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Fort Collins, CO

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Recent fires have spawned intense interest in fuel treatment and ecological restoration activities. Scientists and land managers have been advocating these activities for years, and the recent fires have provided incentives for federal, state, and local entities to move ahead with ambitious hazard reduction and restoration projects. Recent fires also have increased public awareness about the risks and hazards of living in wild areas. The scientific basis for ecological restoration and fuel treatment activities is growing, but remains largely unsubstantiated, with isolated exceptions. Over 300 participants from all over the United States convened in Ft. Collins, Colorado, to learn from 90 oral and poster presentations.

Sponsors

- USDA Forest Service
- Colorado State Forest Service
- Colorado State University
- Joint Fire Sciences Program
- Society of American Foresters
- Western Forest Fire Research Center (WESTFIRE)

Conference Coordinators

- Dr. Phil Omi, Professor, Department of Forest Sciences, Colorado State University
- Dr. Linda Joyce, Research Project Leader, Rocky Mountain Research Station, USDA Forest Service

Editors' Note

Papers presented from the conference were subjected to peer technical review. The views expressed are those of the presenters.

Cover photo: Biscuit Fire, Siskiyou National Forest, 2002. Photo by Thomas Iraci, USDA Forest Service. Courtesy of *Fire Management Today* magazine.

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Foreword

Conference on Fire, Fuel Treatments, and Ecological Restoration: Proper Place, Appropriate Time

Philip N. Omi and Linda A. Joyce, Conference Co-Coordinators

Fuel treatments and ecological restoration are not novel concepts to the land manager. Fuels have been manipulated in southern California since the 1930s, and ecological restoration has been advocated since the 1968 Leopold report re-defined the need for manipulating national park landscapes. Even earlier, advocates for light-burning of the nation's forests or for manipulation of wildlife habitat urged public and private land stewards to restore fire to regional landscapes. Perhaps if we had heeded their words we might have been spared the litany of "worst" fire seasons, such as 1988, 1994, 2000, and 2002. Of course, new circumstances have surfaced in response to these fire seasons, most notably the healthy forests initiative, national fire plan, and cohesive strategies for dealing with the nation's wildfire problems. Unlike previous agency mandates, the national fire plan has been accompanied with dollars to perform treatments—and dollars for which we are owed accountability. Still, numerous questions remain unresolved about the proper place and appropriate time for fuel treatments, to say nothing about suspicions that ecological restoration might be a cover-up for getting the cut out of the nation's forests or other nefarious acts. In short, this conference could not have been better-timed in terms of serving the public interest!

This conference was born of a Joint Fire Science research project at Colorado State University looking at the status of our knowledge regarding fire regimes and fuel treatments. A glance through the contents of these proceedings reveals a much broader perspective—for this we can thank all who responded to our initial call for abstracts. The core of the technical program for this conference focused on fuel treatment for fire hazard reduction, and ecological restoration case studies (including ecosystem effects). In addition, we are pleased to include papers that examine treatment economics and social issues, fire regime considerations, and landscape planning perspectives. In addition to these same topical areas, poster abstracts cover studies of wildfire effects, methodological considerations, and monitoring guidelines. In total, we expect that this collection of papers represent a unique and timely contribution to the literature. Further we hope that these proceedings will provide an important reference for future efforts.

Levels of interest for this conference (i.e., abstracts, registrations, general inquiries) exceeded our greatest expectations, especially since we started planning shortly after September 11, 2001. Participants included nearly 300 scientists, land managers, students, and interested individuals from public and private sectors. Our hope is that the papers included here will shape the future of fuel manipulation and ecological restoration programs here and around the globe.

Special thanks are owed to the steering committee that provided invaluable assistance with planning and implementing conference activities. Members of the steering committee are noted below:

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*We add a special tribute to the memory of Paul Gleason, who served on our steering committee and also coordinated the four conference field trips. Paul had a special passion for the tasks confronting fire and fuels managers. He was an avid reader and discussant about fire research and an eloquent spokesperson for firefighter safety. Above all he will be remembered for his contagious enthusiasm and respect for the land.

Fuel Treatment Performance and Fire Hazard Reduction

Fuel Treatments: Opening Remarks

Wayne D. Shepperd[1] and Sarah Gallup[2]

One person most often quoted (ironically by both sides) in the ecowars of land management policy on federal lands is Aldo Leopold. His classic book on wildlife management was the text for the undergraduate Wildlife Management class at CSU many years ago and sits on many office bookshelves. In it he examined the habitat needs of various upland game species and concluded that many of these animals and birds thrive where there is an abundance of what he termed "edge," where a variety of habitat conditions were available. In today's parlance, we might refer to that as "habitat diversity." Different species of plants and animals require different conditions with no single condition being optimal for all. Therefore, a variety of conditions within a landscape are likely to benefit most flora and fauna occurring within it. Such diversity is also beneficial for *processes* that are active within a landscape. If we take the next step and refer to landscapes containing flora, fauna, and processes as *ecosystems*, we might also refer to those containing sufficient diversity as being in a "properly functioning condition."

Regardless of the term, the thought behind the idea is every bit as valid today as when Leopold observed it – too much of any one condition in an ecosystem is not good. It is as true for plants as it is for animals in an ecosystem. And, it is also true for the disturbance processes that affect ecosystems, including fire. The key for providing for all components of an ecosystem is to maximize diversity – to make room for all. Exactly how we do that on the ground will depend upon the particular ecosystem we are dealing with, the particular mix of conditions that currently exist within it, and the particular species or process of concern. No one approach is ever likely to fit all circumstances. Remembering this will be helpful to readers of this section as we learn in the following papers about various approaches that have been used to reduce the risk of unacceptable fire in forest ecosystems. No single technique is likely to be perfect for all situations. Just as he said when he advised us to "save all of the pieces" when tinkering with ecosystems, Aldo Leopold would likely also advise us to "bring more than one wrench" to do the job. So, let's see what's in the tool kit!

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Performance of Fuel Treatments Subjected to Wildfires

Erik J. Martinson[1] and Philip N. Omi[1]

Abstract—*Fire severity was evaluated in eight recent wildfires with standardized methods in adjacent treated and untreated stands. Sampled sites occurred in a variety of conifer forests throughout the Western United States. Treatments included reduction of surface fuels and crown fuels, both in isolation and in combination. Synthesis of our results indicates that treatment effectiveness is related to differences in tree size (mean diameter) between treated and untreated stands (p<0.001), as well as estimated historic fire frequency (p<0.1). Our results suggest that fuel treatments will be most effective when they complement ecosystem restoration objectives, such as the removal of small trees from ecosystems that historically experienced frequent fire.*

Introduction

Treatments to mitigate fuel accumulation and fire hazard have long been advocated (Weaver 1943). Federal land management agencies have greatly expanded fuel treatment programs in response to increased public attention on wildfire hazards. The unprecedented scale of current fuel treatment activities has intensified debate regarding their means, objectives, and outcomes. Some interest groups maintain that fuel treatments via mechanical thinning are a disguise to expedite timber harvest. Others wonder if potential negative impacts of fuel treatments (e.g., smoke production, exotic invasions, soil damage) outweigh any benefits. Some question whether fuel treatments even decrease fire potential.

Theory does suggest that fire intensity may be exacerbated by fuel treatments (Agee 1996). Canopy reduction exposes surface fuels to increased solar radiation, which would be expected to lower fuel moisture content and promote production of fine herbaceous fuels. Surface fuels may also be exposed to higher wind speeds, accelerating both desiccation and heat transfer. Treatments that include prescribed burning may increase nutrient availability and further stimulate production of fine fuels. All these factors facilitate combustion, increase rates of heat release, and increase surface fire intensity.

However, theory also indicates that treatments can reduce the likelihood of extreme fire behavior involving forest canopies. Crown fire initiation and spread depends on vertical and horizontal fuel continuities (Van Wagner 1977) that are typically reduced by treatment. Thus, treatments that reduce canopy fuels may increase and decrease fire hazard simultaneously. Justifications for expansion of fuel treatment practices are therefore tenuous without empirical assessments of their performance in wildfires. However, the question of fuel treatment effectiveness has received surprisingly little scientific attention.

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Previous Research

An exhaustive literature search for evidence of fuel treatment effectiveness in the United States uncovered just 31 such publications since 1955. Some perspective on the void this represents is gained by considering the number of citations in the Fire Effects Information System (more than 25,000 op. cit. Fischer et al. 1996). Further perspective is gained when these publications are organized by the methods used to assess treatment effectiveness. Fuel treatments in more than half of these publications were not tested by actual wildfires and the treatments themselves are also hypothetical in nearly a third of them. Among the 14 studies of treatments subjected to actual wildfires, just five quantify how fuels were changed by the treatment: a necessity if effective guidance is to be provided for future fuels management. Nonetheless, these publications do indicate with near unanimity that fuel treatments mitigate wildfire behavior and effects (table 1).

However, even among the 10 studies that assess the severity (as opposed to size) of actual wildfires, comparisons are complicated by lack of consistency both in the criteria for evaluating fire severity and in definitions of fuel treatments and controls. Several studies evaluate damage to tree crowns, but some authors define severe damage as more than 50% scorch, while others use 100% scorch or complete consumption as their highest rating. Treatments involve commercial harvest in several of the studies with activity fuels subsequently burned. Some of these assess treatment effectiveness with comparisons to areas where no management activity occurred; others use harvested areas where slash was left untreated. One study (Weatherspoon and Skinner 1995) makes comparisons to both types of areas, allowing interpretation of treatment effects as either positive or negative. This was the only study found that provides any indication that fuel treatments may be ineffective. But the weight of evidence represented in these few studies is far from overwhelming, especially since sampling designs are inadequately described in the earlier publications. Thus, neither existing theory nor available empirical evidence provides much clarity on the question of fuel treatments and the conditions that influence their effectiveness when tested by wildfire.

Table 1—Characteristics and findings of published studies that document the performance of fuel treatments in actual wildfires.

Study	Treatment	Control	Response	Direction[a]
Moore et al. 1955	Prescribed burn	No activity	Crown damage	↓
Cumming 1964	Prescribed burn	No activity	Tree mortality	↓
Wagle and Eakle 1979	Prescribed burn	No activity	Live tree density	↑
Van Wagner 1968	Thin and prune	No activity	Tree survival	↑
Agee 1996	Thin and burn	No activity	Crown fire	↓
Oucalt and Wade 1999	Thin and burn	Thin	Tree mortality	↓
Vihanek and Ottmar 1993	Harvest and burn	Harvest	Soil damage	↓
Hall et al. 1999	Harvest and burn	Harvest	Crown fire	↓
Weatherspoon and Skinner 1995	Harvest and burn	No activity	Crown damage	↑
Omi and Kalabokidis 1991	Harvest and burn	No activity	Crown damage	↓

[a]Direction indicates the amount of the measured response in the treated areas relative to that measured in the untreated control areas. For example, less crown damage was evident in the treated area than in the control area of the Moore et al. (1955) study.

Objectives

A project was initiated in 1995 to begin filling the research void on fuel treatment effectiveness. Eight wildfires have been investigated to date with details previously described (for details see Pollet and Omi 2002; Omi and Martinson 2002). Here we use meta-analytical methods to synthesize the results from these eight study sites. We investigated the ability of several variables (type, age, and intensity of treatments and the historic frequency of fire in the treated ecosystems) to explain differences among study sites in observed treatment effects.

Methods

We identified potential study sites for this research by advertising our interest at professional conferences and over the internet, networking with federal land managers, and initiating contact after large wildfires in areas known to have an active fuels management program. Thirty-eight wildfire areas were considered for sampling, but most of these failed to meet our selection criteria.

Potential study sites were restricted to wildfires that included adjacent treated and untreated areas within the perimeter and where treatment histories were documented and spatially explicit. We chose a narrow definition of fuel treatment that included only non-commercial or pre-commercial activities involving mechanical thinning (i.e., "low thinning"), debris removal, and/or broadcast burning with moderation of wildfire potential as a stated objective. Areas were defined as "untreated" if they had received no management action within the last 20 years, while treatments were applied within the last 10 years. We avoided areas where significant barriers (e.g., cliffs, major roads or drainages) or suppression activities likely impeded fire spread, as well as areas where post-fire salvage activities had taken place or were imminent. We further restricted our sampling visits to forested ecosystems, since these are where treatments are most often applied (Morrison et al. 2001) and where our methods are most applicable. Ten sites met our selection criteria, but two of these were excluded due to their proximity to areas we had sampled previously. Characteristics of the eight sampled sites are provided in table 2.

Data were collected at all sites from variable radius plots (Avery and Burkhart 1994) in adjacent treated and untreated stands. Measurements included stand density and basal area, tree diameter and height to pre-fire live crown, height

Table 2—Characteristics of the eight study sites included in the synthesis.

Site	Treatment type	Treatment age (yr)	Vegetation	Historic MFI[a]
'94 Webb fire, MT	Prescribed burn	4	Ponderosa	14
'94 Tyee fire, WA	Thin and burn	10	Ponderosa	22
'94 Cottonwood fire, CA	Thin, slash removed	4	Ponderosa	28
'96 Hochderffer fire, AZ	Thin and burn	1	Ponderosa	16
'99 Fountainebleau fire, MS	Prescribed burns	1	Slash pine	9
'99 Megram fire, CA	Pile and burn	2	Mixed conifer	59
'00 Cerro Grande fire, NM	Thin	1	Ponderosa	17
	Thin and burn	4	Ponderosa	17
'00 Hi Meadow fire, CO	Prescribed burn	1, 3, 5	Mixed conifer	52
	Thin	9	Mixed conifer	52

[a] Historic mean fire interval (MFI) was estimated for each site from the nearest available fire history information.

of needle scorch and bole char, percent crown volume scorch, and standardized ratings for stand damage and depth of ground char (Omi and Martinson 2002).

We used the standard meta-analytical software Metawin (Rosenberg et al. 2000) to relate fuel treatment effect sizes (i.e., Hedge's standardized mean difference, see Rosenberg et al. 2000) on percent crown volume scorch to each of several site characteristics. These included the type of treatment (in terms of the fuel stratum treated: canopy, surface, or both), treatment age when tested by wildfire (grouped into categories of 1 year, 2 to 4 years, and 5 to 10 years), standardized mean differences in tree densities and diameters between treated and untreated areas, and the estimated historic fire frequency of each site.

Historic fire frequency was estimated for each study site from proximal fire history studies. We identified and selected applicable fire histories from those included in a quantitative synthesis of fire history information (for details see Martinson and Omi, in press). Historic fire frequency was standardized from each fire history by calculating the inverse of the average annual point-specific probability of fire in the period 1710-1779. We estimated the historic fire frequency for each site as a weighted (inversely proportional to variance) average of fire frequencies calculated from the nearest (in terms of latitude, longitude, and elevation) available fire histories.

We employed parametric mixed-effects models in all analyses. Comparison of the size of the random variance component (i.e., variation not explained by sampling error at each location, σ_θ^2) when an explanatory variable is included in the analysis to its size when the predictor is left out provides a measure of the explanatory power (r^2_{MA}) of a parametric mixed effects meta-analytical model (Cooper and Hedges 1994):

$$r^2_{MA} = \frac{\sigma^2_\theta(no\ predictor) - \sigma^2_\theta(predictor\ included)}{\sigma^2_\theta(no\ predictor)} \qquad [1]$$

For comparison, we also report the traditional coefficient of determination (r^2) produced by ordinary regression and analysis of variance, though interpretation of this value is ambiguous in a meta-analysis since it describes the relationship among mean differences, but ignores sampling error. Since pseudoreplication (Hurlbert 1984) was unavoidable at several of the study sites, we did not employ the meta-analytical convention of weighting individual studies by their variance; all sites were given equal weight.

Results and Discussion

Similar to findings from previous research, results from our investigations unanimously indicate that fuel treatments reduced wildfire severity in treated areas. Crown volume scorch averaged 38% in treated areas across the eight study sites, versus 84.5% in untreated areas. Nonetheless, treatment effects among the study sites were variable in their significance. Meta-analysis suggests that much of the variability in the size of treatment effects can be explained by site characteristics, particularly the differences in mean tree diameter between treated and untreated areas (table 3). Mean tree diameter in treated areas was 33.0 cm compared to 23.8 cm in untreated areas. Treatments that increase the average diameter of residual trees through removal of the smallest

Table 3—Variation in fuel treatment effect sizes explained by various study site characteristics.

Explanatory variable	P-value	$R^{2\,a}$	R^2_{MA} [b]
Treatment type	0.45	0.18	0
Treatment age	0.50	0.13	0
Density difference	0.17	0.24	0.20
Diameter difference	<0.001	0.71	1.0
Historic fire frequency	0.08	0.34	0.41

[a] R^2 indicates the amount of variation in mean effect sizes explained by the explanatory variable, but ignores sampling error.
[b] R^2_{MA} indicates the amount of reduction in the random variance component (i.e., variation not explained by sampling error) after inclusion of the explanatory variable.

stems appear most effective. This result illustrates the importance of distinguishing fuel treatments from those silvicultural activities that "thin from above" through removal of the largest trees from a stand (Graham et al. 1999).

Vegetation in untreated areas was denser, on average, than in treated areas: 931 versus 319 trees/ha. But tree density differences between treated and untreated areas were insignificant as a predictor of fire severity differences among our study sites. This could be an artifact of the sampling method that we employed, since variable radius plots may provide inaccurate density estimates for small trees (Stage and Rennie 1994). However, they are more efficient than fixed area plots for sampling the larger trees that are more informative recorders of fire intensity. Nonetheless, the relative insignificance of tree density in our analysis suggests that treatment prescriptions based only on density (or basal area) without diameter specifications may be insufficient from a fuels management perspective. Further efforts to increase small diameter wood utilization are needed.

Though our study sites were limited to ecosystems where historic fires were probably fairly frequent (table 2), our synthesis suggests that historic fire regimes may be an important consideration in fuel treatment applications. Among our study sites, fuel treatments were most effective in those ecosystems where fires were historically most frequent. This result might be expected, since these are the ecosystems where fuel hazard has likely increased the most in the 20th Century (Martinson and Omi 2002). Fuel treatment efficacy in ecosystems where fires were historically less frequent than at our study sites is questionable and remains to be investigated.

The insignificance of treatment type and age as predictors of effectiveness is surprising but primarily indicates a need for additional studies. Particularly scarce is information for treatments more than 5 years old. Currently, variability is too great to distinguish the relative effectiveness of treating surface fuels (e.g., broadcast burning) or canopy fuels (e.g., mechanical thinning) versus combining treatments, but results from individual sites suggest that the safest bet is to treat fuel profiles in their entirety.

For example, little difference in crown fuel conditions was found between treated and untreated areas of the Hi Meadow fire, despite a significant treatment effect on fire severity (Omi and Martinson 2002). Though we were unable to assess pre-fire surface fuel conditions, presumably the treatments sufficiently modified surface fuels to reduce wildfire intensity and effects.

In contrast, thinning treatments in the Cerro Grande fire were equally effective in reducing wildfire severity regardless of whether or not the slash was disposed. We speculate that under the extremely windy conditions during this fire, surface fuels may have had less influence on fire behavior than canopy fuels. Explicit inclusion of weather variables as predictors of fuel treatment effectiveness will be explored in future analyses.

Conclusions

The 20th Century has demonstrated the futility of attempts to eliminate fire from natural landscapes. Society must learn to live with fire and the détente will be realized most appropriately through the medium of fuel treatments. Fuel treatments provide options for landscape management that balance societal preferences with the unavoidable recurrence of wildland fires.

Where fire threatens societal values, fuel treatments can facilitate suppression by providing safe access and egress for firefighters, as well as possible counter-firing opportunities. In wildlands managed to include natural processes, fuel treatments may help restore fire to its historic regime, either by restoring fuel profiles that facilitate safe management ignitions or by buffering the border between values-at-risk and extensively managed areas where natural ignitions are allowed to play themselves out. Results from this synthesis suggest that historic fire regimes are an important consideration in fuel treatment placement and treatments may be most effective when they complement the objectives of ecological restoration.

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References

Agee, J.K. 1996. The influence of forest structure on fire behavior. Pages 52-68 in Sherlock, J. (chair). Proceedings of the 17th annual forest and vegetation management conference. The Conference, Redding, CA.

Avery, T.E. and H.E. Burkhart. 1994. Forest measurements, 4th edition. McGraw-Hill Book Co., New York. 408 p.

Cooper, H. and L.V. Hedges (editors). 1994. The handbook of research synthesis. Russell Sage Foundation, New York, NY. 573 p.

Cumming, J.A. 1964. Effectiveness of prescribed burning in reducing wildfire damage during periods of abnormally high fire danger. Journal of Forestry 62:535-537.

Fischer, W.C., M. Miller, C.M. Johnston, J.K. Smith, D.G. Simmerman, and J.K. Brown. Fire effects information system: user's guide. Gen. Tech. Rep. INT-327. U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 131 p.

Graham, R.T., A.E. Harvey, T.B. Jain, and J.R Tonn. 1999. The effects of thinning and similar stand treatments on fire behavior in Western forests. Gen. Tech. Rep. PNW-463. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 27 p.

Hall, W.L., R.H. Wakimoto, and H.R. Zuuring. 1999. Using fuel treatment and site characteristics to model stand replacement fire in regeneration stands following the 1994 wildfires on the Kootenai National Forest. Pages 151-161 in Proceedings of the Joint Fire Science conference and workshop on crossing the millennium: integrating spatial technologies and ecological principals for a new age in fire management, Volume II. University of Idaho, Moscow.

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

Martinson, E.J. and P.N. Omi. In Press. Historic fire regimes of North America: a geographic model based on quantitative research synthesis. *In* N. Sugihara and L. Brennan (editors). Fire conference 2000: the first national congress on fire ecology, prevention and management. University of California, Davis.

Martinson, E.J. and P.N. Omi. In Press. Relating historic fire regimes to 20th Century fire potential may augment ecological justifications for expanded fuel treatment programs. *In* R.T. Engstrom and W.J. de Groot (editors). Proceedings of the 22nd Tall Timbers fire ecology conference: fire in temperate, boreal, and montane ecosystems. Tall Timbers Research Station, Tallahassee, FL.

Moore, E.B., G.E. Smith, and S. Little. 1955. Wildfire damage reduced on prescribe-burned areas in New Jersey. Journal of Forestry 53:339-341.

Morrison, P.H., K.J. Harma, J.W. Karl, L. Swope, R.K. Allen, I. Standen, A. Workowski. 2001. Initial assessment: year 2001 wildfire situation in the Western U.S. Pacific Biodiversity Institute, Winthrop, WA. Available online: www.pacificbio.org/wildfire2001.pdf [July 24, 2002].

Omi, P.N. and K.D. Kalabokidis. 1991. Fire damage on extensively versus intensively managed forest stands within the North Fork Fire, 1988. Northwest Science 65: 149-157.

Omi, P.N. and E.J. Martinson. 2002. Effect of fuels treatment on wildfire severity. Final Report. Submitted to the Joint Fire Science Program Governing Board. Available online: www.cnr.colostate.edu/~fuel/westfire/FinalReport.pdf [July 24, 2002].

Oucalt, K.W. and D.D. Wade. 1999. The value of fuel management in reducing wildfire damage. Pages 271-274 in Proceedings of the Joint Fire Science conference and workshop on crossing the millennium: integrating spatial technologies and ecological principals for a new age in fire management, Volume II. University of Idaho, Moscow.

Pollet, J. and P.N. Omi. 2002. Effect of thinning and prescribed burning on wildfire severity in ponderosa pine forests. International Journal of Wildland Fire 11:1-10.

Rosenberg, M.S., D.C. Adams, and J. Gurevitch. 2000. MetaWin 2.0: statistical software for meta-analysis. Sinaur Associates, Sunderland, MA. 128 p.

Stage, A.R. and J.C. Rennie. 1994. Fixed-radius plots or various-radius plots? Journal of Forestry 92(12):20-24.

Van Wagner, C.E. 1977. Conditions for the start and spread of crown fire. Canadian Journal of Forest Research 7:23-34.

Van Wagner, R. 1968. Survival of coniferous plantations following fires in Los Angeles County. Journal of Forestry 66:622-625.

Vihanek, R.E. and R.D. Ottmar. 1993. When logged units burn in a wildfire, does slash treatment mitigate effects? Pages 709-714 in Proceedings of the 12th conference on fire and forest meteorology. Society of American Foresters, Bethesda, MD.

Wagle, R.F. and T.W. Eakle. 1979. A controlled burn reduces the impact of a subsequent wildfire in a ponderosa pine vegetation type. Forest Science 25:123-129.

Weatherspoon, C.P. and C.N. Skinner. 1995. An assessment of factors associated with damage to tree crowns from the 1987 wildfires in northern California. Forest Science 41:430-451.

Weaver, H. 1943. Fire as an ecological and silvicultural factor in the ponderosa pine region of the Pacific Slope. Journal of Forestry 41:7-14.

Prescribed Burning and Wildfire Risk in the 1998 Fire Season in Florida

John M. Pye[1], Jeffrey P. Prestemon[1], David T. Butry[1], and Karen Lee Abt[1]

Abstract—*Measures of understory burning activity in and around FIA plots in northeastern Florida were not significantly associated with reduced burning probability in the extreme fire season of 1998. In this unusual year, burn probability was greatest on ordinarily wetter sites, especially baldcypress stands, and positively associated with understory vegetation. Moderate amounts of lightning also were associated with greater burning probability. Factors associated with reduced burn probability included road density and nearby requests for site preparation or seed tree burns, perhaps a proxy for other intensive forest management practices. Alternative tactics may prove more effective than fuel reduction in extreme years.*

Introduction

While La Niña has previously been associated with dry and fire-prone conditions in Florida (Brenner 1991, Brenner and Barnett 1992), the extremely rapid transition from the Super El Niño of 1997-1998 to La Niña in spring of 1998 brought a transition from heavy rains to extremely dry conditions. Dry conditions were especially severe in the St. Johns River Water Management District (SJRWMD) of northeastern Florida. Based on wildfire records from Florida's Division of Forestry (Jim Brenner, Florida Division of Forestry, personal communication), wildfires typically burn about 0.7% of the northeastern Florida landscape per year, but in 1998 they consumed as much area as in the previous 12 years combined (figure 1).

Those records also highlight the unusual importance of lightning as an ignition source during this period, accounting for 89% of acres burned. In 15 of the past 21 years, the incendiary/arson category has accounted for the largest share of ignitions in this populous state. While this combination of lightning and drought was unusual, it would be foolish to count on it never recurring, sparking debate over whether policies that promote increased prescribed burning would be a prudent means to reduce damages should such severe fire conditions recur. The study reported here seeks to inform that debate by testing whether past prescribed burning as implemented in the years prior to 1998 significantly reduced the area burned during that severe six week fire season in northeastern Florida, when taking into account vegetation type, vertical structure, and fragmentation, plus variables related to ignition sources and accessibility.

To empirically test this hypothesis, we develop in this paper a model of the probability of wildfire as a function of both on-site and neighborhood conditions. The resulting statistical tests should help identify strategies and tactics to prevent or minimize damages from fires in future extreme drought conditions in northeastern Florida.

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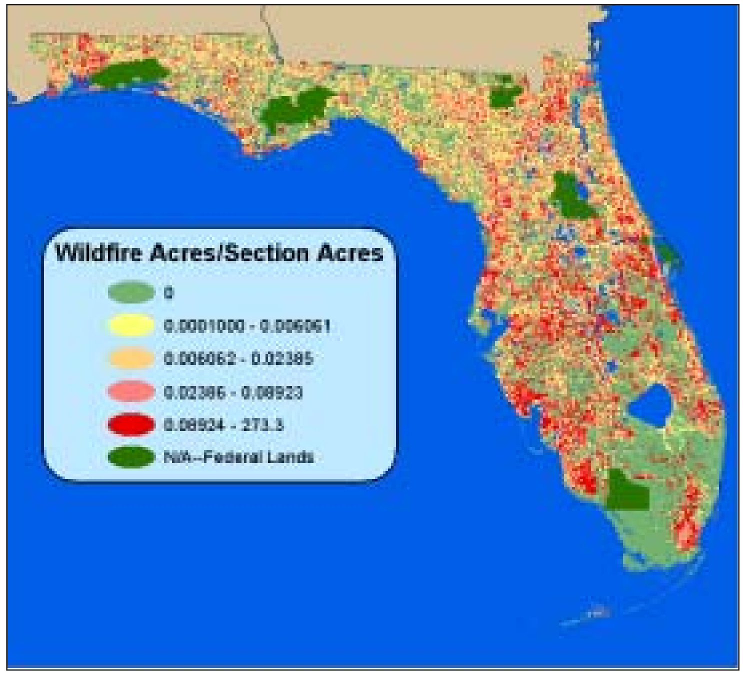

Figure 1—Aggregate historical wildfire risks in Florida 1981–2001.

We assume that stand wildfire risk is related to both on-site and neighborhood vegetation and management conditions, weather / climate, and human factors. Influential site conditions include fuel types and strata, soil moisture content, stand management including prescribed burning, as well as previous wildfires in the stand (which may be considered as a proxy for current aggregate fuel loads). Neighborhood lands can affect wildfire risk through conditions on those lands and by contagion (Chou 1993). Weather affects site wildfire risk through precipitation, evaporation, and wind and by providing a direct ignition source (lightning). Humans affect wildfire risks by (1) development patterns that alter vegetation characteristics and contiguity, thereby affecting wildfire spread and sparking; (2) providing ignition sources, including arson and accidents; (3) suppressing fires once they have begun; and (4) managing fuels and lowering spread rates through vegetation management and building fire breaks.

The SJRWMD includes large areas of actively managed forests, often involving the use of prescribed understory fire to control vegetation and reduce fire hazard, as well as intentional burns to prepare harvested sites for planting or seeding. Recent research suggests that the spatial pattern of these previous burns or treatments may be an important factor in the spread of wildfire (Agee and others 2000, Finney 2001) but these conclusions are based on simulations of fire and management. McKelvey and Busse (1996) had good success stratifying areas at risk based on elevation, slope, and aspect in California's Sierra Nevadas. Nonetheless, they found that some areas reburned more often than expected by chance, notably in areas adjacent to major roadways. As with another Western U.S. analysis (Hyderdahl, Brubaker and Agee 2001), they found statistical relationships between site characteristics and the probability of burning but included little vegetation and no management information in their estimates, and their most important variables of elevation, slope, and aspect are of little relevance in the flat coastal plain of Florida.

In Mississippi, geographically more similar to Florida, Munn, Zhai, and Evans (2003) found that slope was not an important predictor of wildfire.

However, they found that wildfire occurred more often in pine and oak-pine stands than hardwoods, and that wildfire was positively associated with proximity to development. Others have also found human presence to be positively associated with wildfire, increasing the number of ignitions and the number of large fires (Cardille and Ventura 2001; Cardille, Ventura, and Turner 2001). However, Sapsis and others (1996) found that human presence decreased the risk of large fires.

Two studies that may be of particular importance in evaluating fire risk in Florida address riparian areas (Fites-Kaufman 1997) and fuel connectivity (Miller and Urban 2000), although both of these studies evaluated forest fires in the Western U.S. Fites-Kaufman found that riparian areas had an average fire return interval of greater than 20 years, with irregular intervals between fires. Miller and Urban, examining Sierra Nevada forests, found that connectivity in fuels led to increased spread potential. They note, however, that connectivity is likely a minor influence when temperature and fuels are conducive to large fire development.

Other research has focused on identifying the influence of weather and climate, including drought, precipitation, temperature, humidity, and wind. One study (Heyerdahl, Brubaker, and Agee 2001) found that temporal, rather than spatial, climatic variation was the driving force in fires in Oregon. Wind speed was not found significant in predicting large fire development (Potter 1996), though high temperatures and low humidity did contribute to large fires. McKelvey and Busse (1996) found that all of the extreme fire years in the Sierra Nevada occurred during hot, dry seasons, but that not all hot, dry seasons were extreme fire years. The fit from other weather variables was weak.

Model of Wildfire Risk

In light of the previous work on wildfire risk, we specified our model of wildfire risk in this catastrophic season for stand *i* in a population of I stands in year *t*, $R_{i,t}$, as:

$$R_{i,t} = f(S_{i,t}, N_{i,t}, L_{i,t}, H_{i,t}) \qquad [1]$$

where $S_{i,t}$ and $N_{i,t}$ are on-site and neighborhood risk factors, respectively. The $L_{i,t}$ are influences of lightning, and the $H_{i,t}$ are human factors affecting risk in that period. In a particular year, the realization of the risk for stand i is either 0 or 1, so that the occurrence of a wildfire in the stand, $W_{i,t}$, is a binary variable, whose value is influenced by functional (F) relationships between wildfire and influential factors ($x_{i,t}$). Given data on these factors, an empirical representation of this model can be estimated as a binary logit (Greene 1990):

$$P(W_{i,t} = 1 \mid x_{i,t}) = \frac{\exp(x_{i,t}'\beta)}{1 + \exp(x_{i,t}'\beta)} = \Lambda(x_{i,t}'\beta) \qquad [2]$$

$$W_{i,t} = \begin{cases} 1 & \text{if burned in year } t \\ 0 & \text{otherwise} \end{cases} \qquad [3]$$

The equation was estimated using quasi-maximum likelihood covariances and standard errors, robust to varying underlying distributions of the dependent variable. Calculations were performed using EViews (Quantitative Microsoftware 1997).

Data

The unit of analysis used in this study was individual Forest Inventory and Analysis (FIA) forested plots in the SJRWMD (figure 2). Plot locations and stand conditions ($S_{i,t}$) were obtained from the FIA records (USDA Forest Service Southern Research Station, Knoxville, TN).

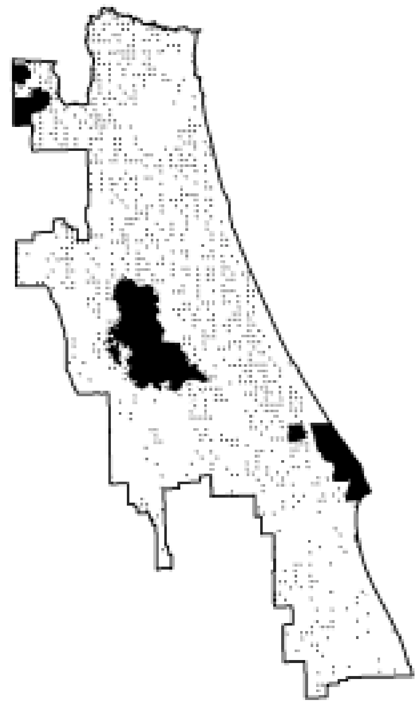

Figure 2—Forested FIA plot locations in the Saint Johns River Water Management District, Florida. Black areas indicate federal lands.

Stand/On-Site Factors

The FIA plot locations in Florida were visited by FIA crews in 1985-86 and reported in the 1987 FIA survey, and visited again in late 1993-1994 and reported in the 1995 FIA survey. From the FIA data, observations of plot conditions for 1993-1994 and observations on activities occurring on the plot between the 1987 and 1995 surveys were used in the model.

FIA field crews reported evidence of wildfire on the plot since the previous survey. They also report evidence of prescribed burning, defined as “the occurrence of fire (excluding wildfire) not used as a site preparation tool.” For our analysis, both the wildfire and prescribed burn variables were coded as 1 if reported to have occurred and zero if not. The FIA surveys also reported a measure of forest-nonforest edge as observed at the perimeter of a 20.2 ha (50 acre) circle. This variable ranged from 0 to 9, with 0 indicating no forest edge and 9 indexing considerable forest edge.

Stands were classified by forest type as (1) cypress [*Taxodium distichum* (L.) Rich.], (2) pine [*Pinus* species], (3) oak-pine, and (4) hardwood types.

FIA reports five measures of vegetation strata: counts of the number of trees in three diameter classes per 0.4 ha of forest in the stand (2.5-5 cm, 5-12.7 cm, 12.8 cm dbh and larger), plus measures of the percentage of space occupied by non-tree vegetation at 0-0.90 m and 0.9-2.44 m above the forest floor. These measures are intercorrelated and thus suitable for recoding into a smaller number of independent variables, which we accomplished using principal components analysis. This procedure produced two orthogonal measures that together explained the majority of variation in the five FIA variables as measured on the different plots. The first measure, referred to as ladder fuel index 1, most strongly reflected the two non-tree vegetation variables. The second measure, ladder fuel index 2, reflected variations in the numbers of small and medium trees. In each measure, higher numbers indicate more vegetation on site.

We identified burn status for each plot (W_s) by overlaying a GIS coverage of approximate FIA plot locations (figure 2) with a coverage of polygons representing areas burned in the SJRWMD between June 3 and July 7, 1998 (figure 3) (Barbra Sapp, St. Johns River Water Management District, personal communication).

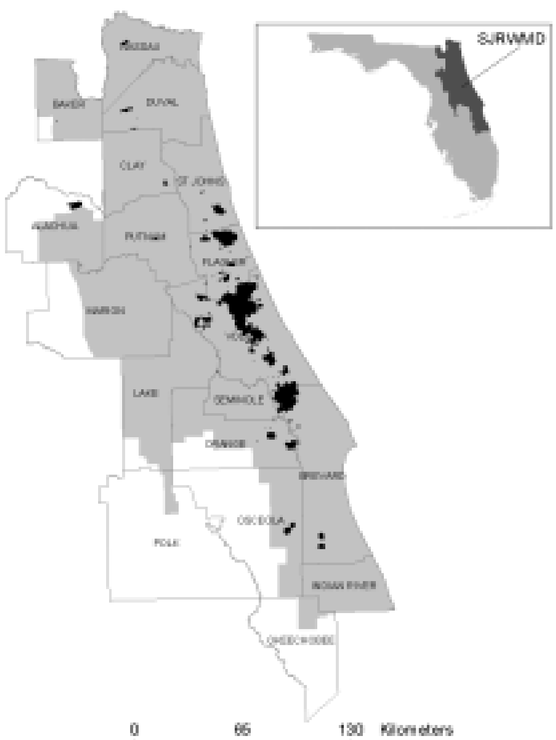

Figure 3—Wildfires in the 1998 wildfire season in the St. Johns River Water Management District.

Neighborhood Factors

To ensure the integrity of the survey process, the locations of plots provided by FIA have been limited to hundredths of a degree. In this region, this corresponds to an accuracy of 1.5 km north-to-south and 1.3 kilometers east-to-west. Unless otherwise noted, this location uncertainty defines the neighborhood size for the following neighborhood variables in this analysis.

Information on wildfire and prescribed burning history was obtained from the Florida Division of Forestry's individual wildfire records, running from 1986 to 1997 in our analysis, and permits for silvicultural burns, which stretched from 1996 to 1998 in our analysis. We chose to focus on the most common types of silvicultural burns: hazard reduction, which we equate with understory burns, and an aggregate of the site preparation and seed preparation burn categories ("regeneration burns"). We omitted wildlife and ecological burns because of their limited use and rangeland burns because they could not be distinguished from burns on croplands. In this paper we refer to both FIA's prescribed burning and Florida's hazard reduction burns as understory burns to avoid confusion with prescribed regeneration burns.

The location of wildfire ignitions and permits for both forms of prescribed burns are reported by Public Land Survey (PLS) section. The wildfire data and permit data were joined to a coverage of PLS sections (David Kelly, Florida Division of Forestry, personal communication) for neighborhood analyses. Wildfire, understory burn, and regeneration burn areas were all expressed relative to forest area in the neighborhood of the FIA plot. These measures were the ratio of the sum of the area of the wildfire or permits issued to the area of forest in a rectangle 1.3 km (east-west) by 1.5 km (north-south) centered around the nominal FIA plot location.

For wildfire, two temporal aggregates were generated: 4 to 12 years (1986-1994), and 1 to 3 years (1995-1997) previous to 1998. These temporal aggregates roughly correspond with the FIA survey cycle and the period between the end of that cycle and the study year. Because the plots with neighborhood regeneration burns in 1997 and 1998 experienced no burning in 1998, only the regeneration burning for 1996 was used in the model.

Two measures of forest surrounding the FIA plots were generated based on Multiple Resource Land Cover data (Riitters 1997). These report the total area of forestland, and the proportion of that forest classified as "woody wetlands" as opposed to "upland forest." Small amounts of forest in the neighborhood provide one indication of fuel fragmentation, along with the FIA measure of forest-nonforest edge.

For information on lightning we used a dataset purchased from WeatherBank, Inc. (Edmond, OK). Originally collected through the National Lightning Detection Network, the dataset contains records of all individual cloud-to-ground strikes covering northern Florida between June 3 and July 7, the most intense period of wildfire activity. Each record reports the location of strike. Converting these into a GIS coverage (figure 4) enabled us to calculate the

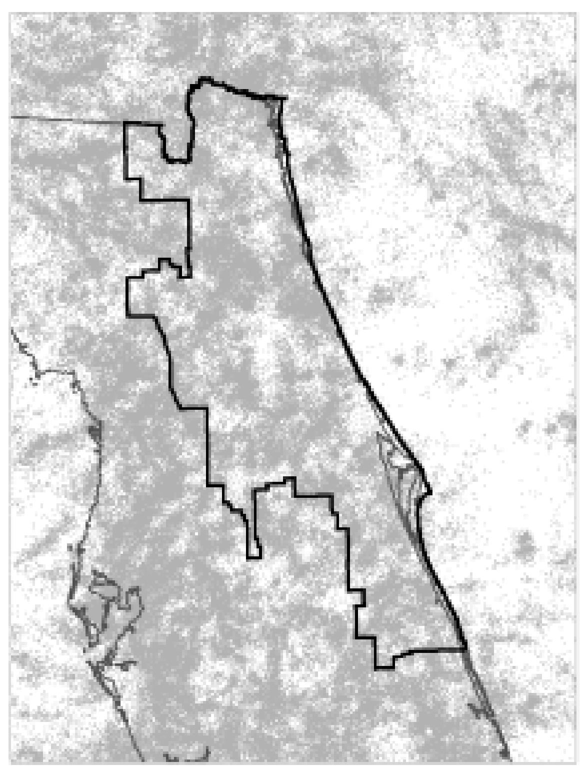

Figure 4—Lightning groundstrikes in northeastern Florida during the subject period, June 3–July 7, 1998.

number of lightning strikes that occurred within 0.833 km of each nominal FIA plot location (1 / 2 mile).

Road density information was derived from a vector coverage of paved roads (ESRI, Inc., Redmond, CA). The coverage was rasterized to 100 m pixels and the density of pixels containing roads was calculated in the vicinity of each FIA plot using the 1.5 x 1.3 km neighborhood. Population density was calculated for a similar neighborhood based on census block population numbers from the 1990 Census, normalized by census land area.

Observations Included or Excluded

Of the 2,948 FIA plots in the SJRWMD, 46%, or 1,346 were classified as timberland. Of these, 81 burned, and 1,255 did not. Logistic analysis requires that factors perfectly correlated with the left-hand side variable be omitted from the analysis. Three factors in this model showed perfect correlations. None of the plots with the following characteristics were judged to have burned in 1998: (1) all plots with wildfire recorded between surveys (23 plots), (2) all plots with neighborhood regeneration burning in 1997 and 1998 (210 plots), and (3) all plots classed as xeric (280 plots). These plots were thus excluded from the logistic regressions. Also excluded were plots with missing data. Florida State wildfire records do not consistently report wildfires on federal lands, thus wildfire history data was unavailable for the 402 plots either in or near federal lands. Also excluded were 13 plots that lacked non-tree vegetation data. Some of the plots are in more than one of the above classes, resulting in a total of 555 usable observations, 59 of which were burned in 1998.

Table 1 shows the mean values for the various independent variables broken out by forest type. It shows, for example, that the 52 baldcypress stands in the sample predominantly occurred on hydric sites, had less non-tree vegetation than other types on average (ladder fuel 1) but more small trees (ladder fuel 2), were surrounded by more wetland forest, experienced virtually no prescribed burning on site, and had few roads and human residents nearby.

Table 1—Mean values and number of observations for FIA plots used in the logistic model, by forest type.

		Forest type			
Variable	**Units**	**Pine**	**Oakpine**	**Baldcypress**	**Hardwood**
Hydric site	proportion on hydric sites	0.0	0.4	0.9	0.4
Ladder fuel 1	index of ladder fuel 1	-9	-45	-95	-25
Ladder fuel 2	index of ladder fuel 2	203	341	429	248
Upland forest	acres/(1.5x1.3 km)	107	89	84	79
Wetland forest	acres/(1.5x1.3 km)	47	52	70	60
Rx burn	1 if prescribe burn, 0 if not	0.1721	0.0217	0.0000	0.0000
Forest edge	1 if little forest edge, 9 if a great deal	3.7	3.8	3.5	3.3
Under.burn96	proportion of forest area	0.0077	0.0164	0.0065	0.0065
Under.burn97	proportion of forest area	0.0001	0.0000	0.0001	0.0001
Under.burn98	proportion of forest area	0.0063	0.0072	0.0211	0.0044
Regen.burn96	proportion of forest area	0.0012	0.0001	0.0019	0.0056
Road density	proportion of pixels containing a road	0.152	0.159	0.087	0.147
Pop. density	persons per acre	0.089	0.120	0.051	0.161
Prev.wildfire1	proportion of forest area	0.0028	0.0013	0.0071	0.0029
Prev.wildfire2	proportion of forest area	0.0047	0.0061	0.0034	0.0031
Lightning	ground strikes/0.25 mi^2	3.23	2.87	2.02	2.74
Count	number of observations	308	46	52	149

Results

The results of the logistic model estimation of the probability of wildfire are shown in table 2. Overall performance of the models was good, with the chi-squared value significant for the model with variables as compared to the model with only a constant. McFadden's R-squared is 0.20, although interpretation of values between 0 and 1 are difficult with this measure (Greene 1993).

In this multivariate model, which regresses the occurrence of wildfire in 1998 on site and neighborhood variables, stand forest type is mildly predictive of burn probability, with baldcypress stands more at risk than other forest types. Baldcypress stands are typically associated with hydric drainage conditions, and at the other extreme from the perfectly and negatively correlated xeric condition.

Pine stands were no more likely to burn than the intercept hardwoods in this unusually severe drought. This may be due in part to the influence of the two ladder fuel measures, which were highly significant and positively correlated with burning in 1998.

While non-tree and small tree vegetation were positive correlates with burning, and previous evidence of wildfire in 1994 was a perfect negative correlate, evidence of understory burning on the site did not exert a negative influence on burning in 1998.

As with the on-site measure of understory burning, none of the three neighborhood measures of understory burning permits had any significant negative influence on wildfire probability in 1998. The only significant burn permit

Table 2—Logit model estimates of wildfire occurrence as a function of site, neighborhood, lightning, and human variables (St. Johns River Water Management District, 1998, Forest Inventory and Analysis plots). [a]

Variable	Coefficient	Standard error	P value
Intercept	-5.03	1.10	<0.0001
Stand/on-site			
Pine forest	0.12	0.49	0.8109
Oak-pine forest	-0.33	0.74	0.6557
Cypress forest	1.33	0.57	0.0213
Hydric site	0.48	0.48	0.3246
Rx Burn (stand)	0.69	0.44	0.1168
Ladder fuel-1	0.009	0.003	0.0048
Ladder fuel-2	0.002	0.001	0.0084
Forest edge	0.11	0.10	0.2736
Neighborhood			
Total timberland	0.02	0.01	0.0001
Proportion wetland forest	-3.12	0.66	<0.0001
Understory burn-96	0.86	2.83	0.7624
Understory burn-97	-327.14	785.36	0.6770
Understory burn-98	1.95	3.12	0.5321
Regeneration burn-96	-124.49	52.97	0.0188
Previous wildfire (1995-1997)	-2.34	7.36	0.7511
Previous wildfire (1986-1994)	19.24	9.69	0.0472
Lightning strikes (1998)	0.16	0.13	0.2251
Ltng. strikes*ltng strikes	-0.02	0.01	0.0487
Population density	-0.41	2.68	0.8774
Road density	-6.09	1.70	0.0003

[a] McFadden's R-squared: 0.20. Log likelihood: -149.59. Model significance level: <.0001.

measure was regeneration burning in 1996, which had a strongly significant negative effect on the probability of fire in 1998. This is consistent with the perfect correlation and exclusion of the plots with regeneration burning in the neighborhood in 1997 and 1998. Regeneration burning appears to have a negative effect on fire risk. While it is plausible that harvesting a stand and then burning the remaining slash and vegetation would at least temporarily reduce fuels and wildfire risk, it is also likely that some of the influence of this measure arises from other management practices associated with intensive forestry. These may include stocking control, herbicide use, and fire breaks, none of which are directly reflected in this model.

The model includes the neighborhood measures of historical wildfire for 1995-1997 and 1986-1994. The existence of wildfires in the last 3 years is not significantly related to the probability of a plot burning, but wildfires in the previous 9 years are significant and positive. This implies that areas that had experienced wildfires more than 3 years ago were again at higher risk of fire in 1998. The exclusion of all plots with recorded on-site wildfires does not allow direct statistical comparison, but the fact that none of these sites burned in 1998 is at least suggestive of a local and countervailing negative effect of previous wildfires.

The fragmentation measure of forest-nonforest edge was not significant, but the amount of forest surrounding an FIA plot was significant, with more forest associated with increased probability of burning in 1998. However, upland forests increased burn probability more than wetland forests, as reflected in the significant and negative effect of the proportion of wetlands. This is in seeming contrast to the elevated burn probability for baldcypress. One possibility is that the baldcypress stands most likely to burn are those at the drier, more upland margins.

Of the two measures of human influence—population density and road density—only road density is significant, and it shows a negative influence on burn probability. As mentioned in the Introduction, the literature on the influence of human presence is inconsistent, but the differences in results may be related to the dominant ignition sources in the dataset being examined. In most years in northeastern Florida, accidents and arson—sources logically associated with roads and people—dominate natural ignition sources. In the 1998 wildfire season, human-caused ignitions played a minor role, allowing influences of roads on detection and suppression to show increased importance. These potential influences include quicker detection, easier access for suppression resources, and greater fragmentation of fuels, each of which could result in less area burned.

Because severe drought conditions spanned the entire SJRWMD in the spring of 1998, we do not attempt to include site-specific weather data. However, because 1998 was highly unusual in the number of lightning caused fires and acres, we included contemporaneous lightning strikes. Studies in the Southwest and Florida (Gosz and others 1995, Shih 1988), have each shown that, in general, lightning is strongly correlated with rainfall, and yet anecdotally we understand that dry lightning was an important ignition source during this period. We attempted to isolate this influence with linear and second order terms. Our results show that lightning has an increasing then decreasing correlation with increased fire risk, with a maximum positive influence reached at approximately eight strikes per square mile. Taken together they suggest that small and intermediate amounts of lightning, perhaps associated with little precipitation, can raise the probability of fire, while high levels of lightning are correlated with suppressive rain events.

Discussion

Taken together, the results indicate that in the extreme 1998 fire season in northeastern Florida, it was forests ordinarily thought of as wet that were most likely to burn—forests on mesic and especially hydric sites, and baldcypress stands in particular. These locations do not match the general pattern of wildfire indicated in the recent FIA survey nor our casual expectations of areas at risk to wildfire. However, baldcypress is not immune to wildfire. At least one source reports that cypress ponds in north Florida typically burn several times a century (Myers and Ewel 1990). It is noteworthy that in this drought, fire risk areas did not merely expand outward from xeric into mesic locations on the landscape. Fire risk simultaneously moved into hydric sites and out of the more typically fire-prone xeric sites, changing rather than augmenting the areas at risk.

Given the above, it perhaps should not be surprising that understory burning was not found to be a significant reducer of risk in this catastrophic year. Xeric sites, even in areas with no previous understory burning in the area, were apparently at low risk during this extreme period. Instead, it was baldcypress stands that were at greatest risk. Based on the FIA data from 1994, baldcypress stands typically have high densities of small trees but little nontree vegetation in the lower strata. However, conditions when FIA crews visited in 1994 may have differed from those during this extreme drought, when areas ordinarily flooded can dry sufficiently to allow understory fuels to first proliferate and then dry out. Baldcypress on the drier margins of wetland forests might be especially prone to such ephemeral conditions, consistent with the negative correlation with percent wetland forest (but see Myers and Ewel 1990 for a contrary fire pattern). Should this be true, understory vegetation on hydric sites would be minimal in more ordinary years, thus precluding use of fuel reduction treatments, whether through fire, chemical, or mechanical alternatives.

While our results do not support the hypothesis that understory burning affects fire risk in extreme drought years, we did not examine whether controlled burns might reduce wildfire intensity or severity. Such activity could reduce damage to the stand, and wildfires in those areas where it is practiced may be safer to control. Understory burns may also reduce wildfire risk in years with more typical rainfall patterns or for shorter periods of time than tested here.

Management Implications

Our motivation for this study was to identify strategies that would mitigate risk during future catastrophic droughts. However, prescribed understory burns apparently do not help, at least as they have previously been conducted. While some potential may exist to increase the protective effects of such burns through better identification of areas of greatest benefit and spatial arrangement, the feasibility of alternative fuel reduction methods in the important baldcypress forests appears discouraging. This suggests we must look at other tactics beside fuel management to mitigate risk on hydric sites during extreme drought conditions. Possible tactics could include constructing and maintaining firebreaks and ensuring defensible spaces around buildings and other areas of value. Suppression capabilities are also important, but given the rare occurrence of these extreme drought conditions, emphasis should be given to maximizing access to suppression resources that are easily mobilized, including unused aircraft and field crews from distant regions.

Acknowledgments

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Literature Cited

Agee, J.K., B. Bahro, M.A. Finney, P.N. Omi, D.B. Sapsis, C.N. Skinner, J.W. van Wagtendonk, and C.P. Weatherspoon. 2000. The use of fuelbreaks in landscape fire management. Forest Ecology and Management 127(1-3):55-66.

Barnett, T. P., and Brenner, J. 1992. Prediction of wildfire activity in the southeastern United States. Southeast Regional Climate Center Research Paper No. 011592. South Carolina Water Resources Commission. Columbia, SC.

Brenner, J. 1991. Southern oscillation anomalies and their relation to Florida wildfires. Fire Management Notes 52(1):28-32.

Cardille, J.A., S.J. Ventura. 2001. Occurrence of wildfire in the northern Great Lakes Region: effects of land cover and land ownership assessed at multiple scales. International Journal of Wildland Fire 10(2):145-154.

Cardille, J.A., S.J. Ventura, and M.G. Turner. 2001. Environmental and social factors influencing wildfires in the Upper Midwest, United States. Ecological Applications 11(1):111-127.

Chou, Y.H., R.A. Minnich, and R.A. Chase. 1993. Mapping probability of fire occurrence in the San Jacinto Mountains, California, USA. Environmental Management 17(1):129-140.

Finney, M.A. 2001. Design of regular landscape fuel treatment patterns for modifying fire growth and behavior. Forest Science 47(2):219-228.

Fites-Kaufman, J. 1997. Historic landscape pattern and process: Fire, vegetation, and environment interactions in the northern Sierra Nevada. Ph.D. dissertation. Seattle, WA: University of Washington.

Gosz, J.R., D.I. Moore, G.A. Shore, H.D. Grover, W. Rison, and C. Rison. 1995. Lightning estimates of precipitation location and quantity on the Sevilleta LTER, New Mexico. Ecological Applications 5(4):1141-1150.

Greene, W. H. 1993. Econometric analysis. MacMillan, New York, NY. 783 p.

Heyerdahl, E.K., L.B. Brubaker, J.K. Agee. 2001. Spatial controls of historical fire regimes: a multiscale example from the Interior West, USA. Ecology 82(3): 660-678.

McKelvey, K.S. and K.K. Busse. 1996. Twentieth-century fire patterns on Forest Service lands. In Sierra Nevada Ecosystem Project: Final Report to Congress, Vol. II, Chap. 44. Davis: University of California, Centers for Water and Wildland Resources.

Miller, C. and D.L. Urban. 2000. Connectivity of forest fuels and surface fire regimes. Landscape Ecology 15(2):145-154.

Munn, I.A., Y. Zhai, and D.L. Evans. 2003. Modeling forest fire probabilities in the South Central United States using FIA data. Southern Journal of Applied Forestry 27(1):11-17.

Myers, Ronald L. and John J. Ewel. 1990. Ecosystems of Florida. Orlando: University of Central Florida Press.

Potter, B.E. 1996. Atmospheric properties associated with large wildfires. International Journal of Wildland Fire 6(2):71-76.

Sapsis, D.B., B. Bahro, J. Spero, J. Gabriel, R. Jones, and G. Greenwood. 1996. An assessment of current risks, fuels, and potential fire behavior in the Sierra Nevada. Sierra Nevada Ecosystem Project. Final Report to Congress, Vol. III. Davis: University of California, Centers for Water and Wildland Resources.

Shih, S.F. 1988. Using lightning for rainfall estimation in Florida. Transactions of the ASAE, American Society of Agricultural Engineers 31(3):750-755.

Riitters, K.H., R.V. O'Neil and K.S. Jones. 1997. Assessing habitat suitability at multiple scales: a landscape-level approach. Biological Conservation 81:191-202.

USDA Forest Service, Forest Inventory and Analysis data. Various years. Southern Research Station, Asheville, NC.

US DOC Bureau of the Census. 1990 Census of Population and Housing, Washington, DC.

Fire Hazard and Silvicultural Systems: 25 Years of Experience From the Sierra Nevada

Scott Stephens[1] and Jason Moghaddas[1]

Silviculture systems influence fire hazard by changing the arrangement and quantities of live and dead biomass. Each system regulates forest growth and development but their long-term influence on fire hazard is largely unknown. We determined if significant differences ($p < 0.05$) exist in fuel loads (1, 10, 100, 1000 hour, duff, litter), surface fuel height, crown cover, and height to live crown base from the single-tree selection, thin from below, overstory removal, group selection, and clear-cut silvicultural systems. Approximately 25 years of data from each of the systems were analyzed from young-growth (approximately 80 years old) mixed conifer forests at the University of California Blodgett Research Station located in the northern Sierra Nevada. These systems were also compared to adjacent 80-year-old unmanaged stands. All areas have experienced a policy of fire suppression the last century. Activity fuels were lopped and scattered with the exception of the clear-cut and group regeneration units that were tractor piled and burned. Significant differences were detected between silvicultural systems with the clear-cut having the lowest average fuel loads. Clear-cut had low surface loads but at 10-15 years of age they produced high horizontal fuel continuity and high hazards. No significant differences were detected between the single tree selection, overstory removal, thin from below, and unmanaged stands, which all averaged approximately 150 metric tons / ha of surface and ground fuels. Activity fuel treatment must be an integral component of silvicultural systems to produce forests with low fire hazards.

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Canopy Fuel Treatment Standards for the Wildland-Urban Interface

Joe H. Scott[1]

Abstract—*This paper describes a canopy fuel treatment standard based on models of crown fire flame size, initiation, spread rate, and firefighter safety. Site-specific prescriptions can be developed with NEXUS or nomograms. A general prescription designed to be effective at 20-ft windspeeds up to 25 mph during drought summer fine dead fuel moisture conditions (1-hr = 4%, 10-hr = 5%, 100-hr = 7%, live = 78%) calls for a crown-fire-free zone (CFFZ) 380 ft wide with a maximum canopy bulk density of 0.10 kg/m*^{3}*. Minimum canopy base height ranges from 2 to 18 ft depending on surface fuel conditions; for fuel model 10 (timber litter and understory), minimum canopy base height is 13 ft.*

Introduction

Houses and other structures can be ignited during a wildland fire by direct flame contact, radiation, or burning embers. The probability of structure ignition can be greatly reduced, but not eliminated, by surface fuel modification immediately adjacent to structures and by adherence to design and construction standards for the structure itself. However, except for an exceptionally well-designed structure, firefighter intervention is needed during the passage of a wildland fire to suppress incipient ignitions. Therefore, when designing fuel treatments for structure protection in the wildland-urban interface we should plan for the presence of firefighters at a structure during fire front passage.

Firefighters need a zone around the structure in which to lay hose, raise ladders to the roof, inspect the home exterior for ignitions, and suppress external structure ignitions. This immediate area around the structure should not allow a spreading surface fire. Surface fuels around this fire-free zone must be treated so that flame lengths allow firefighters to work safely.

Even in full protective wildland clothing, firefighters are more prone to burn injury from flames than a structure is prone to ignition by radiation (Cohen and Butler 1998). In other words, radiation from flames will injure a firefighter or homeowner before untreated wood siding would ignite. Therefore, fuel treatments around structures should be designed to protect firefighters, not structures. Fuels should be treated such that the structure is within a firefighter safety zone. This is the basis for defensible space—the area around a structure where firefighters can safely work (California State Board of Forestry 1996). Many surface fuel treatment standards for creating defensible space exist (for example: International Fire Code Institute 1997, Moore 1981). This paper presents a method for determining the size and characteristics of a coniferous forest canopy treatment.

Firefighter safety zone size, and thus the required area of canopy fuel modification around a structure, is a function of expected flame height. A physical

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heat transfer model suggests this distance must be about 4 times expected flame height, not including a factor of safety (Butler and Cohen 1998). Flame heights are always less than or equal to flame length. To be conservative and simplify, I assume that flame length equals flame height.

Crown fires present special problems for structure protection. They generate huge quantities of burning embers, some of which may travel long distances. Crown fires have very tall flames, which indicate need for a large safety zone and therefore wide fuel modification distance. Modifying canopy fuels to preclude crown fire near a structure is a critical part of ensuring firefighter safety during structure protection.

The guidelines presented here are not intended to be applied by firefighters assigned to structure protection during an active fire—they should be implemented by structure owners before fire threatens.

The Design Environmental Condition

Before prescribing a canopy fuel treatment one must first specify the design environmental condition—the most extreme condition under which the treatment is expected to produce the prescribed result. There are only two factors in the design condition: fine dead fuel moisture content and 20-ft windspeed.

Fine dead fuel moisture content can be set in either of two ways. One method is to identify a threshold condition from a local fire weather database, such as 90th, 95th, or 99th percentile. Firefamily Plus (Bradshaw and McCormick 2000) is a good tool for such analysis. Where such data are not available, the design fine dead fuel moisture content can simply be set to a standard set of moistures, such as those used by Rothermel (1991) in his crown fire spread model (table 1). These values are for the northern Rocky Mountains; other regions may need to use a different set if fuel moistures vary significantly from these.

The design windspeed is more difficult to determine from a fire weather database. Many fire weather databases have a single daily windspeed observation, and this is often a 10-minute average. Windspeeds much higher than reported can occur at the station during other times of the day; the 10-minute average masks significant variability. If using windspeed data, I recommend setting the design windspeed to a value that represents the near maximum (95th or 99th percentile) 1-minute average windspeed that can occur at a site. The design windspeed can alternatively be set to a reasonable value based on expert knowledge of local conditions and the windspeed at which firefighters would discontinue operations. Design windspeeds between 20 and 40 mph are reasonable—I use 25 mph as a default.

Table 1—Rothermel's (1991) fine dead fuel moisture content scenarios. Values are component moisture content (percent).

Timelag fuel component	Early spring before greenup	Late spring after greenup	Normal summer	Drought summer	Late summer severe drought
1-h	8	9	6	4	3
10-h	14	11	8	5	4
100-h	18	15	10	7	6
Live	65	195	117	78	70

Regardless of how the design condition is selected, it is important that this condition be communicated clearly to all responsible parties. It is the limit of effectiveness of the treatment—analogous to the load limit on a bridge.

Size of the Crown-Fire-Free Zone

There are three elements of the canopy fuel treatment standard: required size of the treated area, and within it, maximum allowable canopy bulk density, and minimum allowable canopy base height.

The first element of the canopy fuel treatment standard is the size of the treatment area, called the crown-fire-free-zone (CFFZ). The CFFZ size is a function of the expected flame size. For a site-specific treatment design, determine potential crown fire flame length for the design condition using Rothermel's (1991) nomograms or by using NEXUS (Scott 1999; www.fire.org / nexus / nexus.html), then multiply this estimate by 4 to get the minimum radius of the CFFZ. For designing a non-specific treatment, use the nomogram in figure 1, which uses drought summer fuel moistures (table 1) and requires an estimate of the total available surface and canopy heat per unit area (HPA) in the area surrounding the CFFZ. Surface and canopy fuel HPA have been combined into four categories encompassing the range likely to be encountered (table 2).

At the default design windspeed of 25 mph, drought summer fine dead fuel moisture (table 1), and high surface and canopy fuel HPA (table 2), the CFFZ must be a minimum of 380 ft radius in all directions from the structure (figure 1).

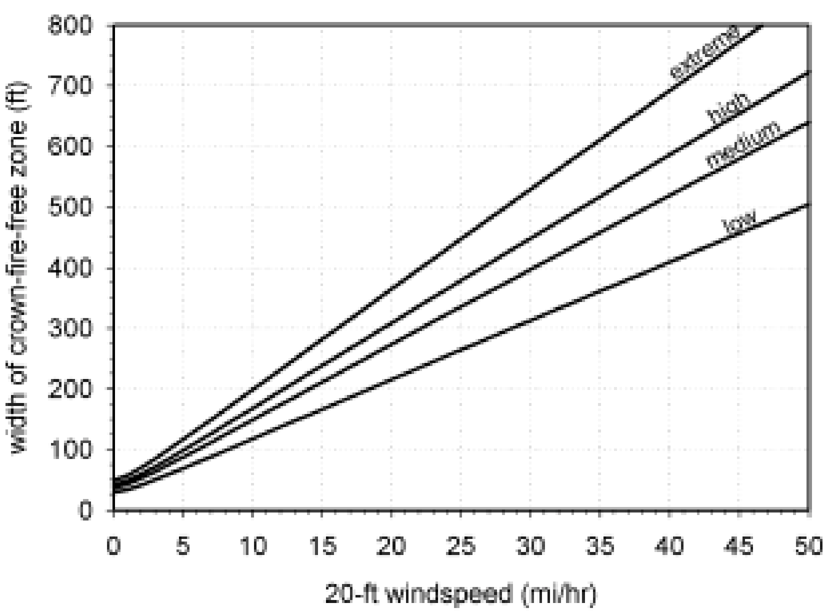

Figure 1—Minimum width of the crown-fire-free zone around a structure as a function of 20-ft windspeed for 4 categories of combined surface and canopy fuel heat per unit area (low = 3500 BTU/ft^2, medium = 5000 BTU/ft^2, high = 6000 BTU/ft^2, extreme = 7000 BTU/ft^2). This chart uses Rothermel's (1991) crown fire spread model to estimate flame length, and assumes level ground, drought summer fine dead fuel moisture conditions (1-hr = 4%, 10-hr = 5%, 100-hr = 7%, live = 78%). Width of the crown-fire-free zone is 4 times the estimated flame length.

Table 2—Stylized surface and canopy fuel loadings and heat per unit area, for estimating crown fire flame length in Rothermel's (1991) correlation. *Drought summer* fuel moisture is assumed (see table 1) for estimating heat per unit area of surface fuels. Available canopy fuel load can be estimated using tables in Rothermel (1991). Fire behavior fuel models (FBFM) are described in Anderson (1982).

Fuel complex heat per unit area class	Surface fuel model	Additional coarse fuels (tons/ac)	Available canopy fuel load (tons/ac)	Heat per unit area (BTU/ft^2)
Low	FBFM 8			
	Compact timber litter		8	3500
Medium	FBFM 10			
	Timber litter and understory		10	5000
High	FBFM 10			
	Timber litter and understory	30	12	6000
Extreme	FBFM 12			
	Medium logging slash		12	7700

Canopy Fuel Characteristics in the CFFZ

Two canopy fuel characteristics must be controlled in the CFFZ: canopy bulk density (CBD) and canopy base height (CBH). Canopy bulk density, the mass of available canopy fuel per unit of canopy volume, must be low enough to cause an active crown fire to cease when entering the CFFZ. Canopy base height must be high enough to preclude initiation of passive crown fire within the CFFZ.

Maximum Allowable Canopy Bulk Density

Fuels within the CFFZ must be managed so that an active crown fire spreading to the zone would cease. This is accomplished by reducing canopy bulk density (CBD) below a threshold value (figure 2). The threshold CBD is determined by linking separate models of crown fire spread rate (Rothermel 1991) and critical conditions for active crown fire spread (Van Wagner 1977).

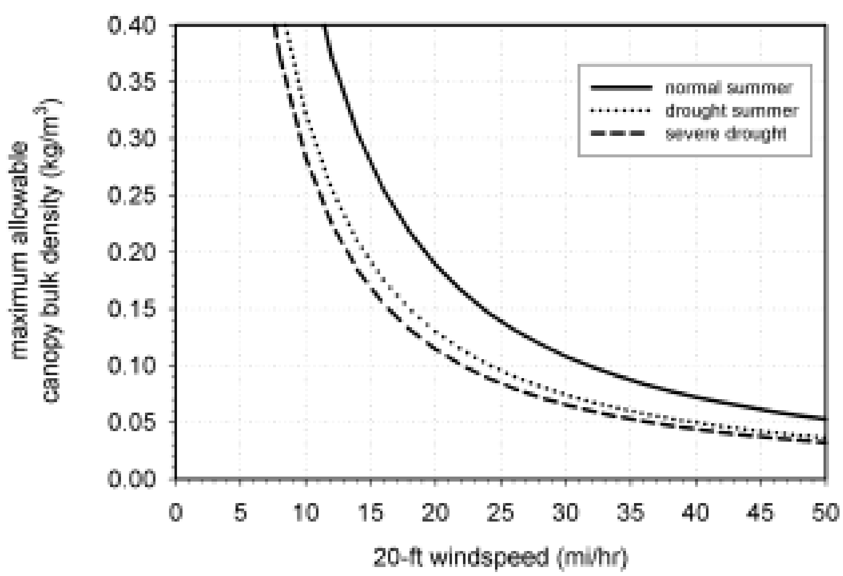

Figure 2—Maximum allowable canopy bulk density as a function of 20-ft windspeed, for a range of fine dead fuel moisture content scenarios. Level ground is assumed.

It is largely a function of the design windspeed (Scott and Reinhardt 2001). At the design condition of drought summer fuel moisture and 25 mph open windspeeds, the maximum allowable CBD is approximately 0.10 kg / m^3.

Unfortunately, CBD is difficult to estimate and prescribe in a treatment. CBD can be determined from indirect measures such as leaf area index, stand biometrics, or from optical sensors (Scott and Reinhardt 2001). Currently, the best method of estimating CBD is to use modified allometric equations that relate tree species and size to crown biomass. The Fuels Management Analyst Plus suite (www.fireps.com) contains the program Crown Mass, the only tool available to help managers estimate CBD from allometric equations. For a site-specific treatment specification that includes fine dead fuel moistures not included in table 1, the user must use NEXUS.

Minimum Allowable Canopy Base Height

To minimize the amount of individual-tree torching that occurs, the CFFZ must be resistant to crown fire initiation. Van Wagner's (1977) model of crown fire initiation determines the fireline intensity necessary to initiate crowning based on canopy base height and foliar moisture content. Foliar moisture content is not a significant factor in the model and can be held constant at 100 percent (Scott 1998b). Because fireline intensity is related to flame length (Byram 1959), we can express the minimum allowable canopy base height as a function of expected surface fire flame length (figure 3). The nomograms produced by Scott and Reinhardt (2001) are useful for computing this threshold crown base height directly from fuel model and fuel moisture condition. NEXUS can be used for conditions not represented by the nomograms. For generic treatment planning, use the chart in figure 4, which assumes drought summer fuel moistures and a wind adjustment factor of 0.25 to represent the more open conditions of a treated stand. For the design condition of fire behavior fuel model 10, drought summer fuel moisture, and a 25 mph 20-ft windspeed, the minimum allowable CBH is 13 ft (figure 4).

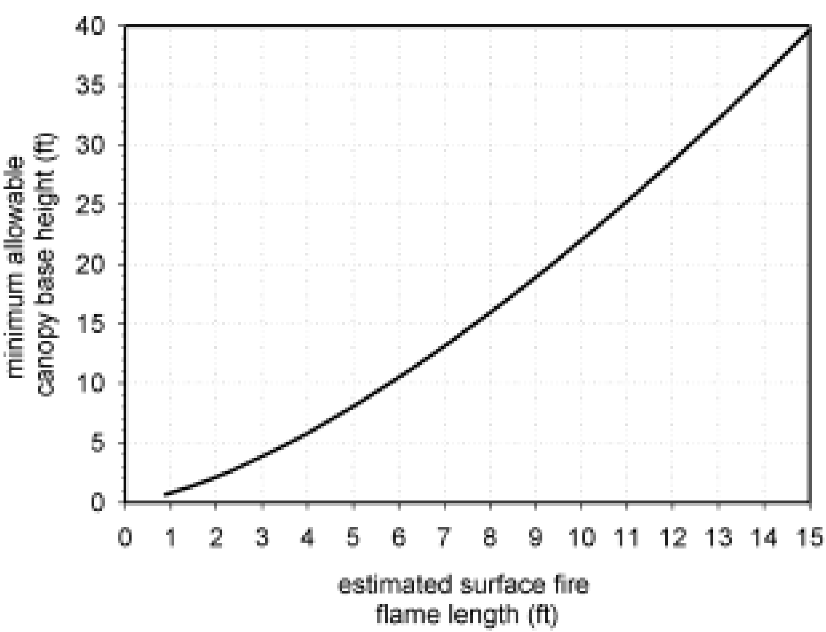

Figure 3—Minimum allowable canopy base height as a function of surface fire flame length, as predicted from Van Wagner's (1977) crown fire initiation model and Byram's flame length model.

