and reliable field joint was achieved by increasing the surfaces of the epoxy glued end joint as shown in figure 8.

By using this joint any length of 100-mm (4 inch) diameter could be produced. Long lengths were used top and bottom with two intermediate layers to span across the 8.4-m (28 ft) roof space. Blocking pieces were used to generate a shear connection between the top and bottom layers, creating the two-way spanning Vierendeel trusses. The continuity of one layer over the side span gives added stiffness to the central span. It also enables a single layer of timbers to span one way across the 3 m (10 ft) between the walls of the side rooms. Instead of steel bolts, 25 mm (1 inch) oak dowels connect the top and bottom chords through a spruce shear block 400 mm (16 inch) long and 80 mm (3 inch) thick that serves as a spacer.

A secondary connector of some interest is the wrapped straps of stainless steel, a standard product, here used to hold two roundwood perlins together as they pass at right angles.

Construction Summary of All Three Buildings

On all three buildings, the roofing materials allow for the variability in form and surface natural to roundwood. On the first two buildings, a flexible membrane surface on a soft insulated blanket was used that fits to the profile shape of the roundwood as well as the double curved surfaces. On the Lodge, boarding was used to support the heavier "green" roof surface of living grass turf, which has already been replaced once. Since the sod is not very thick, it apparently cannot survive even in the damp English climate.

Floors were more difficult to detail. They needed to be covered with level boarding or surfacing and this cannot be achieved with tapered elements. A possible future development would be to use *in situ* concrete on roundwood with shear connectors. Thus the roundwood would act as permanent formwork and as tension reinforcement, with concrete providing the compression element and the finished floor surface.

Evaluations

> The UK visit to the University of Surrey (of an international European technical team) concentrated on a tour of Hooke Park in Dorset, where John Makepeace of the Parnham Trust had set up a school for woodland industry. All the buildings and structures for the school, situated deep in woodland, featured round-pole technology. The buildings are an interesting combination of forestry thinnings with very high-tech ideas carefully engineered by Buro Happold in Bath. They demonstrated ideas of what can be achieved with round-poles particularly when reliance can be placed on epoxy gluing. However, they were seen by the project team to be one-off developmental structures, with only limited relevance to the low-cost higher volume uses required for the round-pole project… (Ranta-Maunus).

This reviewer has similar observations. Although structurally successful, all buildings demonstrated intensive investments of human ingenuity and labor, as well as expensive fasteners in complex high technology joining systems. The similarity of the Happold epoxy resin plug to the American "dowel-nut connection" demonstrates two parallel technical solutions to the same engineering problem. Unfortunately neither seems appropriate to the design of the hogan roof structure. None of the buildings demonstrated use of roundwood for cladding or panel systems. Thus they all have limited lessons for the design of small span, small scale climatically adapted Navajo hogans for northern Arizona. But they are experiments that do not have to be replicated in northern Arizona. Of all those roundwood building ideas that have been built, it is the wrapped straps, and the multi-layer trussed roof at Westminster Lodge that appear to be the most transferable.

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Use of Wood as an Alternative Fuel to Coal and Natural Gas at the Holnam Cement Plant, North of LaPorte, Colorado

Kurt H. Mackes

Abstract—The Holnam Company currently operates a cement plant north of Laporte, CO. The plant is attempting to use wood as an alternate fuel to coal and natural gas. The principal objective of this project is to investigate the extended use of wood as an alternate fuel at the plant. Tests conducted at Holnam indicate that wood is suitable for use at the plant and Holnam could use up to 350 tons of clean wood wastes and residues per day. A substantial network of wood suppliers, drawing wood from landfills, wood processors, and forests, will be necessary to meet overall plant requirements. Successfully converting Holnam to wood would provide economic benefits to both Holnam and the community. Larimer County would benefit because the majority of clean wood will be diverted from the waste stream into its landfill. Using forest residues at Holnam could help reduce the risk of wildfire, increase public safety, and improve forest health. We view converting this plant to wood as only a beginning. If Holnam successfully converts to wood as the primary fuel source, then other plants in the region could similarly use wood wastes and residues as fuel.