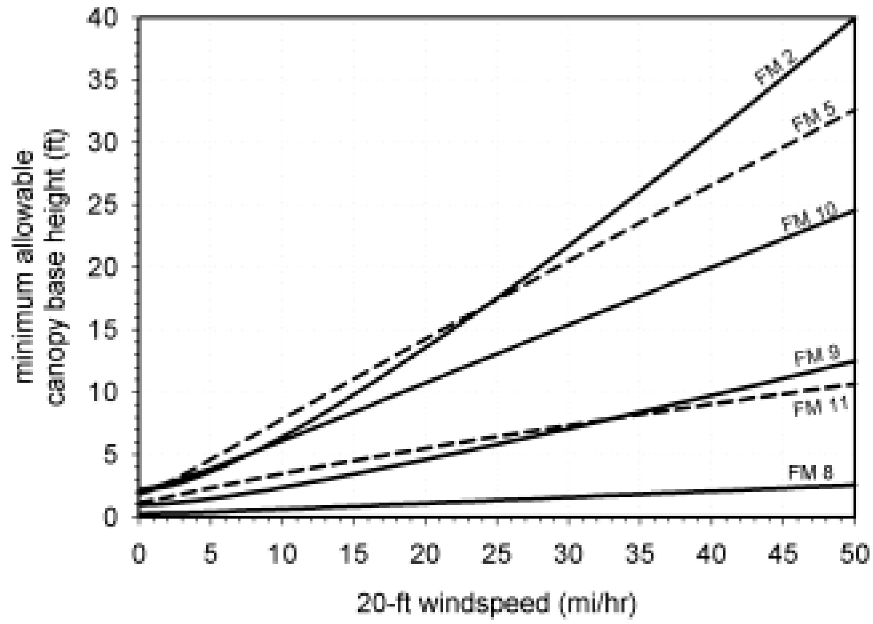

Figure 4—Minimum allowable canopy base height as a function of 20-ft windspeed, for a variety of fire behavior fuel models, assuming level ground, wind adjustment factor 0.25 (represents treated stand condition), and drought summer fine dead fuel moisture conditions (1-hr = 4%, 10-hr = 5%, 100-hr = 7%, live = 78%).

Discussion

Previous work based on similar models suggests fuels need to be treated a maximum of only 130 ft from a structure (Cohen and Butler 1998). However, that recommendation is based on preventing piloted ignition of wood by radiation, not protecting firefighters or the homeowner who may be present during fire front passage. Because structures are so susceptible to ignition by firebrands, if exterior ignitions are to be suppressed, people may need to be present. The treatment standard presented in this paper is designed to protect those people.

The default maximum allowable CBD in this analysis falls within the range of 0.074 to 0.125 kg / m^3 found by Agee (1996) to cause cessation of the crown fires reported by Rothermel (1991). Agee also found an empirical threshold of 0.10 kg / m^3 on the 1994 Wenatchee fires, which coincides exactly with the value from this analysis for drought summer fuel moisture and 25 mph 20-ft winds.

The three elements of the standard do not have an explicit factor of safety attached. There is no built-in safety factor for the prescribed CBD or CBH. However, several factors lead to a built-in margin of safety in the treatment size.

Flame length is used in place of flame height. Models of flame angle and flame height are now being developed, but not yet available. Using flame length in place of height slightly overestimates the required treatment.

The radiation model assumes that the flames radiate directly onto firefighters, but in reality any remaining trees in the CFFZ block some of the radiation before it reaches firefighters at the structure. The amount of blocking is not modeled, but may be significant. Blocked radiation by trees leads to a margin of safety.

The treatment assumes that active crowning is possible outside the CFFZ. If only passive crowning is possible outside the CFFZ, then flame length is over-predicted by the method.

Existing surface fuel treatment standards suggest larger treatment distances on steeper slopes, and generally only on the downslope side of the house. This standard suggests treating the same distance in all directions, regardless of slope steepness. Steeper slopes are well known to increase fireline intensity and spread rate, with the greatest effect in the uphill direction. However, windspeed usually has an even greater effect on fireline intensity and spread rate than slope steepness, but the direction of the effect is not known before the fire. Fire-carrying winds can be from any direction. In many cases a downslope wind would overpower the effect of slope steepness. If the direction of fire-carrying winds is well established for a site, then treatment distance from a structure could be reduced on the downwind side.

The models used in this method suggest very wide canopy fuel treatment areas. However, in many cases the canopy fuels are already within CBD and CBH limits, so no treatment will be necessary. In other cases a light treatment will reduce CBD to the desired level. In no case is complete removal of the forest canopy required to mitigate crown fire potential near a structure.

This standard also makes a structure less prone to ignition from embers. Ember exposure is reduced by eliminating crown fire immediately near the structure. Firebrand research has focused on maximum spotting distance from surface and crown fires, as fire-starting embers can travel very long distances in a convection column. However, little is known about the distribution of spotting distances from a crown fire or torching trees. It is likely that, while long spotting distances are attainable, most embers capable of igniting a structure do not travel very far at all. Therefore, reducing crown fire activity in the vicinity of a structure reduces its exposure to firebrands and thus its potential for ignition.

The cost of modifying canopy fuels to comply with this standard will vary widely. It depends in part on timber markets, terrain and stand conditions, method of treatment, and type of activity fuel treatment. In comparing three alternative treatments to reduce canopy fuels in second-growth ponderosa pine, Scott (1998a) found positive net returns of $156 to $832 per acre treated, depending on logging method, volume of trees removed and type of activity fuel treatment. However, those same treatments applied in different stands might not produce the same revenue.

Modifying canopy fuels as prescribed in this method may lead to increased surface fire intensity and spread rate under the same environmental conditions, even if surface fuels are the same before and after canopy treatment. Reducing CBD to preclude crown fire leads to increases in the wind adjustment factor (the proportion of 20-ft windspeed that reaches midflame height). Also, a more open canopy may lead to lower fine dead fuel moisture content. These factors increase surface fire intensity and spread rate. Therefore, canopy fuel treatments reduce the potential for crown fire at the expense of slightly increased surface fire spread rate and intensity. However, critical levels of fire behavior (limit of manual or mechanical control) are less likely to be reached in stands treated to withstand crown fires, as all crown fires are uncontrollable. Though surface intensity may be increased after treatment, a fire that remains on the surface beneath a timber stand is generally controllable.

If left untreated, activity fuel created while reducing CBD can exacerbate this increase in surface fire intensity and spread rate. Whole-tree harvesting or pile burning or broadcast burning of activity fuel is recommended following canopy fuel treatment to reduce surface fuel flammability.

Conclusion

Existing fuel treatment standards adequately address surface fuel only; they are assumed to be effective in creating defensible space at a structure. This paper presents a simple method of designing canopy fuel treatments likely to protect firefighters and homeowners in the wildland-urban interface. Firefighters attempting structure protection during passage of a fire front can work only within a safety zone. These guidelines create a crown-fire-free zone large enough that the tall flames from an active crown fire are unlikely to injure people protecting structures during a fire.

This method requires the manager to specify the design environmental conditions for the treatment. This design condition represents the limit of effectiveness of the treatment and should be communicated to homeowners and firefighters.

Acknowledgments

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References

Agee, James K. 1996. The influence of forest structure on fire behavior. In: Proceedings of the 17th Annual Forest Vegetation Management Conference; 1996 January 16–18; Redding CA: 52-68.

Anderson, Hal. E. 1982. Aids to determining fuel models for estimating fire behavior. Gen. Tech. Rep. INT-122. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 22 p.

Bradshaw, Larry; McCormick, Erin. 2000. FireFamily Plus user's guide, version 2.0. Gen. Tech. Rep. RMRS-GTR-67WWW. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Butler, Bret W.; Cohen, Jack D. 1998. Firefighter Safety Zones: A theoretical model based on radiative heating. International Journal of Wildland Fire. 8(2): 73-77.

Byram, George. M. 1959. Combustion of forest fuels. In: Forest fire: Control and use, 2nd edition. New York, NY: McGraw-Hill: 61-89.

California State Board of Forestry. 1996. California Fire Plan: a framework for minimizing costs and losses from wildland fires. Sacramento, CA: State of California. 104 p.

Cohen, Jack. D.; Butler, Bret. W. 1998. Modeling potential structure ignitions from flame radiation exposure with implications for wildland / urban interface fire management. In: 13th Conference on Fire and Forest Meteorology; 1996 October 27-31; Lourne, Victoria, Australia. Proc. Fairfield, WA: International Association of Wildland Fire: 81-86.

International Fire Code Institute. 1997. 1997 Urban-Wildland Interface Code. Whittier, CA: International Conference of Building Officials. 47 p.

Moore, Howard E. 1981. Protecting Residences from Wildfires: a guide for homeowners, lawmakers, and planners. Gen. Tech. Rep. PSW-50. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 44 p.

Rothermel, Richard. C. 1991. Predicting behavior and size of crown fires in the Northern Rocky Mountains. Res. Pap. INT-438. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 46 p.

Scott, Joe. H. 1998a. Fuel reduction in residential and scenic forests: a comparison of three treatments in a western Montana ponderosa pine stand. Res. Pap. RMRS-RP-5. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 19 p.

Scott, Joe. H. 1998b. Sensitivity analysis of a method for assessing crown fire hazard in the Northern Rocky Mountains, USA. In: Viegas, D. X., ed.; III International Conference on Forest Fire Research; 14th Conference on Fire and Forest Meteorology; 1998 November 16-20; Luso, Portugal. Proc. Coimbra, Portugal: ADAI. Volume II: 2517-2532.

Scott, Joe. H. 1999. NEXUS: a system for assessing crown fire hazard. Fire Management Notes. 59(2): 20-24.

Scott, Joe H.; Reinhardt, Elizabeth D. 2001. Assessing crown fire potential by linking models of surface and crown fire behavior. Res. Pap. RMRS-RP-29. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 59 p.

Van Wagner, C. E. 1977. Conditions for the start and spread of crown fire. Canadian Journal of Forest Research. 7: 23-34.

Reducing Crown Fire Hazard in Fire-Adapted Forests of New Mexico

Carl E. Fiedler[1] and Charles E. Keegan[2]

Abstract—*Analysis of FIA data for New Mexico shows that 2.4 million acres of ponderosa pine and dry mixed conifer forests rate high for fire hazard. A restoration treatment designed to address altered ecological conditions in these forests increased average crowning index (i.e., the wind speed necessary to maintain a crown fire) by 50 mph, compared to only 23 mph for a thin-from-below approach designed to reduce hazard. After we projected treated stands forward 30 years, only one-eighth of the acres receiving the thin-from-below treatment remained low hazard, compared to over half receiving the restoration treatment.*

Introduction

Recent wildfires provide harsh testimony to the hazardous forest conditions that exist over large areas of New Mexico. The fires of 2000 are especially notable, not just in terms of acres burned, but particularly because of the significant damage to property and associated threats to people. There is now both the public support and political will for major initiatives to address this regional concern (Western Governors' Association 2001). For example, tens of millions of dollars have been distributed through the National Fire Plan, much of it dedicated to reducing hazardous fuels. However, planning to address fire hazard at a strategic level requires understanding the forest conditions most vulnerable to fire and the effectiveness of alternative hazard reduction treatments.

Absence of a detailed, systematic, and uniform forest inventory for all acres and ownerships has heretofore precluded a comprehensive analysis of fire hazard in New Mexico. However, recent availability of Forest Inventory and Analysis (FIA) data, which are collected using consistent inventory protocols across all ownerships, made possible this strategic assessment of fire hazard at a statewide level.

Objectives

The goals of this study were to assess existing forest conditions and fire hazard in New Mexico and to evaluate the potential effectiveness of hazard reduction treatments. Specific objectives were to:

- Quantify occurrence of short-interval, fire-adapted forests in New Mexico.
- Evaluate existing forest conditions for crown fire hazard.
- Develop alternative treatment prescriptions and evaluate their effectiveness in reducing hazard.
- Project existing and treated conditions 30 years into the future and reevaluate fire hazard.

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Methods

Forest Inventory and Analysis data were used to profile existing conditions of forestland in New Mexico. FIA data are collected using a systematic grid, with each inventory point or "cluster" representing approximately 6500 acres. The FIA inventory includes forestlands in national parks and other reserved lands, such as designated wilderness areas. The initial analysis of FIA data for this study included lands classified as "timberlands" (i.e., primarily stocked with traditional timber species) and "woodlands" (i.e., primarily stocked with pinyon pine [*Pinus edulis*], juniper [*Juniperus* spp.], or hardwoods other than aspen [*Populus tremuloides*] or cottonwood [*Populus* spp.]).

The assessment of fire hazard reported here was conducted exclusively on short-interval, fire-adapted forests. In New Mexico, these are primarily comprised of ponderosa pine (*Pinus ponderosa*) and dry mixed conifer forest types. Fire-adapted forests were identified as the highest priority for treatment in "Protecting People and Sustaining Resources in Fire-Adapted Ecosystems — A Cohesive Strategy" (DOI 2001). Frequent, low-intensity fires were the primary agent that shaped these forests historically and kept them resistant to severe fires (Agee 1998). Effective fire-suppression efforts and some logging practices over the last century have resulted in density and structural changes that leave these forests vulnerable to severe damage from wildfire (Covington and Moore 1994).

Fire Hazard

Potential fire hazard was analyzed for individual inventory clusters (n = 261) using the Fire and Fuels Extension (FFE: Beukema et al. 1997, Scott and Reinhardt 2001) to the Forest Vegetation Simulator (FVS: Crookston 1990, Stage 1973, Van Dyck 2001). The FFE extension estimates crown fire hazard based on tree, stand, and site characteristics, and expresses fire hazard/effects in terms of crowning index, torching index, and basal area mortality. Model runs were made assuming a 20-foot wind speed of 20 mph, a moisture level of one (driest condition), air temperature of 90 degrees F, and a wildfire rather than prescribed burning situation.

Crowning index, defined as the wind speed necessary for a fire that reaches the canopy to continue as a crown fire, was the primary variable used to report hazard in this study. Crowning index is largely driven by canopy bulk density, which FFE calculates from individual tree biomass summed to the stand level (Scott and Reinhardt 2001). High-hazard forest conditions were defined as having a crowning index less than 25 mph, moderate hazard from 25 to 50 mph, and low hazard greater than 50 mph. The Central Rockies variant of the FVS model was used to project post-treatment forest conditions 30 years into the future, at which time crown fire hazard (i.e., crowning index) was again assessed using FFE. Two model types were used—model type 1 (southwestern mixed conifer) and model type 2 (southwestern ponderosa pine) for the dry mixed conifer and ponderosa pine forest types, respectively. Default site index values were used (DF 70 for model type 1, and PP 70 for model type 2), and regeneration was turned "on."

Hazard Reduction Treatments

A variety of management approaches can potentially be used to address hazardous conditions in short-interval, fire adapted forests; three contrasting ones are compared in this paper. One approach is low thinning to a given diameter limit, a treatment that has been widely recommended (Dombeck 1997). A diameter limit of 9 inches was used in this analysis (thin-from-below to 9 inches).

A second approach retains all trees larger than 16 inches. This prescription (16 inches diameter-limit) is influenced by concerns that there may be a deficit of trees in the Southwest greater than 16 inches compared to historic levels, and that cutting trees larger than this size is economically rather than ecologically motivated.

A third approach is aimed at initiating restoration of sustainable structure and composition (and ultimately, ecological function). It therefore focuses on the trees to leave in terms of a target density, diameter distribution, and species composition (Fiedler et al. 1999, Fiedler et al. 2001). Under this prescription (restoration), trees are marked for leave in the sizes, numbers, species, and juxtaposition that will go furthest toward restoring a sustainable structure, given existing stand conditions. Most of the 40 to 50 ft^2/acre target reserve basal area is comprised of larger trees, although some trees are marked for leave throughout the diameter distribution, if available. Silvicultural treatments involved in the restoration approach include a low thinning to remove small trees, improvement cutting to remove late-successional species (if present), and selection cutting to reduce overall density and promote regeneration of ponderosa pine. The multiple objectives of the restoration treatment are to reduce fire hazard, increase tree vigor, spur development of large trees, and induce regeneration of seral species.

A common objective of all three treatments is to reduce density (in varying degrees) and create a discontinuity in the vertical fuel profile by cutting the sapling- and pole-sized ladder fuels. Reducing the hazard associated with these smaller cut trees, as well as the tops and limbs of merchantable-sized trees (if any) that are harvested as part of the overall treatment, is an integral part of each prescription. The resulting slash is lopped and scattered, broadcast burned, or piled and burned depending on volume, reserve stand density, landowner objectives, and cost considerations.

All three prescriptions were applied to high hazard conditions (i.e., those with a crowning index <25 mph) in the ponderosa pine and dry mixed conifer forest types. The thin-from-below to 9 inch prescription was only applied to stands that had greater than 50 ft^2/acre of trees larger than 9 inches. For the 16 inch diameter-limit prescription, all trees larger than 16 inches dbh were left, so long as the basal area of these trees was 50 ft^2/acre or greater. If there were less than 50 ft^2/acre of basal area in trees >16 inches , then trees <16 inches were retained from the biggest on down (i.e., 15 inches, 14 inches, 13 inches, etc.) until a basal area density of 50 ft^2/acre was reached.

The restoration prescription differed from the other two prescriptions in that it set a target reserve density of 50 ft^2/acre in all stands designated for treatment. Most of the basal area marked for leave was concentrated in the larger trees, but smaller amounts of basal area were reserved in trees across the full diameter distribution, if available.

Results

Fire Hazard: Existing Conditions

Analysis of FIA data shows that there are approximately 16.6 million acres of woodlands/forestlands in New Mexico. The short-interval, fire-adapted forests that are of greatest concern in terms of fire hazard collectively occupy

4.0 million acres. Approximately 61%, or 2.4 million acres, of these historically fire-adapted forests classify as high hazard, 31% as moderate hazard, and only 8% as low hazard, based on crowning index (figure 1).

The trends in crowning index across density and structural classes were especially notable. For example, focusing on short-interval, fire-adapted forests only, average crowning index declined (i.e., hazard increased) across the range of densities from 39 mph at low density, to 22 mph at moderate density, to 18 mph at high density. In stands with multi-storied structures, 85% were rated high hazard if they were also in the high-density category, whereas only 15% of moderate-density and no low-density stands with multi-storied structure received this rating.

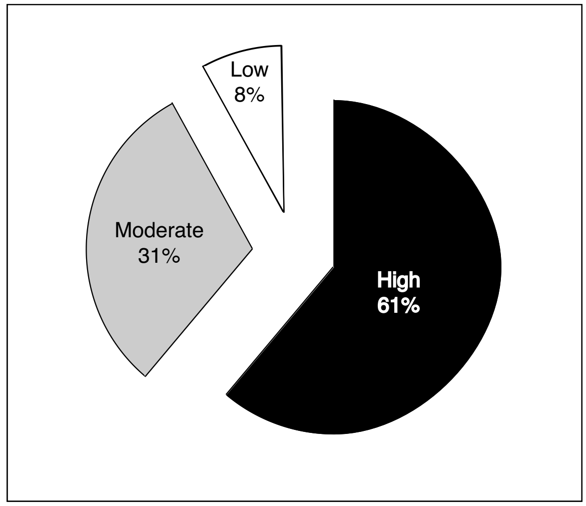

Figure 1—Proportion of New Mexico's short-interval, fire-adapted forests (4.0 million acres) by fire hazard rating.

Fire Hazard: Treated Conditions

Evaluation of the effectiveness of hazard reduction treatments focused on 1.7 million acres of high hazard forests within short-interval, fire-adapted ecosystems (i.e., ponderosa pine and dry mixed conifer forest types). The average composition of these high-hazard conditions is shown in figure 2. Approximately 0.7 million acres were not evaluated because stand composition was such that less than 50 ft^2/acre of basal area would remain following application of one or more of the treatment prescriptions. Both treatment effectiveness and the durability of effects varied by prescription (table 1). For example, thinning-from-below increased average crowning index from 16 to 39 mph, or 23 mph over existing conditions, compared to increases of 45 and 50 mph for the diameter-limit and restoration treatments, respectively (table 1). The Friedman test found significant differences ($p<0.001$) among the distributions of crowning indexes for the four treatments (i.e., existing conditions and three hazard reduction treatments). Multiple comparison tests (Hochberg and Tamhane 1987) also found that all three treatments significantly increased crowning indexes (i.e., reduced crown fire hazard) relative to untreated conditions ($p=0.001$). In addition, both the 16 inches diameter-limit and restoration treatments significantly increased crowning indexes ($p=0.001$) relative to either the thin-from-below to 9 inches treatment or untreated conditions.

The distribution of treated acres in terms of crowning index also varied among prescriptions. While all three hazard reduction approaches increased

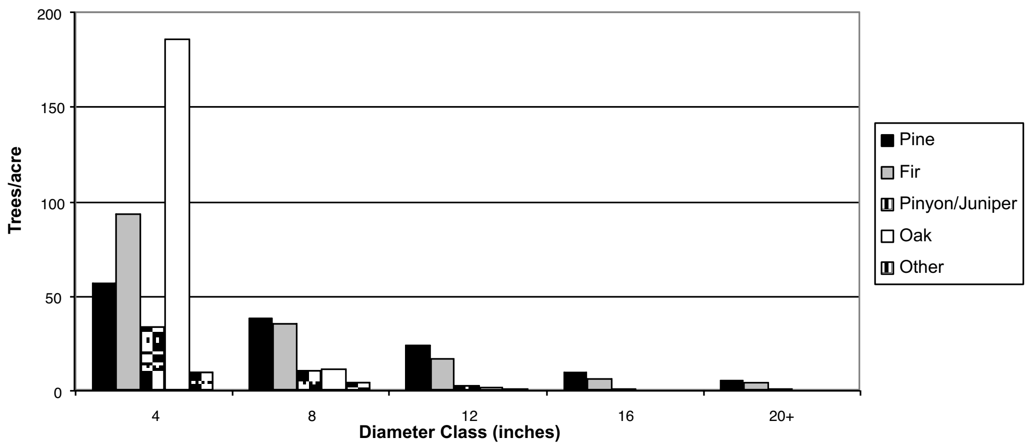

Figure 2—Average composition of existing high-hazard conditions in terms of trees per acre by diameter class and species.

Table 1—Effectiveness of hazard reduction treatments for increasing crowning index within short-interval, fire-adapted forests in New Mexico.

Hazard reduction treatment	Average crowning index before treatment (std. dev.)	Average crowning index after treatment (std. dev.)	Average crowning index 30 years after treatment (std. dev.)
Thin-from-below	16 (6.0)	39 (12.7)	37 (11.0)
Diameter-limit	16 (6.0)	61 (20.6)	57 (18.8)
Restoration	16 (6.0)	66 (20.3)	56 (16.6)

crowning indexes compared to existing conditions, the greater effectiveness of the diameter-limit and restoration treatments is apparent (figure 3).

Crown fire hazard was reevaluated after projecting post-treatment conditions 30 years into the future. Although the effectiveness of all treatments diminished somewhat, differences among treatments tended to persist through time (figure 3). On average, stands that received thinning-from-below remained in the moderate-hazard range 30 years after treatment, with average crowning index declining slightly to 37 mph (table 1). Diameter-limit and restoration treatments retained an overall low-hazard rating, with average crowning indexes above 50 mph (table 1).

The thin-from-below treatment was only effective in moving 18% of treated acres to a low hazard rating (table 2). The diameter-limit and restoration treatments, in contrast, created low hazard conditions on 72% and 79% of treated acres, respectively. The limited effectiveness of thinning-from-below is further illustrated by the small number of treated acres (13%) that remained low-hazard 30 years following treatment. Over 60% retained that classification for the diameter-limit and restoration treatments (table 2).

Table 2—Effectiveness of hazard reduction treatments for creating low hazard conditions in short-interval, fire-adapted forests in New Mexico.

Hazard reduction treatment	Acres rated low hazard after treatment (percent)	Acres rated low hazard 30 years after treatment (percent)
Thin-from-below	18	13
Diameter-limit	72	62
Restoration	79	62

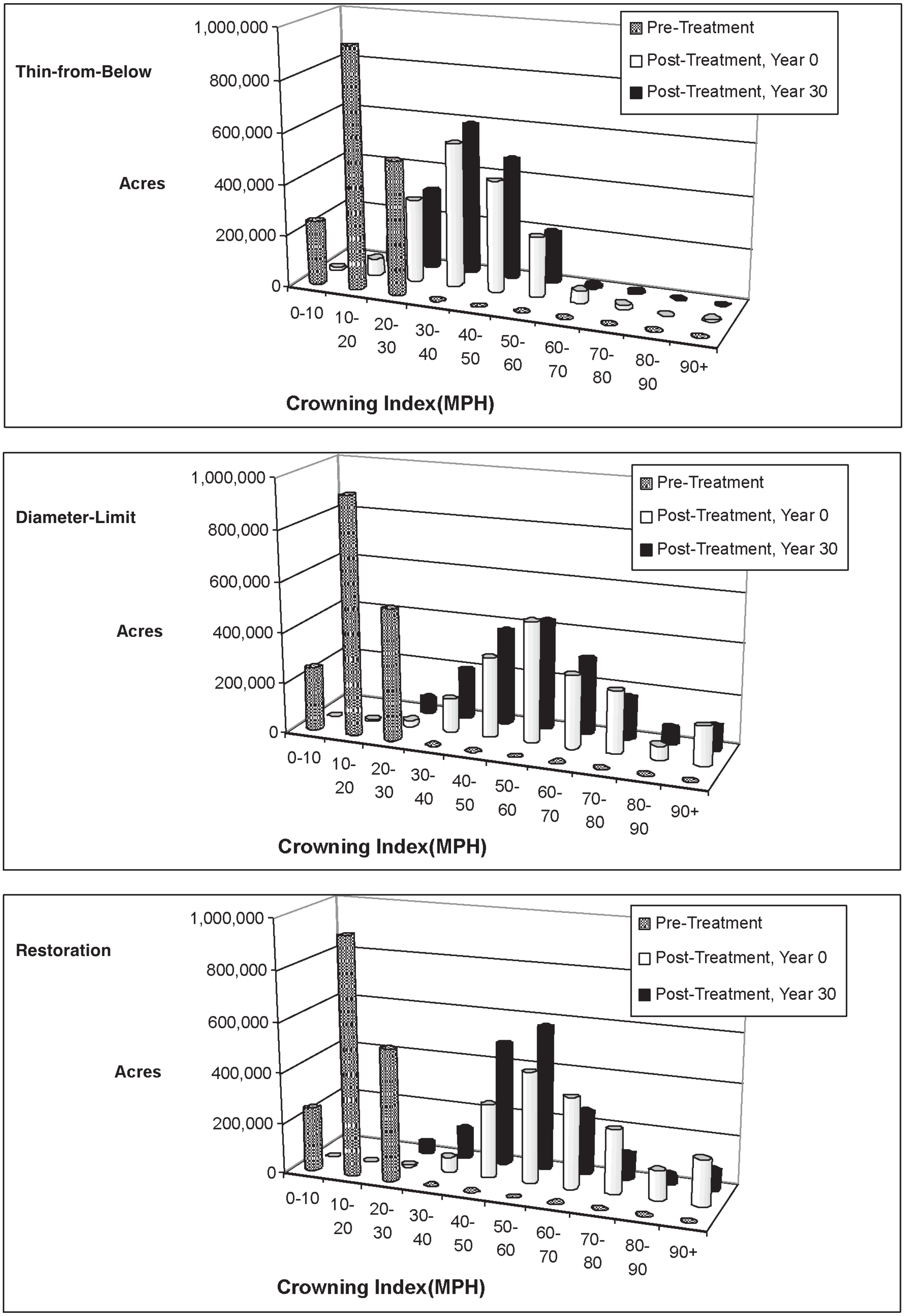

Figure 3—Crowning indexes for pre-treatment, post-treatment year 0, and post-treatment year 30, for three hazard reduction prescriptions applied to high-hazard stands (woodland species removed).

Fire Hazard: Effects of Woodland Species

Pinyon pine and juniper are a common stand component in the ponderosa pine and dry mixed conifer forest types. Whether these species are retained or removed in a given project depends upon treatment objectives, ownership, and the stand context within which they occur. In contrast, gambel oak (*Quercus gambelii*) is typically retained on all ownerships to serve a variety of amenity and wildlife habitat objectives. However, retention of woodland species may have an undesirable effect in terms of increased crown fire hazard. For this reason, post-treatment crowning index values were evaluated for two scenarios—one with woodland species retained and the other with these species removed (table 3). The effect of removing woodland species is substantial, with average post-treatment crowning indexes increasing (i.e., hazard decreasing) by 16, 30, and 26 mph compared to the "retention" scenario for the thin-from-below, diameter-limit, and restoration treatments, respectively (table 3).

Table 3—Effectiveness of hazard reduction treatments for increasing crowning index (C.I.) with woodland species retained and removed.

Hazard reduction treatment	Average C.I. before treatment (std. dev.)	Average C.I. after treatment, woodland species retained (std. dev.)	Average C.I. after treatment, woodland species removed (std. dev.)
Thin-from-Below	16 (6.0)	23 (10.3)	39 (12.7)
Diameter-Limit	16 (6.0)	31 (16.6)	61 (20.6)
Restoration	16 (6.0)	40 (24.0)	66 (20.3)

Discussion

This study represents the first statewide effort in New Mexico to estimate fire hazard and evaluate the effectiveness of various hazard reduction treatments. It can be used both as a strategic planning tool to address broad-scale fire hazard concerns, and as a tactical guide to help managers design effective treatments at the project level.

It is critical that managers carefully evaluate treatment effectiveness before selecting and applying hazard reduction treatments. For example, applying the thin-from-below to 9 inch prescription to high hazard ponderosa pine and dry mixed conifer forests has only a modest effect on lowering average crown fire hazard (i.e., increasing crowning index). Furthermore, this prescription moves fewer than 20% of treated stands into a low hazard condition after treatment, compared to 72% and 79% for the 16 inches diameter-limit and restoration prescriptions. These results underscore the importance of evaluating pre- and post-treatment conditions (stand tables) for fire hazard during the process of prescription development.

Crowning indexes associated with the 16 inch diameter-limit and restoration treatments differed little—either immediately after treatment or 30 years later. However, the ecological conditions and potential sustainability associated with these two treatments will likely differ substantially over time. Under the restoration approach, late-seral species (if present) are preferentially cut to eliminate them as a seed source, and overall reserve density is prescribed sufficiently low to induce regeneration of ponderosa pine, thereby enhancing sustainability. In contrast, the 16 inch diameter-limit approach neither prescribes nor allows removal of late-seral trees greater than 16 inches in diameter—trees large enough to be primary seed-producers. Furthermore, density will generally increase over time under this treatment regime, as more and more trees pass over the 16 inch diameter threshold and become unavailable for cutting. Crown fire hazard will increase, and the resulting conditions will favor establishment of late-seral species in the understory. Over decades, the result could be a fundamental shift in forest type from ponderosa pine to more shade-tolerant (and fire-, insect-, and disease-prone) species. Even if late-seral species are not present, burgeoning density of overstory pines greater than 16 inch diameter will limit establishment and early development of young pines, with fire hazard and vulnerability to beetles increasing commensurately.

A common management view of woodland species in ponderosa pine or dry mixed conifer forests is that they are relatively innocuous in terms of their effects on the growth and vigor of timber species, while providing a variety of ecological, visual, and wildlife values. Results of this analysis show that regardless of hazard reduction treatment, the retention of woodland species leads to a 16 to 30 mph decrease in average crowning index. For this reason, managers should weigh the benefits that would accrue by retaining these species against the substantial reduction of fire hazard that would result from removing them.

Results of this study show that the fire hazard problem in New Mexico is best addressed by forest restoration approaches that recognize the broader ecological context within which hazard occurs. Indeed, Fule et al. (2001) point out that hazard reduction may be viewed as an incidental benefit of restoration treatments given the multiple ecological benefits that accrue to even a partially successful restoration program. Whether degraded, fire-adapted forests are viewed from the standpoint of hazard reduction or ecological condition, an approach that centers on the density, structure, and species composition of the reserve stand is superior to prescriptions that focus only on the size of trees removed. The restoration prescription evaluated in this analysis achieves greater hazard reduction and creates more sustainable conditions than alternative treatments. It is particularly superior when compared to prescriptions with a singular focus on removal of small trees.

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References

Agee, James K. 1998. The landscape ecology of western forest fire regimes. Northwest Science 72: 24-34.

Beukema, Sarah J.; Greenough, Julee A.; Robinson, Donald C.E.; Kurz, Werner A.; Reinhardt, Elizabeth D.; Crookston, Nicholas L.; Brown, James K.; Hardy, Colin C.; Stage, Albert R. 1997. An introduction to the Fire and Fuels Extension to FVS. Proceedings of a symposium; 1997 February 3-7; Fort Collins, CO. Teck, Richard; Moeur, Melinda; Adams, Judy, eds. Gen. Tech. Rep. INT-373. Ogden,

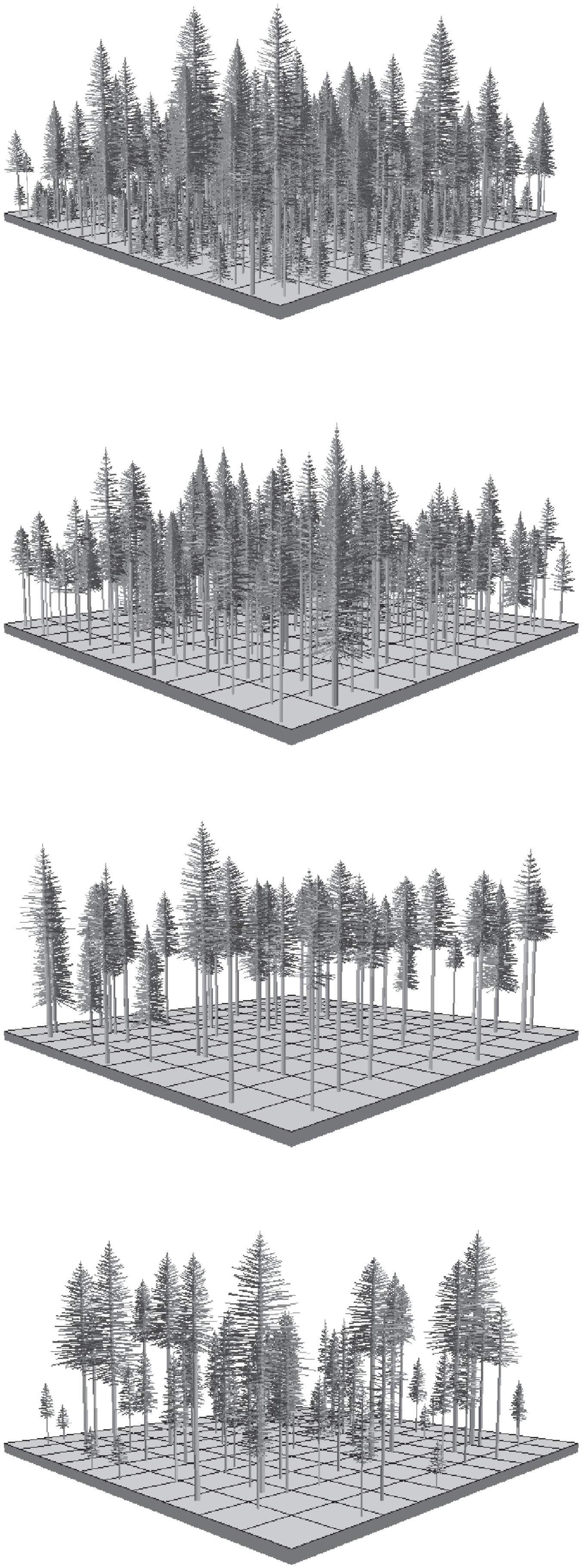

Figure 4—Stand visualizations of:

a) existing conditions in forests rated high for crown fire hazard,

b) after applying a thin-from-below to 9″ prescription,

c) after applying a 16″ diameter-limit prescription, and

d) after applying a restoration prescription.

UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 191-195.

Covington, Wallace W.; Moore, Margaret M. 1994. Southwestern ponderosa pine forest structure: Changes since Euro-American settlement. Journal of Forestry 92: 39-47.

Crookston, Nicholas L. 1990. User's guide to the event monitor: Part of prognosis model version 6. Gen. Tech. Rep. INT-275. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 21 p.

(DOI) Department of the Interior. 2001. "Integrating Fire and Natural Resources Management—A Cohesive Strategy for Protecting People by Restoring Land Health" (http://fireplan.gov/references).

Dombeck, Michael. 1997. Statement to Senate Committee on Energy and Natural Resources, Feb. 25, 1997. U.S. Congress, Washington, DC.

Fiedler, Carl E.; Keegan, Charles E.; Arno, Stephen F.; Wichman, Daniel P. 1999. Product and economic implications of ecosystem restoration. Forest Products Journal 49: 19-23.

Fiedler, Carl E.; Arno, Stephen F.; Keegan, Charles E.; Blatner, Keith A. 2001. Overcoming America's wood deficit: An overlooked option. BioScience 51: 53-58.

Fule, Peter Z.; Waltz, Amy E.M.; Covington, Wallace W.; Heinlein, Thomas A. 2001. Measuring forest restoration effectiveness in reducing hazardous fuels. Journal of Forestry 99: 24-29.

Hochberg, Yosef; Tamhane, Ajit C. 1987. Multiple comparison procedures. John Wiley and Sons, NY. 450 p.

Scott, Joe H.; Reinhardt, Elizabeth D. 2001. Assessing crown fire potential by linking models of surface and crown fire behavior. Res. Pap. RMRS-RP-29. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 59 p.

Stage, Albert R. 1973. Prognosis Model for stand development. Res. Pap. INT-137. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station.

(USFS) United States Forest Service. 2000. "Protecting People and Sustaining Resources in Fire-adapted Ecosystems—A Cohesive Strategy" (http://www.fs.fed.us/pub/fam/).

Van Dyck, Michael G. 2001. Keyword reference guide for the Forest Vegetation Simulator. WO-TM Service Center, USDA Forest Service. Fort Collins, CO. 97 p.

Western Governors' Association. 2001. "A Collaborative Approach for Reducing Wildland Fire Risks to Communities and the Environment—10-Year Comprehensive Strategy" (http://www.westgov.org/wga/initiatives/fire/final_fire_rpt.pdf).

Definition of a Fire Behavior Model Evaluation Protocol: A Case Study Application to Crown Fire Behavior Models

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Abstract—*Model evaluation should be a component of the model development process, leading to a better understanding of model behavior and an increase in its credibility. In this paper a model evaluation protocol is proposed that encompasses five aspects: (1) model conceptual validity, (2) data requirements for model validation, (3) sensitivity analysis, (4) predictive validation (incorporating statistical tests), and (5) model comparison. The proposed protocol was applied to evaluate fire behavior models that were developed to predict crown fire initiation and spread with potential application in fire management decision support systems. The evaluation protocol highlighted the limitations and the distinct behavior of the specific models and the implications of such limitations when applying those models to support fire management decision-making. The model limitations identified through these results helped the authors to characterize deficiencies in the state-of-knowledge of the determinant processes involved in crown fire behavior, thereby identifying pertinent research needs.*

Introduction

Advances in fire behavior science have gradually resulted in the development of fire models to support the decision-making of land managers in a large array of fire management problems (Cohen 1990). The use of such models allows managers to reduce the uncertainties associated with applying fire as a management tool and facilitates proactive management. The complexities associated with wildland fire phenomenology results in a large number of unknowns. These limitations lead researchers to model a specific phenomenon as they rely on distinct simplifying assumptions and include different independent fire environment properties as driving variables. This results in a situation where distinct models attempting to describe the behavior of a particular process respond differently to a given set of conditions. This is noticeable in evaluations of fuel treatment effectiveness in reducing fire potential. Model outputs might be misleading and result in misguided management. Modification of the structure of the fuel complex has been the main, if not the only, method by which fire managers can reduce the fire potential of a given area (Countryman 1974). When the changes that a fuel treatment causes in fire behavior are evaluated, three approaches can be identified: (1) analysis of post fire damage in adjacent treated and untreated stands burned under similar burning conditions (e.g., Weatherspoon and Skinner 1995, Pollet and Omi 2002); (2) monitoring of changes in various fire behavior determinants, such as diurnal and seasonal fuel moisture dynamics, the vertical wind profile, fuel available for combustion, overall fuel complex structure, and experimental fires burns in the various study plots to assess differences in fire

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behavior (e.g., Alexander et al. 1991; Gould et al. 2001; Alexander and Lanoville 2002); and (3) fire behavior simulations integrating fuel descriptors and critical fire weather parameters (e.g., Kalabokidis and Omi 1998, Hirsch and Pengelly 1999).

Due to the nature and complexity of the interactions defining a particular fire environment, it is unlikely that a fire environment monitoring program, unless it was extremely comprehensive, would capture all the variability that would be present in a particular fuel treatment setting. Although this approach would produce the most realistic data, the associated cost limits its applicability. Hence, most evaluations of fuel treatment effectiveness are based on simulations of potential fire behavior that rely on fire behavior models. This approach allows integrating the effects of the determinant input variables through their spectrum of variability. In addition, infrequent combinations of particular fire environment variables and certain fire behavior processes that are not easily measured in the field can accordingly be analyzed. Nevertheless, our understanding of the processes and interactions in the system through the use of models needs to be accompanied with a thorough description of the assumptions and main limitations of the models employed.

Most of the studies that have analyzed the effects of silvicultural / fuel treatment practices on fire hazard at the stand level rely solely on changes in the structure of the fuel complex to infer potential fire behavior (e.g., Fahnestock 1968, Alexander and Yancik 1977, van Wagtendonk 1996, Kalabokidis and Omi 1998, Scott 1998b, Stephens 1998, Hirsch and Pengelly 1999, Fulé et al. 2001, Fiedler et al. 2001). In general, the fire simulations that support these studies have not taken into account the possible effects of changes in fuel moisture gradients and sub-canopy wind flow between treated and untreated situations. Some of these studies produced conflicting results due to degree of comprehensiveness taken in modeling a particular situation and the nature and characteristics of the fire behavior models used to carry out the simulations.

The objective of this paper is to define a model evaluation protocol that results in a better understanding of the use of fire behavior models in these applications, including their limitations and biases, which we hope will eventually increase their credibility when used in evaluating effectiveness of fuel treatments for reducing fire behavior potential.

Defining Criteria for Model Evaluation

We believe that testing and evaluation of models should be a fundamental component of the model development process. These activities assume particular importance in fire behavior science due to the inherent difficulties in measuring and understanding some of fire's determinant processes. In spite of this, model evaluation has not received much emphasis by fire behavior modelers and no comprehensive model evaluation protocol has been applied previously to fire behavior models. Most fire behavior model evaluation that has been done has been limited to two areas: (1) comparisons of model predictions with observed and measured fire behavior data (e.g., Brown 1972, Lawson 1972, Bevins 1976, Sneeuwjagt and Frandsen 1977, Hough and Albini. 1978, Andrews 1980, Brown 1982, Norum 1982, Rothermel and Rinehart 1983, van Wagtendonk and Botti 1984, Albini and Stocks 1986, van Wilgen and Wills 1988, Gould 1991, Hirsch 1989, Marsden-Smedley and Catchpole 1995, Alexander 1998, Grabner et al. 2001); and (2) sensitivity analysis studies (e.g., Bevins and Martin 1978; Trevitt 1991, Scott 1998a, Cruz 1999). Critical examination of these studies reveals that they typically use subjective and / or qualitative measures of model performance and often lack validation standards.

The process of model evaluation has been approached differently by authors due in part to philosophical interpretations as to what constitutes a model. Models cover a large spectrum of idealizations and complexity, ranging from a model being considered as a mathematical interpretation of a theory / hypothesis bound by certain assumptions to models that are regarded as simple algebraic expressions that reflect a certain dataset. An important aspect to consider in model evaluation is the definition of the criteria that should be followed, which depends on the type of model being evaluated and its potential application. When fire behavior science is considered, theoretical models developed to understand certain physical and chemical phenomena (e.g., Grishin 1997, Linn 1997) should be evaluated in a different manner than models built to support fire management decision-making (e.g., Rothermel 1972, Albini 1976, Alexander 1988, Forestry Canada Fire Danger Group 1992). Theoretical models can be viewed as mathematical descriptions of certain physical fire phenomena and so the emphasis on evaluation should be placed on behavioral validation instead of the numerical precision of the result (Bossel 1991). The unknowns involved in some heat transfer and fluid dynamics processes coupled with our inability to reliably measure certain fundamental fire quantities limit any attempt to evaluate these models by direct comparison with independently derived data. In this paper, we emphasize the latter group of models—i.e., simple empirical or semi-empirical models (Catchpole and de Mestre 1986) that combine physical laws with empirical data to generate important model components describing particular physical fire phenomena. When these models are integrated, they produce workable models or systems that can be used to support operational fire management decision-making.

We consider model evaluation as follows: "The substantiation of a model within its domain of applicability possesses a satisfactory range of accuracy consistent with the intended application of the model" (Sargent 1984). Based on an examination of the previously referred fire behavior model evaluation studies and other validation studies of engineering and ecological models (e.g., Mayer and Butler 1993, Oderwald and Hans 1993, Bacsi and Zemankovics 1995), a model evaluation protocol based on the approaches of Rykiel (1996) and Sargent (1984) was defined for the purposes of the present work. This protocol includes the following analyses:

Model conceptual validity—This involves analysis of the conceptual structure and logic of a model. By taking into account the intended use of the model, we aim to determine the validity of the model's various theories and simplifying assumptions in capturing the dynamics of the system.

Data validation—This includes definition of data quality standards, namely the selection of real world data that represent the phenomena of interest for use in predictive validation, statistical validation, and model comparison. This aspect assumes particular importance when analyzing fire behavior data due to the relative inaccuracy and bias that arise from inherent difficulties in capturing reliable fire behavior data from either experimental fires, operational prescribed fires, or wildfires.

Sensitivity analysis—This consists of analyses to reveal the relative influence of model components and input parameters on the behavior of the model overall. This also includes identification of parameters that cause minor or major fluctuations in model outputs.

Predictive validation—This involves comparison of model outputs with an independent dataset of the phenomena under study in order to evaluate model suitability to predict system behavior. This section of the model evaluation incorporates a variety of statistical tests conducted in order to quantitatively evaluate model performance.

Model comparison—This involves a comparison of outputs from several models describing the same phenomena thereby providing an understanding of possible model deficiencies and the limits of applicability. In the present study a comparison of model behavior and validity was done concurrently with the predictive and statistical validations.

In order to better understand the evaluation protocol being proposed, we selected a case study application of the methodology to crown fire initiation and spread models. The models analyzed were Van Wagner (1977), Alexander (1998) and Cruz (1999) for crown fire initiation; and Rothermel (1991), Forestry Canada Fire Danger Group (1992) and Cruz (1999) for crown fire spread rate. The formulation of the two Cruz (1999) models is based on future research needs suggested by Alexander (1998). For the Canadian Forest Fire Behavior Prediction (FBP) System (Forestry Canada Fire Danger Group 1992), we chose fuel type specific fire spread models that can be applied to fuel types in the western United States, namely FBP System Fuel Type C-3 [mature jack (*Pinus banksiana*) or lodgepole pine (*Pinus contorta*)], C-6 [Conifer plantation] and C-7 [ponderosa pine (*Pinus ponderosa*) / Douglas-fir (*Pseudotsuga menziesii*)]. The emphasis of the present study is on establishing a model evaluation protocol rather than a comprehensive analysis of all available models (e.g., Xanthopoulos 1990, Van Wagner 1993). Some fire behavior prediction systems commonly used to evaluate fuel treatments, such as FARSITE (Finney 1998) and NEXUS (Scott 1999), integrate some of the models mentioned above with their specific interpretations of certain fire behavior processes. As a result, the models in these systems can to a certain extent be viewed as distinct from the original model formulations being analyzed here. Nevertheless, the present study is concerned with the core models driving the systems (i.e., crown fire rate of spread and initiation models). We have assumed that those particular interpretations imbedded in FARSITE and NEXUS do not induce significant changes in the outputs to warrant inclusion in the present comparison. Consequently, the evaluation procedures throughout this study are applied solely to the aforementioned models.

Model Conceptual Validity

The analysis of the validity of model theories and assumptions is of particular importance in models intended to support fire management decision-making. Inappropriate use of the models could lead to detrimental and long lasting effects on the ecosystem. Given the unknowns in fire science and difficulty in correctly measuring certain fundamental fire quantities and processes, any analysis of the theories and simplifying assumptions embodied in a model is limited by the current state-of-knowledge, availability of complete and reliable data sets, and our inability to propose more realistic theories. A detailed examination of the underlying theories of the various models considered here is outside of the scope of this paper. Consequently only a discussion of the most restrictive assumptions of the models will be presented.

A brief introductory description of the various models follows. See table 1 for a listing of the inputs involved in the crown fire initiation models. Both the

Table 1—Input requirements for the crown fire initiation models being evaluated[1].

Model	Fireline intensity	FMC	Wind speed	Residence time	CBH/FSG	EFFM	SFC
Van Wagner (1977)	X	X			X		
Alexander (1998)	X	X	X	X	X		
Cruz (1999)			X		X	X	X

[1]FMC= foliar moisture content; CBH= canopy base height; FSG = fuel strata gap; EFFM = estimated fine dead fuel moisture content; and SFC = surface fuel consumption.

Van Wagner (1977) and Alexander (1998) models are based on convective plume theory (cf. Yih 1953) and air temperature (as a surrogate of convective heat flux) decay with height above a linear heat source. The outputs of both the Van Wagner (1977) and Alexander (1998) models are deterministic in nature. The canopy ignition requirements are stated in terms of a critical fireline intensity (as per Byram 1959), as a function of a heat sink (evaluated differently in the two models), and height above the ground. The Alexander (1998) model, although based on the same convective plume theory, relies on a more realistic heat source and heat sink definition and takes into account the interaction between the convective plume and the cross wind with the subsequent tilting of the plume and dilution of the hot gases in the plume. The Cruz (1999) crown fire initiation model is the result of an application of logistic regression analysis to an experimental database (n = 73) and involved both surface and crown fires used in the development of the Canadian FBP System (Forestry Canada Fire Danger Group 1992) in order to predict the onset of crowning. Within this model the vertical stratification of the fuel complex was slightly modified from the commonly accepted definition of a canopy base height (CBH) to the definition of a new fuel complex descriptor, namely fuel strata gap (FSG). This parameter incorporates the effect of dead aerial fuels (canopy base height is associated with live canopy fuels) in reducing the distance between surface fuels and ladder fuels that can support vertical fire propagation. The introduction of this variable was justified by the dependence of canopy base height on just live foliage (Sando and Wick 1972, Kilgore and Sando 1975, McAlpine and Hobbs 1994, Ottmar et al. 1998) and consequently not incorporating the effect of dead fuels in lowering the distance between the surface and canopy fuel layers.

The Rothermel (1991) crown fire rate of spread model is the result of a simple average correction factor relating predicted surface fire behavior by the BEHAVE system (Andrews 1986) using Fuel Model 10 (Anderson 1982) with wind speed adjustment factor set equal to 0.4 and a series of observations of spread rates garnered from wildfires (n = 8). This model does not incorporate any stand structure descriptor in its formulation. The Canadian FBP System models for predicting crown fire rate of spread are based on a sigmoid equation taking into account certain components of the Canadian Forest Fire Weather Index System (Van Wagner 1987) that relate to the potential for fire spread as determined by the moisture content of fine fuels and wind speed. The Cruz (1999) crown fire spread model is based on non-linear regression analysis parameterized using data from high intensity experimental crown fires. This model differentiates crown fires spreading as either continuous or intermittent based on Van Wagner's (1977) spread rate criterion for active crowning.

Data Validation

The very nature of crown fires leads to various difficulties, either institutional, social, or experimental, in acquiring reliable outdoor fire behavior data (Alexander and Quintilio 1990) for model development, calibration, and evaluation. Fire behavior data can be gathered from operational prescribed fires or outdoor experimental fires (e.g., Stocks 1987, 1989) and wildfires (e.g., Alexander and Lanoville 1987). Both types of data sources have inherent limitations. The spectrum of fire environment conditions covered by experimental fire data normally does not include extreme fire weather conditions (e.g., high winds or severe drought). Furthermore, data garnered from wildfires is often incomplete or lacks detail and may not be reliable. Two different types of data were used in the present study: (1) fire behavior data in order to perform the predictive validation and then apply statistical tests and (2) weather and fuel complex data for use in the model sensitivity analysis and model behavior comparisons. The use of experimental fire behavior data would be appropriate in the current evaluation exercise given the generally high reliability that characterizes such data. Both the Cruz (1999) crown fire initiation and crown fire rate of spread models were developed using an extensive experimental fire dataset from published and unpublished sources in Canada supplemented with a few observations from Australia. Unfortunately no experimental crown fire behavior datasets are currently available for use in independently model testing. Consequently predictive validation and statistical tests will be based on wildfire data garnered from published cases studies (Alexander et al. 1983, Simard et al. 1983, Lanoville and Schmidt 1985, De Groot and Alexander 1986, Rothermel and Mutch 1986, Alexander and Lanoville 1987, Stocks 1987, Stocks and Flannigan 1987, Hirsch 1989, Alexander 1991). The use of such data will limit evaluation procedures applied to the Canadian FBP System fuel type specific models, as these were originally parameterized with data from some of those same documented wildfires.

The definition of fire environment scenarios used to characterize baseline data for sensitivity analysis and model comparison was based on well documented burning conditions where all relevant input variables were measured or acceptably estimated. This reliable and compatible data constitutes a benchmark dataset for which model behavior can be compared. For the sensitivity

Table 2—Baseline values used in the sensitivity analysis.

	Kenshoe Lake Exp. Fire 5 (Stocks 1987)	Lily Lake Fire (Rothermel 1983)	PFES Exp. Fire R1 (Van Wagner 1977)
10-m open wind speed (km/h)	29	37	15
Within stand wind speed (km/h)	-	-	5
1-hr TL FM (%)[1]	8	5	10
10-hr TL FM (%)[2]	9	6	11
100-hr TL FM (%)	10	7	12
Live woody FM (%)	75	75	-
FMC (%)	100	100	100
Surface fuel consumption (kg/m^2)	-	-	1 - 2

[1] The1-hr time lag (TL) fuel moisture (FM) content values were estimated according to the procedures described in (Rothermel 1983).
[2] The 10 and 100-hr TL FM content values were assigned values of plus one and two percent points of the value of the 1-hr TL FM as per Rothermel (1983).

analysis, two distinct fire scenarios were chosen: one conducive to marginal crown fire activity (i.e., Kenshoe Experimental Fire 5 documented by Stocks 1989) and the other conducive to extreme crown fire behavior (i.e., 1979 Lily Lake Fire as described by Rothermel 1983 and Alexander 1991). The use of two fire scenarios (table 2) is justified due to multiple interacting factors within the models being evaluated and possible non-linear effects in fire behavior.

The nature of the distinct modeling approaches that characterize the crown fire initiation models under evaluation with their distinct input requirements (table 1) constrains the type of predictive validation analysis and inter-model comparison that can be applied. The evaluation of the Van Wagner (1977) crown fire initiation model requires the monitoring of the transition period to estimate the critical fireline intensity, which is very difficult to accomplish in an experimental fire (Alexander 1998). The inter-model comparison for crown fire initiation models requires the use of a fire situation where the distinct input variables have been simultaneously measured. There is a scarcity of such data in the published literature, one of the limiting factors being the absence of data describing the vertical wind speed profile. The early experimental crown fires carried out at Petawawa Forest Experiment Station (PFES), Ontario, Canada (Van Wagner 1977) offers a complete description of the fuel complex and fire environment characteristics suiting the various model input requirements. Published data from PFES Experimental Fire R1 (Van Wagner 1968, 1977) suits the present data requirements and will be used in the evaluation of the various crown fire initiation models.

Sensitivity Analysis

By quantifying the effect of input variables, sub-models, and model parameters on model output, sensitivity analysis can: (1) expose model components that cause the smallest and largest changes in the model output and (2) assess the degree of uncertainty in the outputs that is associated with inaccurate input estimation. This identifies which input parameters or model components should be most accurately estimated given their influence on the behavior of the system. This is a relevant point in complex model systems, as the interaction between certain variables can induce large changes in the final result. A complete sensitivity analysis scheme should combine the effect of all model components combinations and interactions in a factorial design (Leemans 1991). The complexity of such a process has led to simplified sensitivity analysis schemes (e.g., Bevins and Martin 1978, Scott 1998a). Bartelink's (1998) relative sensitivity (RS) test was chosen for the present study. This parameter can be viewed as an index calculated from the partial derivative of output variables with respect to the perturbation of the input variable. This dimensionless result arises from the following criteria:

$$RS = \frac{V_{+10\%} - V_{-10\%}}{V_{def}\,0.2} \qquad [1]$$

where $V_{+10\%}$ and $V_{-10\%}$ are the resulting value of the critical parameter when the value of the parameter under analysis is changed by 10 percent and V_{def} is the resulting value of the critical parameter under default conditions. The value 0.2 is the relative range of the parameter to be analyzed. The 10% intervals were arbitrarily assigned. A RS score indicates the proportional response of the model to the changes in the perturbed input parameter. A sensitivity scale can be drawn from the results. RS scores less than 1 indicate insensitive (<0.5) or slightly sensitive (0.5 – 1) model responses to inputs; and RS scores larger than 1 indicate model sensitivity, which can be divided into moderate (1 – 2) and high (>2).

The various models were run under the two baseline conditions outlined in table 2 to cover the range of conditions over which crown fire behavior is expected to occur. Although the simplified sensitivity analysis scheme used does not take into account the interactions in a fully comprehensive manner as would be obtained if one were using a full factorial design, the computation of sensitivity scores for the two distinct burning conditions yielded an acceptable range of variability for the relative sensitivity scores. The RS scores computed for the crown fire initiation models (table 3) reflect the distinct modeling approaches that were followed. The non-dynamic nature of the Van Wagner (1977) crown fire initiation model results in no changes in RS scores between the high and very high fire environment severity conditions, with the model results indicating moderate sensitivity to changes in canopy base height (CBH) and foliar moisture content (FMC). Both Alexander (1998) and Van Wagner (1977) show the same sensitivity to canopy base height (CBH) variation, because of their similar formulations for the CBH effect. The sensitivity of the Van Wagner (1977) model to FMC seems to be excessive. This is a result of the pure theoretical formulation of the effect of FMC in increasing the heat sink of a fuel particle. The FMC effect formulation in the Alexander (1998) model is an application of the results of the Xanthopoulos and Wakimoto (1993) laboratory experiments on foliar heating relationships and yields lower sensitivity scores. A more complete analysis of the Van Wagner (1977) and Alexander (1998) crown fire initiation models, both of which are based on convective heat transfer theory, should include a link with a fire spread model for the estimation of fireline intensity, as done by Scott (1999). However, such analysis would include the errors inherent to such models for estimation of surface fire spread (Rothermel 1972, Albini 1976) and fireline intensity (Byram 1959, Andrews 1986), which would confound further analysis. The RS scores from the Cruz (1999) crown fire initiation model should be analyzed considering the shape of the cumulative probability curve that is the outcome of a logistic regression model. The higher magnitude RS scores are relative to the steepest component of the probability curve, which is indicative of transitional behavior, whereas the very low sensitivity values are relative to

Table 3—Relative sensitivity (RS) values associated with the crown fire initiation model outputs for the major input parameters.

Input parameters	Fire environment severity: High	Fire environment severity: Very high
Van Wagner (1977)		
CBH	1.5	1.5
FMC	1.3	1.3
Alexander (1998)		
CBH	1.5	1.5
FMC	1	0.4
Within stand wind speed	0.6	1.4
Flame front residence time	- 0.5	- 1.1
Cruz (1999)		
FSG	- 2.8	- 0.2
Fine dead fuel moisture	- 2.4	- 0.1
10-m open wind speed	2.6	0.1

the flatter regions of the curve, which is characteristic of low (< 0.15) or high (> 0.85) probability scores. This model produces the highest sensitivity scores, making it more prone to amplifying errors due to inaccurate assessment of fire environment input variables.

The crown fire rate of spread models analyzed can be more easily compared due to the commonality of outputs forms and most inputs. Comparisons reveal the models show distinct sensitivities to most of the input variables. The sigmoid equation used in the Canadian FBP System fuel type-specific models result in the same characteristic as the one described for the logistic crown fire initiation model of Cruz (1999). The FBP System fuel type-specific models are extremely sensitive (RS magnitudes between 5.6 to 2.2) to changes in input parameters on the steepest region of the sigmoid curve, which characterizes a transition from surface fire to crown fire spread (table 4). The very high RS scores for the FMC, seen only for FBP System Fuel Type C-6 (3.6 to 1.7), seem unusually high for an unsubstantiated relationship (Van Wagner 1998). Both Alexander (1998) and Cruz (1999) found no evidence for a significant FMC effect on crown fire spread rate. The Rothermel (1991) crown fire spread model responds to environmental changes in the same way as the Rothermel (1972) fire spread model. It shows very low sensitivity to changes in fine dead fuel moisture (RS of -0.2) and moderate sensitivity (RS between 1.3 and 1.4) to wind speed. The Cruz (1999) crown fire rate of spread model,

Table 4—Relative sensitivity (RS) values associated with crown fire rate of spread model outputs for the major input parameters.

	Fire environment severity	
Input parameters	**High**	**Very high**
	Rothermel (1991)	
Fine dead fuel moisture	- 0.2	- 0.2
10-m open wind speed [1]	1.4	1.3
	Forestry Canada Fire Danger Group (1992)	
FFMC[2]	- 5.6 / - 2.3	- 0.6 / - 0.2
10-m open wind speed	5.4 / 2.2	1.4 / 0.3
FMC	- 3.6	- 1.7
	Cruz (1999)	
Canopy bulk density	0.2	0.2
10-m open wind speed	0.9	0.9
Fine dead fuel moisture	- 1.4	- 0.9

[1] 10-m open wind speed was converted into 6-m (20 ft) wind speed by the 15 % adjustment factor as determined by Turner and Lawson (1978).
[2] FFMC = Fine fuel moisture code.

based on a large database of experimental crown fires, is an image of its dataset. This model's RS scores vary from very low for canopy bulk density (0.2) to moderate for fine dead fuel moisture (-1.4). Wind speed has a proportional effect on rate of spread in this model.

Predictive Validation

Predictive validation is applied here to determine: (1) the model adequacy in capturing the behavior of the real world system under study; and (2) if the model accuracy is suitable for its proposed application. Such tests should be done using independent, highly reliable data to decrease the probability of a Type I error (Sargent 1984), rejecting the validity of a valid model. All the high-intensity experimental fire behavior (either near or above the threshold of crowning) in the literature readily available to the authors was used to develop two of the models (Cruz 1999) under analysis. In order to simultaneously compare the various models under a common basis, independent, high-intensity wildfire data was utilized. The characteristics of wildfire derived data generally make it unsuitable for the evaluation of crown fire models (e.g., characteristics lack precise description of fuel complexes and fires spreading under the influence of extreme weather conditions). Consequently, a slightly different approach was followed in the evaluation of crown fire initiation models. The predictive validation of such models was based on the analysis of the model behavior under a well-documented fire situation where all input variables needed to characterize a fire scenario for the various models were either measured and / or acceptably estimated. The crown fire model input requirements from PFES Experimental Fire R1 (Van Wagner 1968, 1977) are presented in table 2. The comparison between models was based on the 10-m open wind speed requirements given variable vertical stratification in the fuel complex, with CBH values of 2, 4 and 6 m. For these fire environment scenarios, the critical fireline intensities for Van Wagner (1977) and Alexander (1998) models, respectively, are as follows: for CBH = 2 m, 475 and 540 kW / m; for CBH = 4 m, 1344 and 1290 kW / m; and for CBH = 6 m, 2470 and 2180 kW / m. The critical fireline intensities for Alexander (1998) model were based on a residence time of 45 seconds as observed in R1, and a constant of proportionality of 16 as determined for needlebed surface fuel complexes (Alexander 1998). The estimation of fireline intensity for use as input in these two models was based on the output of the two principal fire behavior prediction systems used in North America, the BEHAVE System (Andrews 1986) and the Canadian FBP System (Forestry Canada Fire Danger Group). The use of these two systems to predict a fundamental fire behavior descriptor highlights the potential error propagation problem when using the Byram's (1959) fireline intensity as an input variable for determining the requirements for crown fire initiation.

As pointed out previously, any comparison between the crown fire initiation models is hindered by differences in model formulations and the dependence of some of the models on fire behavior quantities that must be estimated in advance as inputs in the prediction. The error propagation

Table 5—The 10-m open wind speeds (km/h) required for crown fire initiation based on variation in CBH (m). Fixed burning conditions associated with PFES Experimental Fire R1 as described by Van Wagner (1968).

	Crown fire initiation models				
	Van Wagner (1977)		Alexander (1998)		Cruz (1999)
CBH (m)	BEHAVE	FBP C-6 [1]	BEHAVE	FBP C-6 [1]	
2	24 (8)[2]	1	27 (9)[2]	0	7
4	48 (16)	4	45 (15)	4	10.5
6	75 (25)	10	69 (23)	9	14.1

[1] FFMC = 92 and BUI = 70 (Alexander 1998).
[2] Mid-flame wind speeds as required for input in the surface fire spread model embodied in the BEHAVE System (Andrews 1986). Conversion from within stand (1.2-m) to 10-m open wind speed based on linear transformation using a 3.0:1 ratio as measured during R1 experimental fire (Van Wagner 1968, Alexander 1998).

problem becomes especially evident when comparing the 10-m open wind speed required to attain the fireline intensity requirements for crown ignition for the two surface fire spread models tested (table 5). Part of that error is produced by the manner in which fireline intensity is estimated and the distinct models used to estimate surface fire rate of spread. The wind speed requirements for crown ignition using the BEHAVE system seem to be unreasonably high at 48 km / h for a CBH of 4 m and 75 km / h for a CBH of 6 m (table 5), a problem that is also evident in Scott's (1998b) analysis and Scott and Reinhardt's (2001) simulations. In contrast, the use of the FBP System Fuel Type C-6 fire spread model yielded what seems to be very low wind speed requirements for crown ignition. The 1 and 4 km / h open wind conditions would result in almost no wind flow within the sub-canopy space, resulting in low intensity surface fires that would hardly induce crown combustion for the fuel complex simulated. These differences in response arise from the fact that the BEHAVE System estimates fireline intensity from the product of reaction intensity (quantity evaluated under no wind, no slope in a laboratory setting) with flame depth (Rothermel 1972, Albini 1976, Andrews 1986). This estimation procedure yields systematically lower values than the original Byram (1959) formulation that was used by Van Wagner (1977) to determine the empirical proportionality constant in his crown fire initiation model. Consequently this inflates the wind speed requirements needed to attain the critical fireline intensities. The Cruz (1999) crown fire initiation model appeared to yield reasonable results for the situations tested, with wind speed requirements varying between 7 and 14 km / h for the CBH range tested. For reference purposes, crowning associated with PFES Experimental Fire (CBH = 7 m) was attained with a wind speed of 15 km / h (Alexander 1998).

The two crown fire initiation models based on convective plume theory (Yih 1953) examined here (i.e., Van Wagner 1977, Alexander 1998) could be regarded as more sound conceptually than the Cruz (1999) model and presumably lead to a greater understanding of crown fire initiation phenomenology. However, some of the limiting assumptions concerning plume theory when applied to free-burning wildland fires and the dependence of Byram's (1959) fireline intensity to define the heat source potentially limits their use as a robust model. Further focus on convective plume theory in the development of new models of crown fire initiation may in fact stifle innovation. The relationship obtained by Yih (1953) through similarity analysis linking the temperature at a certain height above a linear heat source is technically restricted to still-air conditions, although Alexander's (1998) model has attempted to account for the cross-wind case. Neither Van Wagner (1977) nor Alexander (1998) attempted to account for the role of radiant heat flux in the onset of crowning from the flames that typically characterize high-intensity surface fires, although the authors readily acknowledged this shortcoming / possible limitation in their models.

Predictive validation applied to the crown fire spread models was based on a wildfire dataset derived from case studies. No Canadian FBP System model was tested here due to the non-independence of these models and the wildfire dataset used here. Although the published wildfire case studies provide general information on the crown fire runs, fuel types and weather conditions, detailed fuel complex descriptions and quantitative data generally do not exist. For evaluation of the Cruz (1999) crown fire rate of spread model, a nominal canopy bulk density value of 0.15 kg / m^3 was assigned on the basis of knowledge and experience with experimental fires in generally similar fuel types. All the fires were also assumed to be spreading as active crown fires. This assumption is corroborated by the high spread rates observed on the wildfires

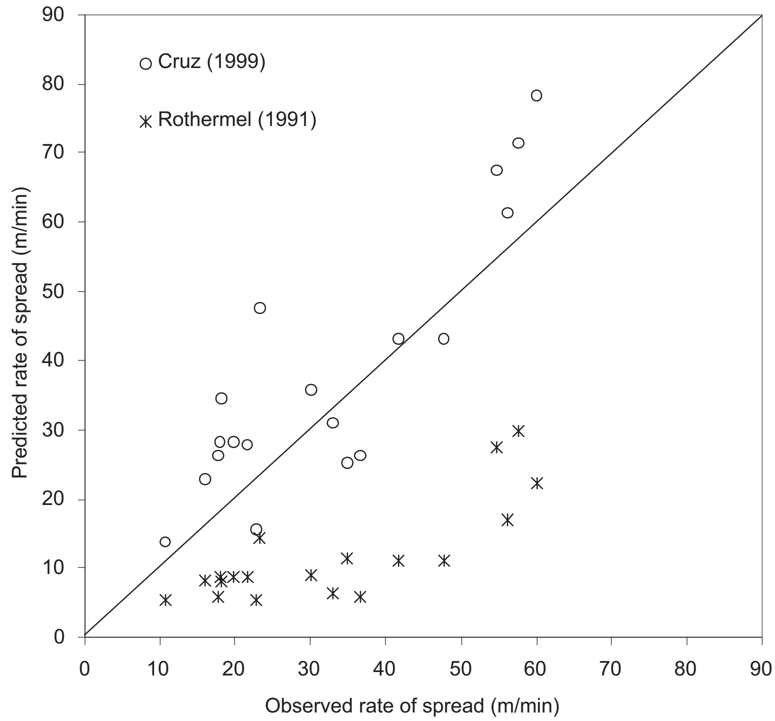

Figure 1—Plot of observed versus predicted rate of spread of crown fires in the Rothermel (1991) and Cruz (1999) crown fire spread models.

selected for evaluation purposes. Figure 1 displays a scatterplot of observed versus predicted rate of spread produced by the Cruz (1999) and Rothermel (1991) crown fire rate of spread models. The results suggest strong under-prediction trends for the Rothermel (1991) model and an acceptable agreement, albeit an over-prediction trend for the Cruz (1999) model. The over-prediction trend of the Cruz (1999) model might arise from the worst-case scenario assumed for the crown fire run simulations that extended for several hours. These scenarios use the lower fine fuel moisture content computed for the fire run, whereas a more detailed fire simulation encompassing fine fuel moisture and wind speed variability over the burning period would probably reduce this tendency to over-predict. Hence we are not certain that the over-prediction trend evident in figure 1 is the result of model bias or the inadequacy of the test data / approach used to replicate real-world conditions. A cursory examination of the scatterplot also reveals an inability for the Rothermel (1991) model to predict high rates of spread for many of the situations considered (i.e., for many of the wildfires the model seldom predicted a spread rate greater than 10 m / min while the observed spread rates varies from 10 to nearly 50 m / min). In order to quantify the adequacy of the model's behavior, two deviance measures were sought: the mean absolute error (MAE) and the mean absolute percent error (MA%E) (Mayer and Butler 1993, Cruz 1999). For the dataset utilized in the present analysis, the mean absolute errors computed were 20 m / min (and MA%E of 62%) for the Rothermel (1991) model and 9.2 m / min (and MA%E of 34%) for the Cruz (1999) crown fire spread model.

Comparing Two Models

The statistical validation procedures complement some of the quantitative results obtained from the descriptive analyses previously performed on the models. Nevertheless, the definition of any statistical validation criteria is hampered by the difficulty in defining adequate tests and appropriate confidence levels for the phenomena under study. Different tests might produce conflicting results, accepting or rejecting the same hypothesis simultaneously. The 0.05 alpha levels commonly accepted for statistical significance in a variety of natural resources studies might not be adequate to analyze phenomena that may vary several orders of magnitude. The previously described limitations in the datasets used here that prevent an independent analysis of the crown fire initiation models also restrict the application of statistical tests to the crown fire rate of spread models. Using the wildfire dataset employed in the Predictive Validation section, the models were analyzed for: (1) their modeling efficiency; (2) linear regression parameters; and (3) simultaneous F-test for slope = 1 and intercept = 0 (Draper and Smith 1981, Mayer et al. 1994). Modeling efficiency, EF, is expressed as follows (from Mayer and Butler 1993):

$$EF = 1 - \frac{\sum (y_i - \hat{y}_i)^2}{\sum (y_i - \bar{y}_i)^2} \qquad [2]$$

where y_i is the observed and $\hat{y}_i$ is the predicted value. This measure provides an indication of goodness of fit, with an upper bound of 1 describing a perfect fit and values less than 0 indicating poor model performance; see Mayer and Butler (1993) for further interpretation of EF. The simultaneous F-test for slope = 1 and intercept = 0 evaluates the null hypothesis, H_0:(b_0,b_1)=(0,1), by the following statistic: $Q=(\beta - b)^{1'}X'X(\beta - b)$, where β is the population parameters to be tested, X'X the matrix term in the independent variable, and b is the vector of regression parameters. The null hypothesis is accepted if $Q \le ps^2F(p,v,1-\alpha)$, where p is the regression degrees of freedom, s^2 is the variance, and v is n – p.

The application of the two crown fire rate of spread models under comparison to the crown fire dataset (see references to the wildfire case studies in Data Validation) produced EF values of –0.14 and 0.68 (table 6) for the Rothermel (1991) and Cruz (1999) models respectively. The slope coefficients (b_1) resulting from the regression analysis (table 6) reflect the slight over-prediction trend of the Cruz (1999) model and the strong under-prediction trend of the Rothermel (1991) model. For both models the simultaneous F-test for slope and intercept resulted in the rejection of the null hypothesis (both $Q > ps^2F(p,v,1-\alpha)$). Given the uncertainty in the input conditions for the wildfire runs and the type of phenomena being analyzed, with rates of spread varying over three orders of magnitude, the results from the F-test should be analyzed with caution, as this test might be too restrictive for the phenomena under study (i.e., the null hypothesis with slope = 1 and intercept = 0, is easily rejected).