Introduction

The Holnam Company operates a cement plant north of Laporte, CO. Founded in 1926, the plant currently produces 460,000 tons of cement annually. The plant utilizes approximately 75,000 tons of coal and 3,600 million cubic feet of natural gas annually in the cement making process. Early in spring 1999 Holnam began considering the possibility of converting the primary source of fuel used at the plant from coal to wood. The principal objective of this project is to investigate the extended use of wood as an alternate fuel source to coal and natural gas at this plant. Several private organizations and government agencies are supporting Holnam in their efforts to achieve this objective, including the city of Fort Collins, Larimer County, the Colorado State Forest Service, the USDA Forest Service (State and Private), and Colorado State University. Included in this publication is project background, a brief description of the cement making process, a discussion of conversion technology, and the process to commercialization. The anticipated benefits and challenges of using wood at the Holnam plant are also discussed.

Project Background

From April until autumn 1999, representatives of Holnam met with Dr. Dennis Lynch (Professor Emeritus, Colorado State University) and Larimer County personnel to discuss the potential of using waste wood from the Larimer County landfill as an alternative fuel at Holnam. In August, John Zerbe (USDA Forest Products Laboratory) visited Holnam to observe the operation and wrote a report regarding the potential for utilizing wood at the plant (Zerbe 1999). A meeting to discuss alternatives for diverting clean wood from the Larimer County landfill occurred in September between representatives of Holnam and the city of Fort Collins Natural Resource Department. In November and December Holnam met with Janelle Henderson (Director of the Larimer County Natural Resources Department) and other Larimer County officials to discuss development of a strategy for diversion of landfill material.

The Coloradoan, a local newspaper, published an article on December 14 on the intent of Holnam to utilize clean wood wastes as an alternative fuel source. On December 16, a public meeting was held in the Larimer County courthouse. Although reviews were mixed, there was generally a positive response from the community toward utilizing clean wood waste at Holman. Also on that day, Channel 9 Television (NBC) of Denver, CO, toured the Holnam plant and filmed a story that aired on that day's 5:00 p.m. newscast. A follow-up article was published by *The Coloradoan* on December 17. Subsequent articles appeared in *The North Forty* newspaper and the Loveland, CO, *Reporter Herald*. Generally, these articles were favorable.

During the public hearing held on December 16, attendees determined by consensus that four principal issues needed to be addressed. These were:

1. Potential fuel sources
2. Transportation
3. Process modifications
4. Air pollution concerns

Four committees were established to investigate these issues. These committees met throughout the winter and spring 2000. Some of the findings are discussed in subsequent sections.

In addition to committee meetings, two public meetings in March and April were held in Laporte, sponsored by the Laporte Business Alliance. The local community had many

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In: Vance, Regina K.; Edminster, Carleton B.; Covington, W. Wallace; Blake, Julie A. comps. 2001. Ponderosa pine ecosystems restoration and conservation: steps toward stewardship; 2000 April 25–27; Flagstaff, AZ. Proceedings RMRS-P-22. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

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concerns, many of which had been expressed in the previous December meeting and which were being addressed by the committees established at that meeting.

One particular new concern pertained to the Title V permit that Holnam was in the process of applying for. This permit is required by the State prior to using wood or other alternative fuels on a production scale at the plant. The initial permit application (which has since been revised) submitted by Holnam listed a broad range of nonhazardous materials that could potentially be used as fuel at the plant. Even though only clean wood had been used in testing to date and there had been minimal opposition to Holnam utilizing clean wood at the plant, there were concerns that Holnam would burn other materials such as plastics and rubber tires if permitted to do so. Therefore, the community was concerned over what materials Holnam would be permitted to utilize. There were articles in local newspapers reflecting these concerns. In particular, there was considerable resistance to using rubber tires at the plant. There was also concern regarding wood from the landfill. While the majority of this wood is clean, there is a percentage of this wood that is contaminated with a variety of substances including finishes, treatments, and adhesives. The impacts of these contaminants on air emissions are uncertain and need to be investigated. Nonetheless, Holnam was focusing on obtaining a Title V permit to use clean wood.

The Cement Making Process

The Holnam plant north of Laporte uses a unique process to manufacture cement. Limestone is crushed to reduce its size and then blended with materials such as shale, sandstone, iron ore, mill scale, fly ash, clay, or sand into a dry powder. The mix is then normally processed in a calciner that burns the organic material in it and drives off CO_2. It is then fed into the elevated end of the 14.5 feet wide by 190 feet long rotary kiln at the plant. Materials in the kiln pass through controlled zones that heat them to 2,450 °F. Flame temperatures in the kiln exceed 3,000 °F. The intense heat chemically alters the materials forming walnut-sized pellets called cement clinker.

The cement clinker is then cooled. Heat from the clinker is extracted by air flow. This heat is used to supplement the heat from fuels used in the kiln. The clinker is mixed with a proportioned amount of gypsum in a finishing mill where steel balls pound the clinker into a fine powder (cement). The cement is packaged in bags for shipping or shipped bulk to retail outlets and construction sites.

Conversion Technology

In 1999 and 2000, research and testing were conducted to determine the best approach for converting the Holnam plant to wood as the primary fuel source. A series of test burns were conducted and some equipment purchased to facilitate testing. Important findings from this work, along with a brief discussion of wood as a fuel are presented.

Wood has a heating value that ranges between 8,000 and 12,500 Btu per ovendry pound, with most species falling between 8,000 and 10,000 Btu per ovendry pound (Inc 1979). Bark ranges between 7,500 and 10,500 Btu per ovendry pound. This compares to a range of 6,900 to 14,300 Btu per pound for coal. Based on the type of bituminous coal typically used at Holnam, which has a higher heating value of 11,000 Btu per pound, approximately 1.3 tons of dry wood yields the same amount of heat as 1 ton of coal.

Although heat production is affected by numerous factors (Ince 1979), one factor that dramatically affects recoverable heat generated by combusting wood is moisture. Because wood is hygroscopic it readily picks up and loses moisture. The higher the moisture content, the lower the amount of recoverable heat available from combustion. Therefore, to maximize recoverable heat it is beneficial to dry wood prior to combustion.

In May 1999, the first test burns were conducted at the plant with good results. A subsequent test incorporating wood into kiln feed to test material flow was conducted in August. Also in August, Holnam arranged for the lease/purchase of two screw augers to feed saw dust and fine wood chips into the calciner. In October, the first calciner test burn using dry sawdust fed into the system with the screws was conducted. The calciner responded positively and no air quality problems were detected. A second burn was conducted in November using one load of wet sawdust and one load of dry sawdust. The calciner did not respond well to wet sawdust; however, the dry sawdust burned well and the coal feed was shut off during this phase of the test, proving that wood can replace coal in the system.

Later in November, a test burn was conducted feeding wood chips into kiln feed to test material flow. During December, 21 tons of green pine chips were supplied by the Colorado State Forest Service for testing. Wood chips were fed into the roller mill with the standard raw rock feed. Although the test was short, results were positive. Later in December, a test was conducted to find out if screw augers would handle wood chips. The chips did not feed well through the screw augers and heavier equipment is required.

Testing continued into January, February, and March 2000, culminating with an extended test using 98 tons of green pine chips processed from beetle-killed pine trees provided by the Colorado State Forest Service. Chips were put into the system with both the raw rock feed and the coal feed. Approximately 28 tons of chips were introduced with the raw rock and the remaining 70 tons were introduced with coal. Wood introduced into the coal feed was mixed with coke at a 1:1 ratio by volume, and then with coal at a feed rate of 15 percent wood/coke and 85 percent coal. Generally, these tests were very successful and the wood worked well with existing systems at the plant.

Significant process modifications may still be necessary to utilize wood at all firing systems of the Holnam plant. There is a need to further study how wood can be fed to firing systems and what modifications are necessary to use it. The main barrier has been developing a feed system for wood chips that works well with existing firing systems at the plant. The screw auger system installed for test burns using sawdust did not work well with 1.5 inch wood chips. Mixing rock and wood together to produce “crusher” feed has been successfully tested as one alternative for introducing wood into the process. However, the plant needs the capability to selectively get sawdust and small wood chips to any of the three existing fuel pumps. Feeding wood into the existing coal bin is possible if mixed at up to 10 percent with coal. This provides some wood feed to all firing points simultaneously; however, this will not be sufficient for complete replacement.

Further testing is scheduled in the future to evaluate systems developed to handle and feed wood. Air emissions from the plant (including stack testing) will be monitored during testing. With the successful completion of this testing, a portion of the plant should be ready to utilize wood on a commercial scale. As noted previously, Holnam is currently in the process of applying for a Title V permit that is required by the State prior to burning wood on a production scale. Conducting this testing is a necessary step toward obtaining this permit. Work required to develop the wood handling systems necessary to completely convert the plant was scheduled to be completed by summer 2000.

Process to Commercialization

Wood as the primary fuel source at Holnam requires that two key issues be addressed. These are completing modification of existing plant processes to accommodate wood (addressed previously) and procuring an adequate supply of low cost wood in a form suitable for use at the plant. To date test burns have shown that wood can be used to fuel plant processes. Although preliminary studies indicate that a sufficient supply of wood exists, there is a need to establish a delivery system for getting wood to the plant, including collection (sorting), size reduction, drying, and transportation.