Table 6—Statistical validation parameters for the Rothermel (1991) and Cruz (1999) crown fire rate of spread models.

		Linear regression				
Model	**EF**	**R^2**	**β_0 (lower/upper 95%)**	**β_1 (lower/upper 95%)**	**Q**	***$ps^2F(p,v,1-\alpha)$***
Rothermel (1991)	-0.14	0.63	12.0 (2.52 / 21.5)	1.75 (1.06 / 2.44)	9231	441
Cruz (1999)	0.68	0.75	4.5 (-4.45 / 13.46)	0.74 (0.52 / 0.95)	1087	283

Concluding Remarks

Models are simplified theories developed to approximate the behavior of real world systems. The process of model evaluation acquires particular importance when the models are used to support decision-making. This shifts the emphasis on validation procedures to the potential applications of these models instead of the model itself (Mayer and Butler 1993). There exists a vast list of validation techniques available to evaluate models but hardly a set appropriate for widespread application covering different modeling approaches. The present set of evaluation procedures was developed to provide information about model performance and how well it replicates the behavior of the real-world system. The theoretical basis of the models evaluated and their distinct modeling approaches make them difficult to compare. The problem of model conceptual validity is one of the most challenging issues. The unknowns in certain fundamental fire behavior processes make it difficult to define which theories are most appropriate. The results of the sensitivity analysis showed how the models respond differently to the various input variables dominating some of the processes involved in crown fire initiation and spread. The main differences were found in the way foliar moisture content affects the susceptibility of canopy ignition. The theoretical model from Van Wagner (1977) is the most sensitive to FMC, followed by the Alexander (1998) application of Xanthopoulos and Wakimoto's (1993) laboratory experiments, which assumes that the duration of heating can be equated to flame front residence time. The Cruz (1999) crown fire initiation model does not incorporate the effect of FMC, reflecting the non-significance of this parameter in the initiation of crown fires within the dataset used in its development. Fuel complex vertical stratification appears to manifest a comparable effect on the various models for crown fire initiation. The various crown fire rate of spread models respond similarly to changes in wind speed, but quite differently to fine fuel moisture content. The Rothermel (1991) model is relatively insensitive to the variation in this parameter whereas the Canadian FBP System models are over-sensitive. This evaluation of model sub-components highlights some areas needing further research and acquisition of new laboratory and field data. Combustion characteristics of live fuels are poorly understood and the effect of foliar moisture content as a heat sink has never been evaluated under heat flux conditions characteristic of wildfire situations. The effect of CBH on heat flux decay in the Van Wagner (1977) and Alexander (1998) crown fire initiation models is solely based on convective heating and no allowance is presently made for a radiative contribution. One would expect that a surface fire burning just below the threshold for crowning will possess an ample flame front depth, leading to a strong effect of the radiative component in heating canopy fuels. Further refinement of existing crown fire initiation models is required in order to accommodate the contribution that radiation plays in the onset of crowning.

The present study has also highlighted the lack of published fire behavior data that can be used in model evaluation. Although the growing complexity of fire management decision-making relies on the extensive use of fire behavior models, there were virtually no studies carried out in the United States that produced suitable data that could be used in the evaluation and calibration of models or systems used for predicting crown fire behavior.

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References

Albini, F. A. 1976. Estimating wildfire behavior and effects. Gen. Tech. Rep. INT-30. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 92 p.

Albini, F. A.; Stocks, B. J. 1986. Predicted and observed rates of spread of crown fires in immature jack pine. Combustion Science and Technology. 48: 65-76.

Alexander, M. E. 1988. Help in making crown hazard assessments. In Fischer, W. C.; Arno, S. F., comps. Protecting people and homes from wildfire in the Interior West: proceedings of symposium and workshop; 1987 October 6-8; Missoula, MT. Gen. Tech. Rep. INT-251. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 147-156.

Alexander, M .E. 1991. The 1985 Butte Fire in central Idaho: a Canadian perspective on the associated burning conditions. In: Nodvin, S. C.; Waldrop, T. A., eds. Fire and the environment; ecological and cultural perspectives. Proceedings of an international symposium; 1990 March 20-24; Knoxville, TN. Gen. Tech. Rep. SE-69. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station: 334-343.

Alexander, M. E. 1998. Crown fire thresholds in exotic pine plantations of Australia. Canberra, Australia: Australian National University. 228 p. Ph.D. Thesis.

Alexander, M. E.; Janz, B.; Quintilio, D. 1983. Analysis of extreme wildfire behavior in east-central Alberta: a case study. In: Postprint volume, seventh conference on fire and forest meteorology. Boston, MA: American Meteorological Society: 38-46.

Alexander, M. E.; Lanoville, R. A. 1987. Wildfires as a source of fire behavior data: a case study from Northwest Territories, Canada. In: Fujioka, F.; Rice, C., comps. Postprint volume, ninth conference on fire and forest meteorology; 1987 April 21-24, San Diego, CA. Boston, MA: American Meteorological Society: 86-93.

Alexander, M. E.; Lanoville, R. A. [In press]. The international crown fire modeling experiment fuel treatment trials. In: Engstrom R. T.; de Groot, W. J., eds. Proceedings of the 22nd Tall Timbers fire ecology conference: fire in temperate, boreal, and montane ecosystems. Tallahassee, FL: Tall Timbers Research Station.

Alexander, M. E.; Quintilio, D. 1990. Perspectives on experimental fires in Canadian forestry research. Mathematical Computer Modelling. 13(12): 17-26.

Alexander, M. E.; Stocks, B. J.; Lawson, B. D. 1991, Fire behavior in black spruce-lichen woodland: the Porter Lake project. Inf. Rep. NOR-X-310. Edmonton, Alberta: Forestry Canada, Northwest Region, Northern Forestry Centre. 44 p.

Alexander, M. E.; Yancik, R. F. 1977. The effect of precommercial thinning on fire potential in a lodgepole pine stand. Fire Management Notes. 38(3): 7-9.

Andrews, P. L. 1980. Testing the fire behavior model. In: Martin, R. E.; Edmonds, R. L.; Faulkner, D. A.; Harrington, J. B.; Fuquay, D. M.; Stocks, B. J.; Barr, S., eds. Proceedings of sixth conference on fire and forest meteorology; 1980 April 22-24; Seattle, WA. Washington, DC: Society of American Foresters: 70-77.

Andrews, P. L. 1986. BEHAVE: fire behavior prediction and fuel modeling system—BURN subsystem, part 1. Gen. Tech. Rep. INT-260. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 130 p.

Bacsi, Z.; Zemankovics, F. 1995. Validation: an objective or a tool? Results on a winter wheat simulation model application. Ecological Modelling. 81: 251-263.

Bartelink, H. H. 1998. A model of dry matter partitioning in trees. Tree Physiology. 18: 91-101.

Bevins, C. D. 1976. An evaluation of the slash fuel model of the 1972 national fire danger rating system. Seattle, WA: University of Washington. 104 p. Thesis.

Bevins, C. D.; Martin, R .E. 1978. An evaluation of the slash (I) fuel model of the 1972 national fire danger rating system. Res. Pap. PNW-115. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 17 p.

Bossel, H. 1991. Modelling forest dynamics: moving from description to explanation. Forest Ecology and Management. 42: 129-142.

Brown, J. K. 1972. Field test of rate of fire spread in slash fuels. Gen. Tech. Rep. INT-116. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 24 p.

Brown, J. K. 1982. Fuel and fire behavior in big sagebrush. Gen. Tech. Rep. INT-290. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 10 p.

Byram, G. M. 1959. Combustion of forest fuels. In: Davis, K. P., ed. Forest fire: control and use. New York: McGraw Hill Book Co.: 61-89.

Catchpole, E. A.; de Mestre, N. 1986. Physical models for a spreading line fire. Australian Forestry. 49(2): 102-111.

Cohen, J. D. 1990. Research has part of the answer. In: Are forests the answer? Proceedings of the 1990 Society of American Foresters national convention. SAF Publ. 90-02. Bethesda, MD: Society of American Foresters: 117-120.

Countryman, C. M. 1974. Can southern California wildland conflagrations be stopped? Gen. Tech. Rep. PSW-7. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station. 11 p.

Cruz, M. G. 1999. Modeling the initiation and spread of crown fires. Missoula: University of Montana. 162 p. Thesis.

De Groot, W. J.; Alexander, M. E. 1986. Wildfire behavior on the Canadian shield: a case study of the 1980 Chachukew Fire, east central Alberta. In: Proceedings of the third Central Region Fire Weather Committee scientific and technical seminar. Edmonton, AB: Forestry Canada, Northwest Region, Northern Forestry Centre. 23-45.

Draper, N. R.; Smith, H. 1981. Applied regression analysis. New York: John Wiley & Sons, Inc. 407 p.

Fahnestock, G. R. 1968. Fire hazard from precommercial thinning of ponderosa pine. Res. Pap. PNW-57. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 16 p.

Fiedler, C. E.; Keegan, C. E., III; Woodall, C. W.; Morgan, T. A; Robertson, S. H.; Chmelik, J. T. 2001. A strategic assessment of fire hazard in Montana. Report submitted to the Joint Fire Sciences Program, Boise, ID. 39 p. [Online]. Available at: http://www.nifc.gov/joint_fire_sci/ummontanarpt.pdf. [March 2002].

Finney, M. A. 1998. FARSITE: Fire area simulator—model development and evaluation. Res. Pap. RMRS-RP-4. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 47 p.

Forestry Canada Fire Danger Group. 1992. Development and structure of the Canadian forest fire behavior prediction system. Inf. Rep. ST-X-3. Ottawa, ON: Forestry Canada, Science and Sustainable Development Directorate. 65 p.

Fulé, P. Z.; Waltz, A. E. M.; Convigton, W. W.; Heinlein, T. A. 2001. Measuring forest restoration effectiveness in reducing hazardous fuels. Journal of Forestry. 99(11): 24-29.

Gould, J. S. 1991. Validation of the Rothermel fire spread model and related fuel parameters in grassland fuels. In: Cheney, N. P.; Gill, A. M., eds. Proceedings of conference on bushfire modeling and fire danger rating systems; 1988 July 11-12; Canberra. Yarralumla, ACT: CSIRO Division of Forestry: 51-64.

Gould, J .S.; Cheney, N. P.; McCaw, L. 2001. Project Vesta—research into the effects on fuel structure and fuel load on behavior of moderate to high-intensity fires in dry eucalypt forest: progress report. In Pearce, G.; Lester, L., tech. coords. Bushfire 2001 proceedings, Australian bushfire conference; 2001 July 3-6; Christchurch, New Zealand. Rotorua, New Zealand: New Zealand Forest Research Institute: 13-21.

Grabner, K. W.; Dwyer, J .P.; Cutter, B. E. 2001. Fuel model selection for BEHAVE in midwestern oak savannas. Northern Journal of Applied Forestry. 18: 74-80.

Grishin, A. M. 1997. Mathematical modeling of forest fires and new methods of fighting them. Tomsk, Russia, Tomsk State University. 390 p.

Hirsch, K. G. 1989. Analysis of the fire behavior associated with three 1988 spring wildfires in Central Canada. In: Maciver, D. C.; Auld, H.; Whitewood, R., eds. Proceedings of the 10th conference on fire and forest meteorology; 1989 April 17-21; Ottawa, ON: Forestry Canada and Environment. 416-425.

Hirsch, K. G.; Pengelly, I. 1999. Fuel reduction in lodgepole pine stands in Banff National Park. In Neuenschwander, L .F.; Ryan, K. C., eds. Proceedings of joint fire science conference and workshop; 1999 June 15-17; Boise, ID. Boise: University of Idaho and International Association of Wildland Fire: 251-256

Hough, W. A.; Albini, F. A. 1978. Predicting fire behavior in palmetto-gallberry fuel complexes. Res. Pap. SE-174. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station. 49 p.

Kalabokidis, K. D.; Omi, P. N. 1998. Reduction of fire hazard through thinning/ residue disposal in the urban interface. International Journal of Wildland Fire. 8(1): 29-35.

Kilgore, B. M.; Sando, R. W. 1975. Crown-fire potential in a sequoia forest after prescribed burning. Forest Science. 21: 83-87.

Lanoville, R.A.; Schmidt, R. E. 1985. Wildfire documentation in the Northwest Territories; a case study of Fort Simpson 40-1983 wildfire. In: Proceedings of the second Western Region Fire Weather Committee scientific and technical seminar. Edmonton, AB: Forestry Canada, Northwest Region, Northern Forestry Center: 17-22.

Lawson, B. D. 1972. Fire spread in lodgepole pine stands. Missoula, MT: University of Montana. 110 p. Thesis.

Leemans, R. 1991. Sensitivity analysis of a forest succession model. Ecological Modelling. 53: 247-262.

Linn, R. R. 1997. Transport model for prediction of wildfire behavior. Los Alamos National Laboratory Scientific Report LA13334-T. 195 p.

Marsden-Smedley, J.; Catchpole, W. R. 1995. Fire behaviour modelling in Tasmanian buttongrass moorlands. II. Fire behaviour. International Journal of Wildland Fire. 5(4): 215-228.

Mayer, D. G.; Butler, D. G. 1993. Statistical validation. Ecological Modelling. 68: 21-32.

Mayer, D. G.; Stuart, M. A.; Swain, A. J. 1994. Regression of real world data on model output: an appropriate overall test of validity. Agricultural System. 45: 93-104.

McAlpine, R. S.; Hobbs, M. W. 1994. Predicting the height to live crown base in plantations of four boreal forest species. International Journal of Wildland Fire. 4(2): 103-106.

Norum, R. A. 1982. Predicting wildfire behavior in black spruce forests in Alaska. Res. Pap. PNW-401. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 10 p.

Oderwald, R. G.; Hans, R. P. 1993. Corroborating models with model properties. Forest Ecology and Management. 62: 271-283.

Ottmar, R. D.; Vihnanek, R. E.; Wright, C. S. 1998. Stereo photo series for quantifying natural fuels. Vol. I: Mixed-conifer with mortality, western juniper, sagebrush, and grassland types in the Interior Pacific Northwest. PMS-830. NFES-2580. Boise, ID: National Wildfire Coordinating Group, National Interagency Fire Center. 73 p.

Pollet, J.; Omi, P. N. 2002. Effect of thinning and prescribed burning on crown fire severity in ponderosa pine forests. International Journal of Wildland Fire. 11(1): 1-10.

Rothermel, R. C. 1972. A mathematical model for predicting fire spread in wildland fuels. Res. Pap. INT-115. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 40 p.

Rothermel, R. C. 1983. How to predict the spread and intensity of forest and range fires. Gen. Tech. Rep. INT-143. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 161 p.

Rothermel, R. C. 1991. Predicting behavior and size of crown fires in the Northern Rocky Mountains. Gen. Tech. Rep. INT-438. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 46 p.

Rothermel, R. C.; Mutch, R. W. 1986. Behavior of the live-threatening Butte Fire: August 27-29, 1985. Fire Management Notes. 47(2): 14-24.

Rothermel, R. C.; Rinehart, G. C. 1983. Field procedures for verification and adjustment of fire behavior predictions. Res. Pap. INT-142. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 25 p.

Rykiel, E. J. 1996. Testing ecological models: the meaning of validation. Ecological Modelling. 90: 229-244.

Sando, R. W.; Wick, C. H. 1972. A method for evaluating crown fuels in forest stands. Res. Pap. NC-84. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest and Range Experiment Station. 26 p.

Sargent, R. G. 1984. A tutorial on verification and validation of simulation models. In: Sheppard, S.; Pooch, U.; Pegden, D. eds. Proceedings of the 1984 winter simulation conference. IEEE 84CH2098-2: 115-121.

Scott, J. H. 1998a. Sensitivity analysis of a method for assessing crown fire hazard in the Northern Rocky Mountains, USA. In: Viegas, D. X., ed. Proceedings of 3rd international conference on forest fire research—14th conference on fire and forest meteorology; 1998 November 16-20; Luso-Coimbra, Portugal. ADAI, University of Coimbra: 2517-2531.

Scott, J. H. 1998b. Fuel reduction in residential and scenic forests: a comparison of three treatments in a western Montana ponderosa pine stand. Res. Pap. RMRS-RP-5. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 19 p.

Scott, J. H. 1999. NEXUS: a system for assessing crown fire hazard. Fire Management Notes. 59(2): 20-24.

Scott, J. H.; Reinhardt, D. 2001. Assessing crown fire potential by linking models of surface and crown fire behavior. Res. Pap. RMRS-RP-29. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 59 p.

Simard, A. J.; Haines, D. A.; Blank, R. W.; Frost, J. S. 1983. The Mack Lake Fire. Gen. Tech. Rep. NC-83. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station. 36 p.

Sneeuwjagt, R. J.; Frandsen, W. H. 1977. Behavior of experimental grass fires vs. predictions based on Rothermel's fire model. Canadian Journal of Forest Research. 7: 357-367.

Stephens, S. L. 1998. Evaluation of the effects of silvicultural and fuels treatments on potential fire behavior in Sierra Nevada mixed conifer forests. Forest Ecology and Management. 105: 21-35.

Stocks, B. J. 1987. Fire behavior in immature jack pine. Canadian Journal of Forest Research. 17: 80-86.

Stocks, B. J. 1989. Fire behavior in mature jack pine. Canadian Journal of Forest Research. 19: 783-790.

Stocks, B. J.; Flannigan, M. D. 1987. Analysis of the behavior and associated weather for a 1986 northwestern Ontario wildfire: Red Lake Number 7. In Fujioka, F.; Rice, C., comps. Postprint volume, ninth conference on fire and forest meteorology; 1987 April 21-24; San Diego, CA. Boston, MA: American Meteorologial Society: 94-100.

Trevitt, A.C.F. 1991. Weather parameters and fuel moisture content: standards for fire model inputs. In: Cheney, N. P.; Gill, A. M., eds. Proceedings of conference on bushfire modeling and fire danger rating systems; 1988 July 11-12; Canberra. Yarralumla, ACT: CSIRO Division of Forestry: 157-166.

Turner, J. A.; Lawson, B. D. 1978. Weather in the Canadian forest fire danger rating system: a user guide to national standards and practices. Inf. Rep. BC-X-177. Victoria, BC: Environment Canada, Canadian Forestry Service, Pacific Forest Research Centre. 40 p.

Van Wagner, C. E. 1968. Fire behaviour mechanisms in a red pine plantation: field and laboratory evidence. Publ. 1229. Canadian Department of Forestry and Rural Development. 30 p.

Van Wagner, C. E. 1977. Conditions for the start and spread of crown fire. Canadian Journal of Forest Research. 7: 23-34.

Van Wagner, C. E. 1987. Development and structure of the Canadian forest fire weather index system. Forest Tech. Rep. 35. Ottawa, ON: Canadian Forest Service. 34 p.

Van Wagner, C. E. 1998. Modelling logic and the Canadian forest fire behavior prediction system. Forestry Chronicle. 74(1): 50-52.

van Wagtendonk, J. W. 1996. Use of a deterministic fire growth model to test fuel treatments. In: Sierra Nevada ecosystem project: final report to Congress. Vol II. Assessements and scientific basis for management options. Davis: University of California, Centers for Water and Wildland Resources: 1155-1165.

van Wagtendonk, J. W.; Botti, S. J. 1984. Modeling behavior of prescribed fires in Yosemite National Park. Journal of Forestry. 82(8): 479-484.

Van Wilgen, B. W.; Wills, A. J. 1988. Fire behaviour prediction in savanna vegetation. South African Journal of Wildland Research. 18(2): 41-46.

Weatherspoon, C. P.; Skinner, C N. 1995 An assessment of factors associated with damage to tree crowns from the 1987 wildfires in Northern California. Forest Science. 41(3): 430-451.

Xanthopoulos, G.; Wakimoto, R .H. 1993. A time to ignition—temperature—moisture relationship for branches of three western conifers. Canadian Journal of Forest Research. 23: 253-258.

Yih, C. S. 1953. Free convection due to boundary sources. In: Proceeding of the first symposium on the use of models in geophysics. Washington, DC: U.S. Government Printing Office: 117-133.

In Situ Soil Temperature and Heat Flux Measurements During Controlled Surface Burns at a Southern Colorado Forest Site

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Abstract—*This study presents in situ soil temperature measurements at 5-6 depths and heat flux measurements at 2-5 depths obtained during the fall/winter of 2001/2002 at seven controlled (surface) fires within a ponderosa pine forest site at the Manitou Experimental Forest in central Colorado. Six of these burns included three different (low, medium, and high) fuel loadings under both a closed-canopy forested site and an open forest with a grassy meadow understory. The fuel loading for the seventh burn was a conical pile of slash about 6 m in height and 9 m in diameter and was intended to duplicate the structure and loading of a slash pile resulting from mechanical harvesting activities. One basic purpose of this initial experiment was to assess how well some commercially available soil heat flux plates would perform at high temperatures. The data presented here include soil temperatures, heat fluxes, and depth and duration of the thermal energy penetration into the soils. The maximum surface heat fluxes were estimated to be about 2400 Watts/meter^2 [Wm^{-2}] at the slash pile burn site, 2300 Wm^{-2} at the high fuel meadow site, and 3000 Wm^{-2} at the high fuel forested site. Extrapolated surface temperatures are about 436 C at the slash burn site, 359 C at the high fuel meadow site, and 95 C at the high fuel forested site. Recovery of a normal daily temperature cycle depended on fire duration and fuel loading. The recovery times were between 16 and 20 hours at the high fuel sites, about half this time at the medium fuel sites, and less that 2 hours at the low fuel sites. However, the recovery time at the slash pile site was about 2 weeks. Although further tests and refinements are planned, the present results suggest not only that soil heat flux can be reliably measured during controlled burns, but that soil temperatures and heat flux can differ significantly with different fuel loadings.*

Introduction

Both natural and prescribed fires play important roles in managing and maintaining most ecosystems in the western United States. In many ecosystems, fire is the major cause of disturbance and change. One of the less visually obvious changes caused by fire, even a relatively small fire, is the heat pulse and associated high temperatures that penetrate the soil. High soil temperatures influence forests and their ability to regenerate after a fire by altering soil properties; killing soil microbes, plant roots, and seeds; destroying soil organic matter; and altering soil nutrient and water status and soil nutrient cycling (Frandsen and Ryan 1986; Hungerford et al. 1991; Campbell et al. 1995; DeBano et al 1998). Some consequences of fire can be subtle and long term (Sackett and Haase 1992; DeBano et al 1998), while others, such as increasing soil erosion and the concomitant effects on water quality and the hydrological cycle, are more immediate and obvious. Because fire is frequently used by land managers to reduce surface fuels, it is important to know how

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soil properties and fuel conditions interact to determine the nature and extent of the soil heat pulse. Ideally, quantifying this coupling between surface fuels and the soil heat pulse requires both observational and modeling studies (e.g., Steward et al 1990), for which both soil temperature and soil heat flux data are necessary for understanding the physical processes governing heat propagation through soils and for verifying model performance. Ultimately, such a combined approach should help managers to maximize the ecosystem benefits of a prescribed fire while minimizing the potential for harm.

This report presents and discusses the initial results of a larger experiment to evaluate how different amounts and geometrical arrangements of fuel loadings influence forest regeneration after fires. Here we specifically detail in situ measurements of soil temperature and heat flux taken during seven controlled burns in Manitou Experimental Forest in central Colorado. The primary motivation for these initial burn experiments was to assess the ability of two commercially available soil flux heat plates to operate at the high temperatures encountered during surface fires. This is achieved in this report by examining the data for internal consistency and by developing a simple method of extrapolating the measured soil temperature and heat flux data to surface values during the fires.

Experimental Design

Site Location and Description

Manitou Experimental Forest is located in Teller County, central Colorado. The site latitude and longitude are 39 ° 04' North and 105 ° 04' West with an elevation of about 2400 m. Vegetation at the burn areas is predominantly ponderosa pine with an understory of bunchgrasses with numerous grassy openings throughout the area. Soils at the burn areas were either Boyett-Frenchcreek-Pendant associated with 15 to 40 percent slopes with a typical bulk density of 1.20-1.35 gm cm^{-3} and an air permeability of about 12 ± 6 (10^{-12}) m^2 or Pendant cobbly loam also occuring on 15 to 40 percent slopes with a bulk density of 1.30-1.70 gm cm^{-3} and air permeability of about 4 ± 2 (10^{-11}) m^2 (USDA and other agencies 1986). Except for the few soils that are derived from red arkosic sandstone, all soils in the area are derived from biotite granite and associated igneous rocks of the Pikes Peak batholith. Annual precipitation on the forest is about 400 mm and the annual mean ambient temperature is about 5 C.

Fuel Loadings and Sensor Deployment

Three sites were instrumented and then amended by adding slash of various densities. The first two sites, consisting of three plots each, were instrumented on March 22, 2001, on the Boyett-Frenchcreek-Pendant type soil. Three plots with low, medium, and high fuel loadings were under a closed-canopy forest (forest site) and the other three plots were within a grassy meadow associated with a more open forest (meadow site), created as a result of a timber harvest two years prior. The forest site was about 130 m west of the meadow site and the plots within each site were adjacent to one another and averaged about 70 m^2. These 6 sites were amended on April 5, 2001, and the controlled burn occured November 9, 2001. The third or slash burn site consisted of one plot and was instrumented on October 12, 2001, amended on October 18, 2001, and burned on January 11, 2002. It was about 200 m east of the meadow site. The slash pile was conical in shape and approximately 9 m in diameter and 6 m in height. The soil type at this burn site was Pendant cobbly loam and it was located within a very large grassy meadow that had been forested two years prior. The forest and meadow burn sites were on south facing slopes of small 2-3 m deep draws. The fires were initiated (as much as possible) at the bottom of the slope. The instruments were inserted horizontally and buried in the soils at relatively more horizontal areas near the top of the slope. The slash pile site was on a gentle west facing slope. Table 1 gives the details on the fuel loadings at each site for pre- and post-burn conditions.

Table 1—Fuel loadings for controlled burns at Manitou Experimental Forest. The first number in each entry is the preburn data and the second is postburn data. NA = Not Available.

Site (loading)	Fuel depth (cm)	Duff depth (cm)	Litter depth (cm)	Total loading (tonsM/hectare)
Meadow (low)	5.0/0.0	1.4/0.0	5.0/0.0	8.80/0.0
Meadow (med)	19.5/2.5	0.63/0.25	5.0/2.5	23.1/7.60
Meadow (high)	30.5/0.85	0.38/0.00	9.3/0.85	68.6/7.94
Forest (low)	5.0/3.4	0.51/0.08	5.0/2.5	1.44/0.78
Forest (med)	22.0/3.4	0.25/0.08	3.4/2.5	15.0/7.11
Forest (high)	21.2/2.5	0.25/0.00	3.4/2.5	31.8/14.4
Slash pile	600[a]/0	NA[b]/NA	NA[b]/NA	560[a]/0

[a] Estimate.
[b] Expected to be similar to average of meadow sites above.

All data were logged on Campbell Scientific CR23X's (Campbell Scientific Inc., Logan, UT). All wires connecting the sensors to the data logger were laid in a trench which was then backfilled (as much as possible) with the original soil. The data loggers were located 10 to 30 m from the sensors. All data loggers were enclosed in a metal fire resistance box. However, at the slash burn site the data logger box was also buried to ensure additional protection from the fire. Data sampling rates before and well after the fires was 1 Hz with average data recorded every half hour. Between 2 and 4 hours before, during, and for about 4 days after the fire, data were sampled at 4 Hz and recorded every 15 s. Table 2 gives the type, number, and depth of the temperature and heat flux sensors. To minimize interference between the sensors, the soil heat flux plates and soil temperature probes were separated laterally by a few centimeters.

On the morning of the fires the moisture content of the litter layers, duff layers, and the uppermost few centimeters of soil were measured at the six forest and grassy meadow sites. The litter layers were 15 percent by weight (meadow) and 23 percent by weight (forest). The duff layer was 33 percent by weight at the meadow sites and 14 percent by weight at the forest sites. Finally, the soil moisture content was about 3 percent by weight at the meadow site and about 2 percent by weight at the forest site. No fuel or soil moisture data were obtained at the slash burn site.

Potential Concerns

Before presenting the observed data it is important to bear in mind the context and the intent of this experiment. All burn sites are representative of slash left after harvesting in the area. Therefore, we would expect the present data are quite typical of many prescribed surface fires within the Manitou

Table 2—Placement and type of soil temperature and heat flux sensors for controlled burns at Manitou Experimental Forest. Here T means a temperature sensor and G_R and G_T are soil heat flux plates. Note: all thermocouple junctions were coated with epoxy (Omegabond 101) for electrical isolation.

	3 closed canopy sites and 3 grassy meadow sites		Large slash pile	
Depth (m)	**Sensor type**		**Sensor type**	
0.02	T[a]	G_R[d]	T[a]	
0.05	T[b]		T[a]	G_T[f]
0.10	T[c]	G_R[d]	T[b]	G_T[f]
0.15	T[c]			
0.30	T[c]	G_R[d,e]	T[c]	G_T[f]
0.50			T[c]	G_R[d]
1.365			T[c]	G_R[d]

[a] Omega Engineering, Stamford, CT: model no. HH-K-24, rated 704 C.
[b] Omega Engineering, Stamford, CT: model no. TT-J-24, rated 260 C.
[c] Omega Engineering, Stamford, CT: model no. TT-T-24, rated 200 C.
[d] Radiation Energy Balance Systems, Seattle, WA: model no. HFT 3.1.
[e] Not included at the low fuel loading sites.
[f] Thermonetics Corporation, La Jolla, CA: glazed ceramic design, rated to 775 C, nominal sensor sensitivity 1250 to 1750 $Wm^{-2}mV^{-1}$, cromel extension wire to data logger: Omega Engineering TFCH-020, temperature rated to 260 C. Note: post-burn examination of extension wire did not reveal any high temperature damage.

Experimental Forest and surrounding areas. However, fuel loadings for wildfires could be quite different and could vary considerably depending primarily on recent fire history and amount of slash from previous logging activities.

Other considerations and concerns involve the measurement of soil heat flux and possible alterations or disruptions of the soil. For example, reliable estimates of soil heat flux require good contact between the soil and the heat flux plate. Any moisture present in the soil will also help maintain this contact. However, when soil temperatures reach about 100 C, soil moisture will be vaporized and the contact between the plate and the soil may degrade, resulting in poor data quality. Furthermore, performance of some of the soil heat flux plates at high temperatures is questionable because they were not specifically designed for these conditions. Finally, there is the possibility that the soil may have been disturbed when amending the plots. For these reasons, therefore, we removed all the soil temperature and heat flux plates several months after the fires and checked the vertical placement of all the sensors and the heat flux plate calibrations.

Results

Observed Soil Temperatures and Heat Fluxes

Soil temperatures and heat fluxes are presented as a pair of graphs for each burn site. The figure denoted with an ‘a’ is temperature and ‘b’ refers to soil heat flux. The first 3 figures are the meadow sites (high, medium, and low amended). The next 3 are for the forest sites (high, medium, and low amended). Figure 7 shows data from the slash burn site. As denoted on figure 7b two different soil heat flux plates were used for this site. None of the data shown in these figures have had any dropouts removed. Here data dropouts are defined as physically unrealistic negative values, such as those shown in Figs. 1a and 3a. These dropouts are associated with checking or resetting the data logger. Figs. 1a and 4a also show some spikes, which are thought to be related to spurious data logger errors. Each of these figures includes one or two days before the burn and two or more days after the burn to emphasize the dramatic change in the daily soil thermal regime caused by the fire and to give some indication of the time required at each site to recover a new daily temperature cycle.

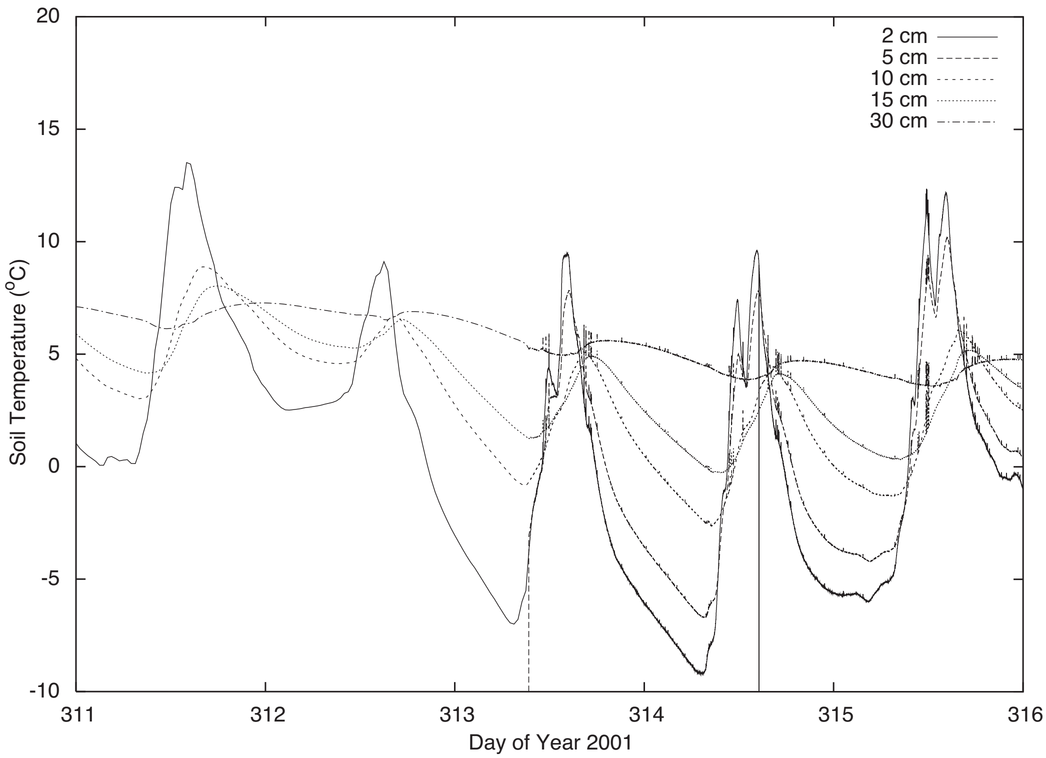
In general, the duration and intensity of the fires were determined primarily by the amount and type of the fuel loading and secondarily by fuel moisture content and rate of spread. These secondary factors are discussed in the next section. Here we examine the primary factors, the amount and type of the fuel loading.

The data show that higher temperatures and greater magnitudes for the soil heat flux are associated with higher fuel loading. For example, the loading at the medium and high meadow sites is about double the medium and high forest sites. Correspondingly, the maximum 2 cm soil temperatures are much higher at the meadow sites for the medium (222 C) and high loadings (260 C) than at the medium (36 C) and high (69 C) forest sites. A preponderance of dry grass in the meadow site versus pine needles in the shaded forest site probably accounts for the more intense, hotter fires at the meadow site than occured at the forest sites. Even more dramatic are the differences in fuel loadings and temperatures between the slash pile site, where the temperature at 2 cm reached about 407 C, and the forest or meadow site. Similarly, soil heat flux is greater (negative indicating heat flow into the soil) with increasing fuel loading, such that the magnitudes at 2 cm reached to about 1500 Wm^{-2} at the high density forest and meadow sites and more than 1700 Wm^{-2} at 5 cm at the slash burn site.

Likewise, the depth of thermal energy penetration and the duration of the heat pulse increased with fire duration and intensity and, ultimately, fuel loading. The duration of the heat pulse can be estimated from the time required for the transient heat pulse associated with the fire to dissipate, re-establishing a daily thermal cycle after the burn. Although these recovery times do vary with depth, in general the data suggest that they were between 16 and 20 hours at the high fuel sites, about half this time at the medium fuel sites, and less that 2 hours at the low fuel forest site. In contrast, the recovery time at the slash pile site was about 2 weeks. The deepest measured penetration of thermal energy occurs at the slash pile site where the temperature and heat flux at 1.365 m began increasing several days after the fire.

The only exception to this general association between the soil thermal response and fuel loading appears to be the low fuel meadow site. At this site it appears that the fire dynamics and the micro structure of the fuel loading prevented the fire from having any measurable influence on the soil. This occured in spite of the fact that the fuel loading was about 6 times the fuel loading of the low fuel forest site. The fire was intense at this site but extremely brief. It flashed over the burial location of the sensors in just a few seconds and had mostly burned itself out in less than a minute or two. Therefore, it appears that the fire duration was too short to have had much effect on the soil (at least at 2 cm and below).

Another possible anomaly occurs with the 30 cm soil heat flux during the slash pile fire. The data are noticeably noisier and for a time are directed upward (away from the soil), rather than into the deeper soil levels. The noisy



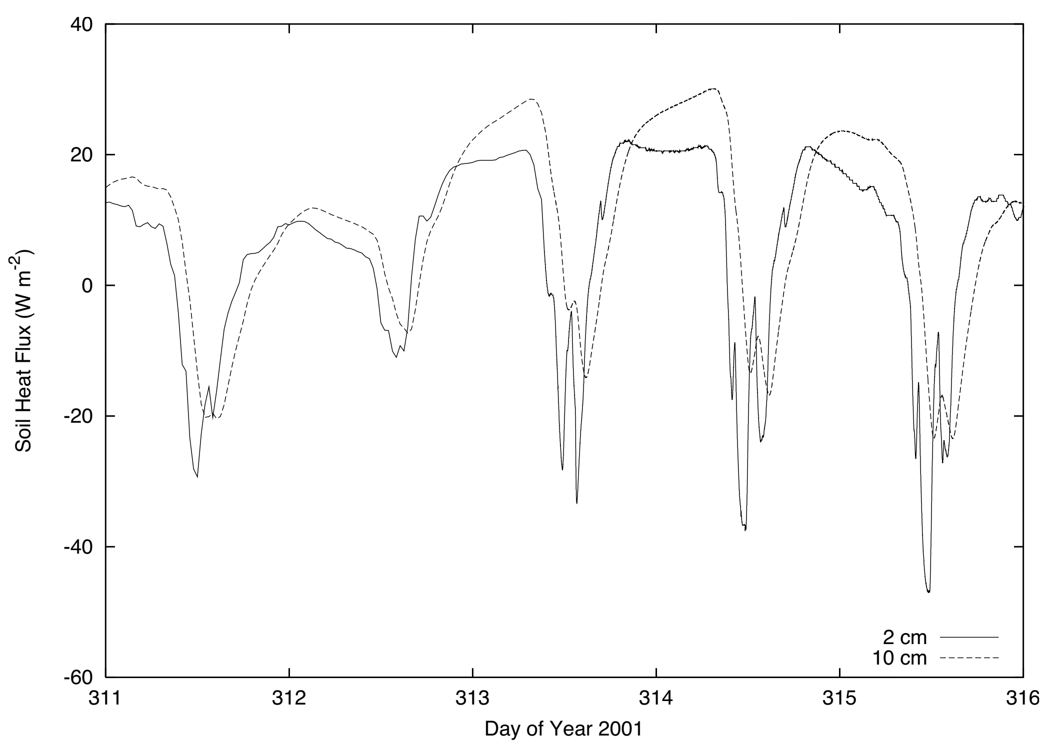
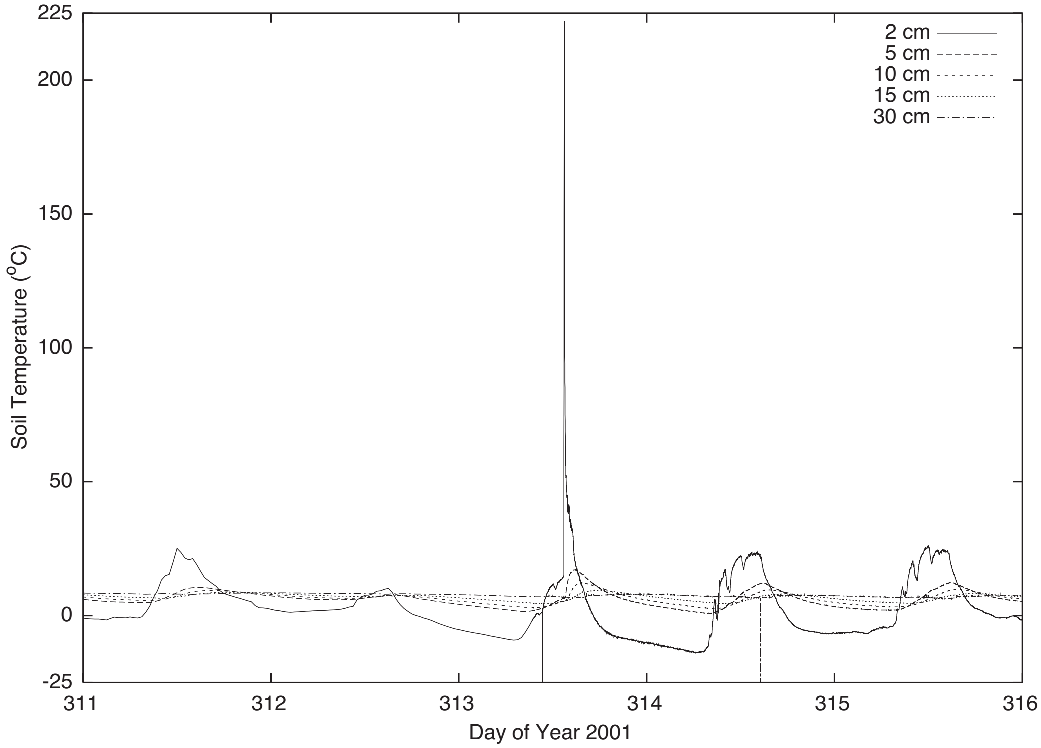


Figure 1—Five days of recorded soil temperatures (a) and heat fluxes (b) at the low fuel meadow site. Time series include two days before the fire and include two days after the fire. The fire was initiated about 1:20 PM MST on day 313 of 2001 and was over by about 1:25 PM. The soil heat flux was measured using Radiation Energy Balance Systems heat flux plates.



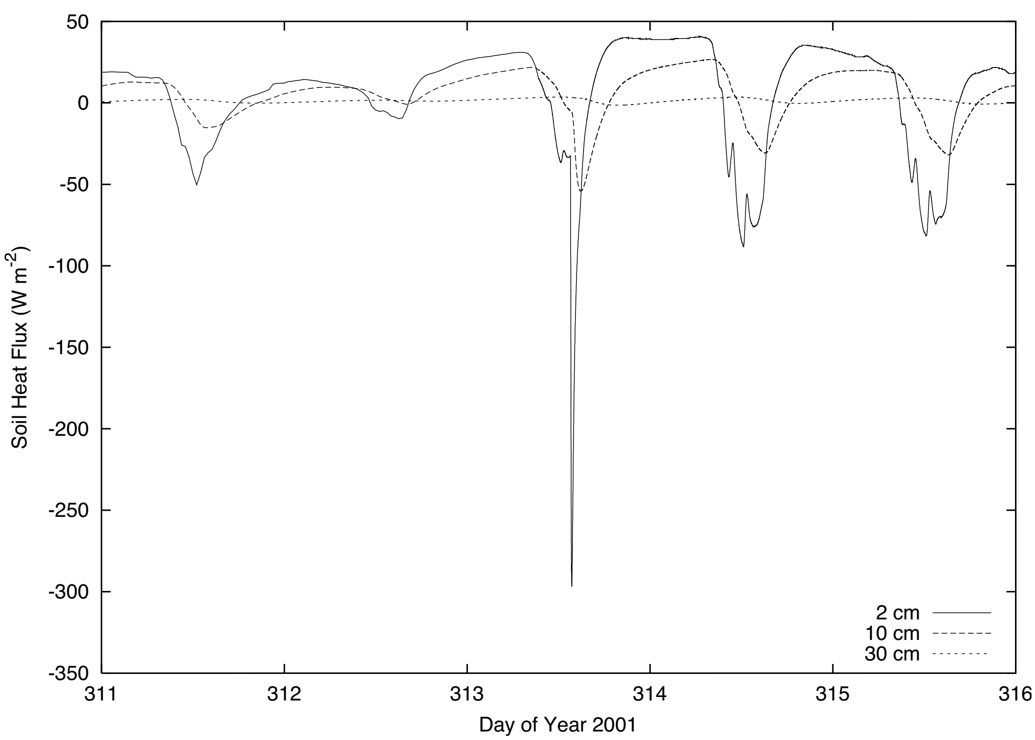


Figure 2—Five days of recorded soil temperatures (a) and heat fluxes (b) at the medium fuel meadow site. Time series include two days before the fire and include two days after the fire. The fire was initiated about 1:30 PM MST on day 313 of 2001 and was over by about 1:40 PM. The soil heat flux was measured using Radiation Energy Balance Systems heat flux plates.

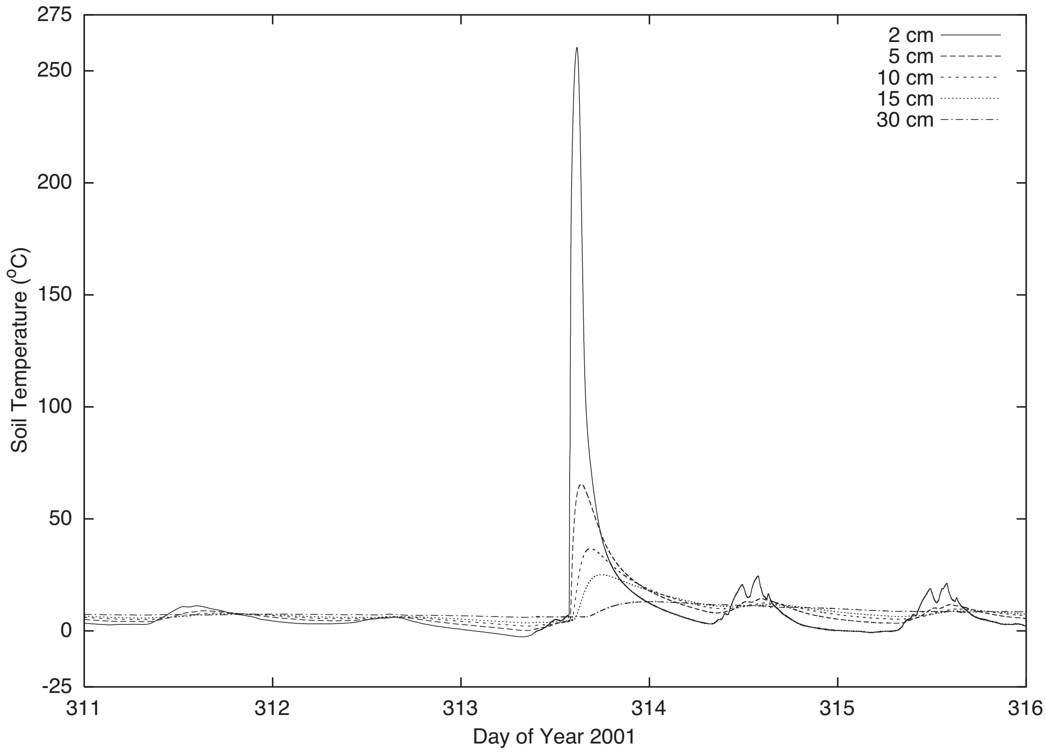
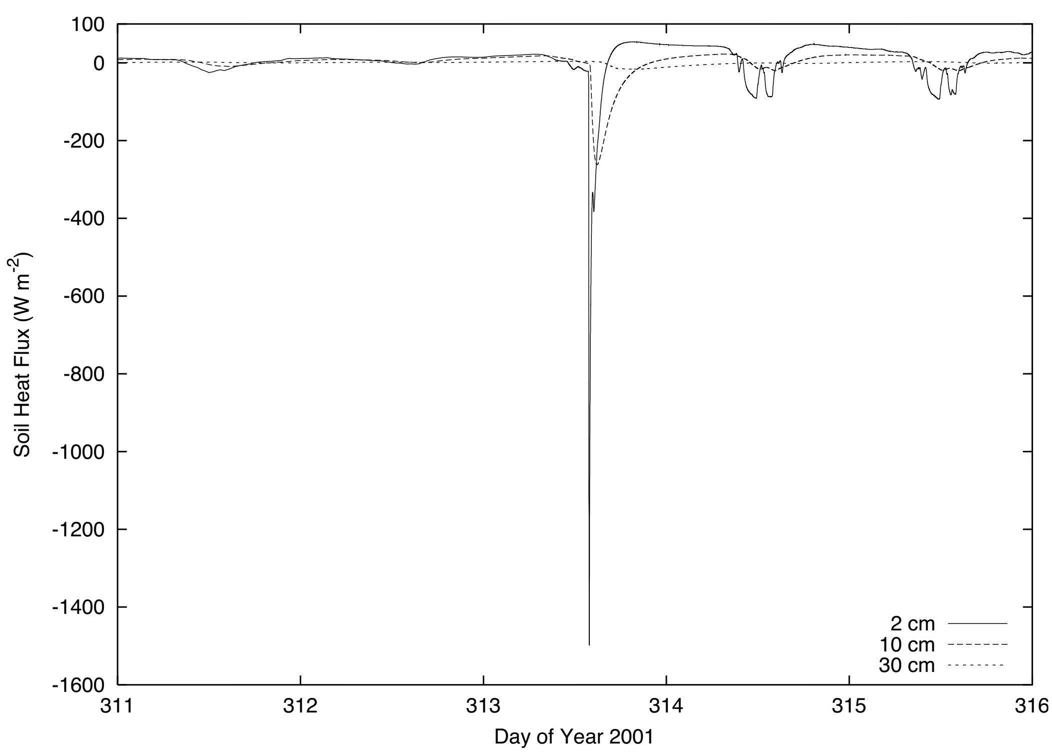


Figure 3—Five days of recorded soil temperatures (a) and heat fluxes (b) at the high fuel meadow site. Time series include two days before the fire and include two days after the fire. The fire was initiated about 1:48 PM MST on day 313 of 2001 and was mostly over by about 2:20 PM. The soil heat flux was measured using Radiation Energy Balance Systems heat flux plates.



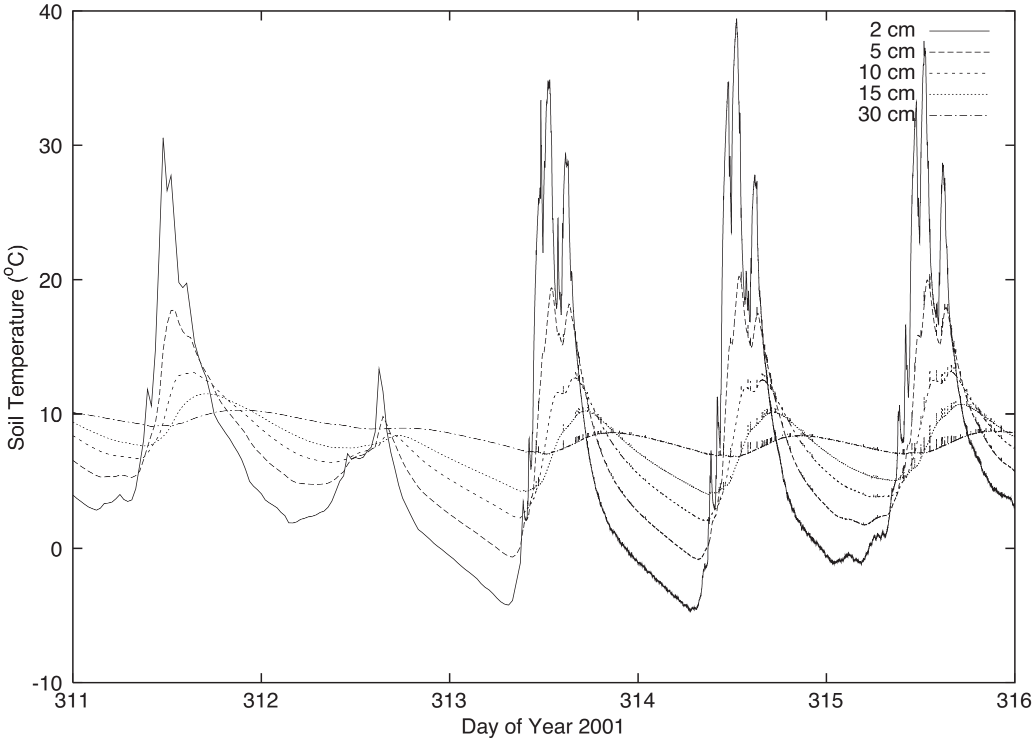
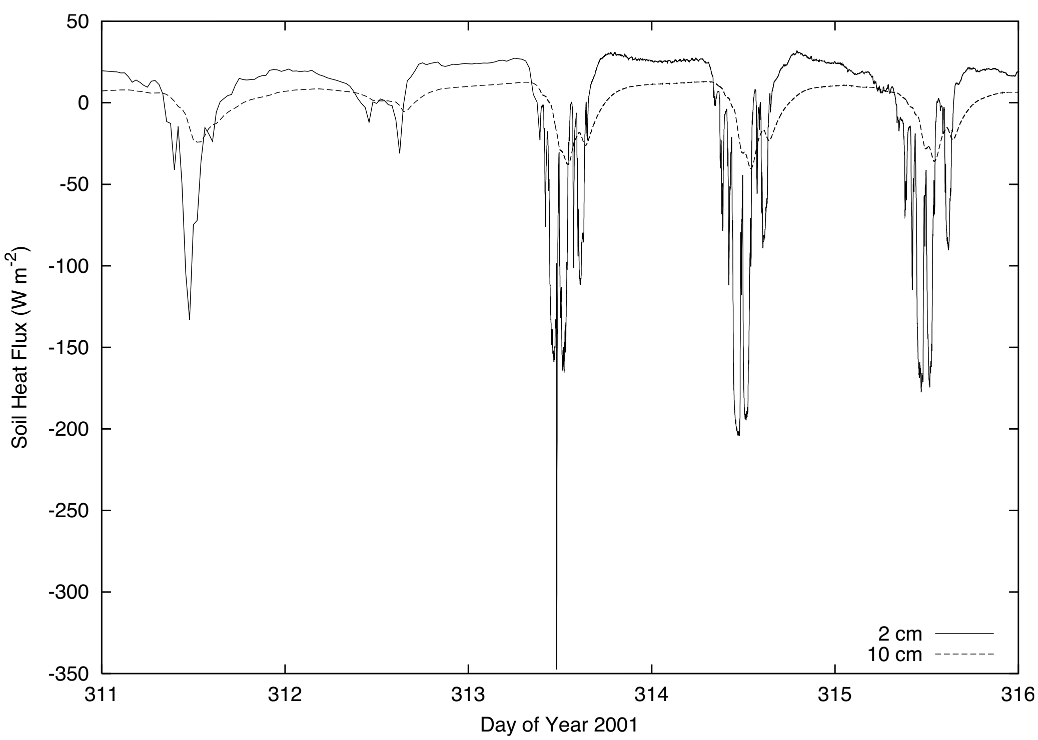


Figure 4—Five days of recorded soil temperatures (a) and heat fluxes (b) at the low fuel forest site. Time series include two days before the fire and include two days after the fire. The fire was initiated about 11:15 AM MST on day 313 of 2001 and was over by about 11:40 AM. The soil heat flux was measured using Radiation Energy Balance Systems heat flux plates.



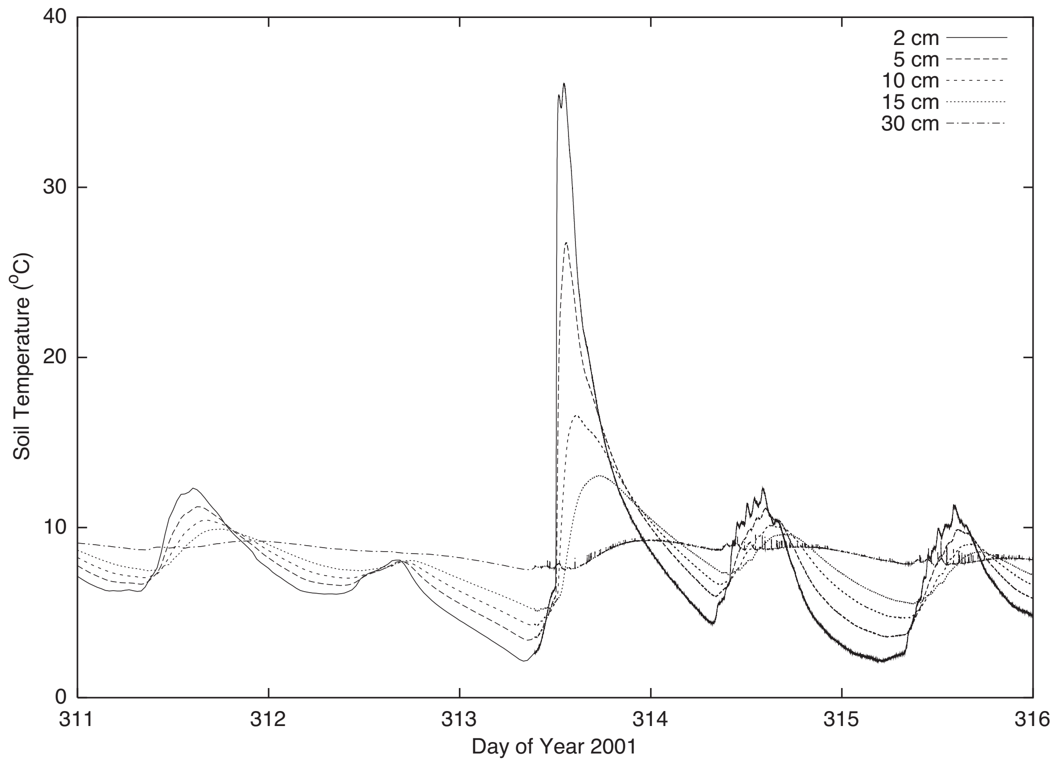
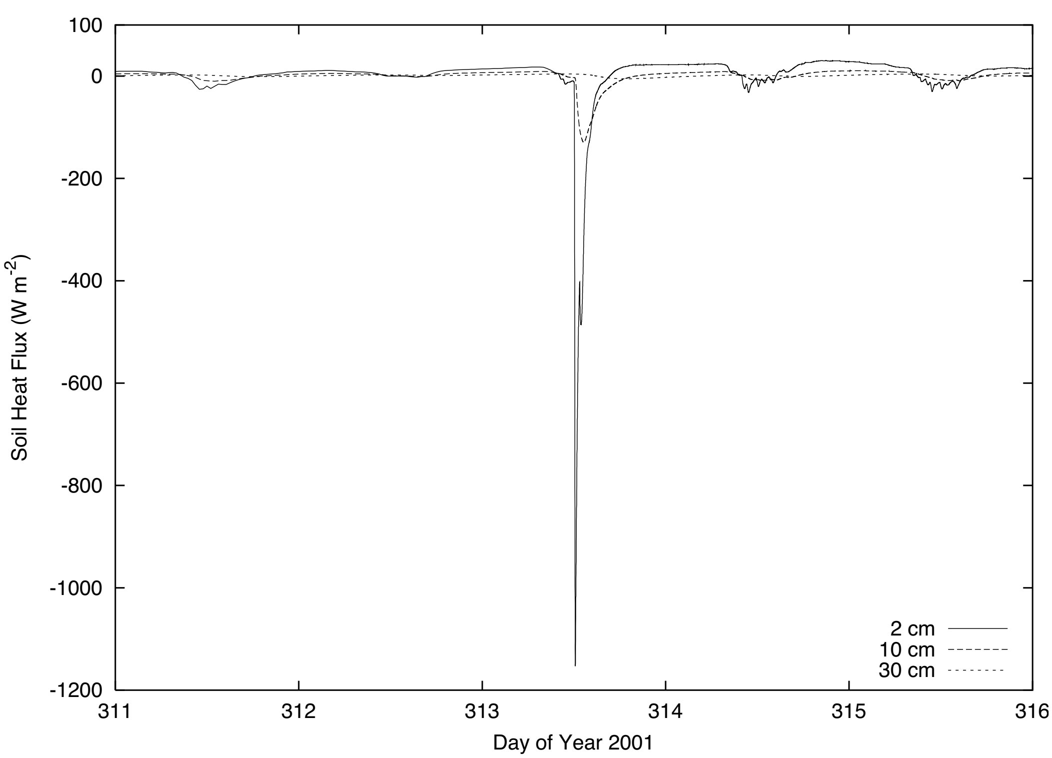
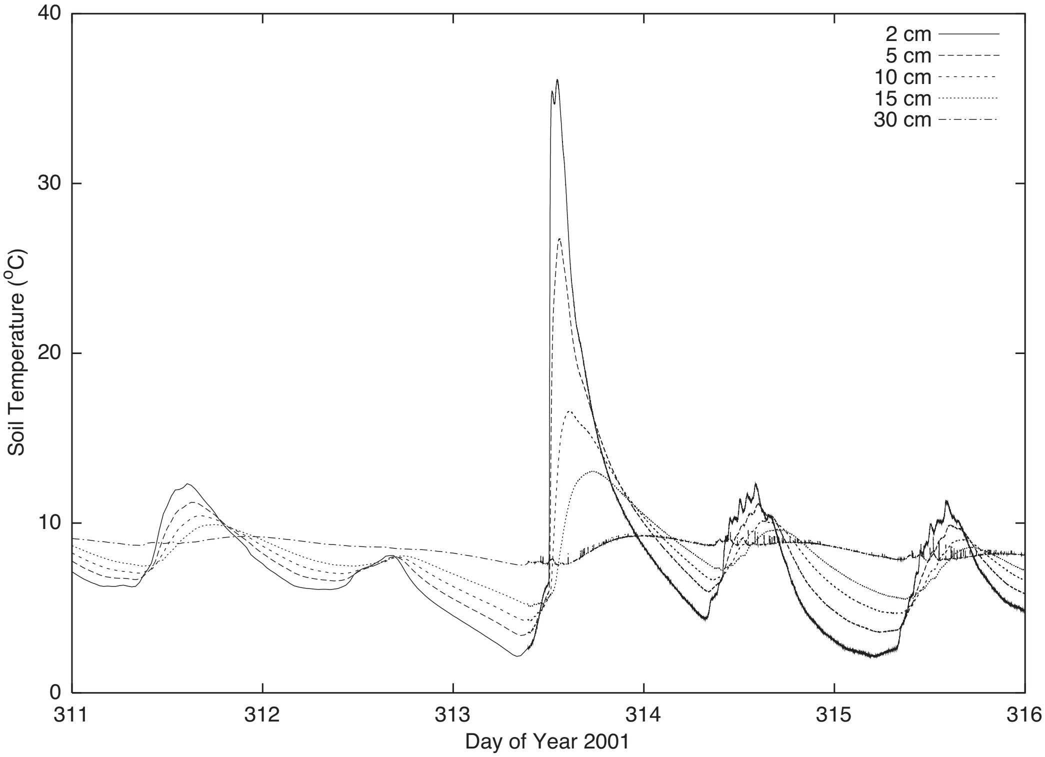


Figure 5—Five days of recorded soil temperatures (a) and heat fluxes (b) at the medium fuel forest site. Time series include two days before the fire and include two days after the fire. The fire was initiated about 11:41 AM MST on day 313 of 2001 and was over by about 12:10 PM. The soil heat flux was measured using Radiation Energy Balance Systems heat flux plates.





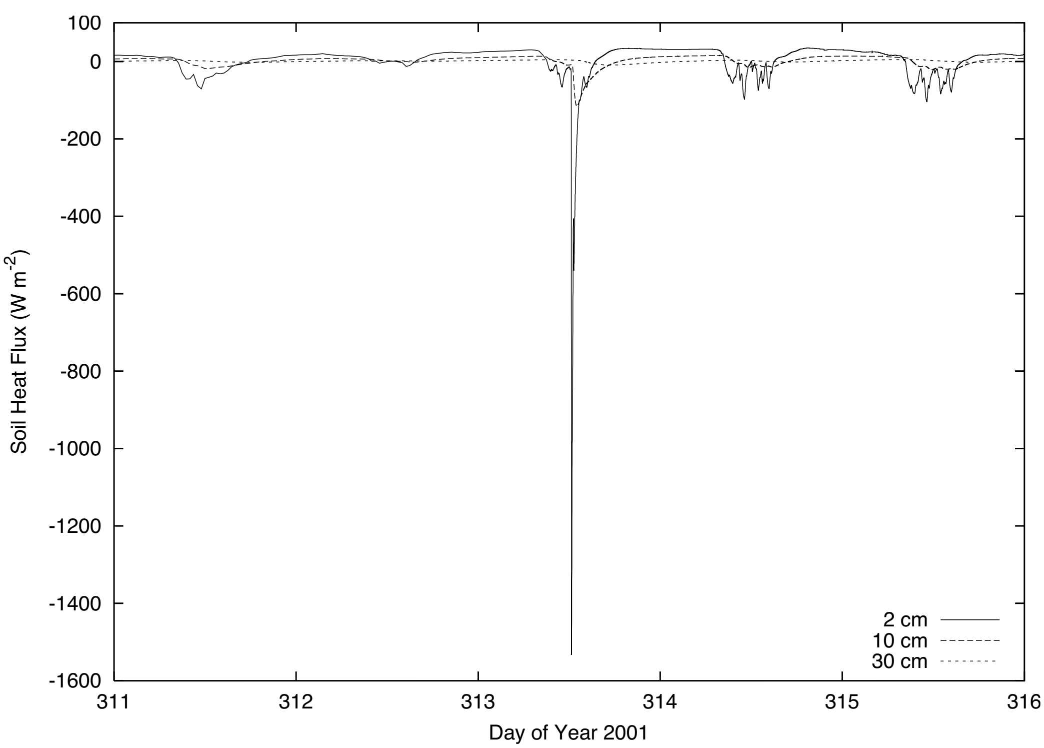


Figure 6—Five days of recorded soil temperatures (a) and heat fluxes (b) at the high fuel forest site. Time series include two days before the fire and include two days after the fire. The fire was initiated about 12:11 PM MST on day 313 of 2001 and was over by about 12:40 PM. The soil heat flux was measured using Radiation Energy Balance Systems heat flux plates.

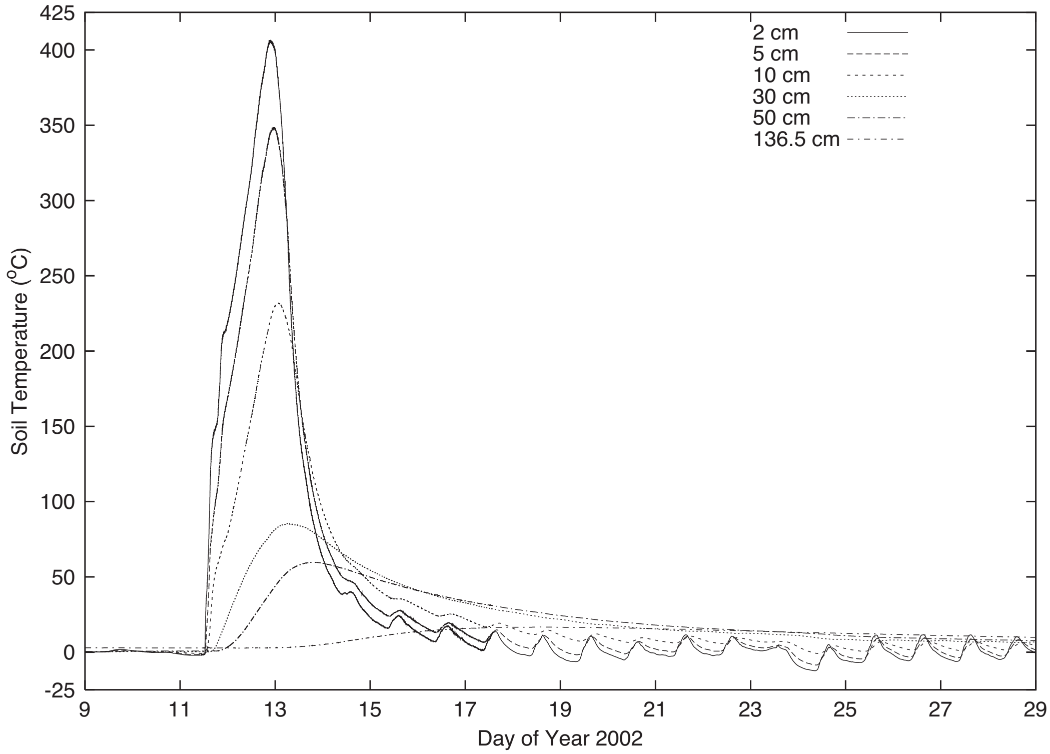
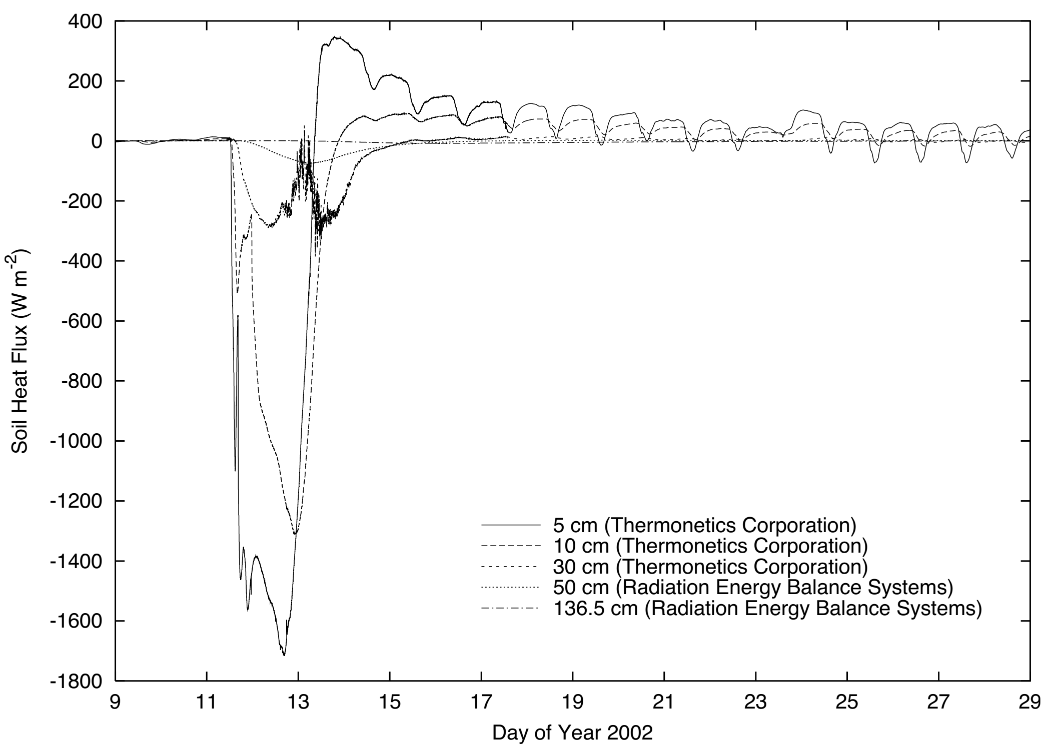


Figure 7—Twenty days of recorded soil temperatures (a) and heat fluxes (b) at the slash pile site. Time series include two days before the fire and include 18 days after the fire. The fire was initiated about 12:20 PM MST on day 11 of 2002 and burned for several hours. The manufacturers of the soil heat flux plates used during this experiment are given in the figure legend.



signal may be caused by seepage of soil moisture or water vapor through cracks or flaws in the glass glaze that seals the heat conductor from moisture. The directional change may also be related to the abrupt change from cooling to warming associated with the collapse of the burn pile about 35 minutes after initiating the fire. Both the 5 and 10 cm heat flux traces show two heat pulses separated by a brief period during which the magnitude of the flux is diminished. Furthermore, the temperature traces also show corresponding periods during which the rate of temperature increase slows considerably. When combined with the heat flux data these temperature data implicate, not only the fire and pile structure dynamics, but also that some of the thermal energy may be vaporizing any remaining soil moisture as the heat pulse propagates through the soil. The final phase of this experiment, discussed in the next section, included retrieving the sensors, rechecking their placement in the soil and the calibration factors of the heat flux plates, and in general examining them for possible defects or damage that may have occured during the periods of installation, the fire, or the site amendment.

Analysis

By using a simple mathematical description of the soil thermal pulse it is possible to estimate (or extrapolate to) the maximum temperatures and heat fluxes that occured at the soil surface during the fire. This information is probably the best way to standardize comparisons between the fires, but it may not be directly measurable. It is also useful for inferring something about the initial soil moisture content and its influence on the soil thermal properties as well as highlighting potential problems with the sensors. To accomplish this we assume a model that (at least approximately) partitions the thermal input to the soil into a dominant fire component and a secondary component representing all other thermal inputs to the soil. The simplest model we can assume relates the vertical profile of the measured amplitude (or maximum) of the heat pulse to soil depth as follows:

$$T_{max}(z) = T_0 + \Delta T\, e^{-z/D} \tag{1}$$

and

$$G_{max}(z) = G_0 + \Delta G\, e^{-z/D} \tag{2}$$

where $T_{max}(z)$ is the maximum observed value of the soil temperature as a function of depth, T_0 is a constant temperature, ΔT is the temperature amplitude associated with of the thermal pulse of the fire, and D is the soil thermal attenuation depth, which is related to the thermal properties of the soil. The symbols $G_{max}(z)$, G_0, and ΔG, used for soil heat flux, have a similar interpretation. By using the model expressed by equations (1) and (2) we follow an approach similar to Raison et al. (1986) who also suggested that the maximum fire related soil temperature decreased exponentially with soil depth. However, the present analysis differs from their study because they did have soil heat flux data nor did they calculate a thermal attenuation depth.

Because we assume that the fire dominates all other forms of thermal input to the soil we should expect that $\Delta T >> T_0$ and $\Delta G >> G_0$ are required for the model to be valid. Therefore, as we depart from this condition the model results will become less reliable. As a consequence, we can state a priori that the larger and more intense the fire, the more useful this model will be for interpretive purposes. Under this analysis scenario, ΔT is the maximum soil surface temperature during the fire and ΔG is the maximum soil heat flux at the surface during the fire.

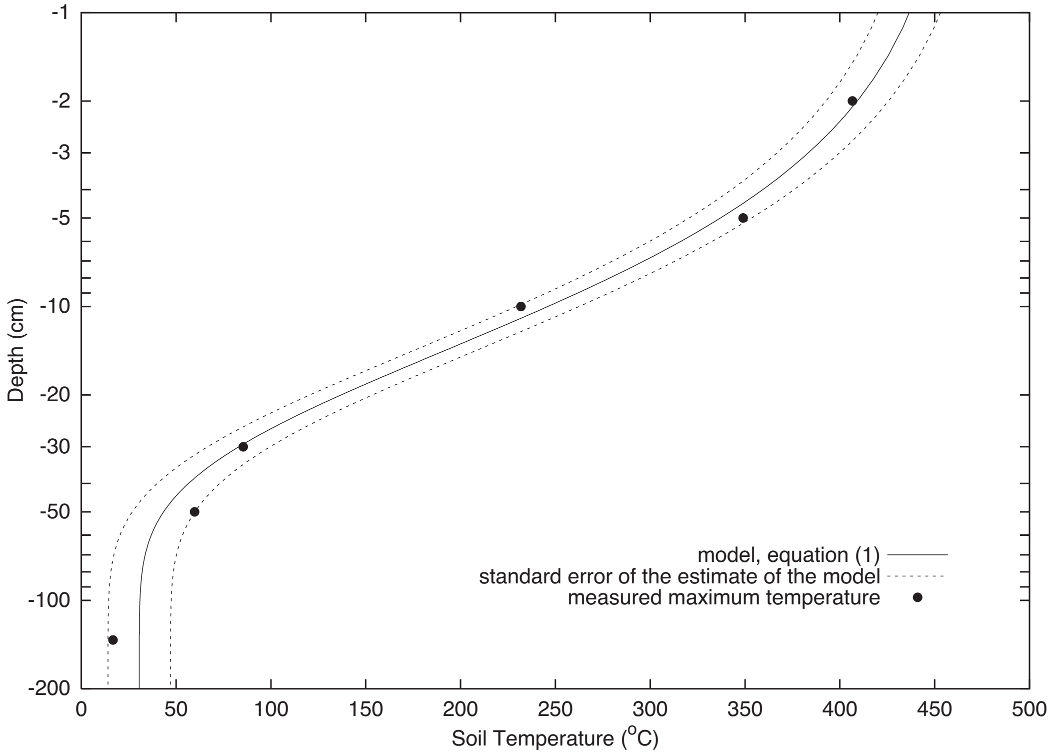
Table 3—Extrapolated surface temperatures, ΔT, and heat fluxes, ΔG, and inferred attenuation depths, D, for burns at the Manitou Experimental Forest. The standard error of the estimates are enclosed in parentheses. I = insufficient number of data points to estimate standard error of the estimate.

Site (loading)	T_0 (C)	ΔT (C)	D (cm)	G_0 (Wm^{-2})	ΔG (Wm^{-2})	D (cm)
Slash pile	31 (13)	436 (21)	14.2 (2.0)	15 (44)	-2350 (121)	16.9 (2.3)
Meadow (high)	24 (6)	734 (140)	1.8 (0.3)	-14 (I)	-2324 (I)	4.5 (I)
Meadow (med)	10 (1)	2049 (467)	0.9 (0.1)	-1 (I)	-454 (I)	4.7 (I)
Forest (high)	12 (2)	95 (7)	4.0 (0.5)	-9 (I)	-2968 (I)	3.0 (I)
Forest (med)	11 (1)	36 (4)	5.8 (1.2)	-5 (I)	-2001 (I)	3.4 (I)

The main benefit to this method of analyzing the fire data is its simplicity. However, one weakness is that it focuses only on the attenuation of the thermal pulse and does not include any aspect of the phase or time lag associated with the propagation of the thermal pulse through the soil. A phase analysis, which requires significantly more mathematical and computational effort, is beyond the intentions of the present study and is the subject of a later study. In this analysis we also treat temperature and soil heat flux independently, although theoretically at any given site the thermal attenuation depth, D, should be the same for both temperature and soil heat flux. The implications of similar and different values of D are discussed after presenting the results of the analysis.

To apply this model and these concepts to the present set of observations we simply fit equations (1) and (2) to the profile of observed maximum temperatures and heat fluxes. Table 3 gives the results of this analysis for the slash pile burn and the high and medium fuel burns at the forest and meadow sites. Note that because we are treating temperature and heat flux independently for this analysis there are two values for D given in table 3. The first (from left to right) is associated with temperature and the second is associated with the heat flux. The low fuel sites, which are not good candidates for this type of analysis, are not included in table 3 because it was not possible with such low intensity fires to distinguish between the thermal signals of the fires and that of the normal background. Figure 8 shows the graphs of the curve fits and observations for the slash pile burn. For this calculation the 30 cm soil heat flux data were not used because it was not possible to unambiguously identify the 30 cm peak magnitude for $G_{max}(z)$. The shaded areas of these figures correspond to the model's standard error of the estimate. Given the size and intensity of the slash pile burn, it probably provides the most reliable results. Overall, for most of these burns the thermal pulse was easily identified and the results confirm the requirement that $\Delta T >> T_0$ and $\Delta G >> G_0$. Only the temperature data at the medium fuel forest site is questionable in this regard.

Table 3 indicates that ΔT varied significantly from site to site. However, the meadow site ΔT values appear surprisingly high, suggesting there may have been temperature sensor problems. On the other hand, these high values may result from the rather high burn rate associated with the meadow sites. But, even before the burn the 2 cm temperature sensor at the medium loading meadow site seemed to be measuring anomalously high temperatures (for example compare figures 2a and 3a). The post fire inspection of the temperature sensors suggested that the nominal 2 cm temperature probes were actually much nearer the surface at both the meadow sites. The cause for this sensor misplacement is unknown, but it may have occured as a result of an



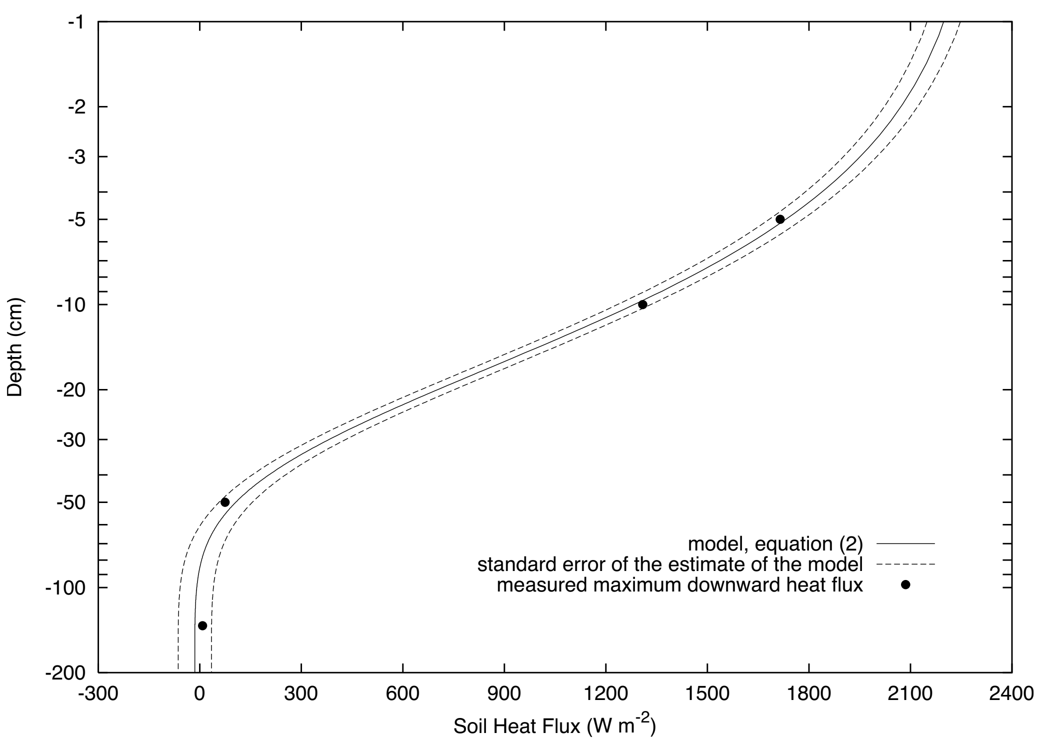


Figure 8—(a) Vertical profile of maximum measured soil temperatures (•) and the corresponding curve fit from equation (1) at the slash pile burn. (b) Vertical profile of maximum measured soil heat fluxes (•) and the corresponding curve fit from equation (2) at the slash pile burn. The shaded area in each figure encloses the model's standard error of the estimate.

Table 4—Post fire reanalysis of extrapolated surface temperatures, ΔT, and heat fluxes, ΔG, and inferred attenuation depths, D, for medium and high density fuel loadings at the meadow site burns. Sensor depth is assumed to be 1.0 cm for the high density plot and 0.25 cm for the medium density plot. The standard error of the estimates are enclosed in parentheses. I = insufficient number of data points to estimate standard error of the estimate.

Site (loading)	T_0 (C)	ΔT (C)	D (cm)	G_0 (Wm^{-2})	ΔG (Wm^{-2})	D (cm)
Meadow (high)	22 (6)	359 (25)	2.4 (0.4)	-12 (I)	-1812 (I)	5.0 (I)
Meadow (med)	10 (1)	253 (5)	1.4 (0.1)	0.2 (I)	-310 (I)	5.8 (I)

unintended disturbance during the site amendment or it may have resulted from a measurement error when the sensors were originally installed. There was no evidence of any misplacement or movement of the 2 cm soil heat flux plates at the meadow site, but the soil heat flux plates are much larger and more easily secured than the soil temperature probes. Table 4 is a recalculation of the model fits for equations (1) and (2) at the meadow site using a post fire estimate of the depth of the sensor probes. The recalculated ΔT values are still quite high at the meadow site, but they appear somewhat more reasonable. The post fire inspection also suggested that the soil sensors at the high fuel loading site were very close to a burned clump of grass, suggesting that this microscale feature of the fire may have contributed toward biasing these results somewhat high. Nevertheless, the recalculated values still indicate that there are significant site-to-site differences in soil temperatures, suggesting the need to examine why temperatures at the grass site were so much higher than the forest site.

These temperature differences likely resulted from differences in fuel type and loading, moisture content, and soil thermal properties. The differences in fuel type and loading between the meadow and forest sites (grass versus pine needles) likely contributed to a faster rate of fire spread across the meadow plots. These meadow burns lasted less than 3 minutes around the area where the sensors were buried, whereas the forest burns lasted between 10 and 12 minutes and the slash pile burn lasted several hours. Therefore, the rate of energy release was probably greatest at the meadow site. This is unquestionably true when compared with the two forest plots because of the much heavier fuel loading at the two meadow sites (table 1). Thus the thermal pulse at the meadow plots was probably confined to the upper few centimeters of soil, which is supported by the fact that the 5 cm meadow data show much less of a thermal pulse than the temperature data above 5 cm. The difference in thermal attenuation depth, D between temperature and heat flux is also greatest at the meadow site. This may be due to greater variation with depth of the soil thermal properties, which are functions of soil moisture and other soil physical parameters, than at the other sites. Finally, the total moisture content of the duff and litter layers at the meadow site was higher than at the forest site. At the meadow site the duff layer was not only deeper than the forest site (table 1), but on the day of the fire it was 33 percent water by weight, whereas the forest duff layer was 14 percent by weight. The same is also true for litter layer. However, on the day of the fire the litter layer at the forest site was slightly higher (23 percent by weight) than the litter layer at the meadow site (15 percent by weight). But, the total mass of litter at the meadow site was more than enough to compensate for the slightly lower water content. However, the moisture content of upper few centimeters of soil was about the same at both sites (between 2 percent and 3 percent by weight). Evaporating the additional moisture in the upper soil layers at the meadow site would also have tended to limit the thermal heat pulse to the upper few centimeters of the soil when compared with the forest site. On the other hand, the soil moisture content at the meadow site would have reduced soil temperatures and heat fluxes relative to what would have occured if the site had been drier (Frandsen and Ryan 1986).

Like the extrapolated surface temperature, ΔT, the extrapolated surface heat flux, ΔG, also shows significant variation. In general however, except for the medium fuel meadow site, ΔG was between 2000 and 3000 Wm^{-2} at all sites, which is in good agreement with the observations of surface soil heat flux made under a burning experimental log pile by Tunstall et al. (1976). Even allowing for the (unsupported) possibility of a meadow site soil heat flux plate positioning problem, the results of the recalculation at the meadow site (table 4) do not significantly alter this general conclusion concerning the surface heat flux, ΔG. Consequently, ΔG at the medium fuel loading meadow plot does appear anomalously low. The recalibration of the REBS soil heat flux plates (table 2) indicated that all sensors were functional and that they had maintained their calibrations in spite of the high temperatures. Possible explanations of the relatively low value for ΔG at the medium loading meadow site are poor contact between the soil and the 2 cm soil heat flux plate or the microstructure of the fire on this plot was such that the soil just above the 2 cm heat flux plate was not exposed to as much heat as the temperature sensors, which were separated laterally from the heat flux plates by 6-8 cm. Poor contact seems more likely, although we cannot completely eliminate the other possibility. The recalibration of the Thermonetics sensors also showed that they survived the fire at the slash pile site without damage, which reinforces our confidence that all other ΔG values are reasonable and that value for ΔG at the medium loading meadow site is in fact related to measurement error.

Although we did not list it in table 2, one of the REBS sensors was destroyed (melted) during the slash pile burn. We had installed this particular sensor at a depth of 10 cm before the fire to compare with the 10 cm Thermonetics sensor. However, the data obtained by this sensor before the fire were not consistent with the 10 cm Thermonetics sensor or any of the other 10 cm sensors. An examination of this sensor after the fire showed that the shrink tubing at the splice joining the sensor wires to the extension wires was not in good contact with the splice, which could have allowed moisture to short the connection and degrade the sensor signal. A similar problem was found with the heat flux plate at the 2 cm low loading meadow plot (figure 1b), which in this case may have caused an underestimation of the soil heat flux at 2 cm. (Otherwise the soil heat flux wave at 2 cm would show more attenuation at 10 cm than it does; see figure 1b.) Again, however, we cannot rule out poor contact as the cause for these two problem sensors. Nevertheless, potential measurement problems before the fire would not have caused the sensor to melt. For the benefit of future studies we note that the maximum soil temperature measured during the fire at this site was 232 C and that this REBS sensor was exposed to temperatures in excess of 150 C for over 48 hours (figure 7a). The 2 cm heat flux plates at the meadow medium and high loading plots were exposed to similar temperatures, but their exposure times were much shorter (figures. 2a and 3a).

Finally tables 3 and 4 also suggest that the attenuation depth, D, varies significantly from site to site. The large differences in the values of D between the slash burn site and the other sites suggests either significantly different soil thermal properties at the slash burn site or that the duration and intensity of the slash burn fire were sufficient to alter the soil thermal properties by evaporating the soil moisture at this site, combusting soil organic matter, and altering soil physical structure (e.g., DeBano et al. 1998). Evaporating soil moisture, at least, is quite likely if the soil contained any water in the upper 30 - 50 cm at the time of the fire. The significant variation in *D* between the temperature data and the soil heat flux data at the meadow site is probably more indicative of some of the previously discussed problems with the temperature and heat flux sensors at this site than it is indicative of any soil physical or thermal characteristics of the site.

Discussion and Conclusions

This study has demonstrated that it is possible to measure soil heat flux during a surface burn. Such data are important for modeling the soil thermal pulse during fires and to help with diagnosing possible soil moisture effects and evaporation during fires. Present results also confirm that soil temperatures and heat fluxes associated with surface fires can vary significantly with fuel loading, duration and intensity of the fire, and (at least indirectly) with the moisture content of the surface litter and duff. This study has also shown that large, long duration fires in heavy surface fuels can cause the thermal pulse to reach deeper into the soil and are more likely to affect belowground biota than shorter and more intense fires. Steward et al. 1990 reached a similar conclusion in their modeling study of surface fires. In addition, results of this study also suggest that surface temperatures of shorter more intense fires may also be high, but that the effects may be confined to shallower soil depths. It is possible, therefore, that such fires may cause greater loss of seeds lying on or embedded shallowly within the soil than larger less intense fires. This last scenario depends of course on the individual and species seeds' heat resistance and the moisture content of the near-surface soil layers, with the more moist soils associated with lower lethal temperatures (DeBano et al. 1998). Nevertheless, during this experiment soil temperatures at all but the lightest fuel loading plots reached nearly 100 C which should prove lethal to virtually all seeds (Hungerford et al. 1991, DeBano et al. 1998).

Management implications of these results for the Front Range of Colorado would indicate that burning surface fuel loadings under ponderosa pine forests and those resulting from lopping and scattering logging slash tend to produce soil heat pulses that are of relatively short duration and have a shallow depth of penetration. Burning large slash piles produce long duration, high temperature heat pulses that penetrate deep into the soil, potentially altering both physical and biotic characteristics of the soil to significant depths. Potential consequences of such large intense fires include the increased risk of erosion and a decreasing likelihood of seeding establishment and survival (DeBano et al. 1998).

References

Campbell, G.S., Jungbauer, J.D., Jr., Bristow, K.L., Hungerford, R.D. 1995. Soil temperature and water content beneath a surface fire. Soil Science. 159: 363-374.

DeBano, L.F., Neary, D.G., Ffoilliot, P.F. 1998. Fire's Effects on Ecosystems. New York: John Wiley & Sons. 333 p.

Frandsen, W.H. and Ryan, K.C. 1986. Soil moisture reduces belowground heat flux and soil temperatures under a burning fuel pile. Canadian Journal of Forest Research. 16: 244-248.

Hungerford, R. D.; Harrington, M. G.; Frandsen, W. H.; Ryan, K. C.; Niehoff, G. J. 1991. Influence of fire on factors that affect site productivity. In Proceedings of the management and productivity of western-Montane forest soils. A.E. Harvey and L. F. Neuenschwander (Eds.). USDA Forest Service, Intermountain Research Station General Technical Report INT-GTR-280.

Raison, R. J.; Woods, P. V.; Jakobsen, B. F.; Bary, G. A. V. 1986. Soil temperatures during and following low-intensity prescribed burning in a Eucalyptus pauciflora forest. Australian Journal of Soil Research. 24: 33-47.

Sackett, S. S.; Haase, S. M. 1992. Measuring soil and tree temperatures during prescribed fires with thermocouple probes. USDA Forest Service, Pacific Southwest Research Station General Technical Report PSW-GTR-131.

Steward, F.R., Peter, S., Richon, J.B. 1990. A method for predicting the depth of lethal heat penetration into mineral soils exposed to fires of various intensities. Canadian Journal of Forest Research. 20: 919-926.

Tunstall, B. R.; Martin, T.; Walker, J.; Gill A. M.; Aston, A. 1976. Soil temperatures induced by an experimental log pile fire: Preliminary data analysis. CSIRO Division of Land Use Research, Technical Memorandum 76/20.

USDA; Forest Service; Soil Conservation Service; Colorado Agricultural Experiment Station. 1986. Soil Survey of Pike National Forest, Eastern Part, Colorado, Parts of Douglas, El Paso, Jefferson, and Teller Counties.

Restoration Case Studies and Ecosystem Effects

Ecological Restoration Thinning of Ponderosa Pine Ecosystems: Alternative Treatment Outcomes Vary Widely

W. Wallace Covington[1]

Ecological restoration prescriptions should be designed using the best available scientific knowledge, have their likely consequences analyzed, and have their consequences monitored. To illustrate these points, three levels of ecological restoration thinning were analyzed for outcomes for a permanent plot established in 1909 on the Fort Valley Experimental Forest. This plot had 47.4 trees per acre in 1876 and 383 in 1997. Simulation modeling suggests that a strict restoration to 1876 densities would produce stand conditions that would support only surface burning with no crown scorch and release 491 btu / ft^2. Leaving three times as many trees would produce stand conditions that would support passive crown fire with 70 percent crown scorch and release 1800 btu / ft^2. Beyond fire behavior, predicted resource values vary widely as well. Strict restoration would produce a stand with high near view scenic quality (Scenic Beauty Indicator of 85), high grass and wildflower production (856 lbs / acre), and relatively high water yield (6.7 inches). The three fold level would produce low scenic quality (SBI 34), lower herbaceous production (134 lbs / acre), and lower water yield (6.1 inches). At its best, ecological restoration can improve ecosystem health, reduce unwanted fire behavior, and enhance human resource values. At its worst, it could be a waste of effort and resources.

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Aspen Response to Prescribed Fire, Mechanical Treatments, and Ungulate Herbivory

Steve Kilpatrick[1], Dean Clause[2], Dave Scott[3]

Abstract—*Land management agencies in northwestern Wyoming have implemented vegetation treatment programs to stimulate aspen (*Populus tremuloides*) regeneration. Treated clones are susceptible to extensive browsing from elk (*Cervus elaphus*) concentrated on adjacent supplemental feedgrounds, wintering moose (*Alces alces shirasi*), and livestock. We sampled eight treated (mechanical cutting and prescribed fire) aspen clones (stands) to determine treatment response 3-9 years post-treatment. A sampling design was tested for monitoring pre- and post-treatment stem densities. Total aspen sucker densities ranged from 3,480 to 29,688 stems/ac (8,600 to 73,360 stems/ha). Two 9-year-old treatments and one 7-year-old treatment achieved > 1,000 stems > 10 ft in height /acre (> 2,710 stems > 3.1 m/ha), the objective for successful clone reestablishment. Mean annual leader growth was 7.2 inches (18.3 cm) and ranged from 4.9 to 12.9 inches (12.4 to 32.8 cm). Treated clones are all expected to reestablish successfully. Stem density, clone homogeneity, and plot size influenced sampling efficiency.*

Introduction

Aspen (*Populus tremuloides)* is found throughout Wyoming's major mountain ranges. The larger stands occur in the Sierra Madre, Wind River, and Gros Ventre mountain ranges (Merrill et al. 1996). Total acreage in Wyoming is estimated at 338,000 acres (Green and Van Hooser 1983). However, it is also estimated the historical acreage (100–150 years previous) was at least double the present. The successional replacement of aspen with conifers, shrubs and herbaceous vegetation continues today (Debyle and Winokur 1985; Bartos and Campbell 1998). Factors contributing to aspen decline and lack of regeneration include fire suppression, livestock grazing, wild ungulate browsing, and natural succession (Krebill 1972; Bartos and Campbell 1998; Gruell and Loope 1974; Mueggler 1989; Romme et al. 1995).

Aspen communities are recognized for their multiple values, including recreation, scenic vistas, water yield, water quality, wood products, habitat for an array of wildlife species, forage for wild and domestic ungulates, and landscape diversity (Bartos and Campbell 1998; DeByle and Winokur 1985). A minimum of 140 mammal and bird species utilize aspen habitat types in Wyoming (Dieni and Anderson 1997). The decline of aspen communities in the West, and throughout the state of Wyoming, is a concern for ecologists and resource managers. Recently, fire managers have also become concerned with the loss of the "asbestos forest type" (Fechner and Barrows 1976). Healthy aspen stands provide natural firebreaks which reduce fire intensity and severity, allowing fire managers additional control options.

Successful aspen regeneration in the West occurs almost exclusively through vegetative propagation (Debyle and Winokur 1985). Reproduction is clonal

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in nature (Barnes 1966) and generally enhanced with disturbances that kill overstory trees and preclude auxin transfer to the roots, thus stimulating suckering (Debyle and Winokur 1985). Management activities that kill or stress overstory trees (e.g., prescribed burning, clear-cutting, herbicide treatments) mimic natural disturbances and enhance aspen regeneration.

The appropriateness of implementing such management actions in areas of intense ungulate herbivory has been questioned. Aspen enhancement projects located near supplemental elk feeding sites (feedgrounds) in northwest Wyoming have resulted in varying levels of success. Krebill (1972) concluded that natural aspen recruitment of 653 stems / ac (1,614 stems / ha) in the Gros Ventre was not sufficient to replace overstory mortality given impacts from herbivory. Hart (2000) re-examined aspen stands evaluated by Krebill and concluded that "herbivory and disease, superimposed on successional events, may be exerting negative effects on the distribution of aspen." Bartos et al. (1994) concluded that a prescribed burn treatment of aspen near elk (*Cervus elaphus)* feedgrounds in the Gros Ventre drainage may have hastened the demise of decadent aspen clones. However, other prescribed burns in the same area that lacked intense herbivory were successful (Bartos et al. 1991). Dieni et al. (2000) concluded that aspen regeneration through clear-cutting on the U.S. Fish and Wildlife Service's National Elk Refuge may have hastened their demise as a result of excessive elk herbivory. Kilpatrick and Abendroth (2000) emphasize aspen treatment site selection. They suggest the following factors, alone or in combination, may contribute to outcome following aspen treatment: site aspect, site distance from concentrations of wintering ungulates and elk feedgrounds, supplemental elk feeding regime, aspen community type, stand vigor, soil type, fire intensity / severity, and level of human disturbance to wildlife.

In this current study, our primary objective was to quantify aspen regeneration post-treatment (mechanical clearing and prescribed burn) at two different areas in northwest Wyoming. Eight post-treatment sites were sampled within the two areas. Both areas are near supplemental elk feedgrounds where herbivory levels could potentially impede aspen regeneration. A secondary objective was to test the accuracy and efficiency of our aspen sampling methodology.

Study Area

This study deals with aspen sampling at eight post-treatment and five pre-treatment sites located south and east of Jackson, Wyoming (tables 1 and 3).

Two post-treatment sites were sampled at the Bryon Flats area located on the Bridger-Teton National Forest approximately 6 miles (9.6 km) southeast of Hoback Junction, Wyoming. Sites are on the east side of Willow Creek, a tributary of the Hoback River. One site was clear-cut in 1994 and the other prescribed burned during the fall of 1995. General treatment goals were to reduce conifer densities, promote aspen suckering, and set back succession. The Wyoming Game & Fish Department operates the Camp Creek supplemental elk feedground approximately 2 miles northwest of the treated sites where 600-1000 elk are fed baled hay during the winter months (December – April). A relatively small moose (*Alces alces shirasi*) population utilized the area rear-round. Mule deer (*Odocoileus hemionus*) utilized the area during the spring – fall. The sites also received summer cattle grazing.

Six post-treatment aspen sites were sampled during 2000 at the Soda Lake site, approximately 7 miles (11.3 km) north of Pinedale, Wyoming. The

Table 1—Aspen post-treatment site location, code, date, herbivory and elevation.

Post-treatment site location and type	Code	Treatment date	Herbivory	Elevation (ft)
Soda Lake - WGFD Rx burn	SLGFBurn	Fall 1991	elk, moose, deer	7690
Soda Lake - Forest Service Rx burn	SLFSBurn	Fall 1991	elk, moose, deer, cattle	7850
Soda Lake - WYGFD mechanical cutting	SLGFCut	Sum. 1991	elk, moose, deer	7500
Soda Lake - Forest Service Exclosure -Rx burn	SLEXBurn	Fall 1991	none	7770
Soda Lake - Spring Creek FS Rx Burn	SLSCBurn	Fall 1991	elk, moose, deer, cattle	8345
Burnt Lake - FS mechanical cutting	BLFSCut	Fall 1997	elk, moose, deer, cattle	8040
Willow Cr. Rx burn	WCBurn	Fall 1995	elk, moose, deer, cattle	6808
Willow Cr. Mechanical Cutting	WCCut	Fall 1994	elk, moose, deer, cattle	6888

Table 2—Post-treatment sampling comparisons, of plot size, number of plots, and stem densities (80% confidence, 20% error).

Plot size	No. plots	Stems/ac
1/500	57	4865
1/500	29	5414
1/500	15	15133
1/500	6	10100
1/500	5	29688
1/500	17	6367
1/300	15	3480
1/100	23	4039

Wyoming Game and Fish Department operates an elk supplemental feedground near Soda Lake. Approximately 800-1,000 elk were historically fed on the north side of Soda Lake until 1993, 1-2 miles (1.6-3.2 km) from the monitored sites. After 1993 the feeding site was relocated south of Soda Lake, which extended the distance to the sampled sites 2-4 miles (3.2-6.4 km). Moose utilized the area year-round and mule deer inhabited the area during spring and fall.

Five pre-treatment aspen sites were sampled during 2001, at the Fremont / Pinyon Ridge site near Pinedale. Two of the sites were located on the north end of Fremont Ridge, approximately 7 miles (11.3 km) north of Pinedale. The remaining three sites were located on Pinyon Ridge, approximately 35 miles (56 km) north of Pinedale.

Methods

A pilot sampling methodology was utilized to evaluate efficiency in acquiring sufficient samples to provide sucker density data sets with 80% confidence and 20% error. These statistical parameters were recommended to land managers for monitoring the success of treatments designed to enhance aspen regeneration (Winward et al. 2000). In addition to density, height and annual leader growth also were monitored. Sampling steps for our study reported here were (see also figure 1):

- Randomly establish a permanent photo point with a 5 ft steel post near the center of the clone / stand. Record GPS coordinates.
- Take photos in the four cardinal directions.
- Select a random azimuth from the permanent photo point.
- Select a random pace distance.

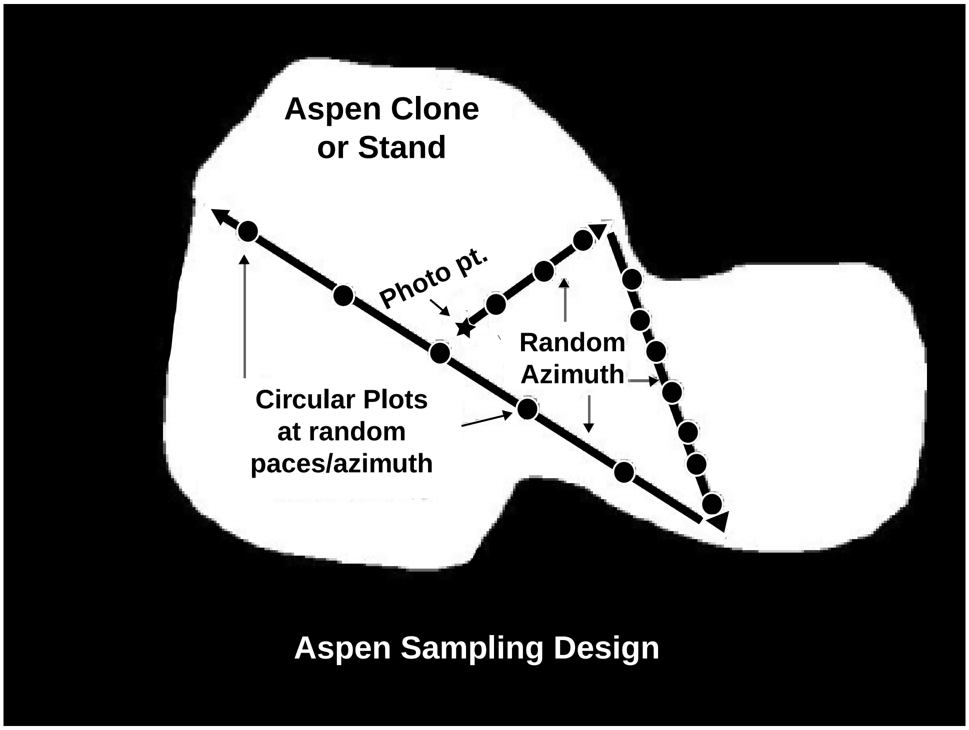

Figure 1—Sampling design used for estimating pre- and post-treatment aspen stem density, height, and current year leader growth.

- Select circular plot size. Recommend 1 / 50 to 1 / 100 acre (1 / 124-1 / 247 ha) for pre-treatment mature tree densities estimated at 150-250 trees / acre (370-618 trees / ha). Recommend 1 / 100-1 / 500 acre (1 / 247-1 / 1,236 ha) plot size for post-treatment densities of 4,000-15,000 stems / acre, (9,884-37,065 stems / ha) respectively. Also, increase the plot size with increased stand / clone heterogeneity.
- Proceed along random azimuth from permanent photo point sampling plots at the random pace intervals. Record within plots: (a) height of stems by 1 ft class (e.g., <1, 1-2, …>10) / plot, (b) number of stems / plot, (c) annual leader growth on two to three dominant leaders of random suckers / plot.
- Stop proceeding along random azimuth when a different community type or ecotone is encountered.
- Select a new random azimuth that intercepts the stand / clone, and select a new random pace.
- Continue the above procedure until the required sample size and statistical reliability is achieved using the following formula (e.g., 80% reliability with 20% error).

$N = (t)^2 * (s)^2 / (P * M)^2$
N = required sample size
t = t table value for desired confidence level (e.g., 80% , 90% C.I.)
s = standard deviation
P = percent error (e.g., 20% = 0.20)
M = mean # stems / plot

Bartos and Winward (2000) recommend the following post-treatment conditions for successful aspen clone reestablishment: >1,000 stems / acre (2,471 stems / ha), >10 ft (3.1 m) in height within 10 years post-treatment. They also suggest mean sucker height should increase by 1-ft / year (0.31 m / year) post-treatment. The above recommendations were transformed into objectives for our aspen treatments.

Areas of mixed-aged aspen and conifers selected for this study were considered to be "at risk" as described by Bartos and Campbell (1998) and were sampled with the above detailed procedure.

Results

Three of the treated sites, SLSCBurn, SLEXBurn, and WCCut, achieved means of >1,000 stems >10 ft in height / acre (>2,471 >3.1 m stems / ha) (figure 2). The fall 1991 Soda Lake Spring Creek prescribed burn (2,200 >10 ft stems / acre; 10,100 mean total stems / acre; figures 2 and 3) was located at the highest elevation and greatest distance from the Soda Lake elk supplemental feedground near Pinedale, Wyoming. Elk herbivory appears to be light during fall and early winter when elk are migrating through the site towards the feedground. Snow depths and human disturbance (winter recreation) are suspected to preclude heavier elk herbivory levels. Cattle use of this site is also considered to be light or moderate during the growing season.

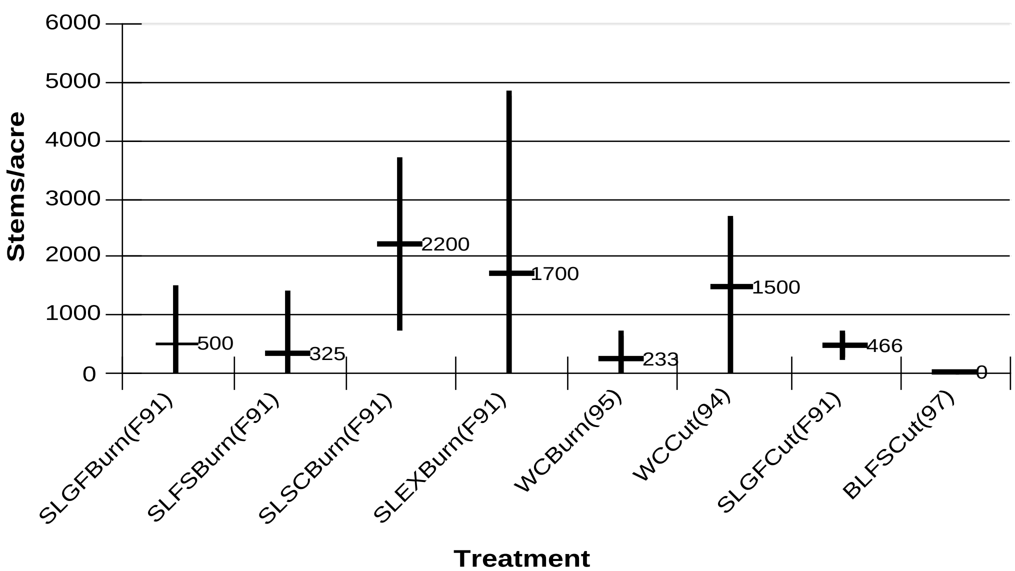

Figure 2—Mean and one standard deviation for stem densities of aspen suckers >10 ft in height at time of sampling.

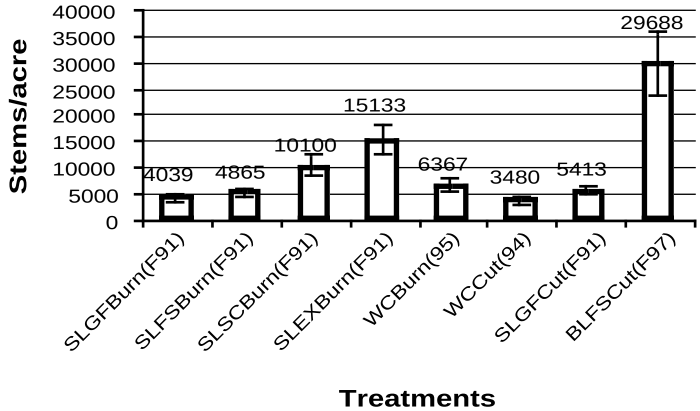

Figure 3—Mean total aspen stems per acre by treatment. Error bars indicate P = .8 confidence interval.

The fall 1991 Soda Lake exclosure prescribed burn (SLEXBurn), located near Pinedale, Wyoming, was the second site appearing to meet the regeneration objectives (figure 2). It is situated at a lower elevation than the above site and is adjacent to the boundary between the Wyoming Game and Fish Department and the USFS. The 8-ft fence surrounding the exclosure was elk and livestock-proof, excluding all herbivory. Mean stems >10 ft (3.1 m) in height and mean total stems were 1,760 and 10,133 stems / acre (4,349 and 25,038 stems / ha) respectively.

The third site having greater mean sucker density and height was the Willow Creek mechanical cutting treatment (WCCut), located near Jackson, Wyoming. It received light wild ungulate herbivory despite being within 2 miles (3.2 km) of a supplemental feedground. Possible reasons for the light herbivory use were location of the site away from a major elk migration route to the Camp Creek supplemental elk feedground and human disturbance from winter recreation. Moose are thought to be responsible for most of the herbivory occurring on the site. Moose are more tolerant of human disturbance than elk and their droppings were evident at the site. Mean total aspen stems and those >10 ft in height were 4,480 and 1,700 stems / ac (11,070 and 4,200 stems / ha) respectively (figures 2 and 3).

The remainder of the treatments with ≥7 growing seasons had total mean sucker densities ranging from 4,039 to 6,367 stems / ac (9,980-75,732 stems / ha). The Burnt Lake Forest Service mechanical cutting site had 29,688 stems / ac (73,359 stems / ha) after its third growing season. All densities were within the 850-19,951 stems / ac (2,100-49,300 stems / ha) reported by others (Bartos et al. 1994; Patton and Avant 1970; Brown and DeByle 1987; Brown and DeByle 1989; Bartos et al. 1991; Kilpatrick and Abendroth 2000).

The median height class of aspen suckers nine years post-treatment ranged from 4-5 ft (1.2-1.5 m) to 5-6 ft (1.5-1.8 m) (figure 5). The median height classes for the Willow Creek burn (six years post-treatment) and mechanical cutting (seven years post-treatment) were 5-6 ft (1.5-1.8 m) and 6-7 ft (1.8-2.2 m) respectively. Median height class for the Burnt Lake mechanical cutting was 1-2 ft (0.3-0.6 m), three years post-treatment.

Current annual growth of dominant leaders did not meet the management objective of 12 inches / year (30.5 cm / year). Mean annual leader growth rate ranged from 4.9-12.9 inches (12.4-32.8 cm) and averaged 7.2 inches (18.3 cm) across all treatments (figure 4). Bartos et al. (1991) documented similar results with average sucker heights increasing 0.8-8.6 inches / year (2.0-22.0 cm / yr) on burned sites near Jackson, Wyoming.

The pre- and post-treatment sample size required to meet the statistical objective of 80% confidence ± 20% error ranged from 15-29 and 5-57 plots, respectively (tables 2 and 3). Stem density, homogeneity of the sampled clone / stand, and plot size influenced the minimum sample size.

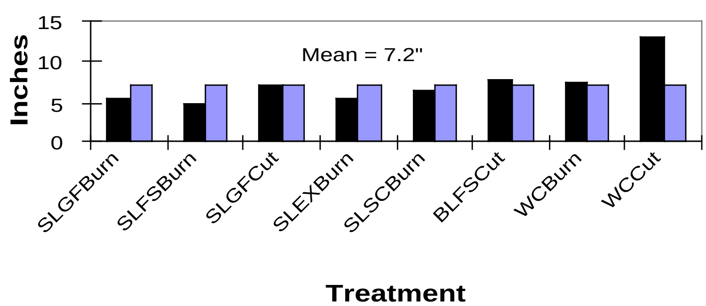

Figure 4—Current annual growth of dominant aspen leaders by treatment site (black bars). Blue bars represent the mean.

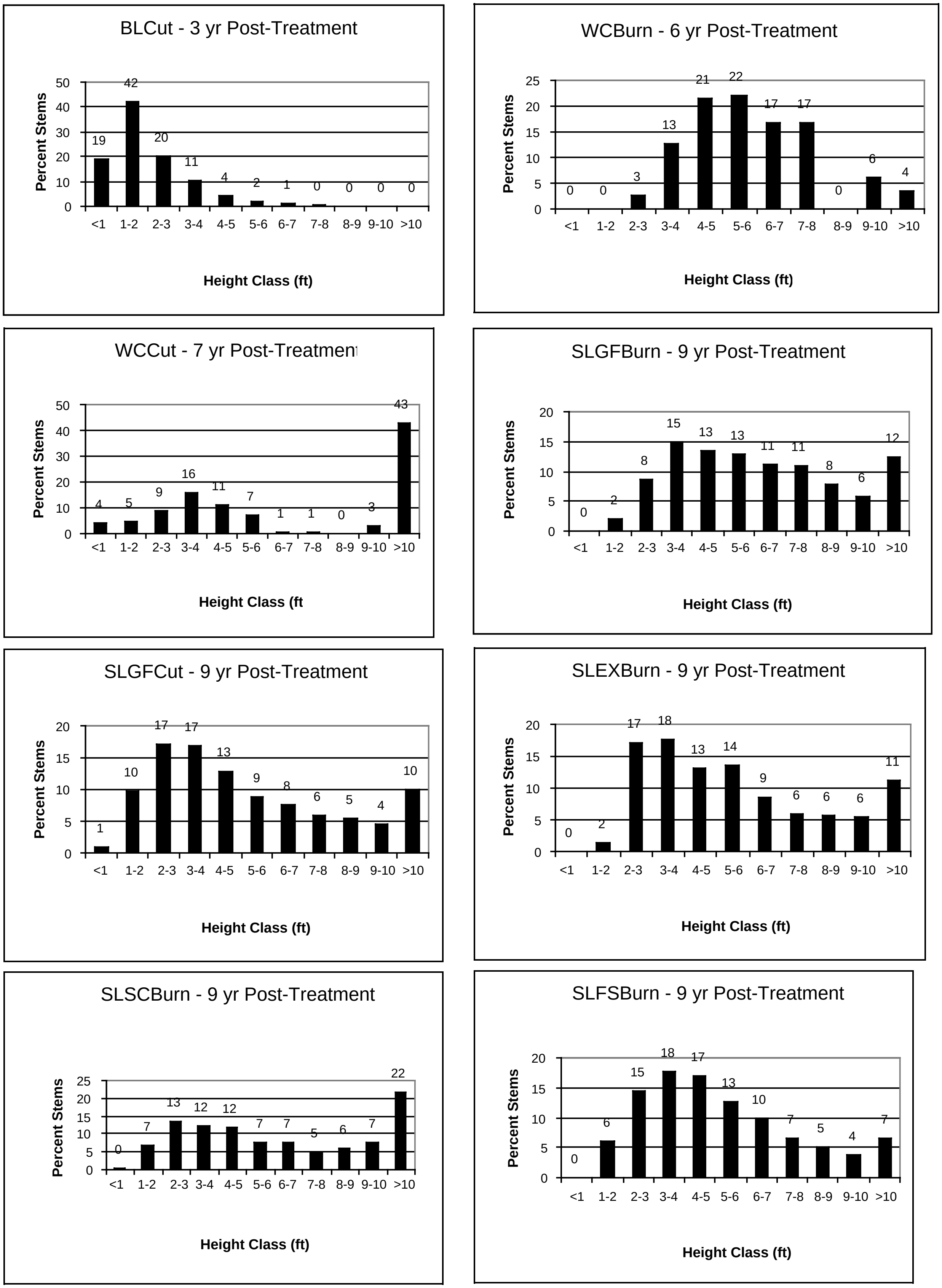

Figure 5—Height of aspen suckers 3-9 years post-treatment at eight different sites in Wyoming.

Table 3—Pre-treatment sampling comparisons of plot size, number of plots, and stem densities (80% confidence, 20% error).

Plot size	No. plots	Stems/ac
1/100 ac	15	1342
1/100 ac	18	1749
1/100 ac	17	206
1/100 ac	22	255
1/50 ac	29	160

Discussion

Successful management-induced aspen regeneration is quite variable within some areas of the West. Resource managers have attempted to enhance aspen regeneration in northwestern Wyoming since the early 1970s. Many factors such as clone vigor, community type, fire intensity / severity, herbivory by wild and domestic ungulates, aspect, elevation, soil type, and moisture regimes have influenced their success. In northwest Wyoming, 25,000 elk are fed supplemental winter rations at 23 different locations. Thus, elk herbivory alone can significantly influence aspen regeneration near these feeding locations. In some areas, successful regeneration cannot be accomplished without clone or stand protection with fencing. Other aspen treatment sites, such as the ones monitored in this study, appear to have promise. Continued monitoring, short and long-term, will help managers identify factors influencing the success of treatments and make appropriate adjustments.

Two of the nine-year (SLSCBurn, SLEXBurn) and one of the seven-year (WCCut) post-treatment sites appear as though they are approaching 10-year post-treatment regeneration objectives. Three of the nine-year post treatment sites (SLFSBurn, SLGFBurn, SLGFCut) may take longer than 10 years to reach proposed objectives of >1,000 >10 ft stems / acre (>2,710 stems >3.1 m / ha). The six-year (WCBurn) and three-year (BLCut) post treatment sites appear to be regenerating successfully but will need continued monitoring prior to making conclusions. All eight treated sites have maintained more than adequate sucker densities for successful clone reestablishment.

Dominant leader growth was only slightly more than half of the management objective. Severe drought conditions during 2000 and 2001 may have resulted in reduced leader growth. Few clones appear to be averaging 1 ft growth per year despite light to moderate ungulate herbivory. Continued monitoring over the next two to four years is recommended to further assess drought effects and evaluate management objectives for successful clone reestablishment.

Both wild and domestic herbivory continue to be an important factor in clone reestablishment. Three of the sites sampled (SLEXBurn, SLGFBurn, and SLFSBurn), were close to each other but received different levels of herbivory. The SLEXBurn was fenced and received no wild or domestic ungulate herbivory. It had the greatest sucker stem density and height of the three sites. The SLGFBurn received only wild ungulate herbivory and was intermediate in density and height. Moving the Soda Lake supplemental elk feedground an additional 1.5 miles (2.4 km) from this treatment site in 1993 appears to have reduced elk herbivory levels on aspen. The SLFSBurn received both domestic and wild ungulate herbivory and had the lowest relative density and sucker height.

Developing an efficient, repeatable, and statistically reliable sampling methodology is important for evaluation of aspen regeneration efforts. The methodology developed in this study to monitor aspen density and height within pre- and post-treatment sites worked well. However, for non-randomly distributed individuals, frequency and density estimates are affected by plot size (Bonham 1989), and an alternative would be to adopt a standard plot size and vary the number of plots to address heterogeneity in particular stands. Also, we will in the future not vary the pace length of each random transect segment since this varies the sampling intensity in different parts of the stand, but instead maintain a uniform pace length. Development of a photo key for estimating pre- and post-treatment stem densities would help in selecting an appropriate plot size for each stand.

References

Barnes, B.V. 1966. The clonal growth habit of American aspens. Ecology. 47: 439-447.

Bartos, D.L.; Brown, J.K.; Booth, G.D. 1994. Twelve years biomass response in aspen communities following fire. Journal of Range Management. 47(1): 79-83.

Bartos, D.L.; Campbell, R.B. 1998. Decline in quaking aspen in the Interior West—examples from Utah. Rangelands. 20(1): 17-24.

Bartos, D.L.; Mueggler, W.F.; Campbell, R.B. 1991. Regeneration of aspen by suckering on burned sites in western Wyoming. Res. Pap. INT-448. Ogden, UT: U.S Department of Agriculture, Forest Service, Intermountain Research Station. 10 p.

Bartos, D.L.; Winward, A. 2000. Personal communications.

Brown, J.K.; Debyle, N.V. 1987. Fire damage, mortality, and suckering in aspen. Canadian Journal of Forest Research. 17: 1100-1109.

Brown, J.K.; DeByle, N.V. 1989. Effects of prescribed fire on biomass and plant succession in western aspen. Res. Pap. INT-412. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station.

Debyle, N.V.; Winokur, R.P., tech. eds. 1985. Aspen: ecology and management in the Western United States. Gen. Tech. Rep. RM-119. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station.

Dieni, J.S.; Anderson, S.H. 1997. Ecology and management of aspen forests in Wyoming; literature review and bibliography. Wyoming Cooperative Fish and Wildlife Research Unit, University of Wyoming, Laramie. 123 p.

Dieni, J.S.; Smith, B.L.; Rogers, R.L.; Anderson, S.H. 2000. Effects of ungulate browsing on aspen regeneration in northwestern Wyoming. Intermountain Journal of Sciences. 6(1): 49-55.

Fechner, G.H.; Barrows, H.S. 1976. Aspen stands as wildfire fuel breaks. Eisenhower Consortium Bulletin 4. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 26 p.

Green, A.W.; Van Hooser D.D. 1983. Forest resources of the Rocky Mountain States. Res. Bull. INT-33. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 127 p.

Gruell, G.E.; Loope, L.L. 1974. Relationship among aspen, fire, and ungulate browsing in Jackson Hole, Wyoming. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Region. 33 p.

Hart, J.H. 2000. Interaction between elk, fungi, and aspen in northwest Wyoming. In: Shepperd, Wayne D.; Binkley, Dan; Bartos, Dale L.; Stohlgren, Thomas J. Sustaining aspen in western landscapes: symposium proceedings; 2000 June 13-15; Grand Junction, CO. Proc. RMRS-P-18. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 197-205.

Kilpatrick, S.A.; Abendroth, D.C. 2000. Aspen response to prescribed fire and ungulate hervivory. In: Shepperd, Wayne D.; Binkley, Dan; Bartos, Dale L.; Stohlgren, Thomas J. Sustaining aspen in western landscapes: symposium proceedings; 2000 June 13–15; Grand Junction, CO. Proc. RMRS-P-18. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 387-394.

Krebill, R.G. 1972. Mortality of aspen on the Gros Ventre elk winter range . Res. Pap. INT-129. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Research Station. 16 p.

Merrill, E.H.; Kohley, T.W.; Herdendorf, M.E.; Reiners, W.A.; Driese, K.L.; Marrs R.W.; Anderson, S.H. 1996. GAP analysis: a biological information system. Cooperative Agreement No. 14-16-0009-1542. Laramie: University of Wyoming, Cooperative Fisheries and Wildlife Research Unit.

Mueggler, W.F. 1989. Age distribution and reproduction of Intermountain aspen stands. Western Journal of Applied Forestry. 4: 41-45.

Patton, D.R.; Avant, H.D. 1970. Fire stimulated aspen sprouting in a spruce-fire forest in New Mexico. Res. Note RM-159. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station.

Romme, W.H.; Turner, M.G.; Wallace, L.L.; Walker, J.S. 1995. Aspen, elk, and fire in northern Yellowstone National Park. Ecology. 76(7): 2097-2106.

Winward, A.; Bartos, D.L.; Shepperd, W.D. 2000. Personal communications.

Experimental Thinning and Burning of Ponderosa Pine Forests in Southwestern Colorado: Effects on Canopy Structure, Understory Composition, and Fuels

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Seven 100-acre stands were treated experimentally by (1) harvesting volume in excess of ca. 50 square feet / acre, leaving mostly larger individuals aggregated in clumps, (2) prescribed burning in spring or fall, and (3) monitoring and assessment of treatment results. Mean tree diameter increased from 14 cm to 29 cm. Lower crown height increased from 4.5 m to 6.6 m. Small woody fuels (1-hr and 10-hr TL) increased with harvest but were reduced to pre-treatment levels (mean 0.07 and 0.92 tons / acre) after burning. Larger fuels (100-hr) remained elevated (mean 3.3 tons / acre) even after burning. Mean herbaceous cover increased from 22 percent to 32 percent and species richness increased. Overall, the restoration treatments succeeded in (1) reducing canopy density and increasing average tree size, (2) reducing the hazard of crown fire and insect outbreak, and (3) rejuvenating the suppressed herbaceous stratum. The treatments failed to (1) reduce dead woody fuel loads on the forest floor, and (2) stimulate pine seedling establishment.

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Effects of Fire Interval Restoration on Carbon and Nitrogen in Sedimentary- and Volcanic-Derived Soils of the Mogollon Rim, Arizona

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Abstract—*Prescribed fire was returned into over-stocked ponderosa pine stands on the Mogollon Rim of Arizona for the purpose of restoring fire into the ecosystem and removing fuel buildups. Prescribed fires have been ignited at intervals of 1, 2, 4, 6, 8, and 10 years to determine the best fire return interval for Southwestern ponderosa pine ecosystems. Two sites were treated: one on volcanic-derived soils, and the other on sedimentary-derived soils near Flagstaff, Arizona, starting in 1976 and 1977 respectively. Samples from upper 5 cm of the A horizons were analyzed for total carbon and nitrogen using an elemental analyzer. Soil carbon and nitrogen levels were highly variable and exhibited an increasing, but inconsistent, concentration trend related to burn interval. High spatial variability measured within treatments is probably due to micro-site differences (location of samples in the open, under large old-growth trees, in small-diameter thickets, in pole-sized stands, next to downed logs, etc.). Stratification of samples by micro-site differences could possibly reduce the within-plot variability but add considerable complexity to the sampling design.*

Introduction

The pre-European settlement ponderosa pine forests of the Mogollon Rim consisted of open stands of uneven-aged trees with a significant grass-forb understory. Light surface-fires occurred on an average interval of 2 to 12 years in Arizona and New Mexico (Weaver 1951, Cooper 1960, Dietrich 1980). These fires consumed forest floor material, burned most of the young regeneration, and promoted growth of a dense, grassy understory. Catastrophic crown fires were rare due to the lack of ladder fuels and the clumpy, widely spaced ponderosa pine canopy (Dieterich 1980, Sackett 1980). Fine fuels reduction from heavy sheep and cattle grazing and then modern forest fire suppression resulted in the development of dense, overstocked stands.

Forest floor fuel loads that were 0.4-4.5 Mg / ha prior to 1870 have since increased by one to nearly two orders of magnitude. Average loadings of naturally fallen fuels were 49 Mg / ha two decades ago with some stands accumulating up to 112 Mg / ha (Sackett 1979, Sackett et al. 1996). Annual accumulations since then have been in the range of 1.3 to 7.8 Mg / ha / yr. Tree densities that were once <130 stems / ha have increased dramatically, especially in dense thickets with more than 2,750 stems / ha (Sackett 1980, Covington and Sackett 1986). Stand basal areas that were <11.5 m^2 / ha prior to removal of fire from ponderosa pine stands on the Mogollon Rim have since increased by a three- or four-fold factor (Marlin Johnson, personal communication). Ponderosa pine stands reached a critical ecological point in 1991. Fuel loads had so increased that by the end of the 20th century wildfires

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consumed four times the area that they did in the period from 1910 to 1990 (Neary et al. 1999).

Carbon and Nitrogen in Ponderosa Pine Ecosystems

Fires can greatly alter nutrient cycles of forest ecosystems depending on fire severity, fire frequency, vegetation, and climate (Neary et al. 1996). Responses of total C and N are variable and depend on the site conditions and fire characteristics. In most soils, the majority of the N pool is contained in the soil organic matter (OM). Mineral forms of N are usually lower but respond to fire. For example, Grove et al. (1986) found no change in organic C in the surface 0-1.2 in (0-3 cm) of soil immediately following burning; however, percent total N increased. Knoepp and Swank (1993) found no consistent response in total N in the upper soil layer, but increases in ammonium N (NH_4-N) concentrations and N mineralization occurred on areas where a burning treatment followed felling.

As would be expected, frequency of burning affects C accumulations. A study was carried out on tropical savanna sites in Africa having both clay and sandy soils that were burned repeatedly every 1, 3, or 5 years (Bird et al. 2000). While the clay sites had greater total C than did the sandy soils, they responded similarly to burning. All unburned sites had 40-50 percent greater C than burned sites. Low frequency burning (every 5 years) resulted in an increase in soil C of about 10 percent compared to the mean of all burned areas. High frequency burning (every year) decreased C about 10 percent. In another study, Wells et al. (1979) reported the results of a 20-year burning study in a pine plantation in South Carolina. They found that periodic burning over a 20-year period removed 27 percent of the forest floor. Annual burning conducted in the summer removed 29 percent of the forest floor as compared to a 54 percent loss resulting from winter burning. The total OM content of the surface soil (0-5 cm) increased in all cases but there was no effect on the 5-10 cm soil layer. Interestingly, when they summed the OM in the forest floor and in the surface 0-10 cm of soil they found that these low-severity periodic burns sites had not reduced, but only redistributed the OM.

Nitrogen in Ponderosa Pine Ecosystems

Prescribed fire has long been viewed as an important tool for restoring ponderosa pine stands in the Southwest (Sackett 1980, Sackett et al. 1996). The purpose of prescribed fire is to reduce fuel loads while promoting a healthy, fire-resistant, and productive forest. Sackett (1980) established a set of studies near Flagstaff, Arizona (Chimney Springs and Limestone Flats), to restore overstocked ponderosa pine stands by introducing prescribed fire at 1-, 2-, 4-, 6-, 8-, and 10-year intervals. Since ponderosa pine growth is often limited by low nitrogen (N) availability, a major concern with frequent prescribed fire is the effect on soil N pools (Powers 1980).

Nitrogen is considered the most limiting nutrient in wildland ecosystems and as such it requires special consideration when fire is managed, particularly in N-deficient ecosystems (Maars et al. 1983). Nitrogen is unique because it is the only soil nutrient that is not supplied to the soil by chemical weathering of parent material. Almost all N found in the vegetation, water, and soil of wildland systems has to be added to the system from the atmosphere. The cycling of N involves a series of interrelated complex chemical and biological processes.

Nitrogen pools can be severely disturbed by soil heating during the combustion process. Volatilization is the chemically driven process most responsible

Table 1—Soil nitrogen loss with increasing temperature (adapted from DeBano et al. 1998).

Stage	Soil temperature (° C)	Soil N loss (%)
1	<200	None
2	200-300	25-50
3	300-400	50-75
4	400-500	75-100
5	>500	100

for N losses during fire. There is a gradual increase in N loss by volatilization as temperature increases (Knight 1966, White et al. 1973). The amount of N loss at different temperatures follows the heating sequence shown in table 1. As a general rule the amount of total N that is volatilized during combustion is directly proportional to the amount of OM destroyed (Raison et al. 1985a). It has been estimated that almost 99 percent of the volatilized N is converted to N_2 gas (DeBell and Ralston 1970). At lower temperatures N_2 can be produced during OM decomposition without the volatilization of N compounds (Grier 1975). The N that is not completely volatilized either remains as part of the unconsumed fuels or it is converted to highly available ammonium nitrogen (NH_4-N) that remains in the soil (DeBano et al. 1979, Covington and Sackett 1986, DeBano 1991).

Estimates of the total N losses during prescribed fire must be based on both fire behavior and total fuel consumption because irregular burning patterns are common. As a result, combustion is not complete at all locations on the landscape (DeBano et al. 1998). For example, total N loss was studied during a prescribed burn in southern California (DeBano and Conrad 1978). In this study, only 10 percent of the total N contained in the plant, litter, and upper soil layers was lost. The greatest loss of N occurred in aboveground fuels and litter on the soil surface. In another study of N loss during a prescribed fire over dry and moist soils, about two-thirds of the total N was lost during burns over dry soils compared to only 25 percent when the litter and soil were moist (DeBano et al. 1979). Although these losses were relatively small, it must be remembered that even small losses can adversely affect the long-term productivity of N-deficient ecosystems.

Monleon et al. (1997) conducted understory burns on ponderosa pine sites burned 4 months, 5 years, and 12 years previously. The surface soils, 0 to 5 cm, showed the only significant response. The 4-month burned sites had increased total C and inorganic N following burning, and an increased C / N ratio. Burning the 5-year-old sites resulted in a decrease in total soil C and N, and a decrease in the C / N ratio. Total soil C and N in the surface soils did not respond to burning on the 12-year-old site.

Nitrogen Losses — An Enigma

It has been conclusively established by numerous studies that total N is decreased as a result of combustion (DeBano et al. 1998). The amount of N lost is generally proportional to the amount of OM combusted during the fire. The temperatures at which N is lost are discussed above. In contrast, available N is usually increased as a result of fire, particularly NH_4-N (Christensen 1973, DeBano et al. 1979, Carballas et al. 1993). This increased N availability enhances post-fire plant growth, and gives the impression that more total N is present after fire. Increased fertility, however, is misleading and short-lived. Temporary increase in available soil N following fire is usually rapidly utilized by plants and microorganisms in the first few years after burning.

The consequences of N losses during fire on ecosystem productivity depend on the proportion of total N lost for a given ecosystem (DeBano et al. 1998). In N-limited ecosystems even small losses of N by volatilization can impact long-term productivity. Consequently, a key ecosystem parameter that has been studied in the ponderosa pine restoration study established by Sackett (1980) is N.

Covington and Sackett (1986, 1992) examined N concentrations in the upper 5 cm of mineral soil at the Chimney Springs burning interval study (Sackett 1980). They found that mineral forms of N (NH_4-N and NO_3-N) made up <2% of the total N pool. Burning at 1- and 2-year intervals significantly increased only NH_4-N levels in the soil. Total soil N in the upper 5 cm was not affected by prescribed fire interval. A later study (Wright and Hart 1997) assessed the effects of the two-year burning interval at the Chimney Springs site. It inferred that repeated burning at two-year intervals may have detrimental long-term effects on N cycling, along with depletion of the forest floor and surface mineral soil C and N pools.

Methods

Study Sites

The original study sites established in 1976 and 1977 were designed to determine the optimum-burning interval necessary to provide continuous fire hazard reduction. These studies are described in greater detail by Sackett (1980), Covington and Sackett (1986), and Sackett et al. (1996). Sites were selected on volcanic soils at Chimney Springs, Fort Valley Experimental Forest, north of Flagstaff, Arizona, and sedimentary soils at Limestone Flats, Long Valley Experimental Forest, near Clint's Well, Arizona. Twenty-one 1.0 ha plots make up each study site. There are three replications of unburned (control), and 1-, 2-, 4-, 6-, 8-, and 10-year prescribed fire treatments. All of the burn rotation treatments, except for the 10-year rotation and controls, were burned the previous October (2001).

Chimney Springs

The Chimney Springs study is located in the Fort Valley Experimental Forest, Rocky Mountain Research Station, Coconino National Forest about 3 km northwest of Flagstaff, Arizona. Soils are Brolliar stony clay loam, a fine, smectic, frigid Typic Argiboroll derived from basalt and cinders (Meurisse 1971). Stand structure and fuels are described by Sackett (1980). The original ponderosa pine stand was virtually undisturbed by wildfire since 1876 but was grazed in the late 19th century and placed under fire control. At the initiation of the study, the ponderosa pine stand consisted of reproduction (976 stems / ha), saplings (2,752 stems / ha), pole-sized trees (771 stems / ha), and old growth (dbh >28 cm, 133 stems / ha). The basal area was 33.0 m^2 / ha in trees >10 cm dbh. The original fuel load of dead surface and ground fuels was 34.0 Mg / ha.

Limestone Flats

The Limestone Flats study is located in the Long Valley Experimental Forest, Rocky Mountain Research Station, Coconino National Forest, about

2 km northwest of Clint's Well, Arizona. Soils are very fine sandy loam textured, fine, smectic, Typic Cryoboralfs. These soils developed from weathered sandstone with limestone inclusions. Stand structure and fuels are described by Sackett (1980). The original ponderosa pine stand was treated with a sanitation cutting in the mid 1960s to remove trees attacked by insects and disease. It was also grazed in the late 19th century, and placed under fire control, but grazing had been eliminated many years prior to 1976. The ponderosa pine stand consisted of reproduction (1,373 stems / ha), saplings (2,881 stems / ha), pole-sized trees (388 stems / ha), and old growth (dbh >28 cm, 82 stems / ha). The basal area was 22.5 m^2 / ha in trees >10 cm dbh. The original fuel load of dead surface and ground fuels was 34.9 Mg / ha.

Soil Sampling

The soils at both the Chimney Springs and Limestone Flats sites were sampled in late December 2002. The initial sampling location was randomly selected within the center 400 m^2 of each plot. The next two samples were located 5 m from the first sample, selected by a randomization process, on two of the cardinal directions from the first sample. The locations were not stratified by stand structure or other site features as was done in the study by Covington and Sackett (1986).

Approximately 500 g was collected from the 0-5 cm depth of the mineral soil. The samples were air dried in the laboratory, sieved to a size of <2 mm, and sub-sampled for analysis. Sub samples were ground to a 40 mesh particle size then oven dried at 40^{o} C.

Carbon and Nitrogen Analysis

Soil total C and N were analyzed on a Thermo-Quest Flash EA1112 C-N analyzer. The computer-controlled instrument oxidizes samples at 1,500^{o} C, separates CO_2 and NO_2, by gas chromatography on a packed column, and determines C and N content with a thermal conductivity detector. Analysis was performed using a standard protocol for this instrument, which includes blanks, certified soil standards, and quality control samples during operations.

Statistical Analysis

Data were analyzed using the SAS univariate ANOVA under the GLM Procedure (SAS 2000) and Tukey's Studentized Range test for means separation of C and N values ($p = 0.05$). The plot design is 6 treatments (burn intervals 1, 2, 4, 6, 8, 10 years) and a control (unburned) times 3 replicates of each treatment and control.

Results and Discussion

Carbon

Total soil C levels in the Limestone Flats and Chimney Springs 0-5 cm horizon exhibit two trends (figure 1). The first is that soil C is significantly higher at Chimney Springs (table 2). The initial forest floor fuel loading (34.0 Mg / ha) was actually lower than the Limestone Flats loading (34.9 Mg / ha). At the start of the study in 1976, the Chimney Springs site had a higher basal area and nearly double the density of pole and old growth trees (Sackett 1980). Covington and Sackett's (1986) stratified sampling indicated higher levels of

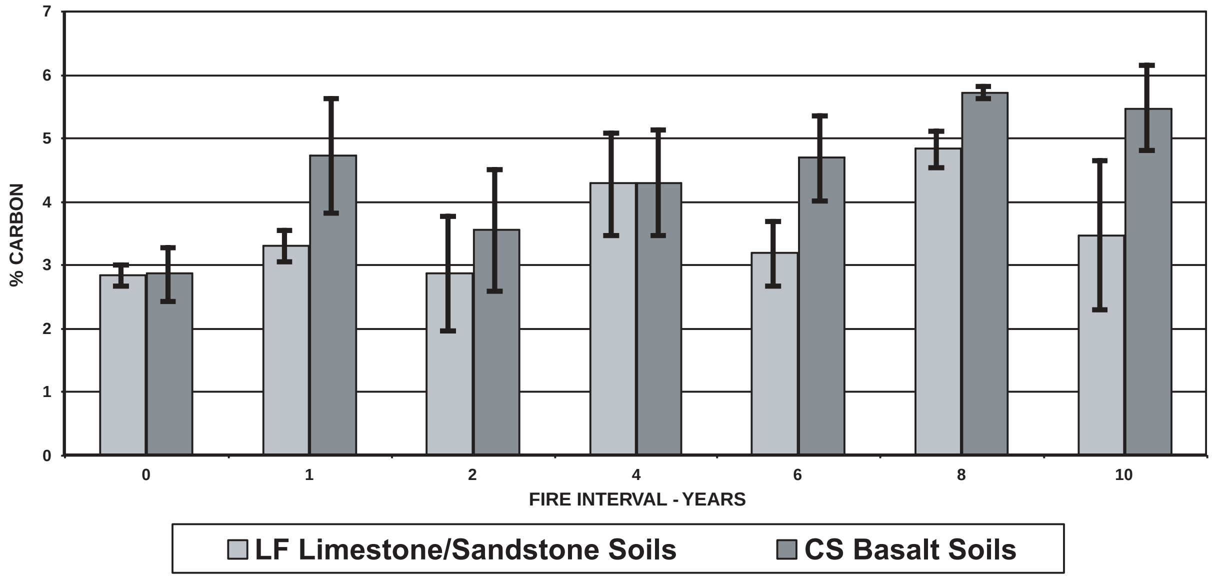

Figure 1—Effect of fire interval on soil total carbon (mean and standard error), Limestone Flats and Chimney Springs burning interval study, Arizona.

Table 2—Studentized Tukey's test for C and N by location, Limestone Flats and Chimney Springs, Arizona, burning interval restoration studies.

Element	Location	Mean (%)	Tukey's test (p = 0.05)	N
Carbon	Limestone Flats	3.543	A	21
	Chimney Springs	4.478	B	21
Nitrogen	Limestone Flats	0.221	A	21
	Chimney Springs	0.287	B	21

N (hence C) in old-growth stands. The random nature of the sampling in this study may have picked up more of the sites at Chimney Springs that Covington and Sackett (1986) identified as "sawtimber" (old growth). Soil classification also explains the difference between the carbon in the Limestone Flats and Chimney Springs soils. The latter were classified as Argiborolls belonging to the Mollisol soil order, indicating that they have naturally higher organic matter contents than the Cryoboralfs (Alfisol soil order) found at Limestone Flats.

The second trend in the soil C data appears to be that burning at the 8-year interval produced statistically significant higher soil C levels than the controls and that burning in general increases total C in the mineral soil (table 3). The C concentration in the soil increased from 2.856% in the control to 5.277% in the 8-year burning interval. However, only the control and 8-year interval are statistically different. These data reflect more of the variability in soil C detected in this random sampling approach than any burning interval trend. It is evident that the prescribed fires reintroduced into the two sites have increased soil C. Sackett et al. (1996) concluded that the best burning interval was 4 years for reducing fuel loads. That interval produced the intermediate C level in the 0-5 cm depth of the mineral soil.

Nitrogen

Total soil N levels followed a similar trend as total soil C (figure 2). Total soil N concentrations were mostly higher across the range of burning

Table 3—Studentized Tukey's test for C and N by treatment, Limestone Flats and Chimney Springss, Arizona, burning interval restoration studies.

Element	Burning interval (years)	Mean (%)	Tukey's test (p = 0.05)	N
Carbon	0	2.856	A	6
	2	3.210	AB	6
	6	3.942	AB	6
	1	4.024	AB	6
	4	4.294	AB	6
	10	4.476	AB	6
	8	5.277	B	6
Nitrogen	0	0.188	A	6
	2	0.212	A	6
	6	0.228	AB	6
	1	0.242	AB	6
	10	0.281	AB	6
	4	0.298	AB	6
	8	0.330	B	6

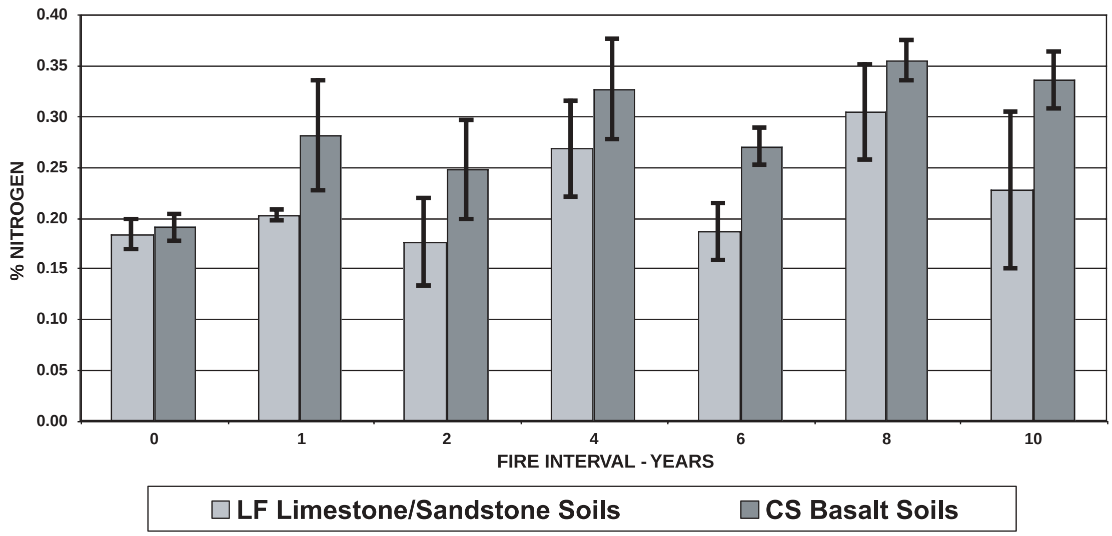

Figure 2—Effect of fire interval on soil total nitrogen (mean and standard error), Limestone Flats (LF) and Chimney Springs (CS) burning interval study, Arizona.

intervals. Concentrations increased from an average of 0.188% in the control plots to 0.330% in the 8-year burning interval (table 3). Soil N at Chimney Springs with Typic Argiboroll soils was significantly different from Limestone Flats with Cryoboralf soils (table 2). Significant differences in total N concentrations where found between control and 2-year burning interval and the 8-year burning interval plot (table 3).