Based on testing conducted at the Holnam plant, clean wood material in the form of sawdust or chips up to 1.5 inches in length work well with existing systems. To maximize recoverable heat from combustion, it is important that the wood be dry, preferably having a moisture content of less than 10 percent. Based on economic constraints, wood will also have to be low cost. Four potential sources of low cost wood have been identified by Ward and others (1999):

- Municipal waste
- Construction waste and demolition debris
- Primary and secondary wood manufacturers
- Forest residues

Some information has been generated on potential sources of wood. A preliminary wood biomass report by Lynch (1999) noted that 187,237 tons of waste was delivered to the Larimer County landfill in 1996. An estimated 29,563 tons of commercial wood waste (primarily construction debris) and 2,410 tons of wood from residential waste were included in this total. In 1997, the Larimer County landfill received 166,683 tons of waste. The reduction was due to the establishment of a new Waste Management landfill near Ault and the majority of their waste streams went to that landfill. In 1997, an estimated 26,318 tons of wood were in commercial waste delivered to the Larimer County landfill, which amounts to approximately 72 tons per day. Additional wood could come from the Waste Management landfill. Waste Management estimated that they currently receive approximately 1 ton of construction wood waste per day.

Because wood is often intermingled with other construction debris, the clean wood suitable for fuel at Holnam would have to be sorted. Lynch (1999) reported that all parties contacted suggested that some type of wood collection system should be possible. Once collected the wood must be reduced to a suitable size through chipping or other more suitable methods. Whatever the method of sorting and size reduction, the consensus is that the tonnage of sorted clean wood suitable for fuel would be something less than the total amount delivered to the landfills. Based on fuel requirements provided by Holnam, up to 350 tons of wood per day will be required to convert the plant entirely to wood; therefore other sources of wood will be required.

Another potential source of fuelwood for Holnam is primary and secondary wood manufacturers along the Front Range of Colorado. Primary manufacturers along the Front Range include several small sawmills, while secondary manufacturers include millwork, cabinet, and furniture companies. Residues from secondary manufacturers have a relatively high percentage of hardwood with good Btu value. In addition, this wood tends to be relatively dry and as a result, burns well. Several local manufacturers have provided wood for testing. In one test where oak and maple residues where used, Holnam was able to cut off the coal feed to the calciner for several hours. While none of the primary and secondary wood manufacturers individually generate a substantial amount of wood residues, collectively they do. Therefore, further investigation is needed to determine the amount of low cost wood that could be procured from these manufacturers. Other possibilities, including used pallets, should also be investigated.

The forests of Larimer County could also be a significant source of supply for Holnam. Lynch (1999) commented that the private, State, and Federal forest lands in the county have more trees existing now than at any time in recorded history. Lynch reported that NEOS Corporation estimated over 1.4 million tons of wood could potentially be removed from Larimer County in forest restoration projects. These materials would have to be removed, chipped, dried, and transported to Holnam, resulting in a cost. Based on preliminary estimates, in the absence of government subsidies, supplying wood from the forest to Holnam within economic guidelines for fuel costs established by the company will be a challenge. However, because of the enormous resource and potential public benefits, there is a need to investigate how this might be accomplished.

Larimer County and the Colorado State Forest Service have also initiated a project to assist private land owners with disposal of residues from their properties. Ten mobile collection sites are going to be established strategically in the county and these sites could present a wood supply opportunity for this project.

Benefits and Challenges

The economics of utilizing wood at Holnam are tied closely to collection, size reduction, transport, and drying costs. The company is anticipating making a significant capital investment (in excess of $1 million) to build a system capable of handling wood. In accordance with corporate guidelines, the payback period must not exceed 3 years. This limits the amount that Holnam can pay for wood wastes and residues. Given size reduction and transport costs, which depending on haul distance could easily exceed economical limits, it is generally thought that the wood will have to be procured at little or no additional cost. Fortunately, there are sources of wood, particularly waste wood destined for landfills, that can be diverted at an acceptable cost. In addition, Holnam will attempt to take full advantage of tax credits associated with using biobased fuels. These tax credits could potentially improve the economics of converting the plant to wood fuel.

Because the availability of wood could be seasonal or periodically in short supply, coal and natural gas systems currently in place at the plant can be used as backup. The plant currently stockpiles 6,000 tons of coal, which is a 1 month supply. Approximately 7,500 tons of wood, a supply for a comparable period, would also be stockpiled, if available.