Covington and Sackett (1986) reported that <2% of the soil N measured in their mid 1980s sampling was mineralized N (NH_4-N and NO_3-N). The data from this sampling conflict with Wright and Hart's (1997) hypothesis that burning at 2-year intervals may have detrimental long-term effects on N cycling, along with depletion of the forest floor and surface mineral soil C and N pools. The 2-year burning interval was not significantly different from the control or other burning intervals, only the 8-year burning interval. Wright and Hart (1997) did not investigate the 1-year burning interval, yet our sampling showed it to be at an intermediate level of N in the 0-5 cm horizon. The

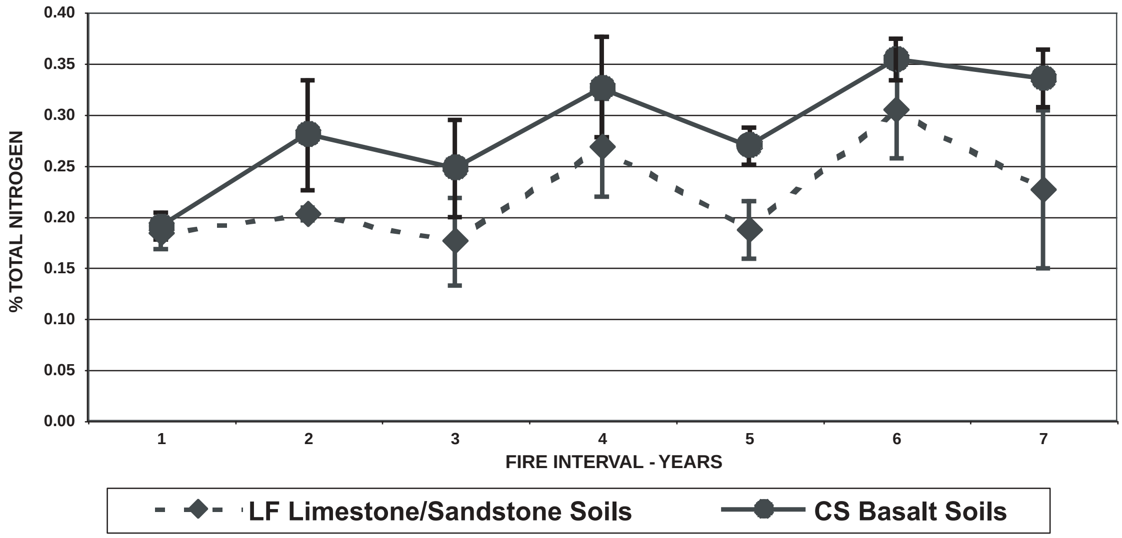

Figure 3—Total nitrogen (mean and standard error) in the A horizon for the Limestone Flats (LF) and Chimney Springs (CS) burning interval study, Arizona.

soil N pool does not provide a readily available source of N to plants and microorganisms because of the slow decomposition rates in these semi-arid ecosystems. This limitation, rather than any declines in the total soil N pool, may account for the N enigma that DeBano et al. (1998) discuss.

Sample Variability

The lack of a strong burning interval response in this study was most likely affected by site variability and the random sampling used. To obtain an understanding of the variability in soil total C and N, individual plot data is quite instructive (figure 4).

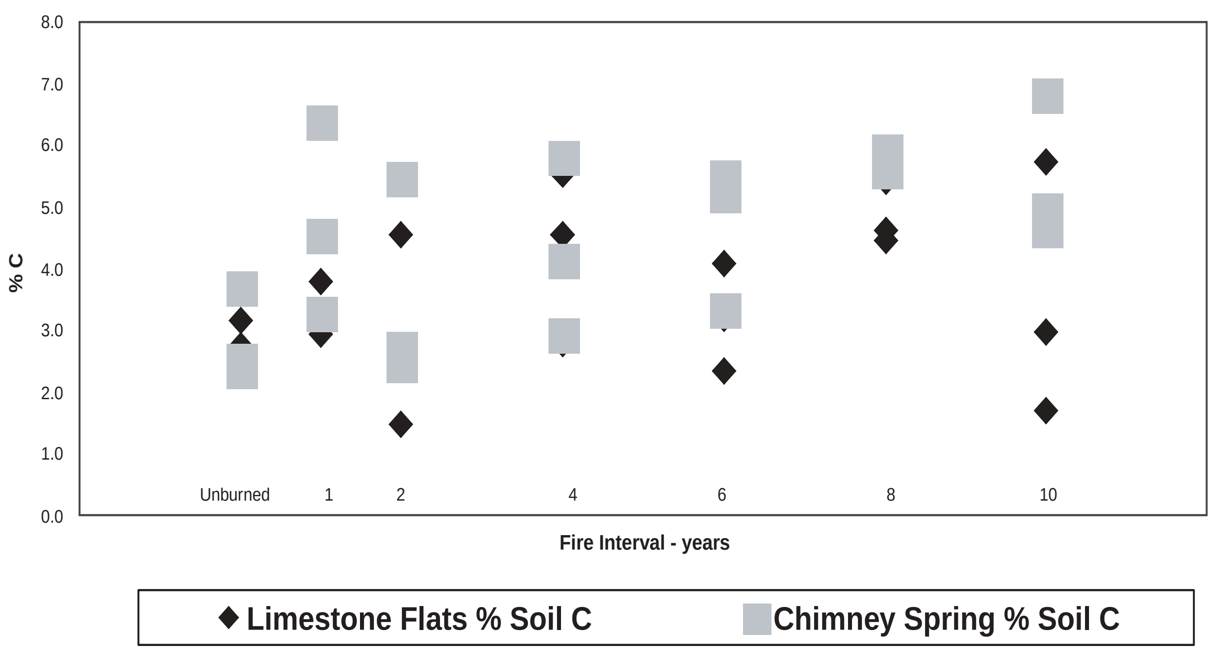

Figure 4—Variability in percent C in mineral soil, Limestone Flats and Chimney Springss burning interval study, Arizona.

The 1-year burning interval plot samples for total C at Limestone Flats ranged from 2.933% to 3.796%, a span of 0.863%. The unburned control samples had a range from 2.630% to 3.160%, a similar span of 0.530%. The 10-year burning interval plots at Limestone Flats had the highest variability. Total soil C in the 0 to 5 cm depth ranged from 1.717% to 5.709%, a span of 3.991%. The unburned control plot samples at Chimney Springs had a range from 2.367% to 3.711%, a span (1.344%) nearly triple that of the Limestone Flats control. Within plot variability was much higher at Chimney Springs than at Limestone Flats (figure 4).

The total C and N variability observed from the random samples at the Chimney Springs and Limestone Flats sites was probably influenced by a number of factors. Covington and Sackett (1986) stratified their sampling at Chimney Springs by stand type (e.g., sawtimber, poles, and saplings). It was very evident during the sampling that there were visually evident differences in the levels of litter accumulations and OM concentrations in the mineral soil under these three different stand types. In addition, several other factors appeared to be important. Samples collected in the middle of clearings and next to decaying, but not completely burned, logs had visually apparent differences in color that reflected OM content. Another factor that could be important, but was not readily discernable on the ground, is the presence of “hot spots” where dead and decaying logs were at some point in time completely combusted by the prescribed fires. These logs would create zones of high fire severity that would burn much of the soil OM and drive off most of the surface mineral soil N (DeBano et al. 1998).

Our recommendation as a follow-up to this study is to resample using Covington and Sackett’s (1986) stand classification approach (i.e., sawtimber, poles, and saplings), but add in areas such as clearings, decaying logs, and high-severity burn spots. Using a composite sample of several cores would also aid in the leveling of variability of the samples. While the classification does allow easy scaling up to stand and landscape levels, the other categories do not. That is why random sampling is still of interest. Some work is still needed to determine sample sizes needed to detect differences between the individual burning intervals, if such differences exist at all.

Summary and Conclusions

The effects of burning intervals for restoration of ponderosa pine stands on total C and N concentrations in the 0-5 cm horizon of two different soil types was examined. The burning intervals (unburned, 1, 2, 4, 6, 8, and 10 years) were provided by a study established in 1976 and 1977 and have been maintained thereafter (Sackett 1980, Sackett et al. 1996). Although there were statistically significant differences between the total C levels in soils of the unburned plots and the 8-year burning interval, there were no differences between burning intervals. There also was a statistically significant difference between unburned and 2-year burning interval and the 8-year burning interval in total soil N. This study determined that burning increased mineral soil C and N, which conflicted with Wright and Hart’s (1997) contention that the 2-year burning interval could deplete soil N and C pools. This study did not examine the mineral fractions of the soil N pool, NH_4-N, and NO_3-N. Although the mineral forms of N are small (<2 % of the total soil N pool), they are very important for plant nutrition and microorganism population functions. It is recommended that the study be repeated contrasting stratified sampling and higher intensity random sampling approaches.

References

Bird, M.I.; Veenendaal, E.M.; Moyo, C.; Lloyd, J.; Frost P. 2000. Effect of fire and soil texture on soil carbon in a sub-humid savanna (Matopos, Zimbabwe). Geoderma. 9: 71-90.

Carballas, M.; Acea, M.J.; Cabaneiro, A.; Trasar, C.; Villar, M.C.; Diaz-Ravina, M.; Fernandez, I.; Prieto, A.; Saa, A.; Vazquez, F.J.; Zehner, R.; Carballas, T. 1993. Organic matter, nitrogen, phosphorus and microbial population evolution in forest humiferous acid soils after wildfires. In: Trabaud, L.; Prodon, P., eds. Fire in Mediterranean Ecosystems. Commission of the European Countries, Ecosystem Research Report 5. Brussels, Belgium.

Christensen, N.L. 1973. Fire and the nitrogen cycle in California chaparral. Science. 181: 66-68.

Cooper, C.E. 1960. Changes in vegetation, structure and growth of southwestern pine forests since white settlement. Ecological Monograms. 30: 129-164.

Covington, W.W.; Sackett, S.S. 1986. Effect of burning on soil nitrogen concentrations in ponderosa pine. Soil Science Society of America Journal. 50: 452-457.

Covington, W.W.; Sackett, S.S. 1992. Soil mineral nitrogen changes following prescribed burning in ponderosa pine. Forest Ecology and Management. 54: 175-191.

DeBano, L.F. 1991. The effect of fire on soil. In: Harvey. A.E.; Neuenschwander, L.F., eds. Management and productivity of western-montane forest soils. Gen. Tech. Rep. INT-280. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 32-50

DeBano, L.F.; Conrad, C.E. 1978. The effect of fire on nutrients in a chaparral ecosystem. Ecology. 59: 489-497.

DeBano, L.F.; Eberlein, G.E.; Dunn, P.H. 1979. Effects of burning on chaparral soils: I. Soil nitrogen. Soil Science Society of American Journal. 43: 504-509.

DeBano, L.F.; Neary, D.G.; Ffolliott, P.F. 1998. Fire’s effects on ecosystems. New York: John Wiley & Sons, Inc. 333 p.

DeBell, D.S.; Ralston, C.W. 1970. Release of nitrogen by burning light forest fuels. Soil Science Society of America Proceedings. 34: 936-938.

Dieterich, J.H. 1980. Chimney Springs forest fire history. Gen. Tech. Rep. RM-278. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 44-48.

Grier, C.C. 1975. Wildfire effects on nutrient distribution and leaching in a coniferous ecosystem. Canadian Journal of Forestry Research. 5: 599-607

Grove, T.S.; O’Connell. A.M.; Dimmock, G.M. 1986. Nutrient changes in surface soils after an intense fire in jarrah (*Eucalyptus marginata* Donn ex Sm.) forest. Australian Journal of Ecology. 11: 303-317.

Knight, H. 1966. Loss of nitrogen from the forest floor by burning. Forestry Chronicle. 42: 149-152.

Knoepp, J.D.; Swank, W.T. 1993. Site preparation burning to improve southern Appalachian pine-hardwood stands: Nitrogen responses in soil, soil water, and streams. Canadian Journal of Forest Research. 23: 2263-2270.

Maars, R.H.; Roberts, R.D.; Skeffinton, R.A.; Bradshaw, A.D. 1983. Nitrogen in the development of ecosystems. In: Lee, J.A.; McNeill, S.; Rorison, I.H., eds. Nitrogen as an ecological factor. Oxford, England: Blackwell Science Publishing: 131-137.

Meurisse, R.T. 1971. Soil report on the San Francisco Peaks area. Flagstaff, AZ: U.S. Department of Agriculture, Forest Service, Elden and Flagstaff Ranger Districts, Coconino National Forest.

Monleon, V.J.; Cromack, K., Jr.; Landsburg, J.D. 1997. Short- and long-term effects of prescribed underburning on nitrogen availability in ponderosa pine stands in central Oregon. Canadian Journal of Forest Research. 27: 369-378.

Neary, D.G.; Klopatek, C.C.; DeBano, L.F.; Ffolliott, P.F. 1999. Fire effects on belowground sustainability: a review and synthesis. Forest Ecology and Management. 122: 51-71.

Neary, D.G.; Overby, S.T.; Gottfried, G.J.; Perry, H.M. 1996. Nutrients in fire-dominated ecosystems. Gen. Tech. Rep. RM-289. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 107-117.

Powers, R.F. 1980. Mineralizable soil nitrogen as an index of nitrogen availability for forest trees. Soil Science Society of America Journal. 44: 1314-1320.

Raison, R.J.; Khanna, P.K.; Woods, P.V. 1985. Mechanisms of element transfer to the atmosphere during vegetation fires. Canadian Journal of Forest Research. 15: 132-140.

Sackett, S.S. 1979. Natural fuel loadings in ponderosa pine and mixed conifer forests of the Southwest. Res. Pap. RM-213. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 10 p.

Sackett, S.S. 1980. Reducing natural ponderosa pine fuels using prescribed fire: Two case studies. Res. Pap. RM-392. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 6 p.

Sackett, S.S.; Haase, S.M.; Harrington, M.G. 1996. Lessons learned from fire use: restoring southwestern ponderosa pine ecosystems. Gen. Tech. Rep. RM-278. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 54-61.

SAS. 2000. GLM Procedure, SAS Institute, Cary, NC. Version 8.1

White, E.M.; Thompson, W.W.; Gartner, F.R. 1973. Heat effects on nutrient release from soils under ponderosa pine. Journal of Range Management. 26: 22-24.

Weaver, H. 1951. Fire as an ecological factor in the southwestern ponderosa pine forests. Journal of Forestry. 49: 93-98.

Wells, C.G., Campbell, R.E.; DeBano, L.F.; Lewis, C.E.; Fredrickson, R.L.; Franklin, E.C.; Froelich, R.C.; Dunn, P.H. 1979. Effects of fire on soil: a state-of-the-knowledge review. Gen. Tech. Rep. WO-7. Washington, DC: U.S. Department of Agriculture, Forest Service. 34 p.

Wright, R.J.; Hart, S.C. 1997. Nitrogen and phosphorus status in a ponderosa pine forest after 20 years of interval burning. Ecoscience. 4: 526-533.

Mt. Trumbull Ponderosa Pine Ecosystem Restoration Project

Ken Moore[1], Bob Davis[2], and Timothy Duck[3]

Abstract–*This paper describes recent and current studies on the Parashant National Monument in northern Arizona. These studies support the Mt. Trumbull Ponderosa Pine Ecological Restoration Project. A major goal of the project is to allow wildfire to resume its historic role in maintaining forest health following decades of declining sustainable conditions.*

Background

The frequent, low intensity fire regime that naturally occurred in southwestern ponderosa pine forests (Covington et al. 1999) was disrupted when European-American settlers arrived in the area in the late 1800s and early 1900s. Livestock grazing removed fine fuels that carry fire and logging altered the age class structure of forests. Wildfire suppression reduced the number of acres burned and increased the fire return interval. As a result, much of the ponderosa pine forest in the southwest is functioning outside of the range of natural variability and is in poor and declining health.

The most famous use of timber from the area was for use in the Church of Jesus Christ of Latter Day Saints' St. George Temple. Due to the technological limitations of logging using mule and oxen to skid in the remote, rugged mountains, as well as those imposed by the use of portable steam sawmills (which were not able to handle the largest trees), a sizeable portion of the old growth was not harvested and remains standing today.

Today the forest consists of the remnants of the pre-settlement forest along with approximately 100 years of regeneration. The previously open, park-like stand, with tree densities around 50 per acre (Mast et al. 1999), has changed to "dog hair," with stand densities over 400 trees per acre. These stands are susceptible to disease, insect infestations, catastrophic wildfire, and drought (figure 1).

Introduction

For the past seven years, the Bureau of Land Management, working with the Ecological Restoration Institute (ERI) at Northern Arizona University and the Arizona Game and Fish Department, have been implementing a large scale, ponderosa pine ecosystem restoration project in the Mt. Trumbull / Mt. Logan area.

The Mt. Trumbull Ponderosa Pine Ecosystem Restoration Project involves public lands managed by the Bureau of Land Management within the newly designated Grand Canyon-Parashant National Monument in northern

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Figure 1—“Dog hair” stand of post-settlement ponderosa pines at Mt. Trumbull, Arizona.

Arizona. As a “sky island,” the approximately 17,000 acre ponderosa pine forest sits atop 7,000-8,000 ft mountains amid a piñon-juniper / sagebrush / grassland community. The forest contains 500-year-old ponderosa pine along with old growth piñon, juniper, and New Mexican locust.

The goals of the Mt. Trumbull restoration project are to 1) restore pre-settlement ecosystem health and function to a ponderosa pine forest in the recently designated Parashant National Monument, just north of the Grand Canyon; 2) reduce fuel loads and disrupt fuel continuity to reduce the risk of catastrophic wildfire; and 3) gather information on a wide variety of ecosystem components and processes in order to understand the effects of restoration treatments. By reducing fuel loads and reestablishing an herbaceous understory we hope to one day restore natural fire processes. Treatments include merchantable and non-merchantable timber harvesting, prescribed fire, and seeding of native herbaceous species. Due to the existing high fuel loads, a variety of techniques have been used to reduce mortality of pre-settlement ponderosa pine trees from treatment activities.

The project includes an adaptive management provision that allows managers to incorporate lessons learned into future treatments. For example, current treatments leave more trees than the earliest efforts. These additional “insurance trees” allow for some mortality from natural and anthropogenic impacts (such as prescribed burning).

The project can be broken out into seven distinct phases or efforts:

1. Project planning and regulatory compliance
2. Ecosystem inventory and timber marking
3. Merchantable and non-merchantable thinning
4. Prescribed burning
5. Restoration of native grasses and forbs by seeding
6. Maintenance of periodic low intensity grass fires in restored areas
7. Monitoring (occurs throughout all phases of the project)

Project Planning and Regulatory Compliance

This project was initiated prior to the designation of the area as a National Monument. Current efforts must balance the needs of the restoration experiment with Monument values. In order to ensure that project impacts do not impair Monument values and to provide our public interests with sufficient information to understand the project and its anticipated effects, we prepare environmental assessments for all treatment activities.

Ecosystem Inventory and Timber Marking

Table 1 shows pretreatment forest stand density data compiled from 128 20 × 50 meter ecosystem monitoring plots established by Northern Arizona University within all existing and proposed ecosystem restoration units. In these plots researchers tallied all stems of ponderosa pine, gambel oak (*Quercus gambelii*), juniper (*Juniperus osteosperma*), and piñon pine (*Pinus edulis*) ≥1 inch dbh. Stems / plot were extrapolated to stems / acre for the entire restoration

Table 1—Pretreatment forest stand density.

Unit(# plots)	Overstory ponderosa	Pole ponderosa	Overstory oak	Pole oak	Overstory: juniper/piñon	Pole: juniper/piñon	Overstory locust	Pole locust	Total
Lava (3)	84	108	46	17	- / 4	- / -	-	11	270
Trick Tank (11)	131	83	82	111	- / -	- / 6	1	15	429
Rye Flat (10)	126	100	42	67	7 / 2	7 / 5	-	-	356
N. Sawmill (8)	229	263	7	44	29 / 2	67 / 27	-	-	668
High Meadow (16)	117	103	74	166	6 / 2	9 / 13	3	6	499
Lower Sawmill (5)	57	26	62	76	26 / 9	29 / 16	-	10	311
EB-1 (4)	55	33	3	5	- / -	- / -	-	-	96
EB-2 (4)	88	57	117	50	- / -	- / -	-	-	312
EB-3 (5)	62	48	-	-	- / 1	- / -	-	-	111
EB-4 (2)	260	577	-	41	10 / -	33 / 33	-	-	954
EB-5 (6)	168	89	22	22	29 / 2	54 / -	-	-	386
Cinder (8)	87	27	33	22	- / -	- / 3	1	-	173
Hi-Lo (13)	134	103	20	73	14 / 7	20 / 15	20	10	416
Nixon (9)	124	100	20	4	18 / 4	38 / 13	-	-	321
Spring (10)	64	26	15	34	30 / 18	23 / 55	-	-	265
Boundary (4)	55	11	16	11	19 / 18	49 / 43	-	-	222
War Club (4)	104	106	1	16	22 / 3	174 / 24	-	-	450
Corner (6)	154	252	2	16	25 / 10	68 / 25	-	-	552
Mean stems/acre	117	117	31	43	13 / 4	31 / 15	1	3	377

units. The table shows overstory stems (≥5.9 in. dbh) and pole stems (≥1 in., ≤ 5.9 in. dbh) and total stems per acre for the restoration units.

Ecosystem monitoring plots used to collect the data shown in table 1 are based on National Park Service long term vegetation and fuels monitoring plots as described in the Western Region Fire Monitoring Handbook (1992). Under conventions described in this handbook, all trees greater than or equal to 5.9 inches dbh are classed as overstory trees. Trees that are greater than 1 inch and less than 5.9 inches are classed as pole trees. Growth increment core samples, used to determine age, are taken from all live pre-settlement trees on each long term ecosystem monitoring plot. In addition, a 10% random sample of all living post-settlement age trees are cored for accurate age determination. This information is available upon request from the BLM, St. George Field Office.

The ecosystem monitoring plots established by NAU in proposed treatment areas and in the control area show that current forest conditions are dense. Overstory, pole, and regeneration data show a mean of 569 trees / acre in the treatment unit plots and 600 trees / acre in the control unit plots. Treatment unit plots and control unit plots show very similar composition before implementation of restoration prescriptions on treatment areas, with 47% and 48% ponderosa pine, respectively and 30% and 33% gambel oak, respectively. Oak is a dominant species in terms of tree density, but makes up only 10.5% and 13% of the basal area in treatment and control unit plots respectively. Juniper and piñon pine have higher densities in treatment unit plots. New Mexico locust appears to have higher densities in the control unit plots but variability is very high due to the patchy distribution of locust. In terms of basal area, juniper, piñon, and locust collectively contribute less than 10% of the stand basal area on treatment unit plots, and less than 3% on control unit plots (NAU 1997).

Diameter distributions show that 50% of current live trees are less than 20 cm (7.8 in.) dbh. Of these smaller trees, ponderosa pine are the major component. Numbers of non-ponderosa pine drop off at the higher diameter classes (NAU 1997). NAU has reconstructed the forest structure from long term ecosystem monitoring plots. 1870 was chosen as the pre-disturbance date based on preliminary fire scar data, which shows the last widespread naturally occurring fire to be in 1870, in addition to logging and grazing records, which place domestic livestock grazing in the Mt. Trumbull area around 1870 (Altschul and Fairley 1989). Increment cores were taken from all live trees identified as pre-settlement in the field, based on conservative size criteria and bark coloration. After laboratory analysis of the tree core data, age data were used to confirm or reject the field determination. For each cored pre-settlement tree of any species, the 1870-1996 radial growth increment was measured on the core and the 1870 diameter at breast height was calculated. For dead trees (snags and logs), the year of death of each tree was estimated based on tree diameter and condition class. To determine the 1870 diameter of dead trees, growth estimates over the 1870 to death date periods were applied. For these preliminary data, NAU used a predictive regression relationship between diameter and basal area increment that was developed for pre-settlement pine trees from samples taken from Camp Navajo Army Depot, Bellemont, AZ.

Results from plots show the reconstructed 1870 forest had an average of 22 trees / acre (55 trees / ha) versus 349 trees / acre (863 trees / ha) in 1996 for all species, while basal area in the reconstructed forest was only 35.7 ft^2 / acre versus 130.7 ft^2 / acre in 1996. These values are within the low range of pre-settlement forest densities and basal areas found in ponderosa pine forests in the southwest. Pine density in the Mt. Trumbull area falls within the range of

7-116 ponderosa pine / ha reported in several early National Forest inventories and other studies in the region (Covington and Moore 1994).

Merchantable Timber Harvest and Non-Merchantable Thinning

BLM uses a combination of timber sales, thinning contracts, and its own workforce to reduce the density of trees within restoration units. Only post-settlement trees are cut. Timber sales remove between 2,500 and 3,000 board ft / ac from the units. The primary products are boards, poles, palettes, grape trellis, sawdust, and fuelwood. Trees less than 18" DBH are logged at a loss, which is amortized by profits from larger trees (up to 28"). Loggers are required to protect residual trees from their harvest operations. Loggers are hauling 12"-28" trees, all post-settlement.

Early in the project BLM reduced the maximum size of ponderosa that was allowed to be cut from 28" to 22" and received no bids; the project lost one year due to the failure to attract any commercial partners. We now include larger (up to 28" DBH) post-settlement trees, although few of the trees cut are of the maximum size. We now mark more leave trees than we did originally to provide insurance against losses during burning and logging operations.

In addition, ladder fuels composed of post-settlement ponderosa pine, piñon pine, or juniper trees of less than 10 inches diameter breast height (dbh) are thinned with power saws. Post-settlement trees are limbed with power saws to reduce fire spread from the forest floor into the canopy. Pre-settlement trees are limbed to a height of approximately 6-8 feet. Snags greater than 12 inches dbh are prepped (e.g., fireline is constructed around these features) in an effort to reduce ignition.

The BLM fuels reduction crew prepares units for burning by reducing the height of any activity fuels, thinning unmerchantable trees, preparing firelines, and raking litter from the base of pre-settlement trees. Logging slash is lopped and scattered, then a low intensity broadcast burn is started under environmental conditions conducive for burning (hot and dry enough to allow for consumption of litter, but not so hot and dry as to allow tree mortality from scorch height or canopy damage).

Consumption of deep layers of forest floor litter generates high amounts of heat energy. Studies at Fort Valley and Long Valley Experimental Forests showed very high mineral soil temperatures during burning. Smoldering combustion takes place in deep layers of forest litter and may continue for durations of 72 hours. Burning for this length of time can result in either temperatures exceeding 140°F, causing cambium death, or lower temperatures for longer durations, which also cause tissue death (Harrington and Sackett 1988).

Prescribed Burning

Following preparation of burn units, BLM introduces low intensity prescribed fires to reduce duff depth from a range of 1-10" to a range of .25-2", and to reduce fuel loading from an average of 10 tons per acre to a range of 1-6 tons per acre. To prevent degradation of the Grand Canyon airshed, favorable winds that carry particulate matter away from the canyon are necessary. Sagebrush that exists in places as a significant understory shrub are not excluded from prescribed fire.

Table 2—Prescription parameters in the use of prescribed fire.

Prescription parameters (forest understory)	Prescribed fire behavior Cool	Hot	Desired	Outside (max.)
Fuel model (s)	9	9	9	9
Live fuel moisture (%)	n/a	n/a	n/a	n/a
1000 hr. moisture (%) used in lieu of duff moisture	16% = 50% duff reduction	6% =90% duff reduction	10% =75% duff reduction	6%
Soil moisture (%)	n/a	n/a	n/a	n/a
Rate-of-spread (ch/hr)	2 chains/hour	2-6 chains/hour	2 chains/hour	2-6 chains/hour
Flame length (ft)	1 foot	2-3 feet	2 foot	2-3 foot
Scorch height (ft)	2 feet	3-4 feet	1 foot	3-4 feet
Type of fire	strip headfire	backing	backing	
Probability of ignition (%)				60%
Spotting distance (mi)				.3 - .4 mile

Table 3—Environmental parameters in the use of prescribed fire.

Environmental parameters (forest understory)	Acceptable range Cool	Hot	Desired	
Air temperature (degrees F)	30˚	75˚	50˚	
Relative humidity (%)	46%	20%	35%	
Wind speed (midflame, mph)	2	8	5	Outside burn block
Wind direction	west	northeast, northwest, east	south, southwest, southeast	at critical holding point—minimum acceptable moisture
1 hr fuel moisture (%)	11	5	7	5
10 hr fuel moisture (%)	n/a	n/a	n/a	n/a
1000 hr fuel moisture (%)	16	6	10	6

Restoration of Native Grasses and Forbs by Seeding

The herbaceous and shrub understory of southwestern ponderosa pine forests is important for many ecological reasons, including food and habitat for herbivores, nutrient cycling, soil formation and stabilization, and its contribution to biological diversity and aesthetic appeal. Understory species composition and cover have been adversely modified since Euro-American settlement and understory production has declined with increasing tree density. Key to restoring the understory is an understanding of the on-site plant material, including above ground vegetation as well as viable seeds in the soil bank.

To obtain a baseline of the species in the current soil seed bank at Mt. Trumbull, soil samples were collected in September 1997 from 5 vegetation types within three of the experimental block units. Soil samples were collected in April 1997 from unburned, lightly burned, and severely burned patches within the 33 acre subunit of the 96-1 restoration unit. Samples were placed in the NAU greenhouse for several months. Germination from these samples was compared to samples taken in a control (untreated) area to determine where recruitment of plants from the seed bank occurs in a restored area. Preliminary results indicate that a large number of seeds survive in the soil seed bank in unburned and lightly burned patches within the landscape mosaic. The survival of these seeds has implications for seeding and recruitment, as well as exotic species control. Samples will be collected in the future from the experimental units to document changes in the seed bank over time and with ecological restoration treatments.

Further research is needed to determine if perennial grasses and forbs utilize the soil seed bank for seed storage. The majority of species that store seeds in the soil seed bank are early successional annuals. In order to restore areas to historic species composition and frequency, seeding of species that have been extirpated from an area may be required, especially if those species are not found in the soil seed bank. BLM has been applying seed to the units after they have been burned in order to reduce exotic weed invasion and to assess the effectiveness of seeding.

Efforts to use local genotypes are made, although local varieties are not always available. Some seed used on the project was collected far way, even though the species is considered to be an historic native of Mt. Trumbull. Seed is applied using hand-held spreaders, usually within six months of the burn treatment. Open sites are "back-dragged" using rakes or chains dragged behind an ATV.

Table 4—2001 Rye Flat seeding. Example of seed mix and application rates used at the Mt. Trumbull Restoration Project.

Species	Total weight (lbs PLS*)	Lbs/acre (80 acres)
Big bluegrass (*Poa ampla*)	50	0.6
Sand dropseed (*Sporobolus cryptadrus*)	40	0.5
Sideoats grama (*Bouteloua curtipendula*)	120	1.5
Utah serviceberry (*Amelanchier utahensis*)	40	0.5
Fringed sagebrush (*Artemisia frigida*)	40	0.5
Wood's rose (*Rosa woodsii*)	40	0.5
Arizona fescue (*Festuca arizonica*)	140	1.75
Prairie Junegrass (*Koeleria cristata*)	80	1.0
Bottlebrush squirreltail (*Sitanion hystrix*)	100	1.25
Slender wheatgrass (*Agropyron trachycaulum*)	100	1.25
Mountain brome (*Bromus marginatus*)	100	1.25
Western wheatgrass (*Agropyron smithii*)	100	1.25
TOTAL	950	11.85

* Percent live seed

Maintenance of Periodic Low Intensity Grass Fires in Restored Areas

Due to delays in implementing the project, re-entries have not yet occurred. The plan is to complete the restoration process phase 1 throughout the area, then re-enter blocks with fire as funding and conditions allow.

Monitoring

The Mt. Trumbull Ecosystem Restoration Project combines landscape scale operational restoration treatments with comprehensive research. The Ecological Restoration Institute at NAU and the Arizona Game and Fish Department are heading up the research efforts tied to BLM's operational treatments. The research / monitoring studies accompanying this project are varied, but focus on the effects of ponderosa pine ecosystem restoration treatments on vegetation structure and composition, and wildlife. To determine the effects of treatments upon vegetation, 250 long term ecosystem monitoring plots have been established across 4,500 acres of ponderosa pine forest. In addition, studies are in place to monitor the effects of restoration treatments upon small mammals, Abert's squirrels, mule deer, bats, Merriams's turkey, reptiles, passerine birds, and insects.

To achieve sustainable management of ponderosa pine forest ecosystems, a scientific basis must be developed for understanding the structure and dynamics of healthy ecosystems. To quantify the effects of present day disturbances, forests where high-intensity wildfires, bark beetle outbreaks, and other factors are resetting the pattern of succession must be measured and compared. However, NAU and BLM believe the threat to current unsustainable ecosystems is so imminent that carefully planned and closely monitored treatments to restore ecosystem health, based upon the best existing knowledge and designed to adapt rapidly as new information is learned need to be carefully initiated now. NAU, AGFD and BLM are providing a program of scientific study which is providing the information and technologies needed by BLM to carry out ecosystem restoration in an adaptive, "learn by doing manner." NAU, AGFD and BLM are providing an integrated plan of research, experimentation, and feedback, intended to lay the groundwork for integrated ecosystem management by providing scientists, land managers, and the public with tested procedures for restoring ecosystem health.

NAU Scientific Study

NAU scientific study consists of three elements:

- systematic analysis of changes in forest structures and disturbance regimes since disruption of the natural fire regime;
- comparison of the effects of contemporary disturbances from high-intensity wildfire, bark beetle mortality, prescribed fire, and understory thinning; and
- initiation of restoration treatments in an adaptive management setting.

Sampling is occurring at a series of study areas, ranging from those with extended fire exclusion to those still under frequent regimes. Sampling is being carried out at Mt. Trumbull, Beaver Creek Biosphere Reserve, Arizona; the Sierra Tarahumara, Chihuahua, Mexico; La Michilia Biosphere Reserve, Durango, Mexico; and other sites within the southwestern ponderosa pine forest. NAU has carefully selected sampling sites that maximize efficient collection of data most valuable to ecological understanding and management planning at regional, national, and international levels. By applying the same sampling and analysis methods on a range of study areas stretching across the southwestern ponderosa pine ecosystem of North America, this systematic approach is aiding in producing a unified, comparable data set for addressing the natural range of variability of pre-disruption forests, as well as the range of contemporary forest conditions. These quantitative data are being used to support the development of operational ecosystem restoration and management plans across portions of the Southwest, not only within the Mt. Trumbull / Logan area.

Specific NAU studies include:

1) Permanent ecosystem monitoring plots for long-term monitoring.

Long-term monitoring is essential to assess current conditions, reconstruct pre-settlement ecosystem structure, and evaluate the effects of restoration treatments. Monitoring involves recordation of pre-treatment data, assessing current conditions, followed by multiple post-treatment recordation of data and analysis on the same plots to determine changes to the ecosystem. Permanent monitoring plots have been established on a 300 × 300 meter grid across Mt. Trumbull. A total of 249 permanent ecosystem monitoring plots are in place across approximately 4,800 acres of ponderosa pine / gambel oak forest.

2) Experimental blocks.

The experimental blocks are designed for detailed measurements of the effects of ecosystem restoration treatments, and complement the landscape scale sampling provided by the permanent ecosystem monitoring plots for long-term monitoring described above. The purpose of the experimental blocks is to quantify restoration effects through intensive measurements at a series of five controlled study sites. Each block consists of a 50-140 acre area divided into two similar units. Treatments—ecosystem restoration and control—have been randomly assigned to each unit. Basic measurements at each experimental block include contemporary and pre-settlement (pre-1870) forest structure (species composition, size distribution, age distribution), current understory composition and density, canopy closure, fuel loading, and photographs.

3) Passerine bird habitat and population responses to ecological restoration.

The diversity and abundance of passerine birds is being quantified with point counts over approximately 2000 acres in the Mt. Trumbull / Logan area using a before-after-control-impact-pairs design. Habitat characteristics, assessed on the permanent monitoring plot grid, are correlated with bird counts. Baseline data collected over three breeding seasons (1996-98) will be compared with post-treatment data.

4) Restoration of grasses, wildflowers, and shrubs.

The objective of this study is to develop practical methods for restoring native understory diversity and productivity. Key to restoring the understory is an understanding of the on-site plant material, including above ground vegetation as well as viable seeds in the soil bank. Soil samples were collected from different treatment areas and germination compared to samples taken in a control (untreated) area.

5) Butterfly response to ecological restoration

Butterflies are an excellent group to study to examine changes in habitat. Because the butterfly larval stage is typically host specific, butterfly diversity can be a good indicator of host-plant species diversity. Different species of butterflies feed on a variety of plants, including trees, shrubs, grasses, and forbs. Because both adult and larval stages are important food sources for birds and other animals, population studies of this trophic level will provide habitat information complementing studies on birds and mammals. Butterfly species diversity and abundance is examined before and after treatment in the restoration units, as well as compared between control and treated units. Primarily, NAU is looking for changes in butterfly community structure in response to the changes in forest structure and herbaceous structure caused by the restoration treatments. In addition, analysis of abundance and species data is occurring, as well as sampling of butterfly communities in pre-settlement like forests.

6) Effects of high-intensity fire.

This study focuses on measuring ecosystem structure (trees, shrubs, grasses, forbs, dead biomass) in sites burned with high-intensity fire, reconstructing pre-burn conditions where possible, and comparing present and forecast conditions over the burned area. Two severe wildfires in the Mt. Trumbull / Logan area were sampled in 1996, the Lava fire (approximately 20 acres), and the Logan fire (approximately 150 acres). This study follows permanent plots over several years in a forest which burned with high intensity following approximately 60 years of fire exclusion.

7) Effect of thinning and sprouting in gambel oak.

The purpose of this research is to examine the potential ecological restoration of gambel oak, specifically by discovering ways in which the growth rate of existing old-growth gambel oak can be increased and how the growth rate of replacement large-sized oak can be enhanced. Two interrelated questions regarding the ecological restoration of gambel oak addressed in this research are: 1) Does thinning within oak clumps result in an increase in the growth of residual large oaks? 2) What is the effect of sprouting on the growth rate of large oak trees?

8) Wilderness restoration study.

Thirty-four ecosystem monitoring plots in the Mt. Logan Wilderness Area were analyzed in 1998, and the data was used to reconstruct the forest structure in 1870, the date of the last natural fire in the wilderness area. In addition, a social survey is being conducted to assess public response to wilderness restoration. A social survey to assess the local populations of St. George, Hurricane, Fredonia, Colorado City, and Kanab on their attitudes towards Mt. Logan Wilderness restoration treatments has been completed.

9) A potential wilderness treatment: forest restoration without wood removal.

This study will apply a variation of a full restoration treatment that does not remove any wood from the sites. All logs and slash generated from the thinning have been left on the ground. Cut trees were limbed to put as much wood as possible in contact with the ground to accelerate decomposition. The logs were bucked into four-foot sections to reduce the chances of an Ips beetle outbreak and slash was moved from under the crowns of the residual trees to prevent fire damage. Comparisons will be made between this technique and conventional wood removal techniques to an untreated area. The effects to be studied include site damage, understory recovery, and fire behavior.

10) Dendroclimatic reconstruction.

Increment cores collected in 1996 from old ponderosa pine trees were cross dated and measured to develop a long tree-ring width chronology for the Mt. Trumbull area. Currently, a third dendrochronologist is checking the chronologies to improve correlations. This chronology will be correlated with weather data to create a model that will reconstruct past climatic trends and will be compared with tree age data and fire history results to search for past climate-plant-disturbance connections and to suggest potential future changes.

11) Fire History.

Frequent, low intensity fire regimes are characteristic of ponderosa pine throughout its range, but specific knowledge of the characteristics of pre-settlement fire patterns in the Mt. Trumbull / Logan area is important to guide the reintroduction of fire and to permit evaluation of the restored fire disturbance regime. Fire history reconstruction based on dendrochronological measurement of fire scarred trees, stumps, and logs is being undertaken in the Mt. Trumbull / Logan area to estimate the frequency and seasonality of pre-settlement fire, determine the date of fire exclusion, and develop a record of post-settlement fires. A key feature of the project is the landscape scale of sampling across the entire ponderosa pine forest, allowing researchers to explore questions of fire size, intensity, and variability within the ecosystem, issues that have exceeded the scope of many previous studies.

12) Modeling forest structure change.

The success of management is largely based upon the ability to predict the consequences of alternative actions. Assessing the future cumulative impacts of ecosystem management practices, involving highly complex and variable interactions, is challenging. NAU is applying a process simulation model, FIRESUM, to forecast the effects of alternative forest structural conditions, understory and fuel characteristics, and fire occurrence. FIRESUM was designed to integrate fire behavior and effects modeling together with ecological process modeling. The model has been calibrated to southwestern ponderosa pine forests and will be initialized with data from the permanent monitoring plots in the Mt. Trumbull / Logan area. Comparisons among past and present ecosystem structures, as well as potential future alternatives, will be made.

13) Habitat relationships of the Kaibab squirrel and other sciurids.

In 1997 a project was initiated to investigate habitat relationships of Kaibab squirrels, rock squirrels, and cliff chipmunks in the five Mt. Trumbull ecological restoration experimental blocks (EB-1 through EB-5). As a continuation of that study, track stations and a tassel-eared squirrel feeding index were used in the five experimental blocks in May and August of 1998 in these same locations which were used in August of 1997. This study will provide information on the nature and distribution of Kaibab squirrel feed trees and their relation to restoration treatments, as well as habitat use information.

14) Indigenous land management practices.

In 1998, a project was initiated to explore indigenous land management practices, focusing on the Kaibab Paiute band. In particular, researchers are seeking to determine the use of fire in the Paiute culture, as well as Paiute knowledge of the ethnobotanical uses of the plants found in the Mt. Trumbull areas. This information will be used to explore relationships between indigenous knowledge and contemporary restoration practices.

Arizona Game and Fish Department Scientific Study

NAU studies focus mainly upon vegetative response to restoration treatments. Given the acknowledged need to incorporate a wildlife component into this research, the Arizona Game and Fish Department (AGFD) has taken the lead in securing additional funding and devising studies to determine the effects of restoration treatments upon a variety of wildlife species. AGFD has assumed responsibility for overall wildlife research coordination and implementation.

AGFD has identified a number of focal wildlife species and species groups for evaluating the effects of forest restoration treatments. Focal species were selected and evaluated with respect to the following criteria:

- their use of or association with habitat components that will be altered by restoration treatments;

- AGFD's ability to measure changes in population density and / or habitat use;
- roles in ecosystem function; and
- economic / recreational value.

The selected species and species groups are: passerine birds, mule deer, Kaibab squirrel, Merriam's turkey, and herpetofauna. Sampling arrays for these focal species are overlaid to the fullest degree possible, to maximize efficiency and increase AGFD's ability to assess the effects of restoration treatments on all forest wildlife.

Specific objectives of wildlife studies are to:

- describe the short-term effects of restoration treatments on forest habitat characteristics;
- quantify short-term effects of restoration treatments on selected wildlife species, including response variables describing diversity, abundance, reproduction, and habitat use;
- identify focal species and areas for long-term monitoring of treatment effects on wildlife populations, communities, and habitats; and
- develop habitat management recommendations for future implementation of restoration treatments in southwestern forests.

Data are being collected at two scales, reflecting the home range sizes and anticipated responses of focal species to the restoration treatments. For passerine birds and herpetofauna, a large number of individual home ranges have been encompassed within individual treatment blocks, as well as the control area. Consequently, detectable responses in population and community parameters may occur within these subunits, which is the focus of sampling. Mule deer, Merriam's turkey, and Kaibab squirrel have home ranges encompassing multiple treatment blocks and / or occur at relatively low densities on the study area. For these species, responses to treatment are only detectable at larger scale and may primarily be reflected in patterns of habitat use, rather than population parameters. Sampling for these species is occurring over larger aggregate units of treatment blocks.

Passerine bird studies

AGFD will extend the NAU passerine bird studies into the post treatment phase. Point count and nest monitoring procedures conducted during the pre-treatment phase in 1996 and 1997 will be repeated following treatments in the area of the study for 2-3 years. The impacts of the treatments upon avian abundance, diversity, and reproductive success will be analyzed. The preliminary phase of this study is now complete and results are available in *Final Report—Heritage Grant I96008, Bird Abundance and Diversity Prior to Restoration Treatments for Old Growth Ponderosa Pine*.

Aerial photographs and GIS vegetative covers

In spring of 1997 low altitude, high resolution aerial photographs were taken of the entire RCA. Cartographers at the Bureau of Land Management National Applied Sciences Center will scan, ortho-photographically correct, and prepare digital copies of the original photos. Once completed, AGFD will incorporate these into a GIS Arc / Info data base for the Mt. Trumbull RCA. AGFD will employ photo interpretation, site visits, and GIS overlays of NAU's 300 m grid vegetation data to define and delineate the major vegetative cover types present. This GIS information will then be applied to various wildlife / vegetation studies in the RCA.

Mule deer studies

Mule deer have been captured and radio-telemetered at Mt. Trumbull. Following collection of field data, habitat selection analysis will be conducted. Landscape scale habitat selection analyses will be conducted on the first two years of data once aerial photographs for the area are entered into GIS. Telemetered deer were located once monthly during the winter of 1998 to learn where they travel during this season. General and micro-habitat data collection components of the study are ongoing.

Kaibab squirrel studies

Tests of a track count index in ponderosa pine habitats on the Coconino National Forest (Dodd 1996) have shown extremely high correlations (r=0.98) between squirrel abundance and habitat use on the Mt. Trumbull restoration study area. Count data is obtained from track plate arrays established in both treatment and control areas. Each array consists of a randomly located grid of track plates. Treatment and control arrays are sampled concurrently, for a 48 hour period, annually. Feeding and sign counts (fungi digs, cone cores, peeled twigs, and terminal clippings) are counted in a sampling grid. To date, 6 sampling grids have been installed in untreated areas, and sign counts and population estimates have been completed for the areas of sampling.

Herpetofauna studies

These studies are being conducted to document the effects of forest restoration treatments on reptile species richness and relative abundance. Data collection relies primarily upon pitfall trapping. Pitfall arrays are overlaid on the 300 m × 300 m grid system placed on the study area by NAU personnel. In addition to pitfall arrays, plywood shaders will be placed on the ground in conjunction with artificial PVC tubing burrows. Lizards caught or seen in the arrays / substrates are included in data tallies. Data has currently been collected at 76 pitfall arrays. Several arrays in the study area have never captured lizards, suggesting that lizards are influenced by the varying habitat types present in the RCA, and that certain current habitat conditions are not suitable for them.

Migrant songbirds

The objectives of the migrating songbird study are similar to those of the reptile study. AGFD is determining the current habitat relationships of autumn and spring migrant songbirds at the site through pre-treatment censusing. Then, AGFD will evaluate the response of migrating songbirds to ponderosa pine forest restoration at Mt. Trumbull. Pre-treatment data are being collected at 136 survey points, which are superimposed on NAU ecosystem monitoring plots. As treatment units are restored to pre-settlement conditions, post-treatment surveys will be conducted at the same grid points. Seventy-nine bird species were recorded on the RCA during fall surveys.

Nestling provisioning rates and fledging rates

In 1998 AGFD initiated a study of parental food provisioning rates to nestlings for several bird species that are currently common breeders in the area, and which may be affected by ecosystem restoration treatments. AGFD determined hatching dates, number of chicks, growth rates, food provisioning rates, fledge dates and fledging success for western bluebird nests and white-breasted nuthatch nests. In addition, vegetation will be measured at nest sites for comparison with available habitat values.

Information on this subject is available in the report *Short-term Wildlife Responses to Ponderosa Pine Forest Restoration Treatments in the Mt. Trumbull Area, Arizona*, reports for 1997, 1998, and 1999.

BLM bat roost study

BLM's primary role in this ecosystem restoration project is implementation of operational treatments and assistance to NAU and AGFD studies where necessary. However, BLM is conducting one bat roost study that will complement NAU and AGFD's research.

Objectives of the Northern Arizona Bat Roost Inventory are to identify and map key use areas for bats on BLM-administered public lands on the Arizona Strip. Key use areas included roosts in trees, snags, caves, mines, cliff faces, bridges, and culverts; watering areas at springs, stock tanks, and catchments; and foraging areas. Priority was given to bat species of concern as listed on the AGFD's Wildlife of Special Concern in Arizona (AGFD 1997) and the Arizona BLM Sensitive Species List (BLM 1998). Within a 15 mile radius of the Mt. Trumbull Ecosystem Restoration project; the study was designed to:

- locate bat roosts in all habitat types;
- identify characteristics of trees / snags used as bat roosts;
- map the availability of suitable roost trees / snags;
- develop a model to predict which trees / snags are most likely to be occupied roosting sites during the active season;
- apply the model to control areas, pre-treatment, and treated areas within the context of the Ecosystem Restoration Project;
- evaluate the impacts of ecosystem restoration treatments on bat populations and the availability of roost habitat;
- where possible, document roost-switching and / or evidence of roost site fidelity; and
- evaluate the level of bat use of wildlife water developments in the area.

Future work will again focus primarily on Myotis volans in the Mt. Trumbull area in a continuing effort to document the impacts to forest dwelling bats from the ecosystem restoration project, and to develop the predictive model for identifying potential roost trees / snags. A minimum of 20 additional snag roosts will be required to lend statistical credibility to the sample size (Herder and Jackson 1999).

Results from all wildlife project components will be compiled by AGFD and project cooperators into a comprehensive final report. This report will assess the short term effects of restoration treatments, and identify research and management options for future treatments. Results will also be presented at appropriate professional meetings and conferences, and submitted for publication in peer reviewed journals.

Conclusion

BLM intends to continue implementing the treatments in the Mt. Trumbull and Mt. Logan area, while AGFD and NAU-ERI will continue their research on the impacts of those treatments. Some units will receive second and third burn and seed treatments. Land use planning for the newly designated Parashant National Monument will provide new goals and objectives for the area. BLM

Figure 2—Post treatment example from Mt. Trumbull.

fire and resources staff, along with our colleagues at NAU and AGFD, hope to restore the ponderosa pine forest ecological integrity. Ultimately, our goal is to allow wildfire to play its historic role in maintaining forest health.

Literature Cited

Altschul, J.H. and Fairley, H.C. 1989. Man, models, and management: An overview of the archaeology of the Arizona strip and the management of its cultural resources. Report prepared for USDA Forest Service and USDI Bureau of Land Management by Statistical Research, Plateau Archaeology, Dames and Moore, Inc. Contract #53-8371-6-0054.

Arizona Game and Fish Department. 1997. Wildlife of special concern in Arizona. Arizona Game and Fish Department. Phoenix, AZ.

Bureau of Land Management. 1998. Arizona BLM sensitive species list. Bureau of Land Management, Arizona State Office. Phoenix, AZ.

Dodd, N.L. 1996. Tassel-eared squirrel research project: April 1996 trapping results assessment. Arizona Game and Fish Department, Research Branch, Phoenix, AZ. Unpubl. Rep.

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

Covington, W.W., and M.M. Moore. 1994. Southwestern ponderosa forest structure: Changes since Euro-American settlement. Journal of Forestry. 92: 39-47.

Harrington, M.G. and S.S. Sackett. 1988. Using fire as a management tool in southwestern ponderosa pine. Panel paper presented at the conference, Effects of Fire in Management of Southwester Natural Resources. Tucson, AZ.

Herder, M.J. and J.G. Jackson. 1999. Impacts of the Mt. Trumbull ecosystem restoration project on forest dwelling bats. Preliminary report. Bureau of Land Management, Arizona Strip Field Office.

Mast, J.N., P.Z. Fule, M.M. Moore, W.W. Covington, and A.E.M. Waltz. 1999. Restoration of presettlement age structure of an Arizona ponderosa pine forest. Ecological Applications. 9(4): 228-239.

National Park Service, Western Region. 1992. Western region fire monitoring handbook. National Park Service publication. San Francisco, CA.

Northern Arizona University. 1997. Mt. Trumbull Ecosystem Restoration Project - Progress Report, April 1997. Forest Ecology Lab, School of Forestry, College of Ecosystem Science and Management, Northern Arizona University. Flagstaff, AZ.

Wildlife Responses to Alternative Fire Management Treatments: The National Fire/ Fire Surrogate Study Approach

Steve Zack[1] **and Kerry Farris**[2]

The National Fire / Fire Surrogate Study involves parallel studies with common treatments and research protocols at 13 sites across the country, with most sites in the West in ponderosa pine dominated forests. All sites are applying three replicates each of three treatments (prescribed fire alone, thinning then fire, and thinning alone [the "fire surrogate"] plus a control (continued fire suppression). The wildlife study we are directing of this effort emphasizes understanding the responses of songbirds, woodpeckers, and small mammals to these treatments. Particular emphasis is placed on evaluating the foraging response to fire treatments of woodpeckers and other "bark-gleaner" (nuthatches, creepers, etc.) birds. General hypotheses of predicted wildlife responses have been developed and will be tested with a national meta-analysis of results. Early results from our NE Oregon site will be presented, and the context and constraints of this novel, national effort will be presented.

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Comparing Two Methods of Identifying Ecological Restoration Opportunities

Jimmie D. Chew[1]

Abstract—*Two methods for identifying ecological restoration opportunities in the Northern Region of the Forest Service are compared. Different analysis methods are often used to address issues at different planning scales. The first method is a nonspatial characterization of current vegetation conditions using Forest Inventory and Analysis (FIA) plot data grouped by potential vegetation for both the state of Montana and smaller landscape areas. The second method uses a spatially explicit model on two landscapes of one-half million and 1 million acres. Similar rule sets are used in both methods to compare current vegetation condition to historic vegetation conditions. This analysis indicates that spatially explicit modeling for determining restoration opportunities would not always support the same decisions made through a nonspatial analysis. For both types of land on the Pintler District area, suitable and unsuitable, the spatially explicit modeling indicates a greater need for more costly restoration and conversion treatments and less maintenance treatments compared to FIA plots for the entire state and just those within the area. For the three land classes on the Bitterroot Face area, only the suitable land had a similar mixture of treatment needs compared to the FIA plots for the entire state and those within the area. The differences in the levels of the treatment opportunities between the methods and scales can have a significant impact on the level of treatment needs and associated budgets identified for programs.*

Introduction

The departure of current vegetation from the conditions maintained by historic fire regimes has had an impact on our ability to meet many management objectives. Increases in the activity of bark beetles, root diseases, and defoliators are examples of insects and pathogens capitalizing on a large food base of susceptible host trees. Increases in fire severity and fire sizes are considered a result of the lack of fire disturbance across landscapes (Monnig and Byler 1992).

Identifying the level of treatment opportunities and the types of land that they occur on is an essential step in identifying funding to meet management objectives. To identify the level of management opportunities that would be needed to restore historic vegetation conditions, the Northern Region of the US Forest Service developed a procedure to analyze the Montana portion of Forest Inventory and Analysis (FIA) data (working paper on file with Region Office, Region One). This approach is based on a set of rules for comparing current vegetation conditions on plots with historic fire regimes. The second approach also uses similar rules to make a comparison, but it utilizes a spatially explicit landscape scale model, developed at the Rocky Mountain Research Station. The model is used to project the trend of current inventoried vegetation and to create a representation of historic vegetation. This paper compares the results of these two approaches.

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Treatment opportunities used for both methods were categorized in three condition classes: maintenance, restoration, and conversion.

- Maintenance opportunities are identified when there is little or no change required in forest cover type or density. Maintenance treatments are relatively low cost operations such as deferring treatment for greater than two decades or using wildland prescription fire for resource benefit with no pre-treatment of fuels.
- Restoration opportunities are identified when there is little need to change the forest cover type but the density needs to be changed. When the associated vegetative conditions are compared to management objectives, pre-treatment such as thinning is required prior to the use of wildland prescription fire.
- Conversion opportunities are identified when the current cover type is substantially different from historic conditions. Conversion opportunities have higher investment costs such as replacing a current cover type due to its relative departure from historical conditions. Treatments of regeneration harvests or stand replacing fire followed by reforestation of intolerant species such as ponderosa pine, western larch rust resistant western white pine, whitebark pine, or aspen are needed to return to historical conditions.

FIA Plot Analysis Method

The Forest Inventory and Analysis is a national, strategic scale program conducted on a state by state basis with a set of field sample locations distributed with approximately one sample location (FIA plot) every 6,000 acres (Gillespie 1999). Treatment condition classes were assigned to the set of Montana plots based on the potential vegetation setting with an inferred historical fire regime from the literature (Hardy et.al. 1998). Table 1 is an example of the condition class rule sets or assignments given by historic fire regimes. The current cover type existing on a potential vegetation group (PVG) setting was determined by species with a plurality of basal area. A current structure class was determined: seedlings / saplings are <5 inches dbh; poles are 5-8.9 inches dbh; medium is 9-14.9 inches dbh; large is 15-20.9 inches dbh; and very large is >21 inches dbh. S is single story, 2S is two stories, and multi is three or more stories. A current density class was assigned: L is 10-39%, M is 40-69%, H is ≥70%. Finally, a treatment class was determined: A maintenance class (M) is assigned if the current cover type was consistent with the historical fire regime within the PVG setting, indicating relatively low departure from historical conditions. A condition class of restoration (R) was assigned if departure was not great (density or layering increased, but cover type had not changed). A conversion class (C) was assigned if a greater departure from historical conditions was indicated by an increase in density and a change of cover type. All three classes are treatment opportunities with different tools and costs associated as in the descriptions above.

Plots were stratified into land classes. The three land classes used for this comparison are suitable, non-suitable, and wilderness. The suitable and unsuitable classes refer to the suitability of National Forest land for the production of commercial timber products. Only forested FIA plots on National Forest System Lands were used in this analysis. A total of 2,343 plots were used for all of Montana, 94 for the Pintler District, and 43 for the Bitterroot Face.

Table 1—Treatment class for historical fire regimes of 35-100+ years, mixed severity. See test for explanation of abbreviations.

Existing cover type/group/a	Structure class/b	Density class/c	Treatment class
Limber pine (*Pinus flexis*)	single, two story	L	M
Limber pine	multi	M-H	R
	seedling/sapling	M-H	R
	pole	M-H	R
	medium	M-H	R
	large	M-H	R
	very large	M-H	R
Ponderosa pine (*Pinus ponderosa*)	S,2S	L	M
Ponderosa pine	S,2S,multi	M-H	R
	seedling/sapling	M-H	R
	pole	M-H	R
	medium	M-H	R
	large	M-H	R
	very large	M-H	R
Eastside Region One DF (*Pseudotsuga menziesii*):			
Eastside Region One DF (xeric)	S,2S	L	M
Eastside Region One DF (xeric) <25%	S,2S,multi	M-H	R
Eastside Region One DF (xeric) >25%	S,2S,multi	M-H	M
	seedling/sapling	M-H	R
	pole	M-H	R
	medium	M-H	R
	large	M-H	R
	very large	M-H	R
Lodgepole pine (*Pinus contorta*) <80 years	all	All	M
Lodgepole pine (upper subalpine) >80 years	all	All	M

Landscape Modeling Approach

The modeling system used is SIMPPLLE (Simulating Pattern and Process at Landscape Scales) (Chew 1995, 1997). SIMPPLLE uses stochastic disturbance processes to model vegetation change. Vegetation is described by cover type, size and structure class, and density classes. Long-term simulations (300-500 years) that start with the current vegetation conditions are made without fire suppression to re-create a representation of historic conditions for a specific landscape. Multiple simulations are used to create the frequency of occurrence for the cover type attributes. Multiple short-term simulations (50 years) with fire suppression are also made on the inventoried current vegetations conditions to generate frequency of future vegetation conditions. Similar rules sets are used to compare the simulated historic vegetation conditions with the simulated current vegetation conditions trends.

If there is no difference between these simulated vegetation conditions, the treatment class of maintenance is assigned. If the two most frequently occurring cover types from the simulated current trends are consistent with modeled historic cover types, but either density or size class differs, the restoration class is assigned. If the cover type does not match between the two sets of simulations, regardless of the other two attributes, the conversion class is assigned.

Comparison

The comparison is made using the percents of the condition classes for all of the Montana FIA plots, percents for the FIA plots within the two landscapes, and the percents from the SIMPPLLE modeling of two landscapes. Existing vegetation conditions on the Bitterroot Face on the Bitterroot National Forest is described by field inventories. The Pintler Ranger District from the Deerlodge Forest is described by satellite imagery. The comparison of the percent of treatment opportunity class on suitable land for the Pintler District is in figure 1. The comparison for the suitable land on the Bitterroot Face is in figure 2. For the nonsuitable land, figure 3 describes the Pintler District landscape and figure 4 describes the Bitterroot Face. For the wilderness land, figure 5, only the wilderness portion of the Bitterroot Face landscape was used.

For all except the suitable land class on the Bitterroot Face, there is less maintenance treatment opportunity class identified by the modeling approach. To gain insight into what accounts for the difference, individual plant communities from the Pintler District landscape are examined. These communities were selected from one type to use for the comparison because of the simplicity

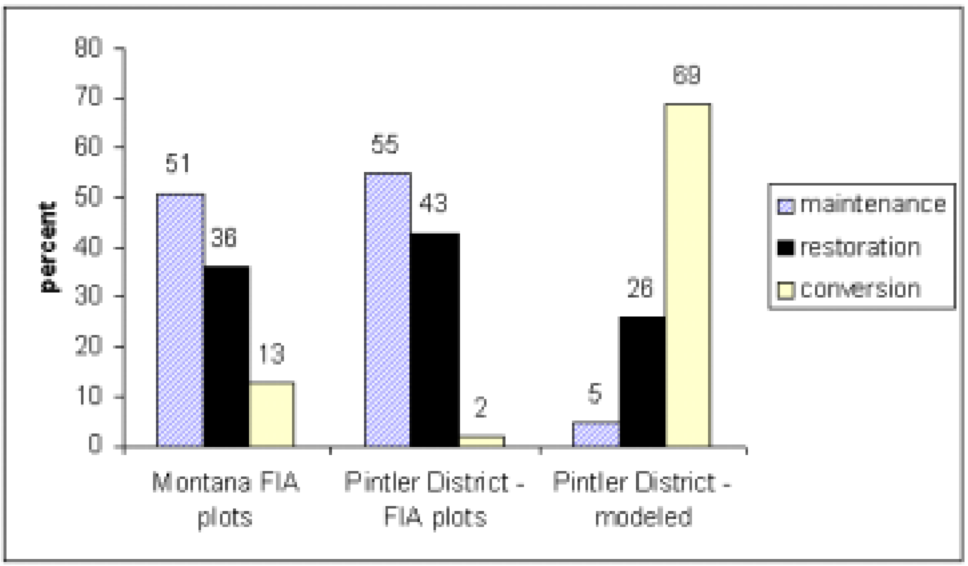

Figure 1—Comparison of percent of suitable land by treatment opportunity for the Pintler District.

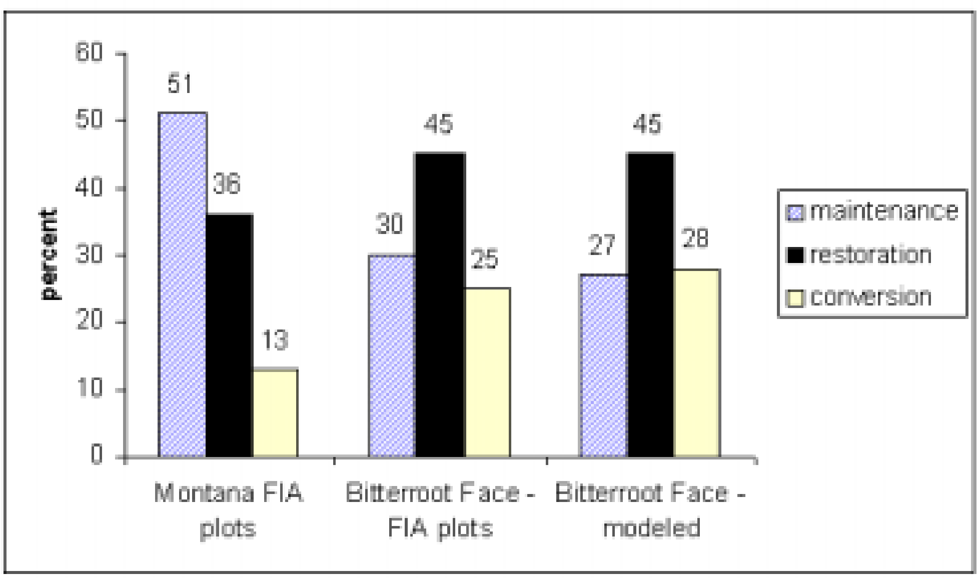

Figure 2—Comparison of percent of suitable land by treatment opportunity for the Bitterroot Face landscape.

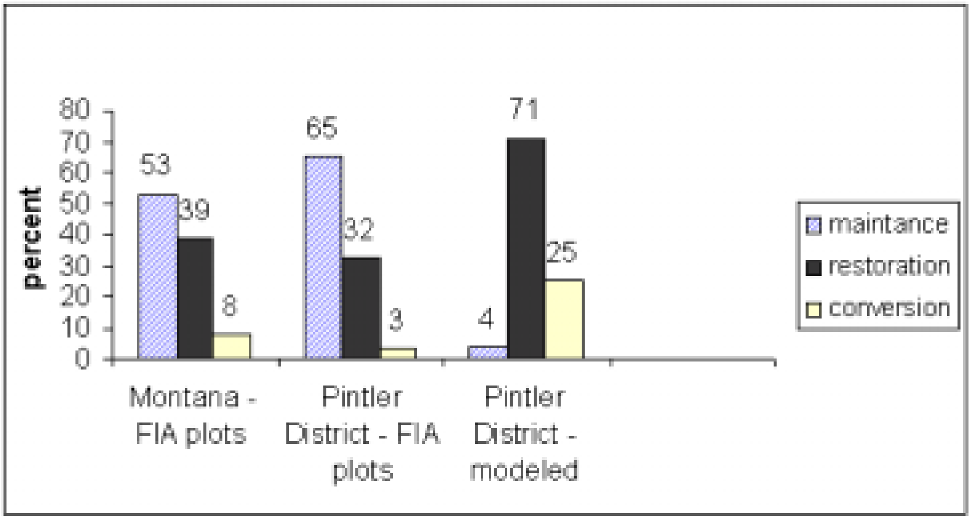

Figure 3—Comparison of percent of nonsuitable land by treatment opportunity for the Pintler District.

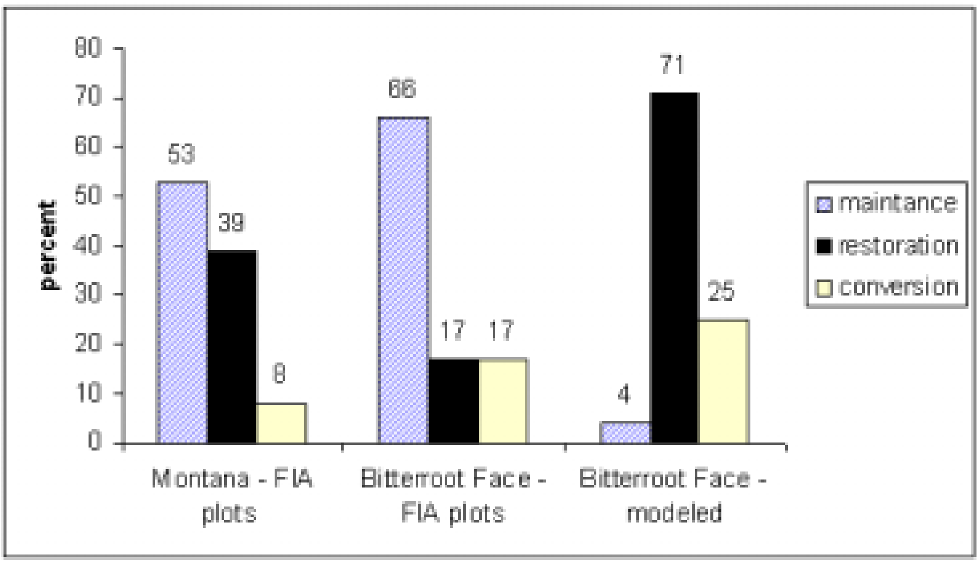

Figure 4—Comparison of percent of nonsuitable land by treatment opportunity for the Bitterroot Face landscape.

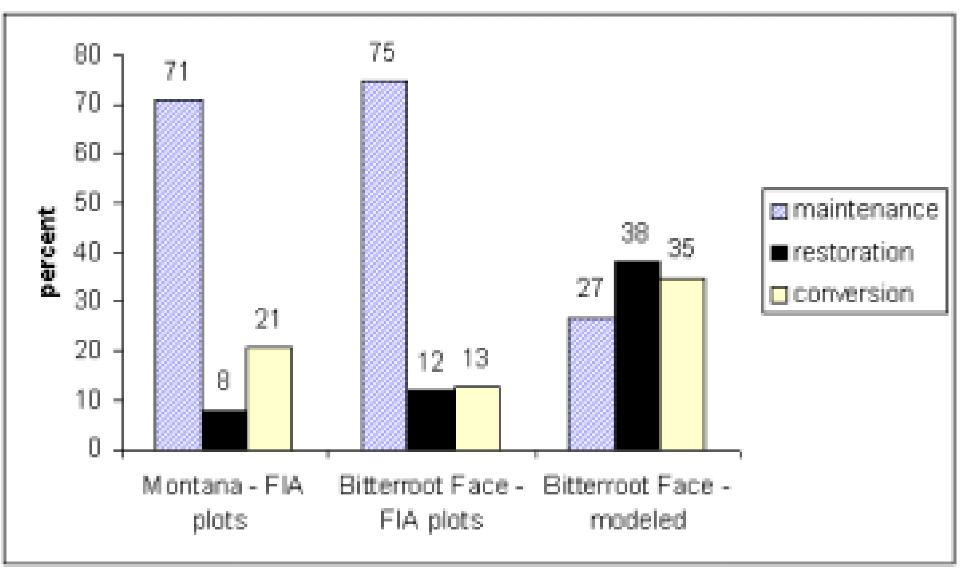

Figure 5—Comparison of percent of wilderness land by treatment opportunity for the Bitterroot Face landscape.

within the Rule Sets (table 1). Xeric Douglas-fir, on slopes greater than 25%, all medium size class, and medium to high density are always put in the restoration condition class. This means that the cover type does not need to be changed but some change in density and structure may need to be made.

Within the Pintler District, using the modeling approach, the Douglas-fir cover type, medium size class, and density 3 on xeric sites have only 2% classified as restoration. The conversion treatment class was assigned to 98%. The difference in results from the SIMPPLLE modeling approach are from being spatially explicit. The arrangement of different cover types present in the current landscape has an impact in the multiple simulations that are made to represent both current trends and possible historic conditions. Different plant communities can be in the same PVG and be assigned the same historic fire regime, but be surrounded by different cover types. The timing of fire events with the size class development of the different cover types and the presence or absence of seed sources all play a part in determining how cover types can vary from one fire event to another.

Figure 6 is taken from a portion of the Pintler District and displays a gradient of Douglas-fir stands intermixed with nonforest communities, to solid Douglas-fir, to Douglas-fir adjacent to lodgepole pine. Three plant communities are selected for a comparison.

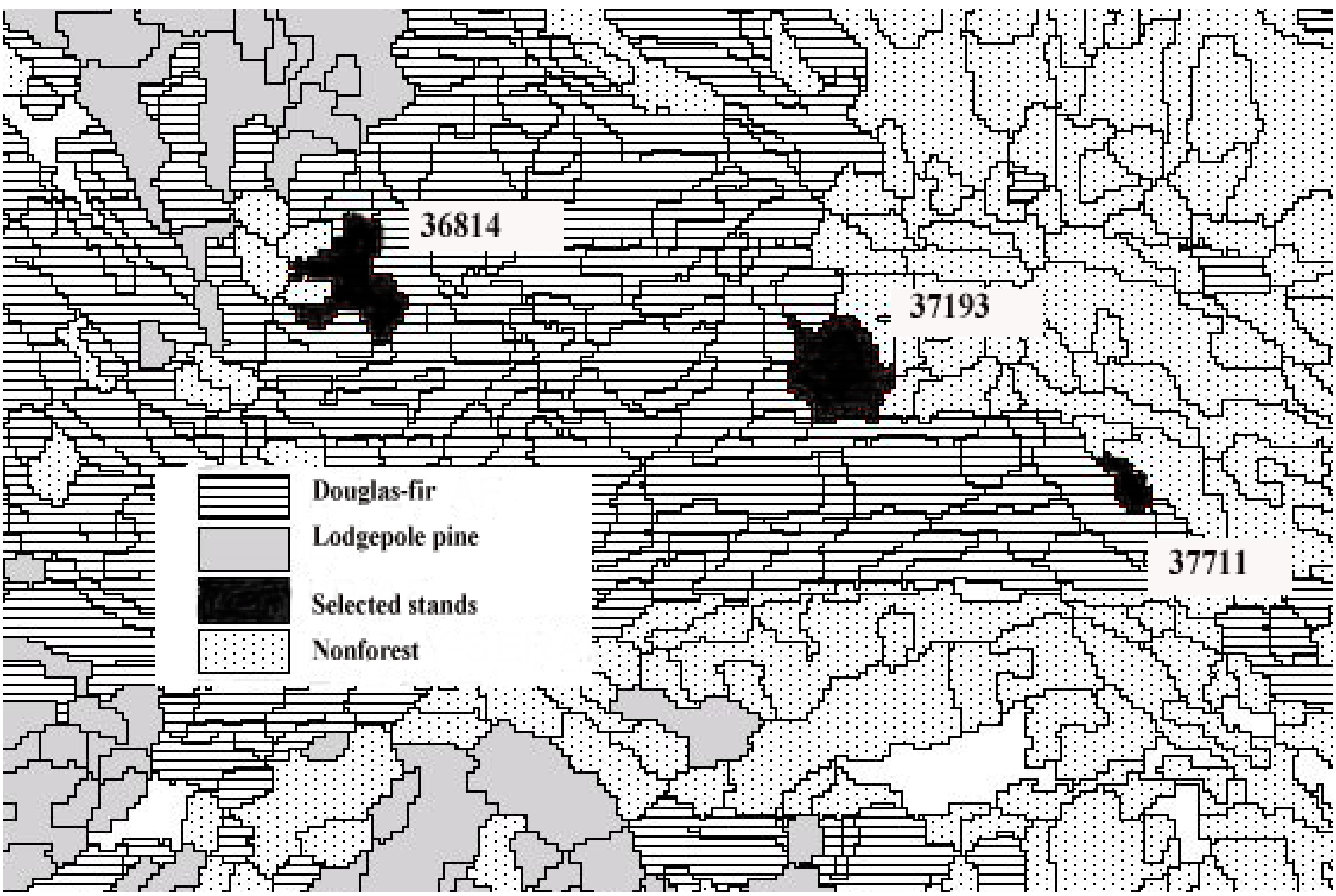

Figure 6—Map showing three plant communities selected for detailed examination of modeled results.

From the modeling method, plant community 37193 is assigned a conversion treatment need. In the modeled current trend, the plant community stays the same type and size class over the next five decades, but increases in density. Historic conditions have the two most dominant cover types being upland-grasses and lodgepole pine. With lodgepole pine it did not grow out of the seedling sapling stage before it burned again. The high level of fire in historic simulations minimizes the acres of the Douglas-fir cover type. Acres of both upland-grasses and lodgepole pine are expanded.

The modeling method resulted in plant community 37711 also being assigned a conversion need. The current trend simulations have the plant community staying the same cover type and size class but increasing in density over the next five decades. Historic conditions have the plant community always in shrub cover type. The level of simulated historic fire has the nonforest communities occurring in areas that have been invaded by Douglas-fir.

Plant community 36814 is assigned a restoration need. The simulated current trend has the plant community staying Douglas-fir and size class medium but increasing in density. The simulated historic conditions are always a Douglas-fir cover type but the size class is predominately pole with frequently becoming pole two-story. The density stays class 3 ninety percent of the time.

Discussion

The FIA process for the entire state was intended to be a broad, nonspatial classification of treatment opportunities appropriate for a Regional assessment. The method of using a spatially explicit model such as SIMPPLLE is often considered more appropriate at the landscape scale for Forest or District planning. Planning processes are often considered hierarchical, with the results of finer level, more detailed planning expected to fall within the context of the decisions made at broader assessment scales. The types of decisions made using a broad scale analysis would continue to be supported as more detailed analysis and planning are conducted. However, this comparison indicates that spatially explicit modeling for determining restoration opportunities would not always support the same decisions made through a nonspatial analysis. For both types of land on the Pintler District area, suitable and unsuitable, the spatially explicit modeling indicates a greater need for more costly restoration and conversion treatments and less maintenance treatments compared to both the FIA plots for the entire state, or those within the area. For the three land classes on the Bitterroot Face, only one, the suitable land, had a similar mixture of treatment needs compared to either the FIA plots for the entire state or those within the area. Would budgets and programs developed for restoration treatments needs from the broad nonspatial analysis underestimate the needs for restoration and conversion? Or are the needs for restoration and conversion overestimated from the spatial analysis that may be completed at other planning levels?

The answer to these questions depends on the confidence in the analysis method used. Does the spatially explicit modeling system overestimate how fast current conditions may change? Does the modeling method overestimate the variability that existed historically? Or does the use of historic fire regimes by PVG's underestimate the historic variability? The differences we see in this comparison can raise additional questions about differences in assessment methods used at different planning scales. Where broad, nonspatial analyses are made, should they be adjusted by using "sample areas" that are modeled spatially? Or should broad scale assessments be composed of spatially explicit analyses on a size of areas that can be added to represent the total area of interest?

References

Chew, J.D. 1995. Development of a system for simulating vegetative patterns and processes at landscape scales. Missoula: University of Montana; Ph.D. dissertation. 182 p.

Chew, J.D. 1997. Simulating vegetation patterns and processes at landscape scales. Pages 287-290. In Proceedings of Eleventh Annual Symposium on Geographic Information Systems, Integrating Spatial Information Technologies for Tomorrow. 1997 Feb 17-20; Vancouver, British Columbia, Canada. Published by GIS World. 640p.

Gillespie, Andrew J. R. 1999. Rationale for a national annual forest inventory program. Journal of Forestry. 97(12): 16-20.

Hardy, C.C.; J.P Menakis, D.G Long, J.K Brown, D.L. Bunnell. 1998. Mapping historic fire regimes for the Western United States: Integrating remote sensing and biophysical data. In: Proceedings of the 7th Biennial Forest Service Remote Sensing Applications Conference. 1998 April 6-9; Nassau Bay, TX. Bethedsa, MD: American Society of Photogrammetry and Remote Sensing. 288-300.

Monnig, E. and J. Byler. 1992. Forest health and ecological integrity in the Northern Rockies. FPM REPORT 92-7, USDA Forest Service, Missoula, MT.

The Post-Burning Response of Bark Beetles to Prescribed Burning Treatments

David J. Ganz[1], Donald L. Dahlsten[1], and Patrick J. Shea[2]

Abstract—*Ecologists and fire scientists have recommended reintroducing fire in fire-dependent ecosystems to achieve the twin goals of restoring pre-settlement forest conditions and reducing catastrophic fire risk (McKelvey 1996, Parsons 1995). Early work by forest entomologists (Miller 1927, Miller 1960; Rasmussen et al. 1996, Salman 1934) established a direct relationship between fire injury and subsequent insect attack in burned-over areas. Initial concern has centered on the primary tree killers* Dendroctonus *spp. and* Scolytus ventralis *LeConte. This research is also finding that* Dendroctonus valens *and* Ips pini *are causing tree mortality with both fall and spring prescribed burns. Post-burning bark beetle induced mortality can be quite significant as demonstrated by two case studies presented here from Lassen Volcanic National Park and Spooner Summit, Lake Tahoe Basin Management Unit. From these two sites, inferences are made on the effect of seasonality for predisposing trees to particular bark beetles. In comparing these two populations, there was no significant difference in the mean number of trees killed by insects in each seasonal window. As case studies, not enough replicates existed to merit a more rigorous analysis. As such, management implications of post-burning bark beetle response are discussed given the information available on fire-insect interactions during these two seasonal windows.*

Introduction

Bark beetles are recognized as a significant factor in California forest ecosystems (Bradley and Tueller 2001, Mutch 1994). Susceptibility of forest stands to bark beetle attack has increased through combined effects of climate change, past forest management, fire suppression, drought, and other pests. The accelerated use of prescribed fire in California ecosystems may or may not alleviate these conditions. Hence, an increased comprehension of fire-insect interactions is necessary for more effective science-based management of forest ecosystems (McCullogh et al. 1998).

Fire may increase the risk of tree mortality from bark beetles by compromising tree defenses against beetles (Geiszler et al. 1984). Charring of the lower bole may damage the tree's vascular cambium, and provide large areas for bark beetle attack by rupturing the resin ducts that help defend pines from attack. Bark beetles, which bore through the tree's outer bark to feed and reproduce within the phloem, are inhibited by oleoresin, resulting in negative relationships between resin production and beetle attack success (Coyne and Lott 1976). Oleoresin is produced by specialized epithelial cells within the xylem and stored within vertical resin ducts in the xylem and in bark resin canals. As a result of heat trauma to these tissues, oleoresin production and defense may be reduced in the lower bole. Alternatively, it has been suggested that some trees which have a long evolutionary history of interactions with fire and bark beetles respond to fire with increases in resin flow to counteract the increased risk of bark beetles (Feeney et al. 1998). Contrary to earlier

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works, it has been reported that increased resin flow does not necessarily result in a decrease in insect activity following fire (Santoro et al. 2001).

Prescribed fire treatments applied in the spring and fall in immature ponderosa pine stands have resulted in significantly different mortality (Harrington 1993). Fires were ignited in the late spring, midsummer, and autumn. Mortality of trees scorched in the spring and summer was about 2.5 times greater than the autumn for similar crown damage. Most trees greater than 18 cm diameter at breast height (dbh) survived injury even with greater than 90% crown scorch. Following spring and summer injury, trees smaller than 10 cm dbh with greater than 50% crown scorching died, but about 90% crown scorch was required to kill large trees. Differences in mortality within the two seasonal windows are likely due to contrasts in physiological activity and to carbohydrate storage (Harrington 1993). Physiological activity was greatly reduced at the time of autumn burns in response to short day lengths, cool air, and cool soil temperatures (Fritts 1976, Kozlowski et al. 1991). Dormant protected buds are likely to survive within scorched crowns and will produce new foliage the following spring (Ryan 1990).

The best indicator of crown injury appears to be the proportion of the crown scorched or killed by fire (Peterson 1985, Ryan 1982, Ryan et al. 1988, Ryan and Reinhardt 1988, Wagener 1961). Empirical evidence has shown that over a wide range of conditions, mortality increases with the square of the fraction of the crown killed (Ryan 1990). Other factors that can be used in mortality prediction modeling include dbh, species, scorch height, bark char height, and local fuel consumption.

Objectives of Study

The Forest Service is looking to use prescribed fire as a tool to reduce the fire hazard and improve the physiological condition of the ecosystem. Jack Ward Thomas, former Chief of the USDA Forest Service, asked the agency to “increase mechanical and prescribed fire treatments to 3,000,000 acres per year in fire dependent ecosystems by the year 2005” (Thomas 1995). The National Fire Plan has since stated that Federal agencies will “jointly develop programs to plan, fund, and implement an expanded program of prescribed fire in fire-dependent ecosystems.”[1] This policy has led to a significant scientific commitment to understanding the impacts of prescribed fire operations.

The purpose of this paper is to compare the extent of tree mortality and bark beetle-induced mortality from two different prescribed fires used in the fall and spring burning windows. The goal is to provide forest managers in this region with an idea of the severity of tree mortality associated with prescribed fires burned under two cases with some similarities in environmental and fuel conditions. In addition, management implications of post-burning bark beetle response are discussed given the information available on fire-insect interactions during these two seasonal windows.

Study Sites

Lassen Volcanic National Park Roadside Unit Spring Burn

The Lassen Volcanic National Park Roadside burn is located 18 km from Manzanita Lake, about 100 m from Lost Creek Campground. The exact

[1] National Fire Plan Website http://www.fireplan.gov/.

location within the park is T. 31 N., R. 4 E., Sec. 10, NE ° (figure 1). The Roadside burn is 102 acres and includes all aspects at elevations of 1725-1798 m (5660-5900 ft).

The objectives of the spring burn were to reduce hazard fuels, monitor fire effects, and restore fire as a natural process. In addition, the hope was to kill a large number of white firs to eventually initiate more pine regeneration. Two permanent square plots were set up using fuels sampling protocol in Western Region Fire Monitoring Handbook (1992). One plot is located in the white fir-dominated portion and the other in ponderosa pine-dominated portion of the spring burn. A total of 93 trees with permanent tags were revisited over the three years since 1999. In the fir-dominated plot, white fir represents at least 60% of the composition of overstory species. Other dominant trees are ponderosa pine, Jeffrey pine, and incense cedar. Sugar pine snags are located in the burn but only in small numbers and do not show up in overstory sampling. In the pine-dominated plot, the overstory species is predominantly ponderosa pine, and together with white fir, these two comprise 80% of the overstory with the remaining component incense cedar and Jeffrey pine.

The precipitation at Lassen Volcanic National Park falls primarily during November through May in the form of snow and rain. Annual air temperatures have an average 14˚C as the daily high and as the low as 0˚C. Annual

Figure 1—Site Locations within California.

total precipitation averages 1067 mm (42 in) per year and snowfall 4800 mm (188 in) per year (Manzanita Lake Weather Station 1949-2002). Relative humidity is fairly high in the spring (during the burning window), ranging from 25% to 85%.

Spooner Summit

Spooner Summit is located in the Lake Tahoe Basin Management Unit, approximately 35 km northeast of South Lake Tahoe (figure 1). Spooner Summit is a steep montane environment covering 50.6 hectares (125 acres) and ranging from 1981 to 2176 meters (6500-7140 ft). Spooner Summit makes up part of the Lake Tahoe's forests that were last cut 140 years ago to supply fuel and timbers for mining the Comstock Lode near Virginia City, Nevada (Taylor 1998). In the early 1900s, fire eradication on federal land became mandatory with state government following suit with the enactment of the California Forest Protection Act in 1905. The regrown forests are now 120 years old and are two to 10 times denser than the original forests (Taylor 1998). These trees now compete intensely for resources, especially water. Drought, combined with the absence of fire which thins forests, has predisposed the trees to bark beetle attack. Under such a scenario, thinning and the reintroduction of fire have been suggested.

In 1992, Forest Pest Management (of Region 5 of the Forest Service) initiated a study to evaluate the effectiveness of Jeffrey pine beetle suppression by comparing the number of trees killed by Jeffrey pine beetle in areas where infested trees were removed annually (treated areas) with the number of beetle-killed trees in areas where the infested trees were not removed (untreated areas). A summary of preliminary results to date indicate that, for selected areas in Lake Tahoe Management Unit between 1993 and 1995, mortality was reduced by 87% in the treated areas, and mortality in the untreated areas increased 182% (table 1). These findings led to the helicopter logging of Spooner Summit in 1996 down to densities averaging 222 trees per hectare (90 per acre).

Table 1—USDA Forest Service Forest Pest Management study on bark beetle infested Jeffrey pines in the Lake Tahoe Basin Management Unit, 1992–1995.[1]

Status and location	1992-93	1993-94	1994-95	Increase/decrease
Treated				
Nevada Beach and Zephyr Cover	909 trees	240 trees	121 trees	-87%
Untreated				
Spooner Summit	1213 trees	1966 trees	2213 trees	+182%

[1]Forest Pest Management website: http://www.r5.fed.us/fpm/fh_94-95/m261e.htm.

Spooner Summit's climate varies from average high temperatures in July of 26 ˚C to lows of 2 ˚C in January. Annual rainfall is very low with an average of 212 mm and average annual snowfall is 5500 cm (216 in). Slopes are very steep, ranging from 10% to 50%. Soils are rocky with shale-like volcanic parent material. A total of 235 trees were tagged and measured. Average tree dbh is 33.6 cm (SD =17.5, CI (95%) = .08) and average tree height is 12.8 m (SD= 4.2, CI (95%) = .09). Pre-fire fuel loads averaged 79 tons / hectare. *Ceanothus prostratus* dominates the foliar cover of the understory.

Site Differences

Two distinctive sampling units, using the Park's Fire Monitoring system, stratified the Roadside burn as Pine and Fir units. Although the species composition, precipitation, location, and management histories are quite distinct, the elevation, size of burn, and forest structure of Pine unit was similar to Spooner Summit making the Roadside spring burn a suitable comparison (table 2). These environmental factors may play a stronger role than seasonality; therefore, in comparing these two cases, only inferences can be made as to why these populations (and their fire-insect interactions) differ.

Table 2—Comparison of metadata from Lake Tahoe Basin Management Unit's Spooner Summit Fall Burn and Lassen Volcanic National Park's Roadside Unit Spring Burn. Preburn data collected in 1996 for Spooner and 1997 for Roadside.

	Spooner Summit	**Roadside Unit, LNP**
Size of burn	50.6 ha	41.3 ha
Month and year of burn	October 1996	June 1999
Elevation	1981-2176 m	1725-1798 m
Avg. temperature (high)	26 ˚C	14˚C
Avg. temperature (low)	2 ˚C	0˚C
Snowfall	5500 cm	480 cm
Precipitation	212 mm	1067 mm
Pre-fire stocking	222 trees/ ha	300 trees/ ha
Average DBH	33.6 cm	37.3 cm
Species composition	98% Jeffrey pine	Fir - 60% *Abies concolor* Pine - 73% *Pinus ponderosa*

Methods

Fire Effects Monitoring

Pre-burn surface fuel loads at Lake Tahoe Basin Management's Spooner Summit were determined using a grid of 28 plots. Plot centers were placed systematically on 4 transects that were installed directly upslope in the Spooner Summit fall prescribed fire unit. At each of the 28 plot centers, surface fuels were inventoried using three Brown (1974) transects at random azimuths (total of 84 transects installed). One and 10 hour fuels were sampled from 0-2 meters, 100 hour fuels from 0-3 meters, and 1000 hour fuels from 0-10 meters. At 10 randomly located grid points, all trees greater than 5.9 inches (15 cm) dbh within a one acre circular plot were tagged, identified to species and dbh and heights measured (dbh to the nearest 0.1 cm and height to the nearest 0.1 m).

Plot locations at Lassen Volcanic National Park's Roadside were selected utilizing a stratified random sampling design. Data were collected within two 20 m × 50 m plots pre-fire, immediately post-fire, and 1- and 2-years post-fire. All overstory trees ≥5.9 inches (15 cm) dbh were recorded within these two plot areas while pole trees between 1.0-5.9 inches (2.5-15 cm) dbh were sampled within one 10 m × 25 m quarter of the plot. All sampled trees were tagged, mapped, identified to species, and recorded as live or dead in accordance with Western Region Fire Monitoring Handbook protocols (1992). Fuel load was measured along four 15.2 m (50 ft) transects per plot using the planar intercept method (Brown 1974). Woody fuel load includes: 1-hour (0-0.24 inches or 0-0.63 cm in diameter), 10-hour (0.25-0.99 inches or 0.64-2.53 cm) 100-hour (1.0-2.99 inches or 2.54-7.61 cm), and 1000-hr (≥3 inches or ≥7.62 cm) fuels. Total fuel load also includes duff, which consists of the layer of partially decomposed, consolidated organic matter below the litter layer. Litter, which is defined as the freshly cast organic matter still retaining its morphological characteristics, was measured but is not included in the total fuel load calculation. Data were analyzed utilizing the Fire Monitoring Software version 3.0. This software provides a platform for data entry and storage while also performing functions including minimum plot calculations and analyses of change over time.

At both of these prescribed burning locations, four duff pins were placed (using the four cardinal directions) around the boles of randomly chosen trees at 30.5 cm (1 ft) from the main stem. Three to five trees were chosen at randomly chosen distances and azimuths (from grid point centers) using a table of random numbers. These trees were thus geo-referenced to plot centers for post-fire visits. Duff stakes were driven flush with the top of the duff and then the fine fuels consumed were measured following the burn. These marked trees served as a subsample of the population for a closer analysis of associated damage to the root collar and *D. valens* LeConte concentrations around the injuries.

The maximum height of bark charring was recorded on all trees greater than 5.9 inches (15 cm). Ocular estimates of three codes of bark char were recorded to the nearest 0.1 of a meter. The maximum height of each charring code was measured on the highest fire-charred side and then again on the opposite side of the tree. If the bark was uncharred, then a zero was recorded. The three levels of charring, designed for use with a similar study at Blacks Mountain Experimental Forest, Lassen County, are defined as follows (Oliver 2000):

- Code 1-Bark is black but not consumed and the bark fissures are not black.
- Code 2–The entire bark including the bark fissures is black, but the bark has not been consumed by fire.
- Code 3–The entire bark is black including the fissures and a significant degree of bark consumption is evident. Bark consumption often is indicated by a “smoothing” of the original bark profile of ridges and fissures.

Four crown measurements were taken to determine the percentage of live crown scorched. These four measurements are defined as follows:

- Measure the height above ground to the base of the living crown if the crown is unscorched or to the base of the crown that was living before the burn but is now dead, in the case of crown-scorched trees.
- Measure the height above ground to the base of the crown that had scorched needles but is still living.
- Measure the height above ground to the base of the unscorched portion of the crown.
- Measure the crown width.

Using these four measurements, the overall percentage of scorched crown was estimated. Overall crown surface area (C^a) is calculated assuming the crown is a solid geometric shape with a measured crown depth (L) and crown width (D). In this study, the amount of crown scorched assumed that the crown resembled a conoid shape. The equation used for crown surface area is as follows:

$$C_a = \frac{\pi D}{2} * \sqrt{L^2 + (\frac{D}{2})^2}$$

For a comparison of burning effects on tree mortality compared to non-burned splits, a paired T-test was used to evaluate the two normally distributed populations. This statistical test was also used to compare a difference in means, in this case the difference of mean mortality between burned and non-burned populations, or between spring and fall burning treatments. It assumes the following hypothesis:

$$H_O : \mu d = 0 \quad \text{and} \quad H_A : \mu d \neq 0 \quad \text{or also expressed} \quad H_A : \mu d \neq \mu_O$$

It was particularly useful in these two cases where there was not adequate replication in the design although inferences were made through comparing the two populations.

Insect Activity Monitoring

Post-burn visits to each plot were performed in the late fall of each year following the burn to determine bark beetle-induced mortality. Spooner Summit performed in 1996 was revisited from 1997 to 2001 and Roadside spring burn performed in 1999 was visited from 1999 to 2001. At both locations, permanent control plots were set up in non-burned areas to compare the effect of burning itself on insect activity. For this comparison, 112 unburned trees, 50 at Roadside and 62 at Spooner Summit, were monitored for the duration of the study.

For the spring burn, multiple visits throughout the growing season were performed to determine the insect invasion pattern for *D. valens*. For this insect, the number of attacks as evidenced by large red pitch tubes and / or granular frass at the root collar was counted and woodpecker foraging noted. The presence of frass, gallery patterns and / or woodpecker activity was used to determine activity of other *Dendroctonus* beetles, mainly Jeffrey pine beetle (*D. jeffreyi* Hopkins), mountain pine beetle (*D. ponderosae* Hopkins), and western pine beetle (*D. brevicomis* LeConte). Woodpeckers working on trees infested by *D. brevicomis* shave the bark to a more or less uniform thickness (Otvos 1965). This enables a visual identification decipherable from the other *Dendroctonus* species. Other insects like the pine engraver (*Ips pini* Say) as well as the California flatheaded borer, *Melanophila californica* Van Dyke, have a pronounced role in mortality after fire (Lyon 1970) and these insects were identified by intrusive sampling (bark removal) of the bole when outside verification was not feasible. The role of these insects and other typical secondary infestation beetles were documented in this project. *Scolytus ventralis* LeConte activity was indicated by white fir pitch streamers and top kill. Within white fir, mortality from *Tetropium abietis* Fall was also documented. A major assumption of this study is that phloem-feeding insects will not infest trees with no viable cambium. Therefore a distinction is made by those trees killed by fire and those predisposed by fire and subsequently killed by insects.

Results

Fire Monitoring

Although burned in different seasons, both Spooner Summit and Lassen Volcanic National Park Roadside prescribed burns have shown similar pre-fire

Table 3—Summary of fire monitoring for Lake Tahoe Basin Management Unit's Spooner Summit Fall Burn and Lassen Volcanic National Park's Roadside Unit Spring Burn. Pre-fire data collected in 1996 for Spooner and 1997 for Roadside.

	Spooner Summit	LNP Roadside Burn
Season of prescribed burn	Fall	Spring
Pre-burn fuel loading	5.13 tons/acre (SD = 6.8)	7.9 tons/acre (SD = 6.5)
Post-burn fuel loading	4.87 tons/acre (SD = 5.3)	4.74 tons/acre (SD = 5.5)
Fine fuels consumed	5.48 cm (SD = 2.3)	3.95 cm (SD = 2.9)
Bark char front code 1	4.25 m (SD = 3.7)	3.77 m (SD = 3.9)
Bark char front code 2	1.77 m (SD = 1.9)	1.86 m (SD = 2.1)
Bark char front code 3	0.55 m (SD = 0.6)	0.70 m (SD = 0.8)
% live crown scorch	60	56

fuel loads and standard deviations (table 3). In comparing the averages from the 235 burned trees of Spooner Summit and the 88 burned trees of Roadside, the fire severity measures of bark char and crown scorch are remarkably similar (table 3). Although the standard deviations for all of these averages are quite large, the ranges follow a similar trend in both burns.

The Lassen Volcanic National Park Roadside Unit has two monitoring plots: one within a white fir-dominated area and the other within a ponderosa pine-dominated area. The white fir-dominated portion of this prescribed burn suffered only mild fire severities with scorching averaging 36% of live crown and bark char (Code 1) reaching heights of 2 m on the high side and .75 m on the low side. Unlike with the pine species, bark char is probably a better severity measure than crown scorch for predicting mortality in white fir.

The ponderosa pine-dominated portion of the Roadside prescribed burn had slightly more severe fire effects due to higher fuel loadings (some fuel loadings were 80 tons/acre). As a result, bark char heights averaged 7 m on the front side and 3 m on the backside. Scorch heights were higher averaging 76% of live crowns scorched. Prior to prescribed burning, many of the ponderosa pines had the needle cast, *Elytroderma dispars*, which may have contributed to higher live crown scorch measures.

At Spooner Summit, 26 trees with complete crown scorch had significant green bud break one year after the fall fire season of 1996. Only one tree was blatantly killed by fire with 100% crown scorch and complete bark char (Code 3) down to the cambium. Bark char heights averaged 7 m on the front side and 3 m on the backside. Live crown scorch heights were a bit higher than the Roadside spring burn averaging 60% of live crown scorched. Using a simple paired T-test to compare fire severity measures at both sites, the difference in bark char codes for the back of the tree and the percentage live crown were significant with a 95% confidence interval. All three front bark char codes were not significantly different in these two populations. Different weather conditions and ignition patterns alone may have caused these differences in back bark char codes and percentage live crown scorched.

Insect Activity Monitoring

After the Spooner Summit fall burn, 31% of the Jeffrey pines were hit by *Ips pini* and *D. valens* in the first year. *D. valens*, with a range of 1-25 attacks per tree, hit 53% of the Jeffrey pines (125 trees). The trees hit by *D. valens* averaged 30.2 cm in dbh (SD of 16) and six *D. valens* attacks per tree (SD of 5) while those plots outside the burn only had one incident of a tree attacked by *D. valens*. Fifty-nine percent of the burned trees attacked by *D. valens* (73 trees) were also hit by *Ips pini*. Only 12 trees were hit by *Ips pini* without the presence of *D. valens*. These 12 trees were hit by *Ips pini* without the presence of *D. valens* (representing only 13% of the *Ips pini* activity). This might indicate some type of predisposal or chemical attractants with *D. valens*.

In this Lassen Roadside Unit spring burn, six trees of the total sample (87) were initially killed by the fire. In year one, 13 were outright killed by *Ips pini* (15%), one by *D. jeffreyii* and 11 by *S. ventralis*. Twenty-three trees were hit by *D. valens*, averaging 16 attacks per tree. By 2001, 50 trees had been killed by one of these four insects. Eight trees within the white fir-dominated area had *S. ventralis* and *T. abietis*, all of which shared extremely high severity measures with crown scorch of 0-90% and bark char heights shown of 5 m front side and 1.75 m backside for Code 1 and 1.6 m front side and 0.73 m backside for Code 3.

Sampling for the first season shows 35% of pines hit with *D. valens* and 24% with *Ips pini* by late-August. Each tree hit by *D. valens* more than 20 times was also subsequently hit by *Ips pini* in this first sampling season. Trees hit by *D. valens* averaged 20 attacks per tree, ranging from 2 to 131 pitch tubes. Although not the intent of this study to document the timing and duration of insect activity, field observations noted two distinct time periods when newly injured trees are susceptible to *D. valens* attack: first early in the summer and then again at the end of the summer months.

Trees at the Lassen Volcanic National Park's Roadside spring burn had steady increases in the number of *D. valens* attacks throughout the first season (figure 2). Here, initial tree attack took place on June 23 (24 hours after the

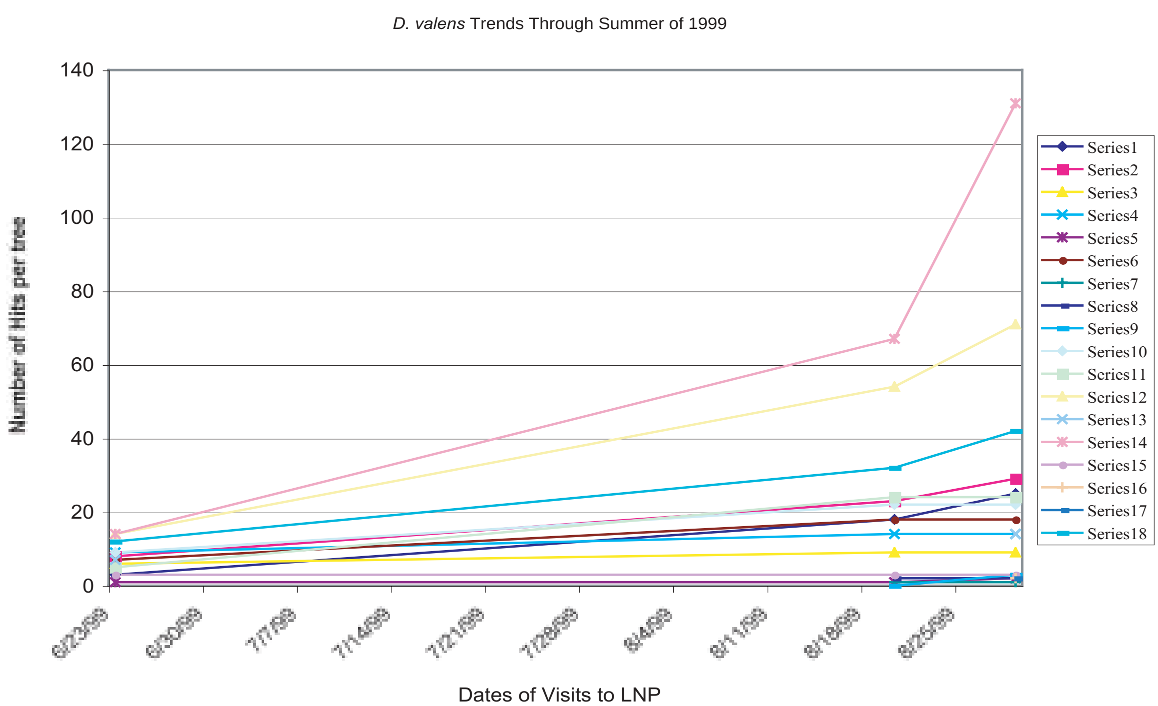

Figure 2—Red turpentine beetle activity during the summer months of 1999 after spring prescribed burning within Roadside Unit, Lassen Volcanic National Park. Initial tree attack, denoted by series, took place on June 23 (24 hours after the burn) and subsequently no new trees were infested until the last sampling on August.

burn) and subsequently no new trees were infested until the last sampling period on August 25 (figure 2). At Blodgett Research Forest in El Dorado County, Owen (1985) found that the largest pulse of *D. valens* took place in the first three weeks of May when daily temperatures had increased. The Roadside spring burn is at higher elevations with a range of 1725-1798 m (5660-5900 ft) compared with that of Blodgett Forest at 1200-1550 m (3900-4800 ft) and therefore the *D. valens* pulse may be delayed due to cooler temperatures. Other environmental factors may be playing a role in determining when such a *D. valens* is likely to occur in a given year. As this is a poorly studied insect in California, more work on the flight periodicity of *D. valens* is needed.

For a comparison of burning effects on insect-induced tree mortality, a paired T-test was used to evaluate the populations of burned and non-burned trees at Spooner Summit and Lassen Volcanic National Park (comparing 323 burned trees with 112 unburned trees outside of the two prescribed burns). The null hypothesis for this test is there is no difference between means of those trees killed by insects (KBI) between burned and unburned populations. The test rejected the null with 95% confidence and a T-value of 8.58 with 433 degrees of freedom and a P of 0. There is a significant difference between the two population means, which in this case, is the proportion of 1's since KBI is binary (0 = alive, 1 = dead). Since it is the proportion or rates of mortality that is of interest to this study, this is a suitable test to run. Ordinarily with such binary data, a nonparametric test that can be applied to two independent sets of sample data would be used (like a Z-test). This Z-test was thus applied to this data. The Z-distribution is a standard normal distribution and the Z-test statistic is calculated by Z=(X-mean of X) / (standard deviation of X), i.e., number of standard deviations away from the mean for any normal data, X. Using this test, the Z-value for KBI is significant with Z = 8.5033 and P = 0.

For a comparison of seasonal effects on insect-induced tree mortality, a paired T-test was used to evaluate the populations of those trees killed by insects in both fall and spring populations of Spooner Summit and Roadside prescribed burns. The null hypothesis for this test is there is no difference between means of those trees killed by insects (KBI) between fall and spring populations. The test accepted the null with 95% confidence and a T-value of -1.6384 with 321 degrees of freedom and P = 0.1023. There is a no significant difference between the two population means, which in this case, are the percentages of trees killed per seasonal window.

Discussion

At the Roadside Unit, the white fir component experienced considerable mortality in the smaller size classes (5-10 cm) while the ponderosa pine component experienced mortality in the larger size classes (five size classes from 20-70 cm). Bark beetles are contributing to most of this pine mortality with only six trees killed by the prescribed fire outright. Given the remaining live overstory component (figure 3), it is probable that white fir will retain dominance over the growing space of the site. It will probably take two to three more fires of this intensity to restore the pre-fire species composition as indicated by the pine snags in the area. In the pine-dominated area of the burn, the higher mortality rate is most likely attributed to a high water table. Old Park Service maps show this area as a grassy opening or a high elevation meadow. Under a more frequent fire regime, this area is likely to revert back to a high

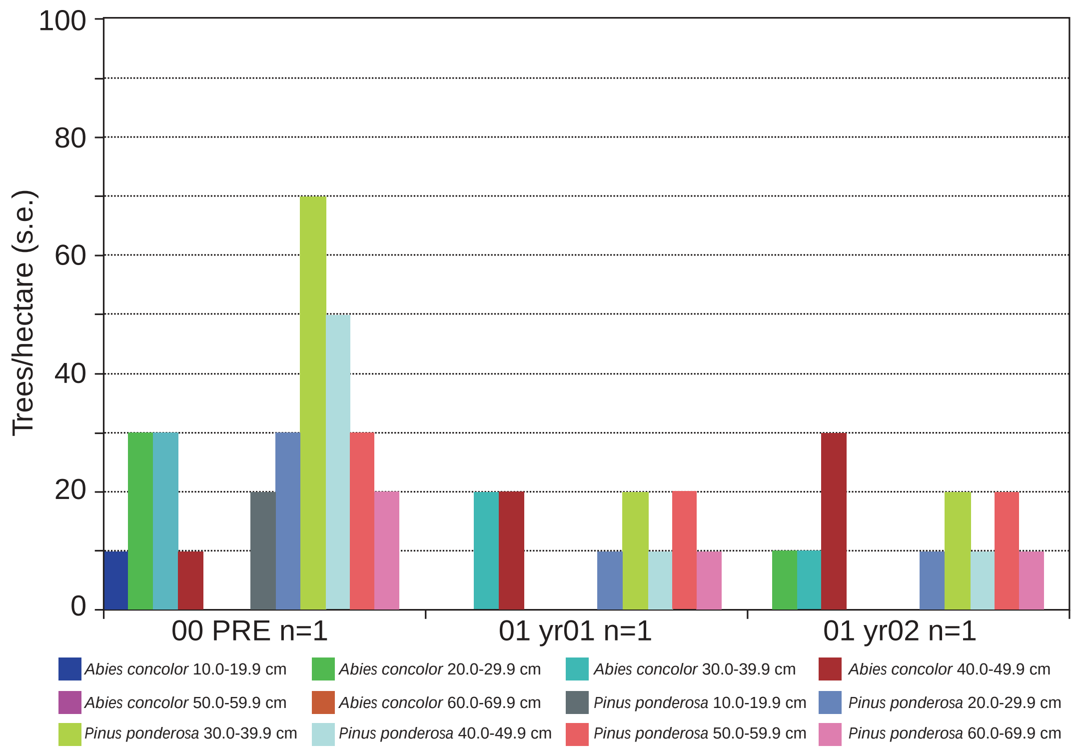

Figure 3—Live overstory tree density by species and size classes within the Lassen Park Roadside spring prescribed burn, Lassen Volcanic National Park, Lassen County, CA. Pre-burn data sampled in 1997, post-burn in 1999-2000.

elevation meadow. With fire suppression, pines have established themselves but not with high vigor. This assessment was made by looking at live crown ratios which are remarkably low for this aspect exposure. A general rule of thumb is that trees living in an open, relative flat environment, such as the one at Roadside, would be expected to have at least 30% of its height in vigorous green growth[2]. However, the trees inspected had only 20% of their total height in a yellowish crown that was riddled with *Elytroderma dispars*. It is speculated that these trees were going to die with or without prescribed burning but at a slower time scale. Therefore, the increased mortality rates from this spring burn can be potentially seen as speeding up succession back towards a meadow landscape.

Historically, *D. valens*, otherwise known as the red turpentine beetle, has not been considered a primary mortality agent in ponderosa pine and Jeffrey pine (Jenkinson 1990). As a lower bole specialist, this insect has been prolific in cutover stumps and predisposed trees but never of much concern to forest managers. Only under rare conditions was it seen overwhelming trees (Smith 1961), usually with exotic introductions and / or other human interventions such as overwatering, fertilizing, soil displacement around root systems, etc. Fire-scorched trees frequently have had sizable populations of this beetle (Eaton and Lara 1967). Mitchell and Martin (1980) speculated that a program of systematic prescribed burning could stimulate a population of *D. valens* to levels where it could overwhelm trees or predispose trees to other mortality agents. Owen (1985) observed a spring prescribed burn at Blodgett Forest, El Dorado County, where 80-90 scorched trees were heavily attacked by *D. valens*. On a wildfire 16 km east of Blodgett Forest in the Rubicon River drainage, Owen again observed unusually high attack rates with 80% of *Pinus ponderosa* and *Pinus lambertiana* attacked (Owen 1985). Ferrell (1996) goes

[2] Monitoring Tree Health on Private Woodlands, University of Idaho Extension Website http: / www.ets.uidaho..edu / extforest / .

further in identifying the probable increase in *D. valens* in the Sierras following the increased use of controlled burning.

In the case of the Spooner Summit fall burn, Lake Tahoe Basin Management Unit, 31% of the Jeffrey pines were attacked by *Ips pini* and *D. valens*. These results are comparable with another study in the Tahoe Basin which found 23% of its sampled trees attacked within the first two sampling seasons (Bradley and Tueller 2001). In another study on the Harvey Mountain spring burn, Lassen County, large open grown trees with high fire severity measures had large lower branches 15.2 cm (6 in) in diameter infested with pronounced *Ips pini* and *D. valens* activity on the lower bole (Ganz 2002). It could be that fire-injured small diameter material is creating a reservoir for both of these species but especially for *Ips pini*, which usually attacks slash piles and tops of trees (Furniss and Carolin 1977).

High variability in fire effects to trees was observed throughout the two prescribed burning study areas, consistent with the range of thermal effects mentioned by Schmidt (1996). *D. valens* and *Ips pini* activity at both sites was particularly high, perhaps due to the high intensity of burns and a stressful environment. Spooner Summit has a southwestern exposure, shallow soils, steep slope, and heavy winds. Beetle-induced mortality was already evident as early as July 9 when plot layout was performed. As previously mentioned, a high water table is speculated to play a role in the Roadside pine-dominated area. Regardless of these two site's predisposition to bark beetle attack, there needs to be a re-evaluation of traditional perceptions of *Ips pini* and *D. valens* and their ecological niches in forested ecosystems following the disturbance of fire. *Ips pini* and *D. valens* are contributing to high levels of pine mortality.

Research from Blacks Mountain Experimental Forest (BMEF) has used the same fire severity measures and has found that the bark char measures are useful for deciphering between those trees killed by fire and those killed by insects (Ganz 2002). In this study, the combination of percent live crown and bark char codes greatly increased the ability of models to predict tree mortality, both from the fire itself and from subsequent insect attack. In all models, these two fire severity measures had a positive relationship with increasing probability of tree death. In general, the models for KBI performed better with higher predictability using the fire severity measures than those models for killed by fire (KBF). Also noteworthy from this study was the fact that those models for KBI performed best with the use of Bark Code 2 (front and back) while those models predicting KBF performed better with Bark Code 3. Biologically, this inherently makes sense as insects are less likely to invade material that has no nutritional value (which Code 3 probably indicates). Also relevant to these two cases at Lassen Volcanic National Park and Spooner Summit was the finding that the same fire severity measures performed differently from tree species to tree species and between individual insects within one tree species (Ganz 2002). For instance, the *Ips pini* response model on the BMEF data set performed differently than the *Dendroctonus ponderosae* response model with regard to the emphasis of particular fire severity measures. Given that Lassen Volcanic National Park and Spooner Summit have very distinct forest composition, the insect responses may differ even more than the BMEF results. Differences in years and seasonal windows (although not found in this particular case comparison) are likely to further convolute these results. Interactions with insect parasites, insect predators, and avian predator complexes, which are known to affect bark beetle populations (Berryman et al. 1970), will vary from spring to fall and from year to year. Further work is recommended to determine the role of these interactions, specifically with different pine species and their post-burning mortality rates.

Conclusion

A primary goal of restoration treatments in ponderosa pine forests is to create more open-stand structures, thereby improving tree vigor and reducing vulnerability to insects, disease, and severe fire. Alternatively, the use of prescribed fire has increased the concerns about the detrimental impacts of fire on tree vigor, especially for its potential to predispose large trees to bark beetle attacks. In the same manner that a management tool has the potential to increase tree vigor and site productivity (site quality), it also has the potential to reduce it through misapplication and site degradation. Prescribed fires have been shown to have such a potential. Growth losses on commercial tree species have been reported in response to fire stress. These losses are usually attributed to root, cambial, and crown damage, leading to a decline in the physiological condition or vigor of the tree (Hare 1961). Other potential sources of losses may be attributed to fuel consumption of organic material and topsoil loss through the process of erosion (Agee 1973). This study has documented some of the effects of cambial and crown damage from two different prescribed burning treatments.

Insect-induced mortality following prescribed burning may be further assessed with specific knowledge of the insect's ecology and the post-burning response. Such knowledge intrinsically can add to the value of prescribed burning as a tool for creating heterogeneity in post-burning tree survival and subsequent recruitment. Although not found in this particular two case comparison, insect-induced mortality differences are likely to occur by seasonal windows and by year (especially under drought conditions). The use of prescribed burning with this embedded knowledge of insect-induced mortality will allow managers to change stand densities, size distribution, and community composition. For instance, many low intensity fires have little effect on the relative density of the stand because the average tree size will change little although the density will decrease through the loss of small stems both from the fire itself and the presence of bark beetles specializing in small diameter classes. Alternatively, the opposite may take place with delayed mortality in the larger diameter classes (due to opportunistic bark beetles) having a pronounced impact on relative densities. An increased comprehension of these fire-insect interactions is necessary for more effective science-based management of forest ecosystems (McCullogh et al. 1998). Given these two cases from Lassen Volcanic National Park and the Lake Tahoe Basin, tree mortality from bark beetles can reduce stocking levels of pine trees which, given the management objectives of the prescribed burn, may be desirable or undesirable. Many have claimed the use of prescribed burning as a means to restore the competitive advantage to the pine species by thinning out shade-tolerant, understory white fir trees. This would create the growing space for pine regeneration or allow remnant large pines to retain dominance and vigor with less water stress resulting from the thinning of neighboring competition. But the two study cases indicate otherwise, further demonstrating the need to consider the management implications of post-burning bark beetle responses.

References

Agee, James K. 1973. Prescribed fire effects on physical and hydrologic properties of mixed conifer forest floor and soil. University of California - Berkeley, School of Forestry. Dissertation.129 p.

Berryman, A. A.; Otvos, I. S.; Dahlsten, D. L.; Stark, R. W. 1970. Interactions and effects of the insect parasite, insect predator and avian predator complex. In: Stark, R. W.; Dahlsten, D. L., eds. Studies on the population dynamics of the western pine beetle, *Dendroctonus brevicomis* LeConte. Berkeley: University of California, Division of Agricultural Sciences. 174 p.

Bradley, Tim; Tueller, P. 2001. Effects of fire on bark beetle presence on Jeffrey pine in the Lake Tahoe Basin. Forest Ecology and Management. 142: 205-214.

Brown, James K. 1974. Handbook for inventorying downed woody material. Gen. Tech. Rep. INT-16. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 24 p.

Coyne, J. F.; Lott, L. H. 1976. Toxicity of substances in pine oleoresin to southern pine beetles [*Dendroctonus frontalis*]. Georgia Entomological Society, Journal of Entomological Science. 11: 301-305.

Eaton, Charles B; Lara, R. R. 1967. 3-red turpentine beetle, *Dendroctonus valens* LeConte. In: Davidson, A. G.; Prentice, R. M., ed. Important forest insects and diseases of mutual concern to Canada, the United States, and Mexico. Canadian Deartment of Forestry and Rural Development. 248 p.

Feeney, Shelly R.; Kolb, T. E.; Covington, W. W.; Wagner, M. R. 1998. Influence of thinning and burning restoration treatments on presettlement ponderosa pines at the Gus Pearson Natural Area. Canadian Journal of Forest Research. 28: 1295-1306.

Ferrell, George T. 1996. The influence of insects pests and pathogens on Sierra forests, Sierra Nevada Ecosystem Project, Final Report to Congress, Vol. II, Assessments and scientific basis for management options. University of California at Davis, Centers for Water and Wildland Resources: 1117-1192.

Forest Pest Management Web site: http://www.r5.fed.us/fpm/fh_94-95/m261e.htm, Jan.29, 2002.

Fritts, Harold C. 1976. Tree rings and climate. London; New York: Academic Press.

Furniss, Robert L.; Carolin, V. M. 1977. Western forest insects. Misc. Publ. 1339. Washington, DC: U.S. Department of Agriculture, Forest Service. 654 p.

Ganz, David J. 2000. The post burning response of bark beetles to prescribed burning treatments in northern California. University of California, Berkeley, College of Natural Resources, Department of Environmental Science Policy and Management. Dissertation. 198 p.

Geiszler, Daniel R.; Gara, R. I.; Littke, W. R. 1984. Bark beetle infestations of lodgepole pine following a fire in south central Oregon. Zeitschrift fur Angewandte Entomologie. 98(4): 389-394.

Hare, Robert C. 1961. Heat effects on living plants. Occasional Pap. 183. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Experiment Station.

Harrington, Michael. G. 1993. Predicting *Pinus ponderosa* mortality from dormant season and growing season fire injury. International Journal of Wildland Fire. 3: 65-72.

Jenkinson, James L. 1990. Jeffrey pine. In: Burns, Russell M.; Honkala, B. H., ed. Silvics of North America. Agic. Handb. 654 Washington, DC: U.S. Department of Agriculture, Forest Service.

Kozlowski, Theodore T.; Kramer, P. J.; Pallardy, S. G. 1991. Physiological ecology the Physiological Ecology of Woody Plants. Academic Press. San Diego, New York. xix + 657 p.

Lyon, Robert L. 1970. California flatheaded borer. Forest Pest Leaflet 24. Washington, DC: U.S. Department of Agriculture. 8 p.

McCullogh, Deborah.G.; Werner, R. A.; Neuman, D. 1998. Fire and insects in northern and boreal forest ecosystems of North America. Annual Review of Entomology. 43: 107–127.

McKelvey, Kevin.S.; Skinner, C. S; Chang, C.; Erman, D. C.; Husari, S. J.; Parsons, D. J.; van Wagtendonk, J. W.; Weatherspoon, C. P. 1996. An overview of fire in the Sierra Nevada. Sierra Nevada Ecosystem Report. Davis: University of California, Wildland Resources Center. II: 1033-1040.

Miller, John M.; Keen, F. P. 1960. Biology and control of western pine beetle. Misc. Publ. 800. Washington, DC: U.S. Department of Agriculture. 381 p. plus illustrations.

Miller, John M.; Patterson, J. E. 1927. Preliminary studies on the relation of fire injury to bark beetle attack in western yellow pine. Journal of Agricultural Research. 34: 597-613.

Mitchell, Robert G.; Martin, R. E. 1980. Fire and insects in western pine culture in the Pacific Northwest. In: Sixth conference on fire and forest meteorology; 1980 April 22-24. Society of American Foresters: 182-190.

Mutch, Robert W. 1994. Fighting fire with prescribed fire—a return to ecosystem health. Journal of Forestry. 92: 31-33.

National Fire Plan Web site http: / / www.fireplan.gov, January 14, 2002.

Oliver, William W. 2000. Ecological research at the Blacks Mountain Experimental Forest in northeastern California. Gen. Tech. Rep. PSW-GTR-179. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 66 p.

Otvos, Imre S. 1965. Studies on avian predators of *Dendroctonus brevicomis* LeConte (Coleoptera: Scolytidae), with special reference to Picidae. Canadian Entomology. 97: 1184-1199.

Owen, Donald R. 1985. The role of *Dendroctonus valens* and its vectored fungi in the mortality of ponderosa pine. University of California, Berkeley, Department of Entomological Sciences. Dissertation. 64 p.

Parsons, David J. 1995. Restoring fire to the giant sequoia groves: what have we learned in 25 years. In: Brown, James K.; Mutch, Robert W.; Spoon, Charles W.; Wakimoto, Ronald H., eds. Proceedings—symposium on fire in wilderness and park management. Gen. Tech. Rep. INT-GTR-320. Ogden, UT. U.S. Department of Agriculture, Forest Service, Intermountain Research Station. INT-GTR-320: 256-258.

Peterson, David L. 1985. Crown scorch volume and scorch height: estimates of post-fire tree condition. Canadian Journal of Forest Research. 15: 596-598.

Rasmussen, Lynn. A.; Amman, G. D.; Vandygriff, J. C.; Oakes, R. D.; Munson, A. S.; Gibson, K. E. 1996. Bark beetle and wood borer infestation in the Greater Yellowstone area during four postfire years. Gen. Tech. Rep. INT-GTR-487. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 10 p.

Ryan, Kevin C. 1982. Evaluating potential tree mortality from prescribed burning. In: Proceedings of symposium on site preparation and fuels management on steep terrain; February 15-17, 1982; Spokane, WA. Pullman: Washington State University, Cooperative Extension Service. 167-178

Ryan, Kevin C.; Steele, B. M. 1990. Cambium mortality resulting from broadcast burning in mixed conifer shelterwoods. In: Proceedings of the 10th conference on fire and forest meteorology; April 17-21, 1989; Ottawa, Canada. Ottawa, Canada: Forestry Canada: 108-116

Ryan, Kevin C.; Peterson, D. L.; Reinhart, E. D. 1988. Modeling long-term fire caused mortality of Douglas-fir. Forest Science. 34: 190-199.

Ryan, Kevin C.; Reinhardt, E. D. 1988. Predicting postfire mortality of seven western conifers. Canadian Journal of Forest Research. 18: 1291-1297.

Salman, K. A. 1934. Entomological factors affect salvaging of fire injured trees. Journal of Forestry. 32: 1016-1017.

Santoro, Alyson E.; Lombardero, M. J.; Ayres, M. P.; Ruel, J. J. 2001. Interactions between fire and bark beetles in an old growth pine forest. Forest Ecology and Management. 144: 245-254.

Schmidt, R. Gordon. 1996. Can we restore the fire process? What awaits us if we don't. In: Hardy, C. C.; Arno, S. F., ed. Use of fire in forest restoration; Seattle, WA, September 14-16, 1995. Gen. Tech. Rep. GTR-INT-341. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 86 p.

Smith, Robert H. 1961. Red turpentine beetle. Forest Pest Leaflet 55. Washington, DC. U.S. Department of Agriculture. 8 p.

Taylor, Alan H. 1998. Reconstruction of Pre-European forest structure, composition and fire history in the Carson Range, Lake Tahoe Basin Management Unit. Final Report for Cooperative Agreement 0024-CA-95. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 27 p.

Thomas, Jack W. 1995. USDA Forest Service "Course to the Future: Positioning Fire and Aviation Management." In: USDA Forest Service Position Statement. Washington DC. Statement before House Agriculture and Natural Resources Committees, Oversight Hearing.

University of Idaho Extension Web site http://www.ets.uidaho.edu/extforest/Aug,99.htm, January 16, 2002.

Wagener, Willis. 1961. Guidelines for estimating the survival of fire damaged trees in California. Berkeley, CA: U.S Department of Agriculture, Forest Service, Pacific Southwest Research Station. 11 p.

Western Region Fire Monitoring Handbook. 1992. Prepared by the Western Region Prescribed Natural Fire Monitoring Task Force. San Francisco, CA: U.S. Department of the Interior, National Park Service, Western Region. 134 p.

Root Pathogens and Fire: Silvicultural Interactions in "Exotic" Ecosystems

William J. Otrosina[1], Susana S. Sung[1], Charles H. Walkinshaw[1], and Brian T. Sullivan[1]

Much attention is now given to risks and impacts of exotic pest introductions in forest ecosystems. This concern is for good reason because, once introduced, an exotic pathogen or insect encounters little resistance in the native plant population and can produce catastrophic losses in relatively short periods of time. Most native fungal pathogens of forest trees have co-evolved for eons with their hosts and have reached a sort of balance between them and populations of susceptible tree species. Recent studies on various forest types have indicated a higher incidence of certain fungal pathogens than were previously thought to occur. These pathogens are either the type not normally thought of as highly virulent or are those that have not been previously reported as a serious problem on a particular host. For example, pathogenic fungi belonging to both the *Leptographium /Ophiostoma* complex and *Heterobasidion annosum* are associated with mortality after prescribed burning in certain longleaf pine stands. Yet, this tree species has traditionally been ranked as highly tolerant to these fungi. In some *Sequoia giganteum* sites, fire suppression has led to encroachment by true firs. The firs developing in the understory have been shown to harbor *H. annosum* which infects the *Sequoia* via root contacts. Could these observations reflect some manifestation of "exotic ecosystems," whereby the conditions under which particular tree species evolved are no longer present or are altered in some way that increases their susceptibility to these fungi? With the current emphasis on ecosystem restoration and alternative silvicultural regimes, it is critical to address questions dealing with impacts resulting from implementation of ecosystem restoration and other management objectives order to avert losses in forest productivity.

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Treatment—Economics

Social Sciences and the Economics of Moderation in Fuels Treatment

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Abstract—*Fuels management is conducted in the context of the social sciences, which bring the science of the human element into the analysis. Of the social sciences, economics addresses the enhancement or improvement in the human condition by improving our ability to allocate scarce resources. The current fuels treatment situation suggests very ambitious ends in the context of limited means. Scarce resources mean that we will not have all of the fuels treatments that we would like. Therefore we will have to choose wisely among alternative treatment options. Socially desirable treatment programs will recognize both the costs and benefits of treatments to seek desired treatment levels. Further, social science investigations will reveal unintended side effects of fuels treatment policies. The challenge upon us is to combine the information on treatment standards, ecology, and technology into line with the social sciences to attain socially desirable treatment systems.*

This page marks a transition in the conference proceedings from the physical and ecological perspectives to considerations of the social sciences. Also, the following papers primarily address the results of specific case studies. This shift in focus is fundamental, so a few contextual remarks regarding the social sciences and their role in fuels treatment are appropriate.

Social sciences use the scientific method to address laws and hypotheses of human values and behavior. Of the social sciences, economics addresses the connection between resource allocation and social welfare. By social welfare, economists mean improving the human condition, or making people better off. We benefit in many ways from fuels treatments, but they are also costly. In a world of scarce resources and limited budgets, difficult choices are required. Resource allocation has long been an important consideration in fire management. As stated by fire historian Steven Pyne et al. (1996):

> "Economic theory has long enjoyed special privilege as a mechanism of reconciling fire management's limited means with its ambitious ends."

Papers in these proceedings, recent studies by the U.S. Government Accounting Office, and the President's National Fire Plan are among many indicators that the ends may never have been so ambitious relative to the means. The potential for substantive contributions from the social sciences may also have never been greater.

Fuels management for purposes of ecosystem preservation, maintenance, restoration, and wildfire suppression are conducted in a context of scarce resources. Scarcity means that we cannot have all of the things that we would like. This pervasive condition applies as well to ecosystem restoration promoted by fuels treatments. In a world of scarce resources, choices are imperative and many choices can be difficult and frustrating.

For federal programs, budget limitations are a practical reflection of scarcity. Each of the fuels management activities noted above is subject to limited budgets that require difficult choices. These choices all require the careful consideration of the treatment benefits versus treatment costs. Such consideration suggests a principle of moderation. The principle is simply stated as: "When

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there are both benefits and costs of an endeavor, there is typically a socially optimal level of the activity." Extreme solutions are impractical and wasteful. For example, social scientists have long known that a pristine environment is economically and socially undesirable. It is too expensive. It would detract from the achievement of too many other things that society values thereby reducing overall social welfare. As a society, we have chosen to accept a less than pristine environment in exchange for other goods, services and opportunities. The hard question and the relevant question is "What is the level of environmental quality that will best benefit society?" Ecologists, engineers, and economists have been hard at work on this problem for years.

Adoption of the 10 AM policy in 1936 cast aside the principle of moderation expressed in the 1916 and 1925 works of Headly and Sparhawk. The works of Headly and Sparhawk were the first expressions of the tradeoff between fire management costs and fire damage. These works sought a balance between the two costs that would minimize total cost. The 10 AM policy stipulated fire control by 10 AM the morning following the report of a fire. It was an extreme response to extreme conditions. As such, it disregarded costs, lacked moderation, and was therefore socially undesirable. Pyne et al. (1996) compared the policy to panic legislation promulgated by savage droughts in the early 1930s.

Concerned that costs were spiraling out of control, the early economic works of Headly and Sparhawk were summoned to advance a more cost-responsive policy. The Forest Service replaced the 10 AM policy in 1978 by what became known as "cost plus net value change" or C+NVC. Like the early works, C+NVC embodies the tradeoff between the costs of wildfire management and its benefits (Rideout and Hesseln 2001). It therefore directly embodies the notion of scarcity and seeks to promote an optimal extent of wildfire management and resource damage. This shift from an extreme and reactionary policy to a thoughtful recognition of tradeoffs marked a fundamental advancement in wildfire policy.

The fully restored ecosystem over a broad scale is potentially analogous to the 10 AM fire suppression policy. Such restoration would seem to preclude the notion that treatments, including fuels treatments, are conducted in the context of scarcity. While full ecosystem restoration may be desirable, in a world of scarce resources we are ultimately forced to focus on optimal restoration. If, for example, funds are available to only treat 5 or 10 percent of the problem, as many federal land managers have suggested, then how are those funds best allocated? Which lands would be best to restore and to what level? How to design optimal treatments for different ecosystems and social conditions will address many of the key questions of this era.

Ecosystem restoration through fuels treatment can be viewed as a project in environmental quality. The basic economic model of restoration is shown in figure 1 and is identical to the correctly formulated theory of C+NVC. When the benefits of restoration are considered in context with the costs, the principle of moderation is illuminated. The principle suggests an optimal level of restoration consistent with R* in figure 1. Here, the marginal cost of restoration equals its marginal benefits. The extreme response of full restoration on a broad scale would be at the right-hand edge of the figure where costs are excessive relative to benefits.

Figure 2 illustrates the tradeoff between the costs and benefits of restoration in a budgeting context. The optimal restoration budget would correspond with the budget provided by the area under the MC curve up to point R*. (Ignoring fixed costs for the moment does not alter the overall point.) A presumption that restoration is limited by the budget is perhaps correct, but it may ignore the social issue of moderation. Anecdotal evidence often suggests that budgets are consistent with the level indicated by B^1. They are too low

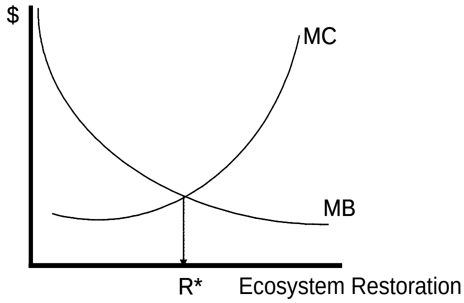

Figure 1—Optimal ecosystem restoration.

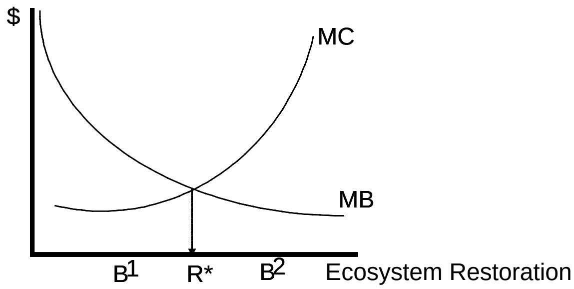

Figure 2—Optimal ecosystem restoration and a budget.

relative to benefits. Unfortunately, the dearth of social research has made it difficult to confirm this suggestion. Likewise, the excessive budget resulting in level B^2 has not been soundly rejected with the current level of research in the social sciences.

The principle of moderation embodied in figures 1 and 2 would suggest that a different set of questions need to be addressed if we are to advance the fuels treatment paradigm beyond the equivalent of the 10 AM policy. Many of these questions address what economists would refer to as the intensive and extensive margins of fuel treatments.

The *intensive* margin suggests questions regarding how fully ecosystems are restored. For example:

- Is full restoration always necessary or desirable?
- What is the design criteria for less intensive treatments?
- What are the beneficial and detrimental ecological effects of partial treatment or restoration?

Questions regarding the *extensive* margin would include:

- Which ecosystems are best suited for full and partial restoration?
- In a given ecosystem and / or ecosystem condition, how many acres should be treated?

Ecological studies that have focused on extremely intensive treatments from small case-study sites provide valuable information and effectively benchmark the full-treatment option. However, ecological research could add much to the field by further addressing the implications of managing the intensive and extensive margins. While the questions above provide a sample of the relevant questions suggested by social considerations, clearly they are interrelated. For example with limited budgets, treating fewer acres means that acres that are treated can be treated more intensively.

The literature and research on fuels treatment has been largely biocentric. For example, in their chapter on fuels treatment and in their section on fuel management, Pyne et al. (1996) contains no discussion of the social sciences. (This is provided as evidence of the lack of research in the field and not as a comment on the book or the authors.) With a biocentric history in fuels research, it should come as no surprise that fuels management and policy searches for cohesive social paradigm.

Non-market benefits, including those associated with ecosystem restoration, can be problematic to assess on a broad scale. In effect, the location and shape of the marginal benefit curve in figure 1 may be impractical and too costly to accurately estimate. While this does not detract from the philosophical strength of the model, it suggests that measures of treatment effectiveness, such as acres changing condition class, may need to be introduced as surrogates for benefit data. This is another area where research in the social sciences has the potential to enhance the fuels treatment paradigm.

Social scientists are trained to reveal unintended side effects of new programs and initiatives. For example, federal standards required air-bags in automobiles with the intent of improving highway safety, but early data showed that fatality rates were unresponsive. Why? Because drivers increased their speed and took more risk knowing that their level of protection had improved.

Similarly, extensive fuels treatments in the wildland urban interface will have unintended social side effects. We are just learning about the unintended side effects of intensive treatments in the interface. For example, as we lower the risks of living in the interface, its desirability will increase. More homes will be built and people will build more expensive homes. The value of property to protect may increase partly as a result of treatment efforts. Just as there are still fatalities on the highway, there will still be homes that burn in the interface. This is the principle of moderation of treatment at work—like a pristine environment, a fail-safe highway is simply too expensive and so too is a fail-safe interface.

So this morning I offer a challenge. Can we combine information on treatment standards and technology with the social sciences to manage the problem in a socially desirable way—in a way that recognizes that we live in a world of scarce resources with diverse public opinions and dynamic social values? Only by integrating the ecology of treatments with the social sciences can we properly address the problems of fuels treatments in ways that will advance societal welfare. The papers that follow on the social sciences will explain the results of studies on cost analysis, social values of fire, recreation, big game, collaboration, community involvement, optimal treatment location, and more.

Literature Cited

Headley, R. 1916. Fire Suppression District 5. USDA Forest Service. May 1, 1916. 58p.

Pyne, S.J.; Andrews, P.L.; Laven, R.D. 1996. Introduction to wildland fire. 2nd ed. New York: John Wiley and Sons, Inc. 769 p.

Rideout, D.B.; Hesseln, H. 2001. Principals of forest and environmental economics, 2nd ed. Fort Collins, CO: Resource and Environmental Management, LLC. 327 p.

Sparhawk, W.R. 1925. The use of liability rating in planning forest fire protection. Journal of Agricultural Research. 30: 693-792.

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Costs for Reducing Fuels in Colorado Forest Restoration Projects

Dennis L. Lynch[1] and Kurt Mackes[1]

Abstract—*The costs to either mechanically remove or mechanically treat forest fuels are examined for various Colorado sites. In the ponderosa pine and mixed conifer zones, no ideal treatment system exists yet for forest restoration through fuel reduction. Each site requires its own ecological analysis. Costs for forest restoration varies by ecological prescription, forest and terrain conditions, and market availability for potential products. In most cases, it may cost too much to remove material or treat areas. Logging may be the most economical method if markets for products exist and processing facilities are nearby.*

Introduction

What does it cost to reduce fuels in forest restoration projects in Colorado? Our research discusses such costs at several locations across the state. A key objective of forest restoration is the reduction of the number of trees and other fuels on the site. An ecological prescription, written by a professional other than the authors, was used to direct each project studied and was specific to the forest type and fuel conditions on the site. In most cases, fuel materials in the form of logs, posts, poles, pulpwood, and chips were removed from the site to reduce fuel loading and achieve desired restorative effects. In a few cases, fuel arrangement was more of a problem than fuel loading and vegetation was mechanically processed and left on the site. A variety of equipment was used for these mechanical treatments. Equipment varied by owner-operator and the type of treatment required.

This paper focuses on costs to mechanically remove fuels from each site or to mechanically treat fuels on a site. It does not include costs associated with wildlife habitat or watershed improvement, for example, which may be a part of a forest restoration program. It does not, therefore, attempt to distribute costs to multiple benefits associated with forest restoration or fuels reduction, a problem reported on by others, such as Gonzalez-Caban and McKetta (1986). In most cases, mechanical removals for forest restoration are followed by the use of prescribed fire to reduce fuel loading on the site. This paper does not report on costs associated with the use of prescribed fire. The following is a summary of project studies we have completed through 2001.

Projects Where Fuels Were Removed From the Site

In the following projects, trees were cut and products were removed from the site to reduce fuel loading and achieve restoration objectives. The types of products varied depending on the markets available at that time.

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Ponderosa Pine Partnership Project

This study has been extensively reported in several publications (Preston and Garrison 1999, Richard and Burns 1999, Lynch, Romme and Floyd 2000, Lynch 2001). It was the first forest restoration project in Colorado, located on the Dolores District of the San Juan National Forest near Cortez, Colorado. Since it was an experimental project, it was sold as an administrative research timber sale to Montezuma County who subcontracted removal of trees to local loggers. The County was concerned that the area was at substantial risk to insect attack and catastrophic fire that could threaten lives, property, and livelihood of county residents. Major data collection efforts occurred during 1996 to 1998 although project development preceded and monitoring followed those dates. Dr. William Romme, who also conducted follow-up ecological monitoring, wrote the ecological prescription for this ponderosa pine forest (Lynch, Romme, and Floyd 2000). This prescription called for reducing the number of trees per acre. Prior to the project there were 280 to 390 trees per acre versus 40 to 50 trees per acre in the 1900 pre-European settlement forest (Romme, Grissino-Mayer, Floyd, Hanna 1999). The key objective of this project was the reduction of trees per acre toward the lower 1900 density to reduce fuel loading and improve fuel arrangement. The largest trees on each site were retained. Those trees were used to identify clumps that varied from a group of six trees up to as much as one-third acre in size. Openings of varying sizes were created between clumps. Snags were retained for wildlife. Phil Kemp, USDA Forest Service, designed silvicutural techniques and marking guides to carry out Dr. Romme's prescription. Dr. Dennis Lynch conducted economic data collection and financial analysis. Mechanical removal of trees was accomplished by Ragland and Sons Logging of Dolores, Colorado. Trees were cut using chainsaws and a JD 743 harvester. Logs were skidded using either a JD 540 or a CAT 518 rubber tired skidder. Logs were loaded on conventional log trucks using a knuckle boom loader for transportation from the forest to several mills producing a variety of finished products.

Lynch (2001) reported very detailed cost information associated with mechanical removal for the 492.6 acres treated in this project. Costs included equipment use, fuel, fluids, workers' wages, administration, insurance, bookkeeping, and fees paid by the logger. All trees that were cut and removed were counted by diameter class. All products removed were weighed and all revenues paid to the logger by mills and other buyers were recorded. Table 1 summarizes the percent of material removed by diameter class in each unit, tons removed per acre, number of acres per unit, and treatment cost per acre for each unit.

The average per acre treatment cost for all units in this project was $869.24. Products such as sawlogs, waferwood logs, posts, poles, and pulpwood were removed from this area and revenues were generated from their sale. The

Table 1--Ponderosa pine project removals, acres, and cost per unit.

			Percent removal by diameter class					
Unit	Acres	Species	3" to 4.9"	5" to 7.9"	8" to 11.9"	12" +	Tons/acre	Cost/acre
Unit 1	125	Ponderosa	No data	50.1%	39.2%	10.7%	29.6	$ 893.22
Unit 4	95	Ponderosa	No data	48.4%	41.1%	10.5%	33.3	$ 1,173.35
Unit 5B	108	Ponderosa	No data	40.9%	51.8%	7.3%	35.0	$ 985.96
Unit 5E	65	Ponderosa	No data	44.3%	51.4%	4.2%	23.5	$ 794.77
Joyce	100	Ponderosa	No data	22.0%	26.0%	52.0%	21.7	$ 497.26

average revenue from such sales was $882.35 per acre. Therefore, the logger who conducted the work had an average profit of $13.11 per acre with a total for the entire sale of $3,852.36 before taxes. This was less than 1% profit (0.8%) before taxes on revenues. However, the USDA Forest Service received stumpage payments (a part of the costs reported above) for the material removed that resulted in a return of $79.08 per acre. These payments included fees for the trees removed, rock replacement for the road, and a collection for slash disposal which was used to assist with prescribed burning.

Mixed Conifer Project

This project followed the Ponderosa Pine Project but has received relatively little publicity. Lynch and Jones (1998) published detailed information on the ecological prescription, costs, and revenues in their summary report. Dr. Wayne Shepperd, Rocky Mountain Research Station, and Bruce Short, current USDA Forest Service Region 2 Silviculturalist, designed the ecological prescription for this mixed conifer forest. The project was located in the Gordon Creek drainage of the Pagosa Springs District of the San Juan National Forest. The untreated forest consisted of an overstory of large ponderosa pine and Douglas-fir with a dense understory of small white fir and scattered clumps of aspen. The prescription called for retaining large overstory trees and removing all white fir within 100 feet of any ponderosa pine or Douglas-fir overstory tree as well as cutting aspen clumps to induce sprouting. Rue Logging, Inc. of South Fork, Colorado, did the mechanical treatment during October and November of 1998. Chainsaws were used to cut trees and logs were skidded with a JD 648 and two JD 650 skidders. A JD 550 skidder was used to clean the large volume of slash created from the white fir trees. A Prentice 410C knuckle boom loader loaded logs on conventional log trucks for transportation to local mills. All trees cut were counted by diameter classes (table 2). All harvesting costs, including administration and fees paid, were collected in cooperation with the logger and converted to an average per acre cost (table 2). The white fir material that was salvageable was taken to the stud mill at South Fork, Colorado where the logger was paid on a per ton basis. The aspen logs were taken either to the excelsior plant in Mancos or the paneling plant near Dolores, Colorado, where revenues were paid on a per ton basis.

In this project, revenue from white fir logs for studs, aspen logs for paneling, and aspen logs for excelsior were used to offset costs. Revenues realized from the sale of white fir and aspen products were converted to an average per acre revenue of $1,272.16 resulting in a profit to the logger of $62.97 per acre before taxes. This was a 4.95% profit before taxes on revenue. The USDA Forest Service charged stumpage fees for the material removed that resulted in a return to the agency of $217.86 per acre. These fees included charges for the trees removed, rock replacement for the road, and a collection for slash disposal, which was used for prescribed burning.

Table 2--Mixed conifer removals, acres, and costs.

			Percent removal by diameter class					
Unit	Acres	Species	3" to 4.9"	5" to 7.9"	8" to 11.9"	12" +	Tons/acre	Cost/acre
Gordon Creek	75	White fir	49.1%	15.1%	17.5%	18.3%	36.1	$1,209.19
		Aspen	--	30.2%	69.8%	--	10.7	

Fox Run Regional Park Project

The Fox Run Project was the first attempt at forest restoration on the Front Range of Colorado. This project was located in the Black Forest near Colorado Springs. It was a cooperative venture between El Paso County Parks and the Woodland Park District of the Colorado State Forest Service. To return the Park's forest to a more natural condition, forest restoration research from the southwest was modified by Chuck Kosteka and Dave Root of the Colorado State Forest Service into an ecological prescription that also met Park objectives. Specific management objectives were:

1. Restore the health and vigor of the forest by selective thinning of trees.
2. Reduce the severe fire hazard and improve insect and disease resistance.
3. Enhance the esthetics and recreational enjoyment of the park.
4. Reduce the uniformity of the stand to enhance the number and variety of wildlife species.
5. Provide a demonstration area with interpretive signs.
6. Retain enough trees around the picnic area to provide screening between the tables.

The ponderosa pine forest contained 548 trees per acre with a basal area of 128 square feet and a spacing of approximately 6 feet between trees. The average dbh of the stand was 6.5 inches and the average height was 42 feet. Fire hazard was rated as severe. The prescription called for reducing the basal area of the stand to 80 square feet per acre with a spacing of approximately 13 feet between larger trees retained in the stand.

There were 24 acres in this forest area that contained approximately 162 cords to be removed. We estimated that this amounted to approximately 18 tons per acre. There was no established market for the material removed and it was the responsibility of the company doing the work to remove all material 4" in diameter or larger. Since an established market for the wood to be removed did not exist, the project was offered for bids in July 1998. An urban tree removal company was the low bidder for the project and was paid $779.17 per acre to complete the work. The bidder used chainsaws and hand skidding to perform the work. He reportedly intended to attempt marketing the material removed for firewood. We do not have direct information on his success, but anecdotal reports suggest he had difficulty with sales.

The Cheesman Reservoir – Trumbull Project

Following the disastrous impacts to Denver's water supply from the catastrophic Buffalo Creek fire, the Denver Water Board hired a forester from the Colorado State Forest Service and began a forest restoration and fuel break program on its lands. A 158.6 acre thinning project was conducted during May into July 2000 on three ponderosa pine forest areas near Cheesman Reservoir and the village of Trumbull on the South Platte River near Deckers, Colorado. The objectives of the project were to reduce fire hazard on these properties and restore forest health and diversity. The project was experimental and followed an ecological prescription based on the work of Dr. Merrill Kaufmann (Kaufmann et al. 2001).