Environmentally, there should be a reduction in air emissions from the plant if wood is burned in place of coal. Zerbe (1999) reported that combustion of wood should be relatively free of sulfur, heavy metal, and particulate emissions when compared to coal. The absence of sulfur could be a major benefit because Holnam currently scrubs sulfur from coal with calcium oxide (CaO). Although CaO is abundant, reducing the level of scrubbing could potentially increase cement yield and reduce the amount of calcium sulfate that must be disposed of. Although clean wood is relatively free of heavy metals, there is potential for introducing contaminated wood into the process; for example, lead paint found in demolition debris from older buildings. It will be important to follow EPA regulations to separate clean wood for recycling from contaminated wood. Particulate emissions from complete wood combustion are anticipated to be relatively low because the ash content of wood is small, typically around 1 percent. This compares to greater than 5 percent for the coal currently used by Holnam. In either case, ash becomes an integral part of the plant product.

Additionally, there are direct benefits to Larimer County and local city governments. Using this wood as fuel provides an alternative to disposing of wood in the landfill. At the Larimer County landfill, removing wood from the stream of trash into the landfill will save space, extending the life of the landfill. Based on the estimated value of space at the landfill determined by the county, removal of wood wastes could save over $1 million annually. Another potential benefit to the landfill is a reduction in methane gas emitted from wood decomposition. If done soon enough, this could eliminate the need for a $2 million methane recovery system.

Using forest thinnings and residues would also provide benefits to the community. Forests found in the region are typically composed of dense overcrowded homogeneous stands of small diameter trees with little or no commercial value. Thinning out these stands and promoting the concept of defensible space around homes and other structures would increase public safety by reducing fire risks. Reducing stand densities would also improve forest health. Proper management of both private and public forest lands is often hindered by the lack of places to take forest residues. They are typically burned, which contributes to poor air quality, or are sent to a landfill. The establishment of 10 mobile collection sites throughout Larimer County should help.

Once capital costs are recovered, the plant should see a significant reduction in the cost of doing business, because energy represents a significant portion of that cost. Therefore, the plant should become more competitive and profitable to operate. Improving the profitability of the Holnam plant has many potential benefits to the local community. Even though plant processes are continually upgraded and modernized, this plant is relatively old, so there are always concerns of the plant closing because of an inability to compete with cement manufactured at newer state-of-the-art facilities. Plant engineers estimate that there are sufficient raw materials available at this location to operate for at least 50 years. Successfully completing this project should help keep the relatively high paying jobs at the plant in the area and stabilize the local economy for many years.

There will probably be a need to form a company to supply wood to the Holnam plant. This company would likely operate as a small private business employing four to five people. It would collect, sort, and process wood into a size suitable for use at Holnam. The company would probably operate from a base located rurally in the area around Laporte, using two or more trucks to transport wood.

Concerns expressed at public meetings were related primarily to the impact of wood combustion on air emissions from the plant and the impact of increased truck traffic on the local road system and community. As discussed previously, it is thought there should actually be an improvement in air quality over that experienced with burning coal. Testing and monitoring of stack emissions by an independent consulting firm will be necessary to verify that the plant is in compliance with air quality standards. A study needs to be conducted to minimize the impact of transporting wood on local roads and community. The study must consider the location of wood suppliers in relationship to the plant, establishing truck routes and travel times that minimize impact. This evaluation will be ongoing as the number of suppliers and the flow of wood into Holnam increases.

There was also some concern expressed about introducing wood in certain phases of the cement making process. Zerbe (1999) stated that there was the potential for fire (and a remote possibility of explosion) in the roller mill operation at the plant if wood was introduced at this location. This was because of the dust cloud created by the milling operation and the high temperature of the gas stream flowing into the mill. In extreme conditions the gas stream entering the mill can exceed the flash point of wood, and if concentrated wood dust came in contact with the gas at this point, a fire (or explosion) could result. However, mixing the rock feed with the wood and then introducing it into the roller mill appears to minimize this risk because limestone is relatively inert and comprises 92 percent of the material mass. This coupled with additional safety precautions (primarily reducing the temperature of the gas stream into the mill, which has been done) should allow for the safe introduction of wood into the process at this point.

Conclusions

If Holnam successfully converts to wood as the primary fuel source, other cement plants throughout the region could similarly investigate using wood wastes and residues as fuel. Because of the preliminary research at Holnam, several other facilities have already expressed an interest. There are also Holnam plants in Canon City, CO, Montana, Utah, and Oklahoma that could potentially use biomass as fuel if the LaPorte plant successfully converts.

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