Chuck Dennis, forester for the Denver Water Board, translated the prescription into silvicultural marking guides for the units. The thinning was accomplished by Brandt Logging, Inc. of Lazear, Colorado, using a Timbco feller-buncher with sawhead, a JD 650 tracked skidder, a Kobelco loader, and conventional log trucks. All material removed was transported to the Louisiana-Pacific oriented strand board plant in Olathe, Colorado, a haul distance of

Table 3--Cheesman-Trumbull removals, acres, and costs.

Unit	Acres	Species	Percent removal by diameter class 3 to 4.9"	5 to 7.9"	8 to 11.9"	12" +	Tons/acre	Cost/acre
Cheesman	83	Ponderosa	No data	56.5%	31.8%	11.7%	14.5	$1,084.74
Trumbull #5	42	Ponderosa	No data	57.1%	32.9%	10.0%	9.9	$727.77
Trumbull #6	33	Ponderosa	No data	62.0%	28.4%	9.6%	14.8	$995.38

approximately 255 miles. The logger was paid for delivered material on a per ton basis.

Table 3 summarizes the results of the detailed cost study conducted by Don Rogers and Dr. Dennis Lynch (2000). The average cost per acre was $970.61. However, revenue per acre was only $479.60 resulting in an average loss per acre of $491.01. The total project loss was $77,873.15, which was absorbed by the logger. The agency did not charge any stumpage fee for the material removed.

Air Force Academy Project

Following an outbreak of mountain pine beetles coupled with mortality from dwarf mistletoe, Air Force Academy forester Jim McDermott planned a ponderosa pine beetle tree salvage removal and forest restoration project covering 136 acres of Academy lands. For the most part, the project involved treating small volumes per acre. Justin Anderson and Dr. Dennis L. Lynch collected data and analyzed results from three units of this project. Two of the units reflect beetle tree salvage removal and one unit reflects a forest restoration portion of the project. McDermott, using research reported by Dr, Merrill Kaufmann et al. (2000), developed an ecological prescription for the restoration unit. The material in the area to be treated was of low value and generally of poor quality, the Academy did not have labor and equipment to treat the area, and businesses to process such material are lacking in the area adjacent to the Academy. Therefore, the project was prepared for bid. The bidder had the option to remove trees that might make useable products from the site without a stumpage fee. The low bid to do the mechanical treatment was $679 per acre submitted by Morgan Timber Products of La Porte, Colorado.

Table 4 summarizes the removals and acres for the three units. The Sample unit data came from a 10.7 acre sample of the 76 acre forest restoration unit. The diameter classes shown in the sample data from the 76 acre unit are uncharacteristically skewed, for restoration projects, toward larger trees. This is due to the nature of the stand treated, the prevalence of beetle attack in the area, and the restoration prescription.

Table 4--Air Force Academy removals and acres.

Unit	Acres	Species	Percent removal by diameter class 3 to 4.9"	5 to 7.9"	8 to 11.9"	12" +	Tons/acre
Ridge	11.2	Ponderosa	7.7%	38.6%	44.3%	9.3%	5.0
House	2.2	Ponderosa	23.6%	49.7%	22.7%	4.0%	6.0
Sample	10.7	Ponderosa	15.5%	17.8%	29.0%	37.8%	12.8

The bidder reported that his bid was influenced by what he perceived as an opportunity to utilize some products from the restoration unit. The restoration unit yielded a total of 14 loads of long logs hauled to Longmont, Colorado, 5 1/2 loads of short logs (8' bolts) hauled to La Porte, Colorado, and 19 loads of wood chips hauled to Florence, Colorado. A sawmill for lumber and dimensional products utilized the long logs, the short logs were either sawn into dimensional timbers or used in an experimental roundwood building project, and the wood chips were stockpiled for a biomass energy test. Volumes per load and revenues for these products are not available to us. No additional products were realized by the bidder from the beetle tree salvage units in this project.

Projects Where Fuels Were Left on the Site

On some forest sites, fuel loading may be quite low or sites may actually need more organic material on the soil surface. South-facing Pikes Peak granite soils, for example, are often characterized by such conditions. Pinyon-juniper woodland sites may also contain examples of limited surface organic material. In such forested areas, fuel arrangement or crown closure may be of primary concern in fire hazard mitigation planning. Under such conditions, fuels may be removed from the overstory and distributed over the surface of the ground without detrimental fuel loading. We conducted cost studies on several sites with such conditions. The following is a summary of projects where data have been collected.

Flickenstein Gulch Project

This project occurred during March 2001 in a 9.8 acre stand of mixed ponderosa pine and Douglas-fir near Deckers, Colorado, on Denver Water Board property. The project was an understory removal of ladder fuels. Chuck Dennis, forester for the Denver Water Board, designed, contracted, and supervised the project. The prescription, based on research by Dr. Merrill Kaufmann (Kaufmann et al. 2001), called for thinning from below those trees less than 8 inches dbh. A total of 1,683 trees were cut in the unit. No material was salvageable for products. Slash from cut trees were treated by lopping and scattering to a depth of not greater than 18 inches. The project took 6 hours and 15 minutes to accomplish by two men using chainsaws. The total payment for the work amounted to $990 or a cost of $100.70 per acre or $0.58 per tree. Table 5 displays the trees cut by diameter class. The forester in charge of this project was pleased with the results.

Table 5--Trees cut by diameter class (dbh) during Flickenstein Gulch project.

Percent removal by diameter class					
< 1"	1 to 2.9"	3 to 4.9"	5 to 7.9"	8 to 11.9"	12" +
22.7%	37.6%	27.4%	10.9%	1.4%	0.1%

Hydro-Ax Projects

Another way of accomplishing mechanical thinning of stands and leaving the residues on the site is by using a Hydro-Ax machine. This machine is basically a large front-end loader mounted with a massive rotary cutting head similar to a rotary type lawn mower. The machine lifts the rotary head to a position above the tree or shrub, achieves maximum speed with the rotating cutting head, and then lowers the head onto the stem. The stem is then pulverized into numerous pieces of varying size depending on the material being cut. The pieces are spread over the surrounding terrain by the force of the cutting head. We studied this machine in the three following situations.

The first study, near Durango, Colorado, was conducted in a 37.7 acre pinyon-juniper woodland, which included clumps of oakbrush. In this unit, the crowns of the trees were essentially touching and very little diversity of ground cover existed. The Hydro-ax was used to cut and pulverize trees and oakbrush clumps to open the vegetative canopy, create vegetative diversity, and improve fuel arrangement. The project area was on relatively flat terrain. The bid cost for this work was $223.34 per acre.

In the second study, decadent mountain mahogany thickets near Cheesman Reservoir were mowed with the Hydro-Ax to re-create openings within the adjacent forest, increase sprouting and stem regeneration. The terrain was rolling with slopes up to 30%. A total of 191.4 acres were mechanically treated at a cost of $124.69 per acre.

In the third study, small dense ponderosa pine stands were mechanically treated to create openings, improve fuel arrangement, and increase diversity on 23.4 acres near Foxton, Colorado. In this case, the bid cost was $203.85 per acre.

Hydro-ax treatments may be acceptable where vegetative conditions require fuel reductions to meet ecological prescriptions and where terrain is such that the machine can operate safely and efficiently. The pinyon-juniper treatment was more time consuming, caused increased wear on the machine, and was more expensive than essentially mowing mountain mahogany shrublands.

Summary and Conclusion

We conclude that there is no *single* optimal treatment system for forest restoration fuel reduction on forest sites encountered in the ponderosa pine and mixed conifer zones of Colorado. Each site will require individual analysis to determine the proper ecological prescription and treatment. This will then lead to a determination of the most effective mechanical equipment.

Costs for forest restoration will vary by ecological prescription, forest and terrain conditions, and the availability of markets for potential products. To date, it is clear that the material being removed from these sites is of low quality and low or negative value. In most cases, it will cost too much to remove material or treat areas. For these reasons, we wish to stress the point that this material is a liability from a fuel hazard standpoint as well as from a financial standpoint.

A well-designed timber sale may be the most economical method for an agency to use for fuels removal if markets for products exist and processing facilities are nearby. This was demonstrated by results of the Ponderosa Pine Partnership and Mixed Conifer Projects, where the loggers received very low profit margins per acre. However, in those cases the Forest Service received stumpage payments, approximating $79 per acre and $218 per acre respectively. If the agency had chosen not to require stumpage payments for the trees, profit margins for the loggers could have increased and made these restoration projects financially more attractive. Unfortunately, after these projects were completed, the primary processing businesses for small diameter material were forced to close due to lack of consistent supply and high resource costs. Therefore, it is unlikely these projects could be repeated today given this critical loss of industry.

Low profit margins and loss of industry quickly translates into non-market alternatives where taxpayer dollars must be used to pay for fuel reduction. The increasing size and costs of catastrophic fires leave us with nagging questions about what to do with excess fuels from small diameter trees in Colorado.

The Front Range, Fox Run Park, Cheesman-Trumbull, and Air Force Academy projects underscore the costs associated with fuel reduction where markets for small diameter material are limited or lacking. In those cases, agency costs ranged from $679 per acre to $1085 per acre to treat fuels where at least some outlets for a portion of the small diameter material existed. What will future costs be if such material has to be landfilled or disposed of in some other expensive manner?

In forest areas where fuels can be left on site without adding to the hazard, agency costs to treat ranged between $101 and $225 per acre. Certainly such sites are limited in number and very dependent on specific forest and soil conditions.

Continuing research in product development is crucial to a market solution of this problem. Research to develop more efficient harvesting systems, new wood products, and market access may help provide revenue streams that will ultimately offset treatment costs. We believe that valued-added product development potential is likely to be highest in the utilization of the 8" to 11.9" material found on these sites. This material, and the 12"+ diameter class material, may be able to offset costs of cutting, handling, processing, and transporting the low value chip material found in the 3" to 7.9" diameter classes.

The non-market alternative is to pay millions of taxpayer dollars to restore forests, reduce fuels, and mitigate catastrophic fire conditions while simultaneously waging very expensive and dangerous battles against large, destructive forest fires that result in severely damaged ecosystems that must be rehabilitated, often with limited success.

Literature Cited

Gonzalez-Caban, A. and C.W. Mcketta. 1986. Analyzing fuel management cost. Western Journal of Applied Forestry 1(4):116-121.

Kaufmann, M. R., P.J. Fornwalt, L.S. Huckaby, and J.M. Stoker 2001. Cheesman Lake—a historical ponderosa pine landscape guiding restoration in the South Platte watershed of the Colorado Front Range. Conference Proceedings RMRS-P-22, pg. 9-18 Rocky Mountain Research Station. USDA Forest Service. Ft. Collins, CO.

Kaufmann, M.R., C.M. Regan, and P.M. Brown. 2000. Heterogeneity in ponderosa pine / Douglas-fir forests: age and size structure in unlogged and logged landscapes of central Colorado. Canadian Journal of Forest Research 30: 698-711.

Lynch, D.L. 2001. Financial results of ponderosa pine forest restoration in Southwestern Colorado. Conference Proceedings RMRS-P-22. Ft. Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 141-148.

Lynch, D. L. 2000. Summary report of the forest restoration thinning project on Denver Water Board property at Cheesman Reservoir and Trumbull sites. Dept. of Forest Sciences. Colorado State University. Ft. Collins, CO.

Lynch, D.L., W.H. Romme, and M.L. Floyd. 2000. Forest restoration in southwestern ponderosa pine. Journal of Forestry 98(8): 17.

Lynch, D. and C. Jones. 1998. Summary report of the mixed conifer demonstration project. Dept. of Forest Sciences. Colorado State University. Ft. Collins, CO.

Preston, M. and C. Garrison. 1999. The ponderosa pine forest partnership: community stewardship in Southwestern Colorado. Montezuma Federal Lands Program. Montezuma County, Cortez, CO.

Richard, T. and S. Burns. 1999. The ponderosa pine partnership. Office of Community Services, Ft. Lewis College, Durango, CO.

Romme, W.H., H.D. Grissino-Mayer, M.L. Floyd, and D. Hanna. 1999. Landscape condition analysis for the South Central Highlands section: Ponderosa pine forests. Unpublished final report to the San Juan National Forest, Durango, CO.

The Effects of Fire on Hiking Demand: A Travel Cost Study of Colorado and Montana

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Abstract—*Surveys were conducted on 33 sites within National Forests in Colorado and Montana to test how forest fires affected recreation demand in the two states. Data were collected on the actual number of visits and on the intended number of visits if the area had been subject to a recent high intensity crown fire, a recent prescribed fire, or an old crown fire (all depicted in photos). A travel cost model was estimated by pooling actual and intended visitation responses in both states. Results indicate that Montana hikers take slightly more trips but have lower net benefits or consumer surplus ($12 per trip) than do Colorado visitors ($55 per trip). Also, the demand functions do not react similarly to prescribed fires. Whereas annual values in Colorado increase over time, there were no significant changes in visitation or net benefits for Montana respondents. However, demand functions do react similarly in response to crown fires, resulting in a decrease in visitation and value over time. This latter result provides evidence in support of increased fuels management as outlined by the National Fire Plan.*

Introduction

Fire managers and recreation managers need cost-benefit information to determine the most effective and efficient fuels management techniques, such as mechanical treatments or prescribed burning. In addition to using accounting costs, a complete economic analysis should include social costs and benefits associated with fire. For example, it may appear that prescribed burning is more cost effective than mechanical treatments given the accounting costs per acre. If burning generates significant negative social impacts in the way of increased health costs from smoke and diminished aesthetics, the economic cost of burning may be higher than the cost of mechanical treatments. It is important to incorporate social values when determining fire management methods, particularly in high-use recreation areas. However, this is difficult given that there are little data available to estimate fire effects on nonmarket amenities.

While a few past research efforts have been conducted to assess the effects of fire on recreation (Englin et al. 1996), much of this work has been in Canadian boreal forests or does not include effects on popular activities including hiking and mountain biking. Furthermore, there is little quantitative information available for fire managers to evaluate the differential effects that wild and prescribed fire have on recreation visitation and values. Several studies have been done that indicate fire effects cause decreases in aesthetic value. Vaux et al. (1984) used the Contingent Valuation Method (CMV) to estimate the economic effects of burned areas on recreation. Results indicated that higher intensity fires negatively affected recreation values. Flowers et al. (1985) conducted similar research with respect to the northern Rocky Mountains

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and determined that there was no clear consensus regarding the treatment of fire duration. Englin et al. (1996) and Boxall et al. (1996) used the travel cost method to assess changes in value on canoeing in Manitoba, Canada. The travel cost method (TCM) was used by Loomis et al. (2001) to evaluate fire effects on hiking and biking in Colorado. They found that there were differential effects on hiking and biking visitation as a result of different fire ages and the presence of crown fires. Similarly, consumer surplus for bikers was indirectly affected by crown fire. Finally, Englin et al. 2001 provide a comparative analysis of fire effects over time in the Intermountain West.

The Colorado survey was replicated in Montana and comparisons were made between states to test whether results are generalizable between states. Therefore the same survey and survey methodology were used to test the differences in fire effects on recreation between Montana and Colorado. Because the survey was designed to estimate demand for recreation in National Forests based on actual trips taken and contingent behavior based on three fire scenarios, the Lolo, the Bitterroot, the Flathead, and the Helena National Forests in Montana were used. The survey instrument was identical except for years trips were taken. An overview is provided of the methodology used in both states, followed by a discussion of the model and hypotheses. Finally, results of the regression models and findings are presented.

Methodology

The travel cost method (TCM) was used to estimate the demand for recreation in Montana as based on the survey by Loomis et al. (2001) conducted in Colorado. Using the resulting demand curve, consumer surplus or net benefits per individual, per trip can be calculated by integrating the area under the demand curve. Actual and hypothetical trips were measured as a function of site characteristics including elevation and species, as well as fire characteristics including the presence of a crown fire, fire age, and percentage of burn observable from the trail, demographics, and travel cost information.

Travel cost data included the cost of gas, camping, and other travel related expenditures. The treatment of travel time is often problematic in the TCM. Omitting travel time can lead to specification errors and an underestimation of the true value of the recreation trip (Allen et al. 1981). A traditional solution to this problem has been to value travel time as a fraction of the wage rate and add it to the monetary cost of travel to create one composite variable. This approach is taken here. To minimize the multicolinearity of travel cost and travel time, travel time is multiplied by 40% of the respondent's wage rate. This approach is used by federal agencies in TCM (U.S. Water Resources Council 1983). Thus the travel cost variable and hence consumer surplus will not exhibit omitted variable bias, but it should not be strongly influenced by the particular value of travel time chosen.

A count data specification of the TCM demand model is employed since the number of trips taken (whether actual or intended) is a non-negative integer. Typical count data specifications include the Poisson and Negative Binomial (Creel and Loomis 1990). These count data models are equivalent to a semi-log of the dependent variable functional form.

Fire Effects TCM

To test differences between Colorado and Montana, a pooled interaction model is used with intercept shifters and slope interaction terms for Colorado observations. The model is specified by equation 1.

$$\text{TRIPS} = \beta_0 + \beta_1(\text{Respondent's age}) + \beta_2(\text{Burn observed}) + \beta_3(\text{Crown fire}) + \beta_4(\text{Time since prescribed fire}) + \beta_5(\text{Time since crown fire}) + \beta_6(\text{Elevation}) + \beta_7(\text{Travel cost}) + \beta_8(\text{Acres}) + \beta_9(\text{Actual vs. hypothetical trips}) + \beta_{10}(\text{Income}) + \beta_{11}(\text{Gender}) + \beta_{12}(\text{Lodgepole pine}) + \beta_{13}(\text{Group size}) + \beta_{14}(\text{Travel time available}) + \beta_{15}(\text{Value of aging prescribed fire}) + \beta_{16}(\text{Value of crown fire}) + \beta_{17}(\text{Colorado}) + \beta_{18}(\text{Travel cost for Coloradoan}) + \beta_{19}(\text{Crown fire in Colorado}) + \beta_{20}(\text{Time since prescribed fire in Colorado}) + \beta_{21}(\text{Time since crown fire in Colorado}) + \beta_{22}(\text{Value of recovering prescribed fire in Colorado}) + \beta_{23}(\text{Value of crown fire in Colorado}) + \beta_{24}(\text{Dirt road access}) + \beta_{25}(\text{Value of aging crown fire}) + \beta_{26}(\text{Value of aging crown fire in Colorado}) \quad [1]$$

Model variables and definitions are given in table 1.

Table 1—Model variables and descriptions.

Variable	Description
Trips taken	Trips planned and trips taken by the respondent.
Acres	Size of fire in acres.
Age	Respondent's age.
Burn observed	Percentage of fire observable on trail.
Crown fire	crown fire = 1, no crown fire = 0.
Time since prescribed fire	Number of years since low intensity fire.
Time since crown fire	Number of years since stand replacing wildfire.
Dirt road	Access was on dirt road = 1, otherwise no = 0.
Elevation	Trailhead elevation above sea level.
Travel cost	Individual share of travel costs plus value of travel time to site.
Actual vs. hypothetical	Actual trip taken = 0, intended trip = 1.
Income	Household income of survey respondent.
Gender	Male = 1, Female = 0.
Lodgepole pine	Lodgepole pine present = 1, other species = 0
Group size	Number of people in group.
Travel time available	Total time available for non-winter vacation (weekends plus paid vacation).
Value of aging prescribed fire	Interaction between travel cost and fire age to test whether individual net benefits per trip changes as prescribed fires recover.
Value of crown fire	Interaction variable between total cost and crown fires to test the effects of crown fires on individual net benefits.
Colorado	Colorado respondent = 1, Montana respondent = 0.
Travel cost for Coloradoan	Interaction between Colorado and travel cost to test differences in individual net benefits between Colorado and Montana respondents.
Crown fire in Colorado	Interaction between Colorado and Crown fire to test how crown fire influences trips taken in Colorado.
Time since prescribed fire in Colorado	Interaction between Colorado and time since prescribed fire to test how trips differ according to fire age.
Time since crown fire in Colorado	Interaction between Colorado residents and areas recovering from crown fires to test how the number of trips taken changes.
Value of crown fire in Colorado	Interaction variable between total cost, crown, and the dummy for Colorado to test the effects of crown fires on consumer surplus for Coloradoans.
Value of recovering prescribed fire in Colorado	Interaction between Colorado, total travel cost, and areas recovering from prescribed fire to test whether individual net benefits change.
Value of aging crown fire	Interaction between travel cost, presence of a crown fire, and fire age to test how value changes in response to recovering crown fires.
Value of aging crown fire in Colorado	Interaction between travel cost, presence of a crown fire, and fire age to test how value changes in Colorado in response to recovering crown fires.

The model is specified to calculate consumer surplus and to indicate whether fire effects have an influence on visitation and value of trips taken, and how this differs between Colorado and Montana. Consumer surplus is the area under the demand curve between current travel cost and the choke price that reduces trips to zero. Because a count data model is used which is equivalent to a semi-log demand function, consumer surplus is calculated as $1/\beta_{\text{Travel Cost}}$ (Loomis et al. 1999). To calculate the consumer surplus per trip for individual Colorado trips, the coefficient for the interaction term is included, which is specified by equation 2.

$$1/(\beta_7 + \beta_{18}) \qquad [2]$$

To test the effects of fire age on consumer surplus we combined travel cost variables with time since prescribed fire for both Montana and Colorado. Specifically, if fire age has an effect on the price slope of the demand curve, the coefficient β_{15} will not be equal to zero. Equations for consumer surplus per trip for Montana and Colorado are given by equations 3 and 4.

$$1/(\beta_7 + \beta_4 \textbf{*Time Since Prescribed Fire}_t) \qquad [3]$$

$$1/((\beta_7 + \beta_{18} + (\beta_4 \textbf{*Time Since Prescribed Fire}_t) + (\beta_{22} \textbf{*Value of Recovering Prescribed Fire in Colorado})) \qquad [4]$$

T-tests are used to test whether there are significant positive or negative effects of the fire variables. Specifically, time since fire age, presence of a crown fire, and time since crown fire, crown fire in Colorado, time since prescribed fire in Colorado, and time since crown fire in Colorado are of interest. Finally, regression results are used to estimate the effects of fire on value per day and the number of trips taken over time. We note that our demand model does not explicitly include a variable for the price or travel cost to substitute sites. Therefore the absolute value of our estimates of visitor net benefits may be overstated.

Data Collection

Sample Design

Three National Forests in Colorado were selected that provided a sample of the possible combinations of fire age and acres burned and were logistically functional to sample. The Arapaho-Roosevelt, Gunnison-Uncompagre and Pike-San Isabel National Forests were chosen. This provides two Front Range National Forests and one interior National Forest. Four National Forests in Montana were selected for this study based on fire history and recreation use. They include the Bitterroot National Forest, the Flathead National Forest, the Lolo National Forest, and the Helena National Forest. Each forest included areas that experienced fire in 2000 and areas without fire to be used as control sites. The mean fire size was 27,000 acres while the median fire size was 1,200 acres. With respect to fire age, the oldest actual fire was 24 years old and the newest, one year. Sites sampled that were not affected by fire were coded as –50 years.

Sampling occurred over 35 days during the main summer recreation season in Colorado in 1998. A total of 10 sites over the three National Forests were sampled. This schedule generally allowed one sampling rotation of two days (one weekday and one weekend day) at nearly all recreation sites during July and August. Twenty-two Montana sites were sampled for a total of 25 days in 2000. Because of fire activity in the Bitterroot Valley, and in Montana in general, all recreation areas were closed for use. Prior to closure, sampling occurred over 11 days. After fire restrictions were relaxed, sampling continued an additional 14 days. The survey was concluded in 2001 after all surveys were distributed. Sampling occurred over 34 days between June and August inclusively and was conducted on both weekdays and weekends in both years.

Survey Protocol and Structure

The interviewer intercepted individuals at each trailhead as respondents were going to or coming from the trails. The interviewer introduced herself and gave her university affiliation and purpose. Respondents were told they could complete the survey on site, or take it home and mail it back in a postage paid return envelope included in the package. Surveys were disseminated to individuals 18 years or older.

Respondents were asked to provide their primary recreation activity and important attributes of the site. Next they were asked to provide travel time, travel distance, and travel cost to the site. Travel cost included gas cost only. Individuals were then asked to provide the number of trips taken to the site, as well as planned trips for the remainder of the year. Finally, respondents were asked how their visitation might change if the cost of their trip increased.

The following section of the survey presented three fire scenarios using color photographs of the following:

- High-intensity crown fire: blackened, standing trees with little greenery.
- Light prescribed burn: underbrush burned, trees burned on the lower portion of the trunk, reddish needles on lower branches, green needles on the majority of the trees.
- High-intensity 20-year-old burn: standing dead trees, white trunks, downed trees mixed with new greenery.

Contingent trip behavior analysis was based on photos that depicted trails in such conditions. Respondents were asked how their visitation to each site would change if half the trail resembled the photo. This enabled efficient conveyance of the effects that high-intensity crown fires, prescribed fires, and older burns have on recreation demand.

Contingent behavior was also assessed based on price using increased trip costs ($3, 7, 9, 12, 15, 19, 25, 30, 35, 40, and 70). Respondents were asked to record the number of trips they would take if travel costs were increased. This provided additional price variability to supplement the natural variability in travel costs due to different originations.

Site characteristics were included to control for variability among sites. Attributes were chosen based on those that were significant in past forest recreation studies (Englin et al. 1996). Site characteristics included elevation, elevation gained on trail, miles of dirt road with respect to access, and the number of recreation activities occurring on the site. Fire history information included fire age, size of burn, and intensity. Finally, vegetation type and the presence of water was recorded.

Results

In Colorado, there were 14 refusals out of 541 contacts made. A total of 527 surveys were handed out. Of these, 354 were returned after the reminder postcard and second mailing to non-respondents for an overall response rate of 67%. The total number of contacts made in Montana was 1,074 of which there were 24 refusals. In total, 1,050 surveys were disseminated, and 559 were returned after first and second postcard reminders were mailed. The overall response rate was 53%.

Of the visitors in Colorado sampled at the trailheads, 59% were hiking and 30% were mountain biking. The remainder of visitors (11 percent) were horseback riding or on motorized vehicles. Of the visitors to the 22 sites in Montana, approximately 78% were hiking, camping, and sightseeing. The next largest categories were biking at 10%, fishing at 7%, and swimming and water related activities at 5%. Only hikers and mountain bikers from each state were included in the analysis for consistency.

In Colorado, visitors drove an average of 77 miles (one-way) and their share of the gasoline costs was $12. In Montana the average distance traveled was 98 miles (one-way) and the average individual cost of gasoline was $9.50.

The demographics of the Colorado sample indicated that 44% of respondents were female, and that the sampled population had an average age of 36.5 years and education level of 16.3 years. The typical household earned $67,232. Demographics for Montana indicated that 49% of respondents were female, the sampled population had an average age of 39 years and education level of 16.0 years. Average household earnings were $55,135. Averages for Colorado and Montana are summarized in table 2.

Table 2—Descriptive statistics of travel survey for Colorado and Montana.

Variable	Colorado	Montana
Trip characteristics		
Travel distance (one way)	77 miles	98.6 miles
Average gas cost per respondent	$12	$9.50
Hikers	59%	77%
Other	41%	23%
Demographics		
Percent females	44%	49%
Age	36.5 years	39 years
Education	16.3 years	16 years
Household income	**$67,232**	**$55,135**

Significant fire variables are displayed in table 3; the model is significant with a p-value of 0.000. The model has an adjusted R-squared value of 25%. There is a significant difference between trips taken in Montana (10 per individual per site) and trips taken in Colorado (7 per individual per site).

Travel cost including the value of travel time is also negative and significant for Montana at $p < 0.01$. Surprisingly, total time available for travel had a negative effect on the number of trips taken and was significant at $p < 0.01$. With respect to site characteristics, LP (lodgepole pine) had a significantly negative effect on the trips taken whereas site elevation was positive and was significant at $p < 0.01$. While aspen, Douglas-fir and ponderosa pine were also evaluated, they were highly correlated with lodgepole pine and therefore omitted from the model. As expected, the coefficient on dirt road access was negative indicating that people take fewer trips if access is not paved. The Actual-Hypothetical variable was positive and significant indicating that respondents overstated the number of trips they would take indicating hypothetical bias for contingent behavior estimates versus the number of actual trips taken.

Table 3—Significant Montana and Colorado fire variables. (Trips = dependent variable.)

	Coefficient	Significance (p)
Acres	-0.0000	< 0.00
Crown fire	0.2107	< 0.00
Time since prescribed fire	-0.0036	< 0.00
Time since crown fire	0.0106	< 0.00
Travel cost	-0.0866	< 0.00
Value of crown fire	0.0002	< 0.00
Value of recovering crown fire	0.0003	< 0.11
Coloradoan	-3.4067	< 0.00
Travel cost for Coloradoan	0.0683	< 0.00
Time since prescribed fire in CO	-0.0155	< 0.00
Crown fire in Colorado	-0.1176	< 0.16
Time since crown fire in CO	-0.0005	< 0.20
Value of recovering crown fire in Colorado	0.0005	< 0.08
Value of crown fire in Colorado	0.0113	< 0.03
Value of recovering prescribed fire in Colorado	-0.0002	< 0.00
R-squared		26%
Adjusted R-squared		24%

Significant demographic variables include income ($p < 0.01$) and gender ($p < 0.01$), which were negative, and group size ($p < 0.01$) and respondent's age ($p < 0.01$), which were positive. As income increases, the number of hiking trips taken tends to decrease. The negative coefficient on gender indicates that females take more trips. The positive coefficient on age indicates that older people take more trips. Surprisingly, group size was positive indicating that larger groups take more trips.

Consumer surplus

Using the coefficient for travel cost (TC), the consumer surplus per trip for Montanans is $11.54 with a 95% confidence interval of $10.89 to $12.28. Using the coefficients for travel cost and the interaction variable between Colorado and total cost to calculate the consumer surplus per trip, the average net benefit for Coloradoans was $54.59 per individual per trip with a 95% confidence interval of $33.79 to $141.94. These results are similar to other studies such as Walsh et al. (1992), who estimate the national average value of hiking to be $29, and Rosenberger and Loomis (2000), who updated the Walsh study arriving at a value of $37.

The effects of fire age on consumer surplus were tested using the value of recovering prescribed fire in Montana and Colorado. Change in consumer surplus was significant in Colorado. As prescribed fires recover, net benefits per individual in Colorado increase. For example, a 25-year-old fire would result in consumer surplus of $89, and for a 50-year old fire, consumer surplus increases to $242. While the value of a recovering prescribed fire was significant in Colorado ($p < 0.01$), the value of recovering prescribed fires in Montana was not significant indicating that consumer surplus in Montana is not affected by time. The increase in annual value for Colorado is 346% over 50 years, whereas over the same time frame, the increase in annual value in Montana is 1.7%. Changes in visitation and value are shown in table 4.

Alternately, the economic effects as a result of a crown fire were statistically significant for both Montana and Colorado, yet the difference in visitation between states was not significant. Given a crown fire, annual individual benefits decrease in Montana by 86.7% and 69.3% in Colorado. These results are statistically significant, and have implications from both policy and management perspectives.

Fire effects

The time since prescribed fire had a slightly positive effect on visitation in Montana and Colorado, and was significantly different between states (p <0.00). However, absolute differences are small enough to have no policy implications when considered alone. For example, trips taken in Montana increases from 11.25 with no fire, to 11.34 with a 25-year-old fire. For a 50-year-old fire, the average number of trips increases to 11.45. Over the same period, trips taken in Colorado increases from 10.28 to 10.30 and 10.33. Thus, the outward shift of the demand curve over the fire recovery interval indicates a very small increase in visitation.

Table 4—Visitor use and benefits with fire age.

Prescribed fire		**0 years**	**25 years**	**50 years**	**% change**
Colorado					
	Trips	10.28	10.30	10.33	
	Value	$54.59	$89.10	$242.20	
	Annual value	$561.18	$917.61	$2,501.92	+346 %
Montana					
	Trips	11.28	11.30	11.45	
	Value	$11.54	$11.54	$11.54	
	Annual value	$130.17	$130.40	$132.13	+1.7%
Crown fire		**0 years**	**25 years**	**50 years**	
Colorado					
	Trips	10.28	10.28	10.28	
	Value	$54.59	$25.66	$16.77	
	Annual value	$561.18	$263.78	$172.40	-69.3%
Montana					
	Trips	10.25	9.98	9.71	
	Value	$11.54	$2.83	$1.62	
	Annual value	$118.28	$28.24	$15.73	-86.7%

The presence of a crown fire was positive and statistically significant (p <0.00), yet there was no difference between states. Trips in Colorado increase from 10.28 to 11.38 given a crown fire, and from 10.25 to 11.48 in Montana. The effect on visitation of time since crown fire was negative and significant for both states (p <0.01). The interaction term indicates that older crown fires receive fewer visits than newer crown fires. Trips to areas with crown fires that are 25 and 50 years old decrease slightly from 9.98 to 9.71 in Montana with no change in Colorado. This may be explained by the initial interest in seeing effects of severe fires.

Conclusion

The average number of individual trips taken per site in a no-fire situation in Colorado was 10.28 with individual net benefits per trip of $55. The number of individual trips taken per site in Montana was similar at 10.25 with individual net benefits of approximately $12.

With respect to fire effects, findings indicate that wild and prescribed fires have varying effects on recreation demand and value in each state. When visitation and value are considered in conjunction, however, prescribed fires result in increased annual values in Colorado (346%). While this is significant, the change in Montana is not (1.7%). Alternately, crown fires in both states result in decreased annual values of 69.3% in Colorado and 86.7% in Montana. While respondents in each state do not behave similarly with respect to prescribed burning, these results provide support for the National Fire Plan (USDI/USDA 2002). In Colorado, prescribed burning not only increases the annual value of recreation over time, but may mitigate increasing social costs resulting from crown fires. In Montana, whereas prescribed fire does not increase value over time, it may have value in terms of mitigating the negative effects on annual recreation values as a result of crown fires.

Because of the rapid pace of education in natural resources, particularly with media coverage of fire, it would be useful to conduct the same survey in the future to test differences over time. While results may be used to generate the social costs of prescribed fires, such costs may fall over time with education and increased knowledge, and may have a different pattern in other states.

Literature Cited

Allen, Geoffrey P., Thomas H. Stevens, and Scott A. Barrett. 1981. The effects of variable omission in the travel cost technique. Land Economics. 57(2): 173-180.

Boxall, Peter, D. Watson, and J. Englin. 1996. Backcountry recreationists' valuation of forest and park management features in the Canadian Shield Region. Canadian Journal of Forest Research. 26: 982-990.

Creel, Michael D., and John B. Loomis. 1990. Theoretical and empirical advantages of truncated count data estimators for analysis of deer hunting in California. American Journal of Agricultural Economics. 72(2): 434-441.

Englin, Jeffrey, John Loomis, and Armando González-Cabán. 2001. The dynamic path of recreational values following a forest fire: a comparative analysis of states in the Intermountain West. Canadian Journal of Forest Research. 31: 1837-1844.

Englin, Jeffrey, P. Boxall, K. Chakraborty, and K. Watson. 1996. Valuing the impacts of forest fires on backcountry forest recreation. Forest Science. 42: 450-455.

Flowers, P., H. Vaux, P. Gardner, and T. Mills. 1985. Changes in recreation values after fire in the Rocky Mountains. Res. Note. PSW-373. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest & Range Experiment Station. 15 p.

Loomis, John B., Armando González-Cabán, and Jeffrey Englin. 2001. Testing for differential effects of forest fires on hiking and mountain biking demand and benefits. Journal of Agricultural and Resource Economics. 26(2): 508-522.

Loomis, John B., Jeffrey Englin, and Armando González-Cabán. 1999. Effects of fire on the economic value of forest recreation in the intermountain west: preliminary results. Gen. Tech. Rep. PSW-GTR-173. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 199-208.

Rosenberger, Randy, and John B. Loomis. 2001. Benefit transfer of outdoor recreation use values. Gen. Tech. Rep. RMRS-GTR-72. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 59 p.

USDI/USDA. 2002. National Fire Plan. www.nfp.gov.

U.S. Water Resources Council. 1983. Principles and guidelines for water and related land implementation studies. Washington DC: U.S. Water Resources Council.

Vaux, H.J. Jr., P.D. Gardner, and T.J. Mills. 1984. Methods for assessing the impact of fire on forest recreation. Gen. Tech. Rep. PSW-GTR-79. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station. 13 p.

Walsh, R., D. Johnson, and J. McKean. 1992. Benefit transfer of outdoor recreation demand studies: 1968-1988. Water Resources Research. 28(3): 707-713.

Linking GIS and Recreation Demand Models to Estimate the Economic Value of Using Fire to Improve Deer Habitat

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***Abstract**—This research combines a Geographic Information System (GIS) based model of deer habitat response to fire with a travel cost method recreation demand model to value the deer hunting benefits of prescribed burning in the San Jacinto mountains of southern California. A statistically significant effect of fire on deer harvest was determined in the GIS based and time-series based production function models. Using the recreation demand model, we estimated the net economic value to hunters of $257 per additional deer harvested. While the initial deer hunting benefit response to the current magnitude of prescribed burning of 1,100 acres ranges from $4,112 to $8,481 depending on the production model, the incremental gains for additional 3,700 acres of prescribed burning are quite similar across production models.*

Introduction

This research compares two models for evaluating the effectiveness of prescribed burning for increasing deer habitat. We also provide a benefit-cost comparison for the San Jacinto Ranger District (SJRD) in the San Bernadino National Forest located in southern California. The methodological contribution begins to answer the challenge posed by Hesseln (2000) in her recent review of the economics of prescribed burning. She stated: "There is a lack of economic models to evaluate short- and long-term ecological benefits of prescribed fire. Without understanding the relationship between economic outcomes and ecological effects, it will be difficult to make effective investment decisions. Research should focus on defining a production function to identify long-term relationships between prescribed burning and ecological effects. Identifying production functions relationships will form the basis for future cost-benefit analysis with respect to prescribed burning" (Hesseln 2000: 331-332). Our study demonstrates two different approaches to estimating production relationships between prescribed burning and deer harvest using time series data and Geographic Information System (GIS) approaches. The production models are linked to the recreational hunting valuation model by including deer harvest as a demand shift variable in the recreational hunting valuation model.

Study Area

The San Jacinto Ranger District (SJRD) is located in Southern California's San Bernardino National Forest near Palm Springs. As noted by the USDA

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Forest Service, "Some of the best deer hunting in Riverside County is found in this area" (Gibbs et al. 1995: 6). The SJRD is an ideal area to demonstrate and compare different approaches to estimating a production function between prescribed burning and deer harvest because prescribed fire has been used for more than 20 years to stem the long-term decline in deer populations since the 1970s (Paulek 1989, Gibbs et al. 1995). Previous research on prescribed burning shows that fire enhances deer habitat and populations (California Department of Fish and Game 1998) but the economic benefits have not been quantified. The results of our analysis should be of some policy relevance as the SJRD plans to increase the amount of prescribed burning by 50% to 100% over the next few years (Gibbs et al. 1995).

In general, Southern California is characterized by a Mediterranean climate, with hot and dry summers and cool, moist winters. There is a significant range of variation in temperatures and local site conditions in the ranger district in the San Jacinto and Santa Rosa Mountains. Elevations in these ranges reach 10,800 feet. The dominant vegetation within the SJRD below 5,000 feet is chaparral. Annual rainfall for the chaparral biome is approximately 15 to 16 inches. Areas above 5,000 feet tend to be dominated by hardwoods and conifers such as live oak and Douglas-fir with annual rainfall of up to 30 inches.

Within the San Jacinto Ranger District, the land is primarily managed by the USDA Forest Service, with small amounts of land administered by the State of California such as the Mount San Jacinto State Park. The land within the San Jacinto Ranger District is an area that evolved with fire as a natural environmental factor. Declining abundance of successional vegetation communities is considered to have the greatest long-term effects on deer populations (California Department of Fish and Game 1998). Historically, fire, either prescribed or natural, has been the primary mechanism for establishing these vegetation communities. Studies in California have noted that after a burn, increased deer numbers can be attributed to individuals moving into the area to feed (Klinger et al. 1989). These increased deer numbers are thought to arise due to increased forage quality and increased fawn survival rates in the recently burned areas. The California Department of Fish and Game has noted a significant increase in buck harvest from 1987 to 1996 in hunt zones that had large fires, versus hunt zones that did not have large fires (California Department of Fish and Game 1998). To improve deer habitat in California, controlled burning has been used in all the major parks and forests for more than a decade (Kie 1984).

Two Production Function Modeling Approaches

The purpose of this study is to test whether prescribed burning has a systematic effect on deer harvest. By examining prescribed burning on deer harvest with two different production models, a macro or aggregate time series approach and a micro, spatial approach (e.g., GIS) can be compared. Using a macro time series approach, we would be able to test the effects of fire, prescribed and natural, across the entire study area over a 20-year time period. Using a micro GIS approach provides greater spatial detail, such as the influence of a meadow or ridge, but this micro data is not available for the 20-year time period. Thus each approach to estimating the production function has its relative strengths and weaknesses.

Estimating a production function that relates deer harvest to acres of prescribed burning must also control for other inputs that influence the production of deer for harvest. This includes wildfire, elevation (used as a proxy for vegetation data that was incomplete), rainfall, temperature, and distance to roads. Thus, multiple regression is an appropriate technique.

Time Series, Macro Scale Production Function

The macro approach is based on a time series regression model to test for a relationship between deer harvest in the SJRD and prescribed fire, controlling for other independent variables such as annual precipitation and temperature during the hunting season. For this approach we used a dataset for SJRD, provided by the California Department of Fish and Game (CDFG) and the USDA Forest Service. The fire records provided data from 1979 for wildfire and prescribed burns within the San Jacinto Ranger District. This ranger district represents the majority of publicly accessible land for deer hunting in Riverside county. Deer harvest data from 1979 were provided by CDFG. The full model is given by equation 1.1. SJRD Time Series Production Function Model:

$$\text{SJRD Deer harvest in year}_t = \text{func}(\text{RXFIRE}_t, \text{WILDFIRE}_t, \text{TOTPRECIP}_t, \text{OCTTEMPt}, \text{YEAR}_t) \quad [1.1]$$

Where:

$RXFIRE_t$ = the acres of prescribed fire in year t
$WILDFIRE_t$ = the acres of wildfire in year t
$TOTPRECIP_t$ = the sum of precipitation for year t
$OCTTEMP_t$ = temperature in October during the hunting season
$YEAR_t$ = a trend variable, with 1979 = 1, 1980 = 2, … 1998 = 19

We estimate a non-linear form of equation 1.1 using the log-log form. The log-log form allows us to interpret the coefficients for fire effects as elasticity's, i.e., the percent change in deer harvest with a 1% change in acres burned.

To model the aggregate harvest for all of the San Jacinto Ranger District, the dependent variable is the total number of deer harvested in year t. The value of the dependent variable is a relatively large number and varies between 80 to 157 deer in any given year. Because of the relatively large values for the dependent variable, Ordinary Least Squares (OLS) is an acceptable approach for the macro time-series modeling.

Micro GIS Approach to Estimating the Production Function

The approach taken in this study uses a geographic information system (GIS) for estimating the deer harvest-fire production relationship. With the GIS approach, the study area is divided into 37 individual hunting zones delineated by California Department of Fish and Game (rather than treating the entire SJRD as one unit). These hunt areas are defined by topographic features such as steep ridgelines or developed features such as towns or major roads. This allowed for the incorporation of other influences on deer harvest that varied spatially across individual hunting areas such as distance to roads and elevation. Because past research indicates that use of burned areas by deer increases dramatically during the following years, (Klinger 1989) a lagged model as shown in equation 1.2 is estimated. This model tests for these effects during the following years by using a lag on the fire variables.

The first step in the GIS analysis was to identify the necessary layers needed to run a regression between deer harvest and fires. A harvest layer was constructed, which contains deer harvest by hunt zones and serves as the dependent variable. Then layers were added for the independent variables including acres of prescribed burning, wildfire acres, average elevation, temperature, distance to trails, distance to dirt roads from each hunting zone, and distance to wildfires from each hunting zone. Vegetation type would have been desirable, but this information was incomplete and will not be completed for the entire area until well into the future. Thus elevation was partially used as a proxy for vegetation composition. We developed two models for the harvest areas to account for the non-uniform size of each hunting zone: (1) include the size of the harvest area as a separate independent variable and use total acres of an area burned and, (2) transform the dependent variable into deer harvest per acre, then use an OLS regression. The total area model (with deer harvest a function of total size of fire, including lags) is shown in equation 1.2:

$$\text{Deer harvest in area}_i \text{ in year }_t = \text{func}\,(\text{Avg\_Elev}_i,\ \text{Ltotal\_Wildfire}_{i\ t},\ \text{Ltotal\_Wildfire}_{i\ t\text{-}1},\ \text{Ltotal\_Wildfire}_{i\ t\text{-}2},\ \text{Ltotal\_Wildfire}_{i\ t\text{-}3},\ \text{Ltotal\_Rxfire}_{i\ t},\ \text{Ltotal\_Rxfire}_{i\ t\text{-}1},\ \text{Ltotal\_Rxfire}_{i\ t\text{-}2},\ \text{Ltotal\_Rxfire}_{i\ t\text{-}3},\ \text{Ldirt\_distance}_i,\ \text{Ltrail\_distance}_i,\ \text{LHvst\_Area}_i,\ \text{Oct\_Temp}_{\ t},\ \text{Year}_{\ t}) \qquad [1.2]$$

Equation 1.2 was estimated using a count data model instead of OLS regression because at the micro level harvest in any limited spatial unit is a small non-negative integer variable. Therefore count data models are statistically more efficient because such models are based on probability distributions that have mass only at nonnegative integers (Creel and Loomis 1990). This is certainly the case for deer harvests as hunters cannot harvest a fraction of a deer and the number harvested in each unit is typically 0,1,2,3… rather than 10 or 50. One of the simplest count distributions is the Poisson process. Given the stringency of the mean-variance equality restriction imposed by the Poisson, a more generalized count model like the negative binomial is often more consistent with the data. The negative binomial version relaxes the mean-variance equality of the Poisson. Both the Poisson and the negative binomial yield the equivalent of a semi-log form where the log of the dependent variable is regressed against the explanatory variables.

An alternative specification to account for the different size harvest areas involved transforming the dependent variable into deer harvest per acre. This results in equation 1.3, which is estimated using ordinary least squares regression since this dependent variable is continuous and is not restricted to integer values:

$$\text{Log Deer\_harvest per acre in year}_{i\ t} = \text{func}\,(\text{Avg\_Elev}_i,\ \text{Ltotal\_Wildfire}_{i\ t},\ \text{Ltotal\_Wildfire}_{i\ t\text{-}1},\ \text{Ltotal\_Wildfire}_{i\ t\text{-}2},\ \text{Ltotal\_Wildfire}_{i\ t\text{-}3},\ \text{Ltotal\_Rxfire}_{i\ t},\ \text{Ltotal\_Rxfire}_{i\ t\text{-}1},\ \text{Ltotal\_Rxfire}_{i\ t\text{-}2},\ \text{Ltotal\_Rxfire}_{i\ t\text{-}3},\ \text{Ldirt\_distance}_i,\ \text{Ltrail\_distance}_i,\ \text{LHvst\_Area}_i,\ \text{Oct\_Temp}_t,\ \text{Year}_{\ t}) \qquad [1.3]$$

Details of GIS Based Micro Regression Variables

Elevations are based on USGS digital elevation models and act as a proxy for vegetation types, which were not available. However, we do not have an expected sign on elevation, but include it to control for elevation differences among the 37 individual hunting areas within the San Jacinto Ranger District. Based on the literature reviewed above, wildfire and prescribed fire are expected to have a positive sign. The distances to road and trail variables are based on the distances from a central point in each hunting zone. Two arguments can be made about the sign of this variable, therefore the expectation is left to be ambiguous. One argument is based on accessibility for hunters, where being close to either a trail or road would make hunting easier and more desirable, which would positively effect deer harvest. The second argument is based on the intrusion of the road or trail possibly fragmenting deer habitat. This perspective would lead to a decline in deer harvest because roads cause a break in habitat and pose as a threat from cars.

The distance to fire variable is based on distance from a central point in each hunting zone to the closest fire in that time period. This variable's sign may be either positive or negative.

Harvest area, which takes into account the size of each hunting zone, is expected to have a positive sign. The argument here is that as hunting areas become larger, then the amount of deer habitat increases, which attracts more deer and increases the probability of hunter success. October temperature and year are the other variables used in the models. October is when hunting season is open and, based on hunter's surveys, when temperatures are high the deer tend to bed down and seek cover. Therefore, harvest rates decline, which gives the October temperature a negative sign. Year is a trend variable to capture any temporally varying effects and we do not know whether it would be positive or negative. Table 1 summarizes the description of the variables and their expected sign, if any.

Table 1—Description of GIS based micro regression variables.

Variable	Description	Expected sign on coefficient
Deer harvest	The dependent variable; the number of deer harvested in a designated hunting zone.	
Avg_Elev	Average elevations, based on USGS Digital Elev. Model and re-classed into elevation categories.	No expectation
TOTWFIRES	Total wildfires in a particular year within the San Jacinto Ranger District (-1,-2,-3 are time lags).	+
TOTRXFIRES	Total prescribed fires within the San Jacinto Ranger District for a particular year (-1,-2,-3 are time lags).	+
DirtDist	The distance to the nearest dirt road, in meters from a central location of each hunting zone.	No expectation
TrailDist	The distance to the nearest trail, in meters from a central location of each hunting zone.	No expectation
Fire_Dist	The average distance from a central location of each hunting zone to the central point of a wildfire.	No expectation
HuntArea	The size of each harvest area, measured in acres.	+
Oct_Temp	The average temperature in October, degrees Fahrenheit.	_
Year	A trend variable to look for systematic changes.	No Expectation

Estimated Production Functions

Macro Time Series San Jacinto Ranger District Equations

Allowing for non-linearity proved to be a better predictor of deer harvest than the linear models (linear results available from the authors) so we present the double log model. Results from preliminary regressions also suggested combining the wildfire and prescribed burn into one variable. In table 2, the coefficient for total fire has a small magnitude of .048, but it has a significant t-statistic of 2.3. The sign on this variable is positive and the coefficient can be interpreted as an elasticity due to the log-log functional form. Therefore, a 1% increase in acres burned will lead to a .048% increase in deer harvest. The

Table 2—Macro time series ranger district log-log model. Dependent variable is the log of SJRD deer harvest.

Variable	Coefficient	Std. error	t-statistic	Probability
Constant	41.80865	11.07	3.776	0.002
ln_Totalfire	0.048735	0.0205	2.371	0.032
Total_Precip	-0.00096	0.0026	-0.366	0.719
Oct_Temp	-0.02703	0.0106	-2.536	0.023
Year	-0.01785	0.0055	-3.199	0.006

R-squared: 0.677	Mean dependent var: 4.808
Adj. R-squared: 0.584	S.D. dependent var: 0.202
S.E. of regression: 0.1303	F-statistic: 6.343
Durbin-Watson: 2.066	Prob (F-statistic): 0.002

other significant variables are October temperature and year. The statistically significant negative sign on the October temperature coefficient is consistent with the opinion of hunters that an increase in temperature results in a decrease in the number of deer harvested. The year variable indicates that a systematic time trend effect exists within the model. This model's explanatory power is reasonably good with an R^2 value of .67. The Durbin-Watson statistic of 2.06 indicates that autocorrelation is not a problem. The same model presented in table 2 was estimated with a 1-year lag but this model did not perform well using the lag. The coefficient on the lag of total fire (-1) was .01 and the t-statistic is .44, which indicates the lag is insignificant. The R^2 value did not change from the model with the no lag (results available from authors).

Summary of Micro Regressions Based on GIS Analysis

The two regression models estimated using GIS derived data are presented in this section: one count data and the other OLS, both of which show prescribed burning had a statistically significant effect on deer harvest. As can be seen in table 3a, total acres of prescribed fire is significant during the year of the prescribed fire, and its significance declines over the next three years in the count data model. During the first year, prescribed fire's coefficient is .044 with a t-statistic of 2.4. Since this count data model logs the fire acreage variables it is equivalent to a log-log model. As such, the .044 is the elasticity, which is remarkably similar to the .048 elasticity in the macro time-series model reported in table 2. The total acres of wildfire variable was not significant for any of the years in this equation. The total area count data model has an R^2 value of .25.

Using OLS as an estimator of deer harvest per acre as a function of fire and the other variables provides a similar pattern of signs and significance as the total area count data equation. In this model, a double log form was also used, but this time the dependent variable acts as a controlling measure for the size of each harvest area by dividing harvest in each hunting zone by the number of acres in each zone. The result of this model in table 3b shows that prescribed burning has a statistically significant effect on deer harvest in the first year with a t-statistic of 2.25. Then during the years following the fire, prescribed burning becomes less significant, which corresponds to the previous count data model. The only time wildfire has a significant impact is during the second year following the burn. The sign of the coefficient for wildfire in the second year is negative and less than one, which would imply a negative effect on deer harvest in that year. Distance to dirt roads is also significant,

Table 3a—Count data model based on GIS using total acres burned with lags. Dependent variable is DEERKILL; n = 825. Method is ML - Negative Binomial Count.

	Coefficient	Std. error	t-statistic	Probability
C	62.96425	23.1157	2.7238	0.0065
LAVG_ELEV	-0.237276	0.13070	-1.8153	0.0695
LTOTWFIRES	0.010712	0.01714	0.6248	0.5321
LTOTWFIRES(-1)	0.008299	0.01701	0.4877	0.6257
LTOTWFIRES(-2)	-0.027728	0.01548	-1.7902	0.0734
LTOTWFIRES(-3)	-0.02466	0.01557	-1.5829	0.1134
LTOTRXFIRES	0.044067	0.01790	2.4608	0.0139
LTOTRXFIRES(-1)	0.027531	0.02701	1.0192	0.3081
LTOTRXFIRES(-2)	0.011491	0.02223	0.5169	0.6052
LTOTRXFIRES(-3)	0.011491	0.01866	0.6155	0.5382
LDIRTDIST	-0.233799	0.03774	-6.1943	0
LTRAILDIST	0.395161	0.04175	9.4633	0
LFIRE_DIST	0.072684	0.04739	1.5335	0.1251
LHUNTAREA	0.940678	0.08699	10.812	0
OCT_TEMP	-0.012073	0.01681	-0.7178	0.4728
YEAR	-0.034733	0.01176	-2.9534	0.0031
Overdispersion parameter:				
Alpha: C(17)	-0.2810	0.1081	-2.598	0.0094

R^2: 0.257	Mean dependent var: 1.7587
Adjusted R^2: 0.242	S.D. dependent var: 2.6113
S.E. of regression: 2.2731	Restr. log likelihood: -1920.6
	LR index (Pseudo-R^2): 0.3051

Table 3b—Least squares deer harvest per acre using GIS data model with lags. Dependent variable is LDEERKILLAC; n=825. Method is least squares.

Variable	Coefficient	Std. error	t-statistic	Probability
C	1.341834	1.58826	0.844	0.3984
LAVG_ELEV	-0.009703	0.00927	-1.046	0.2958
LTOTWFIRES	0.001163	0.00120	0.963	0.3357
LTOTWFIRES(-1)	0.000499	0.00116	0.429	0.6678
LTOTWFIRES(-2)	-0.002231	0.00107	-2.086	0.0373
LTOTWFIRES(-3)	-0.001775	0.00109	-1.627	0.104
LTOTRXFIRES	0.002635	0.00116	2.254	0.0244
LTOTRXFIRES(-1)	0.002134	0.00188	1.134	0.2568
LTOTRXFIRES(-2)	0.001333	0.00154	0.862	0.3889
LTOTRXFIRES(-3)	0.001212	0.00130	0.926	0.3546
LDIRTDIST	-0.012952	0.00250	-5.174	0
LTRAILDIST	0.01825	0.00222	8.213	0
LFIRE_DIST	0.005144	0.00320	1.607	0.1084
LHUNTAREA	-0.008678	0.00615	-1.409	0.159
LOCT_TEMP	-0.050413	0.08604	-0.585	0.5581
YEAR	-0.002827	0.00082	-3.436	0.0006

R^2: 0.138	Mean dependent var: -4.532
Adjusted R^2: 0.122	S.D. dependent var: 0.093
S.E. of regression: 0.087	F-statistic: 8.684
	Prob (F-statistic): .0000

a t-statistic of 5.17 and a negative coefficient of -.012. This means that harvest areas farther away from dirt roads have a lower probability of harvesting a deer. The positive sign on the distance to trails variable implies increases in the probability of a deer harvest the farther the hunting area is from a trail. All the other variables in this model fail to be significant indicators of deer harvest, except for the trend variable, year. Therefore, some unidentifiable systematic temporal change is occurring within the model. Overall this model has a lower level of explanatory power than the total area micro count data model. The R^2 value using OLS is .13 as compared to twice this level of explanatory power in the total area count data model.

Applying the Regression Production Functions

To calculate the incremental effects of different levels of prescribed burning on deer harvest, the acres burned variable is increased from one level to a higher level in the regression model. We use the double-log macro time-series model and the micro GIS-based double-log total area count data models, as these two models have the highest explanatory power. The resulting predicted change in deer harvest will be valued in dollar terms in the next section.

Applying Results of Micro GIS Production Function Model

The results of the "Total Acres Burned" count data model from table 3a provide positive evidence on the desirable effects of prescribed burning programs on deer harvest. The first row in table 4 forecasts the estimated number of deer that would be harvested without having a prescribed burning program. The second row in table 4 represents the current level of prescribed burning. The effect of increasing prescribed burning is calculated by increasing the number of acres burned in each of the 37 hunting areas by 100 acres per hunting area, and then 200 acres per hunting area (for a total of 8,510 acres) to evaluate a wide range of prescribed burning levels in the SJRD. The first level (1,100 acres) is about the average prescribed burning over the last 20 years. Maintaining this level of prescribed burning does provide a significant increase in deer harvest over the no burning level. However, the gain in deer harvest increases more slowly with additional increases in burning in each hunt area.

Table 4—Comparison of deer harvest response to prescribed burning using the macro time series model and GIS micro model.

RX acres burned	Additional acres burned	Macro time series model: # deer harvested	Time series marginal increase in deer harvest	GIS micro model: # deer harvested	GIS marginal increase in deer harvest
1	NA	83	NA	42	NA
1110	1110	116	33	58	16
4810	3700	124	8	66	8
8510	3700	128	4	71	5

Applying Results of Macro Time Series Production Function Model

To estimate the change in deer harvest using the Macro Time Series Production Function Model, the double log model reported in table 2 is used. The total fire variable in this model is increased and the predicted level of deer harvest is calculated at the mean of the other variables. This is done at the same four acreage levels used above.

The results in table 4 suggest there is a substantial gain in deer harvest with the first 1,100 acres burned, especially as calculated from the macro time-series model. However, a very similar diminishing marginal effect is evident from both the macro time-series production function regression and the micro GIS production function regression after burning more than 1,100 acres. That is, regardless of the spatial level of detail adopted, burning an additional 3,700 acres is expected to result in about eight more deer being harvested in the SJRD.

In order to determine the economic efficiency of additional prescribed burning it is necessary to compare the benefits of additional prescribed burning in the form of the economic value of deer harvest against the costs. It is to the development of the valuation data that we now turn.

Valuation of Deer Hunting

According to CDFG, deer hunting is considered to be one of the major outdoor recreation activities in SJRD (Gibbs et al. 1995). Previous research on deer hunting in California showed that increased success rates and opportunities to harvest a trophy deer increase the economic value of deer hunting (Loomis et al. 1989, Creel and Loomis 1992).

Travel Cost Method for Valuation of Deer Hunting

The Travel Cost Method (TCM) has been a primary approach for valuing recreational hunting. The basic concept of TCM is that the travel cost (i.e., transportation cost, travel time) to the site is used as a proxy for the price of access to the site. When hunters are surveyed and asked questions about the number of trips they take and their travel cost to the site, enough information is available to estimate a demand curve. From the demand curve, net willingness to pay or consumer surplus can be calculated (Loomis and Walsh 1997).

Besides variable travel cost or its proxy, travel distance, inclusion of a travel time variable in the demand function is necessary to represent the opportunity costs of time as part of travel costs. Cesario (1976) suggested one-fourth the wage rate as an appropriate estimate of the opportunity cost of time based on commuting studies. For individuals with fixed workweeks, recreation takes place on weekends or during pre-designated annual vacation and cannot be traded for leisure at the margin. In such cases, Bockstael et al. (1987) and Shaw (1992) suggest that the opportunity cost of time no longer need be related to the wage rate. These studies suggest that both the travel cost and travel time be included as separate variables, along with their respective constraints, income and total time available for recreation.

Table 5 contains a list and definition of variables used in the TCM demand model. For this study we chose the TCM variables according to the consumer demand theory and existing literature on deer hunting in California. Individuals who hunt on opening day, belong to hunting organizations, hunted in previous seasons, and had a successful deer harvest may take potentially more hunting trips because such hunters have higher preferences, experience or skill in deer hunting recreation. Because a majority of hunters in our dataset work a fixed workweek, we assume deer hunters maximize utility level subject to their income and time constraints (Shaw 1992). In other words, total time available for recreation is a constraint similar to income for time intensive activities like hunting. The total time budget is constructed for the TCM model using responses to the survey questions regarding availability of

Table 5—Variables included in travel cost model.

Variable	Definition
Dependent variable	
NUMTRIPS	Number of primary purpose of deer hunting trips taken to the SJRD during 1999 deer hunting season.
Independent variables	
AGE	Hunter's age
DEERKILL	Did you harvest a deer in this area during this hunting season? 1= YES, 0 = NO
HUNTOPEN	Did you hunt on opening day of the season? 1= YES, 0 = NO
HUNTORG	Are you a member of a sportsman's organization? 1= YES, 0 = NO
PREVSEAS	Have you hunted in this area in a previous season? 1= YES, 0 = NO
PRIVLAND	Did you hunt on private land? 1= YES, 0 = NO
RTRAVMILES	Round trip travel miles from home to the hunt zone
PCINC	Hunter income
TOTIMEBUD	Total time budget during hunting season
TRAVETIME	Number of hours one-way travel time

vacation time and time periods chosen to hunt (e.g., weekends only versus weekdays). In this study, the total time budget ranged from 8 to 31 days because the deer-hunting season in SJRD lasted for one month only.

Count Data Nature of TCM Dependent Variable

The nonnegative integer characteristic for the dependent variable, number of seasonal trips, is from a count data process. Given the count data form of the dependent variable, a preferred estimation technique would be the negative binomial count model to estimate the demand function (Creel and Loomis 1990). The negative binomial is the more generalized form of the Poisson distribution, which allows the mean of trips to be different from its variance. The count data TCM model is specified in equation 2.2:

$$\begin{aligned} \mathrm{NUMTRIPS} = \mathrm{EXP}\,(&C(1) + C(2)^*\mathrm{AGE} + C(3)^*\mathrm{DEERKILL} \\ &+ C(4)^*\mathrm{HUNTOPEN} + C(5)^*\mathrm{HUNTORG} + C(6)^*\mathrm{PREVSEAS} \\ &+ C(7)^*\mathrm{PRIVLAND} - C(8)\ ^*\ \mathrm{RTRAVMILES} + C(9)^*\ \mathrm{PCINC} \\ &+ C(10)^*\mathrm{TOTIMEBUD} - \mathit{C(11)^*TRAVTIME}) \end{aligned} \qquad [2.2]$$

In equation 2.2, we expected the coefficient for DEERKILL (i.e., C (3)) to have a positive sign, because hunters would likely take more hunting trips if the hunting quality has been good. Also, if hunters hunt on the opening day (i.e., C (4)), private land (i.e., C (7)), and/or previous seasons (i.e., C (6)), and belong to hunting organizations (i.e., C (5)), then we expected a positive effect on the number of trips the hunter takes as these variables indicate a strong preference for the deer hunting activity. For those hunters with a higher income level (i.e., C (9)) and/or higher total time budget (i.e., C (10)) we expect more hunting trips as well due to less binding income and time constraints. However, round-trip travel distance (i.e., C (8)) and travel time (i.e., C (11)) are expected to have negative effects on the number of hunting trips because increases in these two variables result in higher hunter's expense.

Calculation of Consumer Surplus in TCM

The consumer surplus from deer hunting is computed from the demand curve as the difference between what people are willing to pay (e.g., the entire area under the demand curve) and what people actually pay (e.g., their travel costs). Because the count data model is equivalent to a semi-log functional form, consumer surplus from a trip is calculated as the reciprocal of the coefficient on round trip travel miles, expressed in RTRAVMILES scaled to dollars using the cost per mile (Creel and Loomis 1990).

Hunter Survey Data

For cost effectiveness in data collection, a mail questionnaire was sent to a random sample of deer hunters with licenses for zone D19, which includes the San Jacinto Ranger District. Of 762 questionnaires mailed to deer hunters in California during the 1999 hunting season, 7 were undeliverable. A total of 356 deer hunters' responses were collected after two mailings. The response rate is approximately 47%. Among these respondents, 69 did not hunt in the San Jacinto Ranger District portion of Zone D19. The response rate of this study is suspected to be low because many of the hunters that did not hunt in the SJRD portion of the D19 Hunt Zone may not have returned the survey.

Statistical Results

Estimation results are summarized in table 6. There is a negative effect of travel miles, travel time, and income on number of trips taken. Income, in this study, is insignificant. The regression results of this study indicate that hunters which successfully harvested a deer during the hunting season (i.e., DEERKILL), hunted on opening day (i.e., HUNTOPEN), hunted in this area in a previous season (i.e., PREVSEAS), and had a larger total time budget (i.e., TOTIMEBUD) had positive and significant effects on the number of hunting trips taken. Consistent with economic theory, hunters with longer round trip travel miles (RTRAVMILES) and greater travel time (TRAVTIME) tend to take fewer hunting trips.

Table 6—Estimated negative binomial count data TCM demand equation. Dependent variable is NUMTRIPS.

	Coefficient	Std. error	Z-stats	Probability
Constant	1.324485	0.2163	6.1226	0.0000
AGE	0.001395	0.0037	0.3684	0.7125
DEERKILL	0.366571	0.1547	2.3695	0.0178
HUNTOPEN	0.524153	0.1148	4.5640	0.0000
HUNTORG	0.067655	0.1058	0.6390	0.5228
PREVSEAS	0.285282	0.1344	2.1217	0.0339
PRIVLAND	0.038041	0.1314	0.2892	0.7724
RTRAVMILES	-0.002230	0.0008	-2.4906	0.0128
PCINC	-1.00E-06	2.78E-06	-0.359	0.7192
TOTIMEBUD	0.010128	0.0048	2.0994	0.0358
TRAVTIME	-0.289315	0.0867	-3.3340	.0009

R^2: 0.2058, Adjusted R^2 : 0.1685
Consumer surplus: $134.53/trip
90% confidence interval: $81.13 ~ 393.59
Marginal consumer surplus per deer harvested: $257.17/deer
90% confidence interval: $154 ~ 752

In table 6 the consumer surplus is calculated by:

$1/\beta$(i.e., coefficient of distance) * \$0.3/mile (i.e., cost per mile).
= 1/0.002230 *\$0.3 = 448.43 * \$0.3 = \$134.53/trip, where the \$0.30 is the sample average cost per mile.

Finally, the 90% confidence interval in table 6 is obtained by the following equation:

90% confidence interval on consumer surplus per trip = 1/ (β_{DIST} ± 1.64 * 0.000895) * \$0.30/mile = \$81.13 ~ \$393.59 dollars per trip

Estimating the Benefits of Harvesting an Additional Deer

The average number of trips per hunter is 5.56 trips and 10% of deer hunters successfully harvested a deer. To calculate the incremental or marginal value of an additional deer harvest we can use the TCM demand equation to predict the extra number of trips deer hunters would take if they knew they would harvest a deer that season. This essentially shifts the demand curve out by the amount of the coefficient on deer harvest. The equation predicts that each hunter would take 1.9116 more trips each season if they knew they would harvest a deer. Therefore, the marginal value of another deer harvested (i.e., marginal consumer surplus) is equal to \$134.53 * 1.9116 = \$257.17 per deer harvested. Finally, the 90% confidence interval in table 6 for an additional deer harvested is obtained by applying the 90% CI on the value per trip times the additional number of trips taken by the hunter: 90% confidence interval of the value of harvesting an additional deer =

1.9116 * \$81.13 ~ 1.9116*393.59 = \$155 ~ \$752 dollars per deer harvested.

Benefits of Prescribed Burning

Table 7 summarizes this study's main conclusion—the annual deer hunting benefits of additional acres of prescribed burning. While the initial deer hunting benefit response to prescribed burning of 1,100 acres ranges from \$4,112 to \$8,481 depending on the model used, the incremental gains for more than the current acreage of prescribed burning is quite similar across models. That is, the annual economic hunting benefits of increasing prescribed burning from its current level of 1,110 acres to 4,810 acres is \$2,056, regardless of the model used. Likewise for an additional 3,700 acres of prescribed burning to

Table 7—Annual deer hunting benefits from increased prescribed burning: macro time series model and GIS micro model results.

RX acres burned	Additional acres burned	Time series marginal increase in deer harvest	Annual increase in deer hunting benefits	GIS marginal increase in deer harvest	GIS Annual increase in deer hunting benefits
1	NA	NA	NA	NA	NA
1110	1110	33	\$8,481	16	\$4,112
4810	3700	8	\$2,056	8	\$2,056
8510	3700	4	\$1,028	5	\$1,285

8,510 acres, the deer hunting benefits are calculated to be between $1,028 to $1,285 each year, fairly similar despite the different modeling approaches.

Comparison to Costs

Discussions with fire management personnel on the San Bernadino National Forest suggested that their prescribed burning costs range from $210 to $240 per acre. This is a lower total cost per acre than reported by González-Cabán and McKetta (1986), but substantially higher than the direct costs per acre for southwestern National Forests in Wood (1988). Nonetheless, if we use the $210 per acre figure, the full incremental costs of burning the first 1,100 acres would be $231,000, with each additional 3,700 acres burned costing $779,100. The additional benefits of deer hunting benefits represent at most about 3.4% of the total costs of performing the first 1,100 acres of prescribed burning. This finding can be used in two ways. First, the incremental costs of including deer objectives in the prescribed burn should not exceed $8,000, as the incremental benefits are no larger than this. Second, the other multiple use benefits such as watershed and recreation, as well as the hazard fuel reduction benefits to adjacent communities, would need to make up the difference if the prescribed burning program is to pass a benefit-cost test.

Conclusion

This study evaluated the response of deer harvest and deer hunting benefits to prescribed burning in the San Jacinto Ranger District in Southern California. To estimate hunter's benefits or willingness to pay (WTP) for harvesting an additional deer, the individual observation Travel Cost Method was used resulting in a mean WTP to harvest another deer of $257. With regard to the response of deer harvest to prescribed and wildfire, we compared a macro level, time-series model which treated the entire San Jacinto Ranger District as one area and a micro GIS model which disaggregated the Ranger District into the 37 hunting areas delineated by California Department of Fish and Game. The macro time-series model estimated a larger response to burning of the first 1,000 acres than the micro GIS model did, but for increases in fire beyond 1,000 acres, the two models provide nearly identical estimates.

Using the marginal willingness to pay for harvesting another deer calculated from the TCM demand model, the deer harvest response to fire yields annual economic benefits ranging from $4,112 to $8,481 for the first 1,100 acres burned. For an additional 3,7000 acres burned, the gain is $2,056 annually, while for a second increase of 3,700 acres (for a total of 8,510) the increase ranges from $1,028 to $1,285 per year. The costs of prescribed burning on the San Bernadino National Forest range from $210 to $240 per acre. Thus the cost to burn an additional 1,100 acres is $231,000, which is an order of magnitude larger than the deer hunting benefits gained. Specifically, the deer hunting benefits of the first 1,100 acres burned represent about 3.4% of the total costs of conducting the first 1,100 acres of prescribed burning. However, there are probably other multiple use benefits such as protecting watersheds and wildfire hazard reduction. These other multiple use benefits of prescribed burning would have to cover the rest of the costs of prescribed burning if the program is to be economically feasible. Investigating the extent of these benefits would be a logical next step in evaluating the economic efficiency of prescribed burning in the San Jacinto Ranger District.

While fire management practices have been identified as having widespread impacts on deer habitats, many other factors that affect deer habitat exist. These other factors include livestock grazing, timber harvesting, urban development, diseases, habitat loss, and annual weather patterns (CDFG, 1998). This study attempted to take into account as many factors as possible, but the amount of data and time available for modeling were a constraint.

Some future improvements in our modeling effort that may better isolate the effects of prescribed burning on deer habitat include controlling for the severity of wildfire as different fire severities will have different effects on vegetation and soils. Furthermore, including vegetation and soils layers in the GIS model, rather than using elevation as a proxy, could improve the predictive ability of the GIS-based model.

Subject to these caveats, we have demonstrated two approaches to estimate a production function relating prescribed burning to effects on deer harvest. We found positive and significant effects on deer harvest for the two GIS models and a positive impact of fire using a macro-time series model. The USDA Forest Service and California Department of Fish and Game can make use of these approaches for future cost-benefit analysis of prescribed burning.

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References

Bockstael, Nancy E., Strand, Ivar E. and Hanemann, W. Michael. 1987. Time and the recreation demand model. American Journal of Agricultural Economics 69(2): 293-302.

California Department of Fish and Game. 1998. An assessment of mule and black-tailed deer habitats and populations in California, with special emphasis on public lands administered by the Bureau of Land Management and the United States Forest Service. General Technical Report. 89p.

Cesario, F.J. 1976. Value of time in recreation benefit studies. Land Economics 52(1): 32-41.

Creel, M. D. and J. Loomis 1990. Theoretical and empirical advantages of truncated count data estimators for analysis of deer hunting in California. American Agricultural Economics Association 72(2): 434-441.

Creel, M. D. and Loomis, J. 1992. Modeling hunting demand in the presence of a bag limit, with tests of alternative specifications. Journal of Environmental Economics and Management 22(2): 99-113.

Gibbs, P., D. Freeman, C. Harvey, R. Lawson, K. Valenzuela, D. Pumphrey. 1995. Area analysis and plan for prescribed burning. San Jacinto Ranger District, San Bernadino National Forest. 23p.

González-Cabán, A. and C. McKetta. 1986. Analyzing fuel treatment costs. Western Journal of Applied Forestry. 1(4): 116-121.

Hesseln, Hayley. 2000. The economics of prescribed burning: A research review. Forest Science 46(3): 322-334.

Kie, John G. 1984. Deer habitat use after prescribed burning in Northern California. Research Note PSW-369. Berkeley, CA: U.S. Department Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station. 3p.

Klinger, Robert C.; Kutilek, Michael J.; Shellhammer, Howard S. Population responses of black-tailed deer to prescribed burning. Journal of Wildlife Management 53(4): 863-870.

Loomis, J., D. Updike and C. Unkel 1989. Consumptive and nonconsumptive values of a game animal: The case of California deer. Transactions of the 54th North American Wildlife and Natural Resources Conference. Wildlife Management Institute, Washington DC.

Loomis, John and Walsh, Richard G. 1997. Recreation economic decision: comparing benefits and costs 2nd Ed. State College, PA: Venture Publishing, Inc. 430 p.

Paulek, Tom. 1989. Santa Jacinto/Santa Rosa Mountains deer herd management plan. California Department of Fish and Game. Report, Sacramento, California. 65p.

Shaw, W. D. 1992. Searching for the opportunity cost of an individual's time. Land Economics 68 (1): 107-15.

Wood, D.B. Costs of prescribed burning in southwestern ponderosa pine. Western Journal of Applied Forestry 3(4): 115-119.

Co-Firing Wood Biomass With Coal at the Cañon City Power Plant

Daniel Prokupets[1], Dr. Kurt Mackes[2], and Dr. Skip Smith[2]

Abstract—*Green full tree chips produced from small diameter trees can be co-fired with coal. This paper reports some of the challenges and benefits at a Cañon City power plant. Tree chips include the main stem, branches, tops, needles, and bark. If a sufficient supply of low-cost wood is available, the power plant will continue to burn it. Collaboration among agencies and private companies is crucial for continued success.*

Introduction

Past fire management and suppression practices have created conditions favorable for catastrophic fire throughout wildland / urban interface zones of the Intermountain West. These past practices are responsible for greater standing fuel loads that potentially threaten public safety, watershed productivity, and forest health. The National Interagency Fire Center (2002) reported that over 8.4 million acres of forest land burned nationwide during the 2000 fire season, which compares to a 10-year average (1990-1999) of about 3.8 million acres per year. Unfortunately, unless efforts to reduce standing fuel loads are implemented, it appears that conditions will continue to be favorable for large wildfires in future years.

In response to the 2000 fire season, the National Fire Plan (Department of the Interior 2001) was developed. One objective of the National Fire Plan is to implement strategies that mitigate the severe fire conditions existing on forest land throughout the Intermountain West. These strategies revolve primarily around forest restoration (fire mitigation) thinning, prescribed burning, or a combination of both (usually mechanical removal or thinning followed by prescribed burning). Because of inherent risks associated with prescribed burns in the absence of prior fuel reduction, forest restoration thinning is now being utilized throughout the Intermountain West to mitigate fire conditions.

Forest restoration thinning involves removing primarily small diameter trees, which are found abundantly in the Intermountain West. In Colorado, small diameter trees are considered to be less than 12 inches in diameter at breast height (dbh). However, the majority of trees removed during thinning projects are usually considerably less than 12 inches dbh and often less than 5 inches dbh. In addition to being small, these trees tend to have many limbs and correspondingly their wood has many knots usually considered to be defects in solid wood products. Small diameter trees also have disproportionately high quantities of juvenile and reaction wood that further reduce wood quality. As a result, many of these trees are currently unmerchantable.

Therefore, the question has arisen, how can these small trees be utilized? The potential for producing composite products, such as oriented strand board (OSB) or particleboard, and pulp and paper products is limited in Colorado

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by the lack of infrastructure. This is also true for other products such as pelletized fuel. Because of the high capital investment required to build plants for producing these products, it is doubtful that a private company would risk building such a plant unless guaranteed a wood supply. No such guarantees currently exist. Because small trees have little or no value, much of this material is left in the forest lopped and scattered or chipped and blown back onto the forest floor. In addition to being wasteful, the impacts of doing this on soil and ground water are unclear and currently being studied.

Utilizing green wood chips produced from small diameter trees to generate electric power is one alternative. There are currently numerous plants throughout the country that utilize wood biomass as a source of fuel for generating electricity (Bain and Overend 2002). Although there are plants that use only biomass, many also co-fire wood with other fuels such as coal or natural gas. Co-firing is not a “new” technology and many power plants that operate on coal do not even have to modify their handling and boiler systems to utilize some wood. Therefore, these plants can burn wood with little or no capital investment.

Background

The W. N. Clark power plant located in Cañon City, Colorado, is capable of co-firing wood with coal. The plant is an older facility and has experimented with co-firing in the past. Prior efforts were discontinued because of inconsistent wood supply and quality issues with the size of wood particles provided. Wood residues procured from a local sawmill were not of uniform size, which caused handling problems. Also, fine wood particles and dust in the residues had a tendency to become airborne and were considered a potential explosion risk. Nonetheless, the plant expressed interest in co-firing if a sufficient economical wood supply of acceptable quality and size could be procured.

This research was conducted to demonstrate that the W. N. Clark power plant could utilize green full tree chips produced from small diameter trees. Full tree chips included the main stem, branches, tops, needles, and bark of the tree. A resource assessment is also in progress to identify potential sources of low cost wood and to determine the best scenario for wood delivery, processing, and storage. This project was a collaborative effort of Aquila (W. N. Clark power plant), Colorado State Forest Service, Department of Forest Sciences at Colorado State University, City of Cañon City, Fremont County, and Sangre de Cristo Resource Conservation & Development.

Discussion

The W. N. Clark power plant is currently permitted to burn up to 5 percent wood by weight with coal. To demonstrate that the plant could burn green full tree chips for an extended period of time, ponderosa pine chips were supplied from forest restoration thinning projects in the region. The Colorado State Forest Service subsidized the transportation of chips to the plant. Chip deliveries began in September 2001 and the plant began co-firing immediately. Initially a mix of less than 1 percent green wood chips by weight or 1 to 2 tons of wood per day was used. Over 200 tons of green wood chips have been co-fired with coal since September.

Chip moisture content ranged from 20 to 70 percent. The moisture in the chips did not present any significant problems for plant systems except for reducing the amount of recoverable heat. Because of the impact on recoverable heat, wood chips used in this research would have had to be considerably drier (20 percent moisture content or less) to significantly increase the amount used. The plant is currently evaluating ways to increase the percentage of wood burned and there is potential to burn up to 25 tons per day.

Several sources of wood for the plant are being investigated. These include continued supply of small diameter trees from forest restoration thinning projects in the region, local primary and secondary wood processors, and arborists and municipalities in the area.

As part of the National Fire Plan, 38,400 acres are to be treated in Colorado during 2002 (Department of the Interior 2001). Based on an analysis by Lynch (2000) of forest restoration thinning work done at Cheesman Reservoir in Colorado, treating these acres could conservatively yield from 9 to 15 green tons of green biomass per acre. Therefore, 345,000-576,000 tons of green biomass could potentially be available. The primary constraint on availability is the distance that this biomass would need to be transported. Transportation of green wood chips to the W. N. Clark power plant is currently being subsidized and even though the plant can pay for wood, in the absence of the subsidy, it is not likely that chip transport would be economical. Transportation data is currently being collected and maximum haul distances are not yet known, but they are considerably less than 100 miles (more likely in the range of 20 to 30 miles). In addition to acres that are to be treated under the National Fire Plan, more wood biomass could be available through road construction, defensible space efforts, and other logging activities not considered as part of the National Fire Plan.

Primary and secondary wood processors could also be a significant supplier of wood chips. In spring 2001, Ward (2000) surveyed 173 primary and secondary wood manufacturers operating in Colorado. The 75 companies that responded generated 380 tons of residues per week on average. About 83 percent were willing to consider alternatives to current wood biomass disposal practices. More importantly, two sawmills in the immediate area said they would be interested in supplying residue to the plant. However, handling and dust problems that occurred when the W. N. Clark power plant attempted to use sawmill residues in the past would likely reoccur.

Another possible supply of green wood chips could be urban wood residues. This would include residues from work done by local arborists and municipalities, as well as construction and demolition debris going into the local landfill. An advantage of this wood material is that it can often be procured at little or no cost. Results from a survey of local arborists conducted by Prokupets (2002) revealed that they conservatively generate an estimated 2000 tons of wood debris annually, which is currently enough to supply the W. N. Clark power plant. However, a preliminary test burn conducted with wood residues (chips) supplied by a local arborist was not successful. As with wood processing residues, wood particle size was inconsistent and over sized (long, stringy) particles clogged the fuel handling system at the plant. As a result, future research will include evaluating wood processing equipment (chippers and grinders) to determine the most cost effective way of producing a wood chip suitable for use at the plant.

Conclusion

Research to date has demonstrated that green full tree chips can be co-fired with coal at the W. N. Clark power plant. The plant has indicated that it will continue to burn wood if a sufficient supply of adequate low cost wood is available. Although it appears that there is a sufficient supply of low cost wood available to the plant, further research is necessary to determine how best to collect and process the wood biomass into a useful size and form.

Generally, project success depends on overcoming several major challenges. The costs associated with procuring and co-firing wood chips with coal must be economical. The logistics of wood delivery, chipping, and storage must be evaluated to determine the most cost effective methods. Transportation is a major cost of getting wood from forest restoration thinning projects to the plant. Haul distances will likely have to be considerably less than 100 miles and government subsidies may be necessary to cover all transportation costs. Perhaps most crucial to the success of this project is the collaboration of the various agencies, private companies, and individuals who will be involved.

Literature Cited

Bain, Richard L., and Ralph P. Overend. 2002. Biomass for Heat and Power. Forest Products Journal. 52(2):12-19.

Department of the Interior. 2001. National Fire Plan Website: Colorado State Summary from Department of the Interior. 2001. <http://www.fireplan.gov/statebystate/Colorado1.cfm>

Lynch, Dennis L. 2000. Summary Report: Forest Restoration Thinning Project on Denver Water Board Property at Cheesman Reservoir and Trumbull Sites, May 8 to July 2000. Colorado State University. Fort Collins, CO.

National Interagency Fire Center. 2002. Wildland Fire Season 2000 at a Glance. NIFC, Boise, Idaho. <http://www.nifc.gov/fireinfo/2000/>

Prokupets, Dan. 2002. Phone Survey of Arborists in the Cañon City Area. Colorado State Forest Service, Fort Collins, CO.

Ward, Julie E. 2000. Phone Survey of Primary and Secondary Wood Processors along the Front Range of Colorado. Colorado State Forest Service, Fort Collins, CO.

Treatment—Social Issues

Fire Social Science Research: Opening Remarks

Antony S. Cheng[1]

Introduction

Determining the "appropriate place, appropriate time" for fuels treatments in response to forest wildfire risk is a social process that blends scientific information with social values and attitudes. Natural resource scientists and managers must realize that whether they like it or not, they contribute to this process. Social science perspectives and research can complement technical analyses to determine if, when, and where fuel treatments are appropriate.

Why Bother With Social Science?

The forestry community has long been aware of the importance of garnering public support for management decisions. Public land management decisions in particular have been subject to scrutiny and conflicting perspectives. Indeed, opinion polls and more rigorous random sample surveys show that the public is wary of logging on public lands. In recent years, this wariness has transferred even to fuel treatments to reduce the risk of catastrophic wildland forest fires. Even in geographic areas such as the Intermountain West where large forest fires have raised public awareness about unhealthy forest conditions, there are mixed messages about the relationship between the role of fire, forest health conditions, and forest practices. Fuel treatments such as thinning are still relatively new concepts and may conflict with deeply held values.

Perhaps more significant is the varying degrees of trust in resource managers. For whatever reason, public distrust of forestry professionals can be the most significant barrier to implementation, even if the science and economics are sound. For the public's part, there is a lot at stake—aesthetics, property values, and conflicting visions of what forests should look like in the future.

In short, the sustainability of fuel treatment programs turns on public understanding and acceptance. Social science methods can provide insights and methodologies to identify gaps in public understanding and barriers to acceptance. Social science can also provide useful perspectives on institutions, organizational behavior, and decision-making. Social science research can complement forest resource managers' efforts to innovate and adapt according to changing social contexts.

Three Problem Dimensions Worth Exploring

1. Who is the public?

A primary challenge facing forest resource managers is to identify the relevant public stakeholders who may be affected by or interested in the effects

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of fuel treatments. A rich array of social assessment tools and data sources are readily available to address this challenge. Obviously, the general public needs to be understood, especially concerning fuel treatments on public lands. However, resource management decisions are not necessarily popularity contests. Additionally, random sample general public surveys tend to aggregate responses across many important variables, such as knowledge of forest conditions and forest fire risk, risk perception, and geographic context, to name a few.

Public stakeholders that may have a more direct bearing on the appropriate place and timing of fuel treatments are communities, residents, and landowners directly affected by forest fires and fuel treatments. Such stakeholders tend to be more vocal and mobilize in response to activities that run contrary to their values. The challenge faced by forest resource managers vis-à-vis these stakeholders is to listen, understand, and work with the stakeholders towards productive courses of action. Of course, every community is different and the responses of community stakeholders to fuel treatment efforts can differ significantly depending on the context – geographic, social, economic, and political.

The importance of context cannot be overstated. Public understanding and acceptability of fire and fuel treatments tend to be “flashy” or episodic based on recent events. For example, people with immediate experience with the Cerro Grande fire of 2000 will have very different views on fuel treatments than individuals for whom the fire was a distant occurrence. Across communities, there are wide differences in expertise, capacity, and leadership to address unhealthy forest conditions and mitigate fire risk. One can imagine communities lying along a gradient of knowledge, capacity, and leadership. Two communities just 10 miles apart can have very different recognition of problems and understanding of options available to them.

2. Fuel treatments: the solution to what problem?

Despite the recent attention to fuel treatments among forestry professionals in the Intermountain West, questions remain about the true goals of such treatments, especially on public lands. The different perceptions of goals mirror the deep-seated conflicts over the purpose of public lands in general: Are they to be managed for multiple human uses or to be protected as the nation’s remaining biological heritage in the midst of landscapes long dominated by humans?

Again, context matters: specific treatments and prescriptions may be acceptable at a general level, but there are likely significant differences across particular contexts. For example, one may support thinning out dense stands of ponderosa pine across the Rocky Mountain West, but would resist logging “in my backyard.” Why the change of heart?

A medical analogy may be useful. Prescribing morphine to dull intense pain and minimize suffering is generally accepted as sound practice. However, should a doctor prescribe morphine to cure a headache? A morphine prescription is a treatment, but the treatment depends on the nature of the problem. Therein lies a significant difference between forest resource management and medicine: reaching consensus on the nature of “the problem” is often elusive and the source of intractable conflict. Even in the wake of catastrophic wildfires, there remains public debate over the true nature of “the problem.” Is it to protect private property? To protect public safety and welfare? To restore a small area of forest like a domestic watershed or across a large landscape like the entire Interior West? Forest resource and fire managers should not take for granted the existence of a consensus on the definition of the problem. Indeed, a conference entitled, “Reclaiming the concept of forest restoration on public lands" is being held right after this conference in Boulder. It is being organized by a coalition of conservation groups and will highlight key differences in how the "forest health" problem is being defined in the public policy process.

3. Integration of science and public values

Forest management has always been a blend of science and values, yet the process of blending science and values has often been arbitrary and unsystematic. One result has been the increased intervention of the legislative and judicial systems in removing discretion and judgment from resource professionals in favor of highly regimented statutes, regulations, and procedures. A second result is that resource professionals are placed in the role of arbiter among competing stakeholder claims – the classic "loggers versus environmentalist" split being one such competition. Integrating science and public values can become obscured, leading to decisions that are neither technically sound nor socially acceptable.

Efforts are being undertaken across the West to move beyond "analysis paralysis" towards more collaborative approaches to defining and addressing problems related to unhealthy forests and fire risk. However, collaboration is easier said than done. Collaborative efforts are time-consuming and often do not produce expected results – perhaps because expectations are unrealistically high for collaborative processes. Much work remains to be done in designing and evaluating collaborative planning processes and adaptive management strategies. One thing is clear: there is no one universal model. However, collaborative processes hold the promise that determining the appropriate places and times for fuel treatments can be widely supported and readily implemented. The sustainability of forest ecosystems may indeed depend on collaborative processes making honest and earnest efforts.

Summary

Determining the appropriate places and times for fuel treatments to address forest fire risk occurs in a complex social context. Understanding and effectively engaging within this context is imperative for forest resource managers. Social science perspectives and research methods should not be considered addenda or afterthoughts in developing fuel treatment plans, as they can offer insights on public perceptions and strategies for effectively engaging public stakeholders.

People and Fire in Western Colorado: Methods of Engaging Stakeholders

Sam Burns[1], Chuck Sperry[2], and Ron Hodgson[3]

Abstract—*In the context of the National Fire Plan, greater attention should be given to the engagement of communities in mitigating catastrophic wildfires. An overview is presented of a study in Western Colorado based on over 25 focus groups. This study seeks to discover improved ways to foster participation and ownership among local citizens and stakeholders in fire prevention and education efforts. The focus group process addressed local defintions of the "wildfire problem," community values placed at risk by wildfire, conditions and resources that would facilitate greater community participation in dialogue and action, and recommended fire prevention messages and methods of communication and education.*

Introduction

The conduct of social science research about fire behavior and management should be placed in the context of the growing involvement of communities in stewardship improvements on public lands, or what in many circles is being called community-based forestry. In so doing, the focus of community-oriented research shifts from viewing people as mere respondents to a set of study questions toward being participants in a potential or anticipated community engagement process. This reorientation seems especially relevant in the additional context of the National Fire Plan, because of its mandates for greater involvement by citizens in addressing common resource management concerns in the community-public land interface.

Let us consider community-stewardship, civic engagement, and the National Fire Plan as an integrating context for the research project known as "People and Fire in Western Colorado," an inquiry that addresses how more meaningful community conversations might be pursued about catastrophic wildfire prevention and mitigation.

Community Stewardship and Civic Engagement

Attention to community stewardship is a growing phenomenon in natural resource planning and management. At the heart of this process is the basic principle that people, communities, and the surrounding landscapes need to be connected if they are to be mutually sustainable. (See Gray et al. 2001.) Whether the specific form of stewardship relies upon public participation, civic engagement, collaborative learning, community development, alternative conflict resolution, community action, or action research, the fundamental intent is to build new forms of problem-solving relationships whereby community

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members engage authentically with natural resources issues and goals. In other accounts, the theme of linking communities and public lands is paramount. (See the Four Corners Public Land Partnership program descriptions available at www.fourcornersforests.org, or Office of Community Services-Evaluation Reports at Fort Lewis College http: / / ocs.fortlewis.edu.)

Gary McVicker, a person well known in Colorado for his leadership in promoting community-based stewardship efforts, notes that traditional processes of land use planning place the lead planning organization in “the center of competing interests, ... but they have largely failed to win the support and, more importantly, ownership from these competing interests.” McVicker believes that much of public land use planning can be characterized as “more an investment in formal decision-making than in public consensus; demanding more and more information to satisfy public interests; hampered by administrative appeals; costly, and not tied to agency budget processes for implementation; and losing public interest and support” (McVicker, Unpublished paper.)

Margaret Shannon, a leading spokesperson for collaborative stewardship of public lands, in expressing her concerns about how public land interests become framed as private interests, says that “current political institutions which reward interest driven behavior…must be redesigned so as to require civic conversation when the public good is at stake” (Shannon 1992.)

As we empirically consider the range of public perspectives about fire behavior and management in local communities, should we not be thinking of developing a conversation among stakeholders, leading to community stewardship? Should our fire prevention education process not attune itself more clearly to community conversations, rather than stop at the content of the message? (See “A Civic Conversation about Public Lands: Developing Community Governance” by Sam Burns in Gray et al. 2001.)

The Community as the Context of the National Fire Plan

The central focus of the National Fire Plan, the “wildland-urban interface,” is by definition a community issue or concern. Whatever goals, plans, and actions are developed to reduce the risk of destructive fires at the border between public and private lands will require the support of many community groups, interests, or stakeholders. The hope that the National Fire Plan will change management emphasis from fire suppression to fire risk reduction underscores the need for greater participation and ownership on the part of communities.

Not only does this process need to involve communities in a collaborative planning process, it also needs to address long-term stewardship of larger scale ecosystems, and building economic capacity to reduce and utilize fuels removed from landscapes and watersheds adjacent to communities. (See case study on the Ponderosa Pine Forest Partnership, Richard and Burns 1999.)

Greg Aplet, a forest ecologist with the Wilderness Society, emphasizes the critical role of communities in the National Fire Plan when he notes that of the four primary actions needed, three of them are the responsibility of local communities. He notes:

“First, we must protect our communities…

Second, we need to determine where the places are where we can still allow fire to play its natural role…

Third, we must restore fire through prescribed burning in those forests whose structures will allow the safe reintroduction of fire…

Finally, on those parts of the landscape that will not burn safely, we must begin the process of mechanically treating fuels to create a structure that eventually will accept characteristic fire."

Aplet concludes, "As I review these steps, it occurs to me that only one of them, the management of those places where we will allow fire to burn, is primarily a federal responsibility. The other three will require unprecedented cooperation of multiple stakeholders and levels of government to achieve" (Aplet 2001, p.5).

Since the introduction of the principles of "community based ecosystem management," (see Gray et al. 2001), the stewardship capacity of local communities has been increasingly recognized. Furthermore, when placed in the context of the wildland-urban interface goals of the National Fire Plan, such principles become even more paramount, due to the heightened need for public-private cooperation and investment.

The National Fire Plan calls for increased action by communities in both planning and stewardship to reduce the risks of catastrophic wild fires. Citizens and leaders are being given increased opportunities to prioritize where risks exist, where fuel reduction efforts should occur, and the degree and scope of the fuel treatments. In Southwest Colorado, the five counties of Archuleta, Dolores, Montezuma, La Plata, and San Juan have developed, with support from the San Juan National Forest, county fire plans that identify high-risk or fire prone areas on private and public lands and propose a range of collaborative mitigation and prevention actions (available through the Office of Community Services, Fort Lewis College or see on line at Southwestcoloradofires.org). As the National Fire Plan is implemented, and as community interaction and partnerships evolve, there is increasing awareness that the values, attitudes, and knowledge held by community members about natural and prescribed fire are key components to successful mitigation of catastrophic wildfire. Why is this true?

The ways in which citizens and policy makers understand the role and significance of fire, and the condition of surrounding ecosystems, continuously affect the goals, strategies, and actions that they deem appropriate to reduce wildfire risk and to restore forested and rangelands to sustainable levels of health. If no collaboration is created in the wildland-urban boundary area, effective, public-private stewardship will be diluted, if not totally derailed.

Orientation of the Research Project

Previous community-oriented, social science research has tended to focus on public perceptions or acceptability of various fire management strategies. (See Cortner et al. 1981 and Machlis et al. 2002, which addresses previous research public perceptions and acceptability of fire.) Many studies have addressed perceptions of wildfire risks to communities, attitudes toward the role of fire in ecosystems, the degree to which managed fire will be deemed appropriate, and other topics. Since the National Fire Plan calls for resource managers to work cooperatively with communities and citizens to manage fire behavior and effects, it is imperative that multi-party resource stewardship efforts be undertaken. In cross boundary situations between private, local, and federal government entities, a lack of participation by one party will maintain existing hazardous fuels, which will negatively impact all adjacent properties

However, involving a variety of interests in collaboratively planning and implementing prescribed fire on public and private lands is an ambitious goal. To begin with, many of the parties do not share a meaningful common view of fire's role in the natural environment, its effects, or whether public investment in fire mitigation should become a high priority. (The debate over the proper role of thinning and prescribed fire has continued well into 2002, after another major wildfire season; see Kenworthy 2002 and Robbins 2002 as recent examples of the level of disagreement about defining the "problem" or "issue," or potential "solutions.")

To reach the goal of communities and fire managers (professional and volunteer) working alongside each other, improving public safety, and making forest lands more healthy, much more needs to be understood about the values and understandings of fire held by the various interest groups. In February 2002, social scientists (see Appendix 1 at the end of this paper) met to discuss the methodology of the People and Fire in Western Colorado Research Project. It was proposed that a process of collaborative action and convergent understanding needs to be constructed from an array of beliefs and understandings held by diverse interests and groups. With a deeper knowledge of such attitudes, values, and perceptions, there might be a greater likelihood of multiple-party cooperation in addressing wildland-urban fire mitigation. (figure 1).

As indicated by this model, the orientation of the proposed research is to discover from communities how best to establish a relevant dialogue about fire mitigation and prevention; that is, how to better create the civic conversations needed to produce a multi-stakeholder community fire plan. Achieving this civic dialogue has obvious implications for the methods and content of fire prevention education.

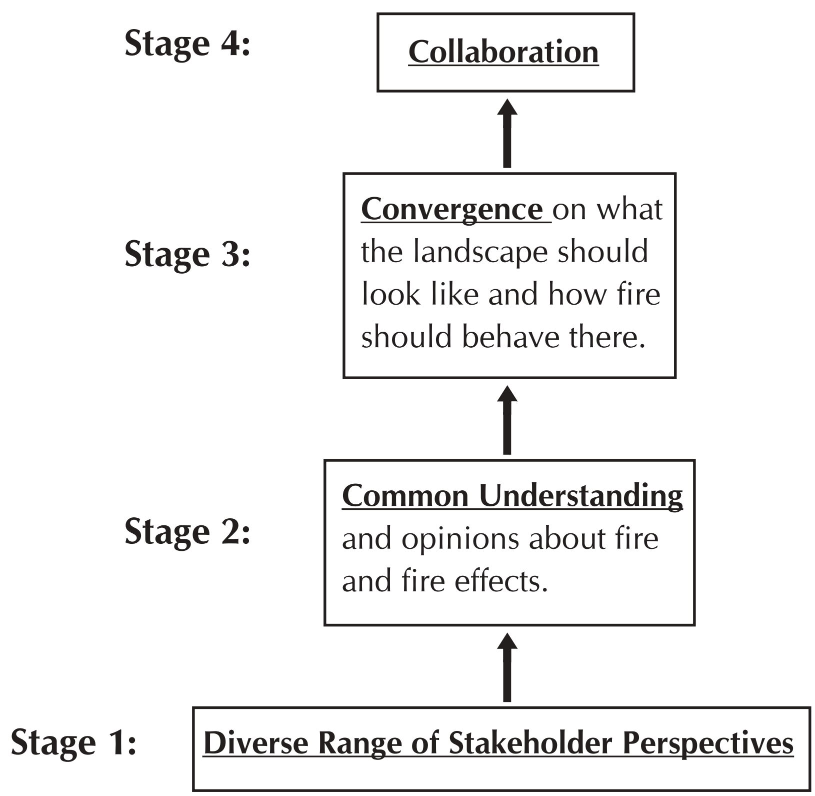

Figure 1—Process for collaborative action and convergent understanding.

Objectives

The West Slope Community Fire Research Project has two specific steps or phases:

• To identify the relevant individuals and organizations that have interests in fire, natural and prescribed; have a stake in how fire occurs and is managed;

and could play an active role in community efforts to reduce risks of catastrophic wildfire.

• To gather and document the values, attitudes, and knowledge held by these interests and stakeholders through facilitated group discussions in a manner which takes into account the social, economic, and cultural diversity of Western Colorado.

These objectives will constitute the two phases of the overall research project. The stakeholders will first be identified, and then a series of discussion or focus groups will be held within several natural, social areas of each subregion or study area (figure 2).

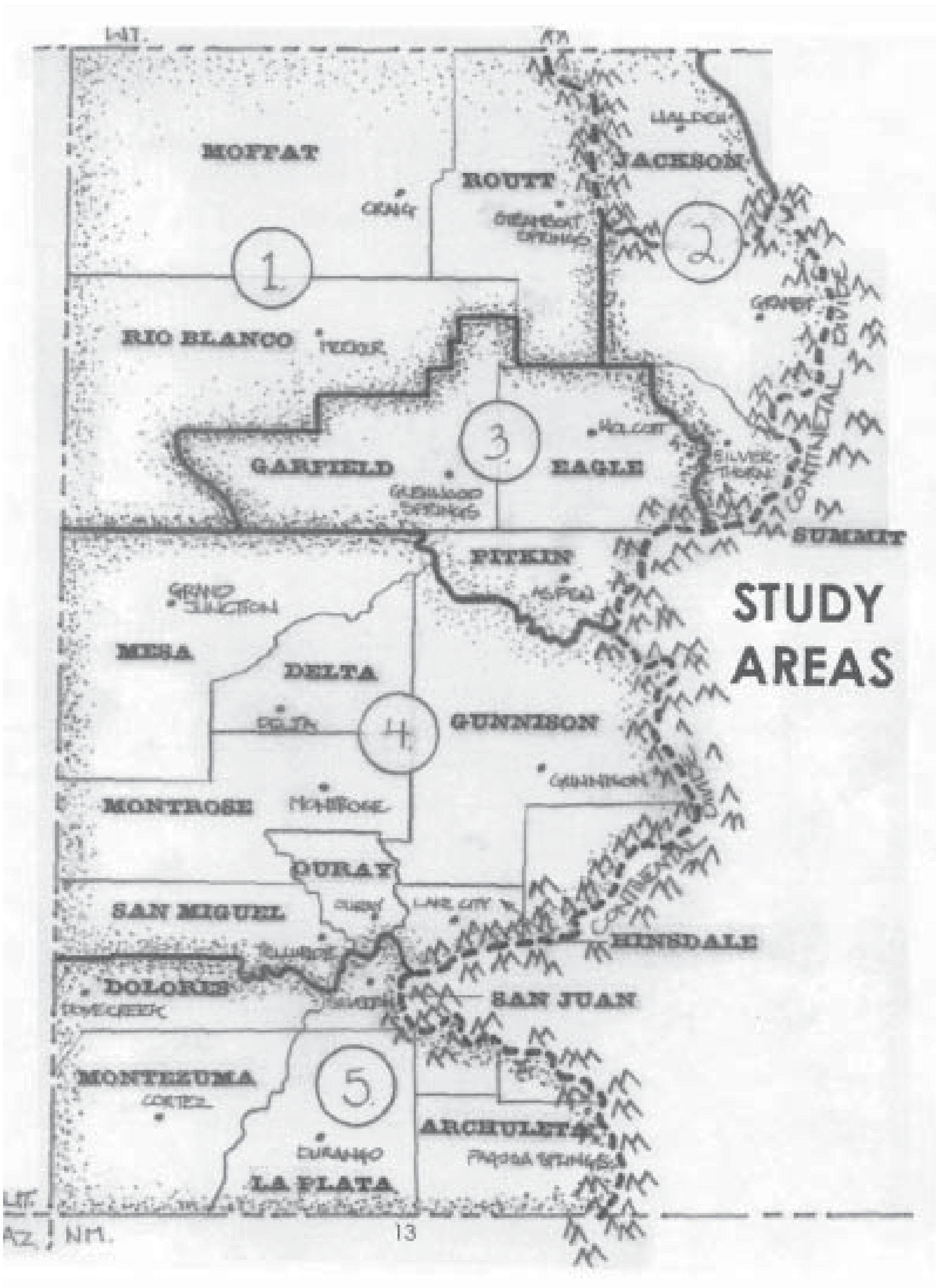

Figure 2—Study areas in Western Colorado.

Strategy

The study region for research on community and social understandings of fire is the Western Slope of Colorado, in essence that portion of the state lying west of the Continental Divide. This area consists of 21 counties, which will be divided into five sub-regions based on watersheds, economic patterns, and demographic and lifestyle characteristics. Throughout Western Colorado, the federal government, under the auspices of the U.S. Forest Service, the Bureau of Land Management, and the National Park Service, manages most of the land base. The tribal lands of the Southern and Ute Mountain Ute Tribes also contribute to a low percentage of privately owned lands, which in some counties can be under 10 percent.

The Western Slope in general can be described as a region in transition from an economy based in agriculture and mining to one linked to tourism, retirement communities, and recreation. However, the pace of this transition is markedly different throughout the Western Slope, which further underscores the need to assess the values and understandings of community members about fire, fire risk, and management within distinct sub-regions.

In order to begin the process of gathering the knowledge and understandings that could become the basis of a convergence-collaborative process, approximately 25 focus groups will be conducted in the 21 counties in Western Colorado. The counties will be divided into five study areas as per the attached map. A single two-hour meeting will be held in selected communities within each natural topographic-social region.

Diverse stakeholders will be chosen from a variety of interest areas such as recreation, wildlife, real estate, and local government. However, and perhaps more importantly, stakeholders will be selected for a balanced knowledge of both community and fire issues. Stakeholders need not be formal community leaders or professional experts about fire, although there could be some of these persons represented. It is preferable that the various interest-oriented stakeholders include persons who know something about citizen concerns about surrounding forest lands, about beliefs regarding natural and prescribed fire management, and about what it would take to reach common understandings about wildfire mitigation planning and decision making.

While focus group members may have strong views of their own about reintroducing fire into surrounding ecosystems, or thinning the lands adjacent to a given community, they should also be open to listening to other viewpoints in a dynamic group discussion. Most strategically, they should be willing to assist in describing what others believe or think about fire and appropriate management solutions, in a manner that could be utilized to build convergence and collaboration around a community based fire management plan.

The sample for this research will not be chosen randomly and evenly throughout Western Colorado. Rather, the participants will represent the attitudes and values of the social and cultural places where they live and work, or what many analysts refer to as a “sense of place.” Places in Western Colorado vary dramatically as a result of recent economic and demographic changes. There are traditional ranching communities like the west end of San Miguel County, and second home enclaves like Aspen. Among these and many other communities, there are quite different relationships with the surrounding forestlands. (See Swanson 2001 and the socio-economic data profiles on communities in Southwest Colorado collected as a part of the San Juan National Forest Plan Revision, 1996-98.)

Focus Group Inquiry

Four areas of inquiry will be pursued with each focus group:

Framing the Issues

- From your perspective, what is the wildfire problem or issue?
- Do you see wildfire as a problem, or merely an issue?
- Do some people in the community not see wildfire as a problem?
- How do various groups in your community view the wildfire problem-issue?
- What terms do people use to frame or describe the wildfire problem-issue?
- To whom do people attribute responsibility for the wildfire problem-issue and / or possible measures to reduce risk or threats, as they see them?

Community Values

- What locally held values cause people to think that wildfire is a concern, in the sense that those values could be threatened or might be compromised by wildfire?
- Do certain groups hold these values in particular? For example from a governmental perspective, or any specific interest group positions?
- Do you have a sense of what the most important community values are related to wildfire and improving community safety?

Capacity for Community Dialogue

- What conditions would need to exist in your community, in order for you and others to develop a productive dialogue on fire issues and / or any actions to reduce community risks? Examples of "conditions" could be a level of trust among key parties, a sense that participation in the dialogue would result in productive outcomes, or having reasonable access to information and knowledge about fire risk and environmental conditions. (There could be many other types of conditions.)

Education

- What do members of your community need to know to begin to talk productively about the wildfire issues and potential measures to improve community safety?
- Where do people prefer to obtain information about community issues of this nature? (Radio, TV, newspaper, workshops, etc.)
- Are any particular means or methods of receiving information more acceptable to community members than others? (Brochures, videotapes, group presentations, field trips, etc.)
- Are you, or others you know, willing to be a part of a monitoring group that would visit sites where efforts are being made to reduce wildfire risks in your community as a part of a learning dialogue?

Engaging Stakeholders

In the conduct of community planning and decision-making processes, it is rather routine to ensure that stakeholder identification is representative with regard to a broad range of socio-economic and demographic characteristics. Typically, stakeholder selection would take into account employment, length of residence, political power, and age, among many other societal dimensions. While the importance of these factors is unquestioned in considering the democratization of resource stewardship, in the context of developing community based fire mitigation efforts, this research project will emphasize a stronger sense of community capacity building, establishing civic dialogue, networking, relationship formation, and public conversation, and will therefore strongly influence the stakeholder engagement process.

In this light, the West Slope Fire Research Project has established a stakeholder identification and selection process based on the following conditions, assumptions, and attributes:

- A priority will be placed on stakeholder knowledge of local communities and their values.
- Similarly, stakeholders will be selected because they have a substantial degree of knowledge about fire and fire management, although they will not necessarily be "experts" in a professional or scientific sense.
- Engagement of stakeholders will occur on the basis of their active participation in envisioning and creating a civic conversation about fire impacts and mitigation measures.
- Stakeholders will be looked upon as representing communities of place, having social, experiential, and historical knowledge of a particular place, rather than as isolated, individual respondents who are merely sources of data or factual information.
- Utilizing local organizations, which facilitate the stakeholder nomination and selection process and serve as conveners of the focus groups, will increase the capacity of individual communities and regions to collaborate with fire management and education staff in the ongoing implementation of the National Fire Plan.
- Stakeholders will be given the option of serving as monitors in subsequent fire-risk reduction and education efforts, if such opportunities become available in their community study area.

Limitations and Challenges

Approaching the stakeholder identification and selection from a community action and development perspective, and seeking knowledge in an evolving local context, also present numerous challenges:

- Consideration has had to be given to coordinating with community fire planning that is in progress in many of the study areas. Some communities are just getting underway and others are nearly completed.
- In some cases, local fire management staff are reluctant to support a project, which they perceive as merely "research," rather than one that actually gets work completed on the ground.
- Simply creating a network of stakeholders on a one-time discussion basis may create longer term public expectations about civic engagement that need to be recognized and appropriately addressed through opportunities to participate in fire management planning.
- Creating a stakeholder group of diverse interests in a region also raises other issues, such as identifying communication barriers among various private, state, and federal jurisdictions, which affect the research outcomes in both short and long term ways.
- Communities may experience large catastrophic wildfires, accentuating in the minds of many the urgency for community action and work with fire managers.
- In essence, when working within an action-oriented stakeholder-based research process, the research step often blurs into "action thinking," to the

extent that in many communities there could be heightened interest in immediate or timely feedback of the project findings.

Summary Perspectives

Over the past 20 or so years, numerous studies have been undertaken with regard to social and communal values concerning fire risk, fire behavior and consequences, and fire management activities. Beginning in December of 2001, these studies were reviewed in order to discover previously asked questions and research findings. This review of previous social science research was compared and contrasted with several contemporary community fire planning efforts to prepare research agendas on the social aspects of fire, as well as recent conference proceedings. These findings were utilized as the basis for determining the focus of this research by the social science advisory team.

Within Western Colorado, five sub-regional study areas were chosen based on an analysis of river basins, social and economic characteristics, and other aspects of social and cultural geography and senses of place. These areas include Southwest Colorado south of the San Juan Mountains; the Uncompahgre and Gunnison Valleys from Ouray north to Grand Junction and running west to the Utah border; the high mountain areas between Glenwood Springs, Aspen, and Eagle along the I-70 corridor; and the northwest quadrant from Rio Blanco County east to the Routt County (figure 2). Within each of these five sub-regions, from three to five communities were chosen to conduct the facilitated group discussions.

Stakeholders were identified utilizing a wide range of networks within each sub-region. These include specific land and resource user groups, local government officials and staff, emergency management personnel, healthy community organizations, civic and non-profit groups, and wildlife and other conservation associations, among others. Stakeholder identification is being completed with an eye towards grounding the research process within local groups and networks, in anticipation that they can continue to participate in follow-up education, fire demonstration, and monitoring activities.

In February 2002, a meeting of social scientists was held in Fort Collins, Colorado, for the purpose of designing a protocol for the focus groups. This advisory team assisted in identifying research topics, key questions, and a scope of inquiry, which will be practical and advantageous to pursue in the group discussions. The discussion protocol served as a guide for the facilitators who conducted the focus groups, while allowing for local adaptability to fit special social and historical conditions.

Summaries of each group discussion were prepared. These were then collated into five study area reports, and finally into a Western Slope (the geographic area of Colorado west of the Continental Divide) set of findings and outcomes. The summaries will be made available to the local and constituent organizations, state and federal natural resource management staff, and fire education specialists for use in ongoing efforts to reduce catastrophic fire risk and implement local mitigation and stewardship practices.

The obvious question we have is whether engaging stakeholders for the purpose of developing community conversations and action about fire management will produce a different type and quality of local knowledge from standard survey research. Will this community-oriented stakeholder identification approach contribute to increased civic engagement and stewardship in the context of the National Fire Plan? Will this process of selecting persons with community knowledge and fire awareness have any implications for how to reach the less attentive and informed public?

We believe that the relevance of this research approach is the potential of creating a model for enhancing community capacity to engage in collaborative, fire mitigation planning in the wildland-urban interface.

References

Aplet, G. 2001. Fire planning and funding needs are local, not Federal. Communities and Forests, newsletter of the Communities Committee of the Seventh American Forest Congress. Fall issue.

Birchfield, Jim, ed. 2000. National Conference on the Social Acceptability of Fuel Treatments on Western Public Lands. October 22-24, 2000. Sponsored by the Bolle Forestry Center, University of Montana.

Cortner, H.; Zwolinski, M.; Carpenter, E.; and Taylor, J. 1981. Public support for fire management policies. Journal of Forestry. 82(6): 359-361.

Gardner, P., Cortner, H., Widaman, K., and Stenberg, K. 1985. Forest-user attitudes towards alternative fire management policies. Environmental Management. 9(4): 303-312.

Gray, Gerald J.; Enzer, Maia J.; and Kusel, Jonathan, editors. 2001. Understanding community-based forest ecosystem management. Binghamton, New York: Haworth Press Inc. (Food Products Press).

Kennworthy, Tom. 2002. US fire policy isn't cutting it. USA Today. August 22.

Machlis, Gary, et al. 2002. Burning questions: A social science research plan for Federal wildland fire management. National Wildfire Coordinating Group.

McVickers, G. No date. Community based partnerships and environmental health: A vision for change. Unpublished manuscript.

Moote, A. 2002. Community-based forest restoration: Three case studies in the Southwest. forthcoming in a guidebook on forest restoration. The Ecological Restoration Institute: Northern Arizona University.

Richard, Tim and Burns, Sam. 1999. The ponderosa pine partnership: Forging new relationships to restore a forest. Durango, CO: Fort Lewis College, Office of Community Services.

Robbins, Jim. 2002. Forest thinning challenged as a tactic to control fires. New York Times. August 27.

Shannon, M. 1992. Community governance: An enduring institution of democracy. Proceedings and Summary of the Workshop entitled Multiple Use and Sustained Yield: Changing Philosophies for Federal Land Management? (Printed for use by the Committee on Insular Affairs, U.S. House of Representatives.) Pp 219-250.

San Juan National Forest Plan Revision. 1996-1998. Socio-economic data / community profiles. Durango, CO: Fort Lewis College, Office of Community Services.

Swanson, Larry. 2001. The West's forest lands: Magnets for new migrants and part-time residents. Changing Landscape. 2(1): 16-21.

Appendix 1

Timeframes

The research project was initiated through an assistance agreement between the Office of Community Services at Fort Lewis College and the Colorado Office of the Bureau of Land Management in September of 2001. The following time frames serve as the general implementation schedule for the project:

- October 2001-March 2002 / Research Design, Resource Identification and Contracting: During this period, previous research was reviewed, key resource persons in Western Colorado were contacted, a science advisory committee met and made recommendations on research questions, and research contractors were selected.
- March 2001-April 2002 / Stakeholder Documentation: The stakeholder identification phase will be completed, utilizing five sub-regions to focus the inquiry.
- May 2002-November 2002 / Discussion Group Analysis: Within each sub-region, a series of facilitated group discussions will be held to describe and document the values and perspectives of the various interests of individuals and groups about fire.
- September 2002-February 2003 / Analysis and Reporting.

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From Analysis Paralysis to Agency-Community Collaboration in Fuels Reduction for Fire Restoration: A Success Story

Timothy Ingalsbee[1]

Abstract—*In 1996, the Ashland Ranger District of the Rogue River National Forest proposed the HazRed Project to expand a shaded fuelbreak system within the Ashland municipal watershed. The original proposal sparked intense community opposition and was withdrawn following administrative appeals. The Forest then proposed the Ashland Watershed Protection Project and used collaborative methods to generate continuous substantive public input. When a final decision was issued in 2001, the Project had gained enthusiastic community endorsement with volunteers helping to implement it on-the-ground. This story offers useful lessons for successfully overcoming "analysis paralysis" in fuels reduction and forest restoration projects.*

Introduction

In the winter of 2001, former chief of the U.S. Forest Service Jack Ward Thomas complained in testimony before Congress that the Forest Service was suffering from "analysis paralysis." The intended message was that it was becoming increasingly difficult for forest managers to implement management projects in a timely fashion due to a burdensome number of conflicting environmental regulations requiring lengthy public processes. In the public policy literature, "analysis paralysis" is a concept referring to an overload of data that makes it difficult to analyze effectively. The problem of the Forest Service, it is argued, is not due to an overload of data, but an overload of public controversy. Controversies are often generated by management proposals that involve commodity timber extraction, especially when these projects are presented as something else such as fire hazard reduction or forest restoration projects.

Individuals and organizations affiliated with the conservation community have been particularly adept at asserting their rights under agency regulations and the nation's environmental laws to prolong environmental analyses and decision-making. Forest Service decisionmakers sometimes misinterpret the public opposition to commercial logging as opposition to all forest management in general. The need for hazardous fuels reduction and forest ecosystem restoration, however, has created new management opportunities for both conflict and cooperation between federal agencies and local communities. The following paper will present the story of how a progressive Forest Service Ranger and a conservation-minded local community were able to teach and learn from each other, and transcend "analysis paralysis" over a contentious timber sale proposal, to eventually reach consensus on a restoration-oriented fire hazard reduction project within a municipal watershed. It promises to become a model of agency-community collaboration in fire restoration work,

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with long-term ecological and social benefits to the local landscape and wider society.

The HazRed Project

Draft Environmental Assessment

The Ashland Ranger District issued a scoping notice on July 5, 1996, for the "Ashland Interface Fire Hazard Reduction (HazRed) Project." The purpose and need for the HazRed Project was to "reduce fire hazard levels in strategic areas to protect values at risk of being lost to large-scale stand-replacing fire." The Draft Environmental Assessment (EA) was issued in February 1997 and proposed to treat 1,631 acres with a mix of commercial logging, manual cutting and piling, and prescribed underburning in order to construct and expand a ridgeline shaded fuelbreak system in the interior of the watershed. The fuelbreak would have reduced canopy closure to 30-40 percent, leaving an average 20-30 foot horizontal spacing between the crowns of dominant and codominant trees. All snags and large downed logs would be removed, and woody material on the forest floor would be reduced to an average of 1.5 tons per acre. The stated purposes of the fuelbreaks were to allow the safe deployment and evacuation of firefighters, increase the penetration of fire retardant through the forest canopy, and reduce the spread of running crownfires [Draft EA; p.4]. The Project would also have involved road and helispot construction in order to facilitate skyline and tractor yarding systems, and to shorten helicopter yarding distances.

The Ashland Municipal Watershed

Commercial logging is highly restricted in the Ashland Watershed and there were several indications that the proposed HazRed Project would spark significant controversy. First, the Project area is managed as a restricted watershed since it is the primary domestic water source for the City of Ashland. The City and the Forest Service have a cooperative agreement dating back to 1929 that requires the agency to consult with City officials prior to any plans to remove timber or other forest products from the Ashland Watershed. The watershed is characterized by steep, unstable slopes of decomposed granitic soils with naturally high rates of erosion, often in mass debris flows that dump sediment directly into streams. Just before the Draft EA was scheduled to be released, the watershed experienced a major rain-on-snow event that resulted in a 30 year flood event on New Year's Day in 1997. Many landslides were triggered alongside logging roads, the downtown commercial district was flooded by several feet of water, and the City was forced to import potable water for several weeks. The Draft EA alarmed the community since many of the proposed fuelbreak units were rated as having extreme landslide hazards and in some cases were located directly above active landslides.

Second, the project area was located inside a critical habitat unit and late-successional reserve (LSR) established by the Northwest Forest Plan to conserve habitat for the northern spotted owl and other old-growth associated species. The Mt. Ashland LSR, coincidentally, is a critical node or "crossroads" in the LSR system, linking the high elevation Siskiyou range of the Klamath mountains with the southern portion of the Oregon Cascades and the Oregon coast range. Commercial logging is highly restricted and intensely controversial in LSRs. Forest Service staff predicted that timber extraction for the fuelbreaks would cause long-term habitat degradation for eight pairs of northern spotted owls.

Third, the Ashland Watershed is located in the Klamath-Siskiyou bioregion, a proposed world heritage site renowned as one of the richest areas of biodiversity in the North American continent. Fire has played a major evolutionary role in shaping species composition, stand structure, and the amount and distribution of live vegetation and dead fuel in the region. The watershed is comprised of a fire-dependent mixed-conifer ecosystem with a natural fire return interval of 8-15 years; however, the Forest Service has managed the watershed for fire exclusion since the early 1900s, resulting in 4-9 missed fire cycles [Draft EIS; pg. I-6] (USDA FS 1999). Part of the expressed need for the HazRed Project was to compensate for the effects from past fire suppression that resulted in excessive hazardous fuel loads. Ironically, the purpose of the fuelbreaks was to increase suppression effectiveness and continue fire exclusion.

Finally, the Ashland Watershed has prime recreational, scenic, and spiritual values for the local community. Known affectionately as "the forest at Ashland's doorstep," local residents have a strong sense of place and personal connection with the watershed. Unlike most other rural communities in southern Oregon, Ashland's economy is not dependent on the timber industry; on the contrary, it is the home of Southern Oregon University and the renowned Oregon Shakespearean Festival which attracts thousands of students and tourists, generating $45 million in revenue each year. Ashland has a reputation for being a wealthy, liberal community supportive of environmental and social justice causes. Any timber sale would have been controversial and engendered the opposition of local residents philosophically opposed to any commodity resource extraction in the watershed for any reason.

For all the above reasons, conservationists recognized the HazRed Project as both controversial and precedent-setting and thus questioned the legality of the Forest Service issuing an EA instead of an Environmental Impact Statement (EIS) for the Project. In each case, though, the Forest argued that the impacts on water quality, soils and slopes, spotted owl habitat, and scenic / recreational values from a potential "catastrophic wildfire" far outweighed the minimal impacts to be caused by commercial logging for fuelbreak construction.

Public Involvement in the HazRed Project

The local nonprofit conservation organizations, Headwaters and the Klamath-Siskiyou Wildlands Center (KSWC), were instrumental in generating community involvement in the HazRed Project. Headwaters organized some early protest demonstrations and public rallies in 1997 and wrote several articles critical of the Project's potential environmental impacts in the organization's quarterly newsletter. The KSWC sponsored public hikes through proposed fuelbreak units where people observed old-growth sugar pines up to 6 feet in diameter at breast height (DBH) had been marked for cutting, along with over 4,400 trees greater than 20 inches DBH, and nearly all of the understory white fir trees. The sentiment of the environmental community was that the marking was excessive and that the Project was essentially a timber sale, not a fire hazard reduction or forest restoration project. A coalition of environmentalists submitted their own alternative during the comment period that would have put a diameter cap of 17 inches DBH for trees able to be extracted. Finally, the Ashland Mayor and City Council submitted a comment letter with a number of suggestions that echoed many of the conservation community's concerns, including the desire to construct no new shaded fuelbreaks and to focus on reducing surface fine fuels and brush rather than extracting large trees.

The community's fear that the HazRed Project would be followed by additional timber sales in the watershed in the future was prompted by the fact that one of the uses of the Project's timber sale revenues was intended to pay for completing an earlier so-called hazard reduction project. The Helikopter Salvage Sale had been logged in 1990/91 but had left behind thousands of submerchantible trees and had left untreated over 200 tons per acre of logging slash, which contrasted with the natural fuels accumulations of 10-35 tons per acre in unlogged sites [Draft EIS; pg. III-1] (USDA FS 1999). As is the case with nearly every proposed hazard reduction project involving a timber sale, the Forest Service intended to do the commercial logging first, then perform the restoration activities later using the timber sale receipts. This only fed into the public's perception that commercial timber extraction was the primary objective of the Project.

The District Ranger responded to the community's concern about excessive timber marking by literally walking the units with her staff and deciding on a tree-by-tree basis which ones specifically contributed to fire hazard, and which ones did not. On her own initiative, the Ranger ordered thousands of trees to be demarked, decreasing in half the amount of 20 inch DBH trees that were going to be logged, retaining all sugar pine and cedar trees, and allowing isolated clumps of large healthy trees to remain uncut [Final EA; APP. B, pg.29] (USDA FS 1998). This good faith gesture struck a responsive chord and personally endeared the Ranger among the Headwaters organization and other members of the local community. But it still did not sway the conservation community's belief that the HazRed Project was simply using the rationale of fire hazard reduction as an excuse to "get the cut out."

Final Environmental Assessment and Decision Notice

The Final EA, Decision Notice (DN), and Finding of No Significant Impact (FONSI) for the HazRed timber sale were issued simultaneously in March 1998. The decisionmaker announced that the Final EA was a complete revision and cautioned readers to avoid comparing it with the Draft. A notable and positive change was the cover of the Final EA. The cover of the Draft EA had displayed a photograph of a huge black, billowing smoke cloud rising above the city of Ashland during the 1959 Ashland Fire. Local environmental advocates were critical of the use of that old photo, perceiving it as an attempt to use scare tactics to win short-term public support for the timber sale at the possible long-term expense of community support for prescribed burning in the watershed. In response to this criticism, the Final EA did not use the photograph again; instead, the cover contained a candid statement from the Ranger declaring that she had not intended the photo to be used as "an alarmist approach to frighten citizens unduly." Inside the document, much of the information was reorganized and included several additional scientific references (including papers by Agee et.al. 1996 on crownfires, and Omi 1997 on fuelbreaks). Another change was that the Ranger selected a new preferred alternative, which had slightly less acres of logging compared to the original proposed alternative. The DN dropped new road construction and reduced the total acreage to be treated down to 1,472 acres, of which 457 acres were to be commercially thinned and 1,015 were to have noncommercial treatments.

While the selected action was slightly modified and the environmental impacts were reduced, the objective of the Project remained the same: to maintain and increase the effectiveness of the existing shaded fuelbreak system using commercial thinning for overstory removal. The Forest Service emphasized that portions of the 1959 Ashland Fire had been successfully contained along segments of a fuelbreak cut by the Civilian Conservation Corps (CCC) in the 1930s; however, the agency downplayed the fact that only understory brush was manually removed during the construction of that CCC fuelbreak, and that the 1959 wildfire had also breached other portions of the fuelbreak. Regardless, the Forest held fast to its argument that a ridgeline fuelbreak system would strategically "compartmentalize" the watershed and accomplish the following fire hazard reduction goals: 1) reduce surface fuel conditions and surface fireline intensity; 2) reduce fuel ladders and crown fuels that contribute to the start and spread of crownfires; 3) allow for penetration of fire retardant; 4) provide safe areas for firefighter deployment and evacuation; and 5) provide control points for prescribed underburning [DN; pg.3] (USDA FS 1998).

The fuelbreak strategy gained the official approval of the Regional Ecosystem Office which provided internal oversight on projects proposed in LSRs. Indeed, the Regional Director of Fire and Aviation Management, Mike Edrington, wrote a letter to the decisionmaker praising the HazRed Project. "We see the HazRed project as an example of the type of treatment that must be implemented on a much larger scale in the Pacific Northwest if we are to improve the health of fire adapted ecosystems for long term sustainability," Edrington wrote. Accordingly, environmentalists considered the HazRed to be a dangerous precedent, essentially functioning as a "Trojan Horse" that would justify timber sales in all other LSRs under the guise of fire hazard reduction.

Despite internal agency approval, critics pointed to new analysis of the local fire history that clearly showed the highest fire risk to the watershed came from ignitions in lower elevations primarily adjacent to residential and recreational areas and roads. Furthermore, topography and prevailing up-valley winds carried fires from the urban interface zone up into the watershed. This was precisely the ignition and fire spread pattern of the arson-caused Ashland Fire in 1959. Never had fires started in high elevation areas of the watershed and then burned rapidly downslope against the wind to reach the City. Critics challenged the logic of locating fuelbreaks in the middle of their watershed, asking the rhetorical question, "Why would you want to build a moat in the middle of your castle?" They preferred to locate fuelbreaks within the wildland / urban interface (WUI) zone on predominantly non-federal land where both fuel hazards and fire risks were rated high to extreme, and where suppression actions could prevent fires from spreading into the LSR and watershed and / or private homes.

Administrative Appeal and Withdrawn Decision

Although the Ranger had voluntarily issued an additional 30 day comment period for the Draft EA, thereby expanding it to a full 60 days, she issued the DN without first circulating the significantly revised Final EA for public comment. Conservationists suspected that the DN was rushed forward without a new comment period in order to avoid an impending injunction against logging projects that was part of a successful lawsuit over the Forest Service's failure to comply with the Northwest Forest Plan's Survey and Manage requirements. The HazRed timber sale generated six appeals by environmental organizations and a local private citizen. A number of substantive and procedural NEPA claims were raised, including the fact that there was no opportunity to comment on the significantly revised Final EA. It was this specific issue that in July 1998 the Regional Appeals Review Officer cited in ordering the DN to be withdrawn.

Conservationists' Critiques of the HazRed Project

It would be instructive to briefly review some of the main fire and fuels-related critiques of HazRed because these critiques have been raised on similar fuelbreak construction, fuels reduction, and forest restoration projects elsewhere in the National Forest System.

Need to Develop a Wider Range of Alternatives

One of the criticisms raised against the Project was that the Draft and Final EAs failed to contain a broad enough range of alternatives. Specifically, conservationists wanted analysis of an alternative that would have used only non-commercial methods for hazardous fuels reduction. In fact, an alternative that would have only used prescribed underburning was originally considered during the development of the Draft EA. The document acknowledged that underburning would have reduced surface and ladder fuels, but this alternative was dropped from further development because it would not have accomplished the desired canopy reduction in mid-to-large sized trees within fuelbreaks [Draft EA; II-7] (USDA FS 1997). In order to remove overstory trees and reduce the risk of crownfire spread, the agency insisted that logging was simply a management "tool," not a goal, and reminded the public that any timber outputs generated by the Project would not contribute to the Forest's Probable Sale Quotient.

The claim that "logging is a tool, not a goal" is being widely repeated in National Environmental Policy Act (NEPA) documents for Forest Service fuels reduction / forest restoration projects. This claim is often met with considerable skepticism by environmental interest groups and members of the public who generally mistrust government agencies. Additionally, non-logging alternatives are rarely voluntarily developed by the Forest Service without enormous public pressure first being applied. Hence, however sincerely believed by proponents, the assertion that "logging is only a tool" is failing to mollify critics who still see timber extraction as the main driver of Forest Service land management decisions. Indeed, the silvicultural prescriptions for the HazRed fuelbreak units emphasized the need for "adequate turn volume" for helicopters—the amount of timber taken in each flight from the logging unit to the landing zone—and specified that only trees 10 inches DBH or above would be removed. Timber markers were instructed to select a minimum of 500 board feet contained in five or six trees within a 50 foot radius for each helicopter load. These prescriptions clearly prioritized profitable logging operations over effective hazard reduction, and harkened back to the Helikopter Salvage Sale that had taken mainly commercially valuable large trees, leaving behind the more flammable submerchantible small trees.

Need to Analyze the Environmental Effects of Fire Suppression

Since the advent of the "forest health crisis" in the early 1990s, the Forest Service has begun acknowledging the adverse ecological effects of fire exclusion on fuel loads, stand structure, and tree stocking levels; however, the adverse environmental effects of fire suppression have never been analyzed or disclosed in a programmatic NEPA analysis. Whenever this analysis has been specifically requested in Project-level NEPA documents, the agency has often claimed as it did in the HazRed Project that "fire suppression is an emergency response activity that does not require environmental analysis to be conducted according to NEPA regulations" [Final EA; APP. B; pg.14] (USDA FS 1998). Conservationists urged the Forest Service, to no avail, to fully disclose the potential indirect and cumulative effects of conducting fire suppression activities within and adjacent to fuelbreaks. Doing so, they argued, would help the agency make the case for the need for proactive fire hazard reduction projects as a means of avoiding reactive fire suppression actions. A fundamental question had failed to be asked by the agency: are the impacts of logging and *using* fuelbreaks for firefighting more or less significant than the effects of wildland fire alone?

The Fire Management Plan for the LSR stated, "Shaded fuelbreaks do not contain and control wildfires on their own. It takes available, trained, skilled suppression resources to take advantage of the shaded fuelbreaks" [LSRA; D-23] (USDA FS 1996). Some of the foreseeable environmental impacts that routinely occur during suppression operations and could happen within fuelbreaks include: habitat tree felling for cutting firelines, helispots and safety zones; soil disturbance by heavy equipment and handcrews constructing firelines; chemical contamination of soil and water by retardant drops and refueling saws, pumps, and vehicles; and severe and / or homogenized fire effects by burnout and backfire ignitions. Moreover, since "worst case" scenarios were used to analyze the effects of future wildfires, critics insisted that "worst case" scenarios should be used to analyze the effects and effectiveness of future fire suppression actions within fuelbreaks. This would entail total, aggressive suppression under extreme weather conditions along the complete length of the fuelbreaks system.

Need to Address Structural Fire Protection in the Wildland/Urban Interface Zone

Despite the fact that HazRed was identified as the "Ashland Interface Fire Hazard Reduction Project," and that "Human Life and Property" was put at the top of the list of values at risk, nowhere else in the NEPA document was the issue of structure protection in the WUI zone addressed. The Draft EA did not explain how human life and private property within the city of Ashland would be protected by the proposed shaded fuelbreak located deep in the interior of the watershed.

There have been several other proposed fuelbreak timber sales on National Forest System lands using the rationale of "community fire protection;" yet, general problems remain from the lack of a precise, science-based definition of the WUI zone and lack of empirical evidence that fuel and vegetation treatments conducted several miles away from communities will in fact help protect private structures from wildfire damage. According to research conducted by the Forest Service's Fire Sciences Lab in Missoula, Montana (Cohen 1999), the prime zone for vegetation treatments to effectively and efficiently reduce home ignitability factors is approximately 200 feet surrounding structures. In the case of the HazRed Project, the proposed fuelbreaks would have been constructed several miles and ridgelines away from the city, offering dubious benefits, if any at all, to structural fire protection needs. This points to the general need for projects to clearly demarcate fuels treatments in wildlands conducted for ecosystem restoration purposes from fuels treatments in the WUI zone conducted for community protection purposes.

Need to Implement the Federal Wildland Fire Policy

The 1995 Federal Wildland Fire Management Policy and Program Review (Federal Fire Policy) and the 2001 Review and Update of the Federal Wildland Fire Policy signify a potentially profound change in federal fire management philosophy. The letter and spirit of the Federal Fire Policy commits agencies to genuinely move away from systematic fire exclusion toward prescribed and wildland fire use for the restoration of fire-adapted ecosystems. The most urgent institutional need and highest priority action item for implementing the Federal Fire Policy was the development of new Fire Management Plans (FMPs). According to the Federal Fire Policy, FMPs are required for every area on federal lands subject to wildland fire, or every acre containing burnable vegetation. These FMPs offer the strategic framework for the full range of fire management projects and actions, from hazardous fuels reduction and forest restoration projects to fire prevention campaigns and fire suppression incidents.

The HazRed Project failed to tier to the Federal Fire Policy or discuss how, if at all, the Project complied with the Policy in terms of fire reintroduction and forest restoration goals. Critics were concerned that the proposed fuelbreak was designed solely for the purpose of containing wildfires, thus continuing the Rogue River National Forest's obsolete fire exclusion-based FMP in the watershed's fire-dependent ecosystem. Critics asserted that implementing a fuels reduction project before the Forest first developed a new Fire Policy-compliant FMP was "putting the cart before the horse." Even worse, the two were disconnected and heading in opposite directions: the HazRed Project was oriented toward continued fire exclusion while a Policy-compliant FMP should be oriented toward fire reintroduction and ecosystem restoration.

Need to Ensure Proper Fuelbreak Maintenance

An inherent challenge with extensive fuelbreak systems is the need for periodic maintenance to retard the growth of flammable native and exotic vegetation that can thrive in exposed, logging-disturbed sites. Without maintenance, fuelbreak sites can convert from a timber fuel model to a grass or brush fuel model and actually result in increased fireline intensity and rate of spread, thus undermining the stated purpose for safe, efficient fire suppression actions. Part of the HazRed Project involved commercial thinning and prescribed burning in portions of existing fuelbreak segments that were logged 10 years earlier. However, an abundance of smaller, submerchantible trees had been left behind from the timber sales, and manzanita brush had rapidly grown in the opened canopies. These small trees and brush made these sites largely ineffective as fuelbreaks. Because the Forest had failed to adequately maintain existing 20-year-old fuelbreaks, this did not give the community much assurance that the proposed new fuelbreaks would be maintained for the next 200 years—the timeframe that the Forest had used to analyze the effects of the fuelbreaks in protecting the watershed from future large-scale "catastrophic" fires. Additionally, the use of chemical, mechanical, manual, and prescribed burning methods for fuelbreak maintenance cause their own cumulative impacts, which needed to be analyzed along with the effects of fuelbreak construction.

The Ashland Watershed Protection Project

Draft Environmental Impact Statement

The Regional Office instructed the decisionmaker to withdraw the DN and issue an additional 30 day comment period for the revised Final EA. At the Ranger's own discretion, however, she decided to conduct a more extensive environmental analysis and develop a new Draft EIS. The HazRed Timber Sale Project was renamed the Ashland Watershed Protection Project (AWPP). The purpose and need for the AWPP was to provide high quality drinking water and maintain large areas of late-successional habitat by creating a "fire resilient landscape relatively resistant to large-scale high severity wildfire" [Draft EIS; S-1] (USDA FS 1999).

The Draft EIS included a range of four action alternatives (instead of only two that had been in the HazRed EA). Alternative One was the No Action alternative required as the baseline for comparing the action alternatives. The objective of Alternative Two was to protect late-successional structure and treat understory vegetation and surface fuels using prescribed underburning as the only treatment method. It sought to replicate, to the extent possible, the historical fire cycles for the Project area and restore historical vegetation conditions in the watershed. Alternative Three stressed protection of soils and site productivity by using manual treatments (e.g., cutting with chainsaws and handtools) and "swamper burning" (continuously feeding material into small piles about 4-6 in feet diameter) to selectively remove shrubs, small trees up to 8 inches DBH, and jackpots of dead surface fuels, rather than mechanical treatments with fellerbunchers and skidders. Cutting with chainsaws and hand tools only and small pile and swamper burning would avoid the impacts on soils from both heavy equipment logging and broadcast prescribed burning. Alternative Four would try to minimize changes to late-successional forest structure while reducing fire hazard. It proposed using a combination of treatment methods including mechanically removing trees up to 17 inches DBH, manual treatments, and prescribed underburning, but would not construct or expand shaded fuelbreaks. Slash and fuels would be burned by a variety of methods and possibly chipped and hauled away for biomass. Alternative Five, the preferred action, was essentially the proposed action in the HazRed Project, and its goal was to maximize fire hazard reduction using all of the above treatment methods and maintaining, expanding, and constructing shaded fuelbreaks, with no diameter limit on the trees slated for mechanical removal.

The AWPP was a significant improvement over HazRed on some but not all issues. As an EIS, the AWPP did provide a much wider range of alternatives, including two alternatives using noncommercial methods. However, conservationists were critical of splitting up the manual treatments and prescribed underburning into two separate proposals. They believed that combining those methods would have successfully reduced surface and ladder fuel loads, raised the crown base height, and increased the average stem diameter in ways that would have significantly reduced the risk of crownfire initiation. This would have reduced the need to extract large trees in order to reduce crownfire propagation.

The Draft EIS also disclosed the existence of the Federal Wildland Fire Policy and its nine guiding principles, but the Ranger dropped from consideration an alternative that would have utilized Wildland Fire Use for Resource Benefits (WFURBs) precisely because the Forest's existing suppression-based fire-exclusion-oriented FMP did not allow for WFURBs in the watershed. It was assumed throughout the analysis that fire suppression would occur, and the Draft EIS even provided some crude estimates of the potential costs of future suppression for each alternative. Unfortunately, the Draft EIS did not analyze the potential environmental impacts of suppression within the fuelbreaks. Later, in the Final EIS, the agency would argue that a Forest Plan Amendment or Revision would be necessary in order to fully implement the Federal Fire Policy and utilize WFURBs [Final EIS; App. I-10] (USDA FS 2001a).

The Ashland Watershed Stewardship Alliance

Prior to the release of the Draft EIS, the District Ranger took the initiative to reduce some of the tension that had flared up in the community over the original HazRed Project and seek active citizen involvement in the AWPP. She contacted the Peace House, a local nonprofit organization affiliated with the National Fellowship of Reconciliation, and asked them to organize a "community dialogue meeting" in February 1999. As a neutral ground with a large amount of credibility in the community, the Peace House brought together the Ranger and environmental activists for face-to-face discussions about the forthcoming AWPP. From the original small gatherings, the group decided to meet regularly and expand the base of participants. Representatives of the City of Ashland, business owners, forest workers, and community organizers met twice a month from March through August when the Draft EIS was released. In September, the group named themselves the Ashland Watershed Stewardship Alliance (“the Alliance”). They began to meet twice weekly and set up four subcommittees that reported to the larger group. About 40 people actively participated in the Alliance meetings, with some gatherings attracting over 100 people sitting in a large circle at the Peace House to discuss competing and complementary visions on how to protect and restore the Watershed. Although the District Ranger did not directly participate after the first few meetings, the community recognized that it was her initiative and tacit ongoing support that kept the Alliance meeting regularly and working for a new, constructive, collaborative relationship between the agency and community.

The Stewardship Alliance was comprised of highly experienced and skilled people, including Headwater’s staff, a member of the Society of American Foresters, the Ashland City Forester, a retired Forest Service economist, as well as other credible scientists, foresters, and environmentalists residing in the community. The Alliance produced a 95 page proposal that was submitted on the last official day for comments on the Draft EIS. The preamble to this document is noteworthy for the spirit of collaboration it conveys:

> “We, involved citizens of the Ashland Watershed, look to deal constructively with conflict and promote a collaborative relationship between the Forest Service and the people of this community. As neighbors within this forest, we share a common interest to begin the work necessary to mitigate the risks of wildland fire within the watershed, restoring a forest ecosystem which will be resilient to periodic natural fire events. We seek to accomplish this goal while maintaining the ecological, social, aesthetic, spiritual, economic and educational qualities which the people of this region value in these forests…Working with the Ashland Ranger District we hope to develop alternatives which can cover the costs, while building a lasting and mutually beneficial relationship with the agency—one which utilizes local expertise, folds local values into the planning process, and builds a culture of long-term stewardship between the citizens of this community and the land” (Ashland Watershed Stewardship Alliance 1999).

The Alliance’s Proposal presented a number of ecological, social, and economic goals and principles that they wished to be applied toward development of a new alternative for the Final EIS. It is beyond the scope of this paper to present the complete list (Ashland Watershed Stewardship Alliance 1999), but the following items are worth emphasizing because they are increasingly being requested by conservationists participating in fire and fuels related Project proposals. Under the category of Ecological Goals and Principles: 1) Focus fire hazard reduction activities primarily on reducing the fuels from the brush and smaller understory trees that have increased above natural densities due to fire suppression; and 2) Accomplish different aspects of the project in a sequence that allows for non-controversial treatments to proceed as soon as possible, so that lessons can be learned and applied later.

Under the category of Social Goals and Principles: 1) Develop and nurture the shared responsibility of the community for the stewardship of the Ashland watershed (including planning, funding, implementation, and monitoring); and 2) Establish a process encouraging participation by all interested parties in an open and transparent manner that leads to understanding and trust.

Finally, under the category of Economic Goals and Principles: 1) Base the decisions for fire reduction work on sound ecological guidelines and not on the extraction of commercial material to meet Project funding needs; and 2) Work with the U.S. Forest Service in developing new funding structures for the fire hazard reduction project and if needed to secure additional funding sources for the work (Ashland Watershed Stewardship Alliance 1999).

Among a number of technical items in the Alliance's Proposal, they recommended a fire management strategy of area-wide vegetation treatments instead of shaded fuelbreaks; preferred the use of manual pre-treatments with prescribed underburning instead of mechanical thinning in order to better conserve soil and vegetation on geologically unstable slopes; demanded an active monitoring plan with secured funding; and desired a phased implementation of project activities over time. The group felt that it was important to proceed immediately on the activities that were non-controversial and enjoyed broad community support. This meant the first phase of the Project would utilize manual treatments to cut brush and small trees under 8 inches DBH in the interface areas directly bordering residential sites. The second phase would then be to combine manual treatments with prescribed underburning. The third and most controversial phase of the Project would be to do maintenance on the existing fuelbreaks, which if necessary could involve removing some large trees. The fourth phase would be to develop a comprehensive long-term fire restoration plan for the whole watershed, based on the lessons learned in the prior three phases. The Alliance gained endorsements for their alternative proposal from all of the area's environmental organizations, a wide spectrum of community organizations and private citizens, and most importantly, from the Ashland Mayor and City Council who gently reminded the Forest Service of their 70-year-old Cooperative Agreement regulating management activities in the watershed.

As mentioned, the Alliance expressed the desire to work with the Forest Service to develop innovative, non-traditional funding mechanisms such as service contracts and special use permits to get the needed work done. For example, the Alliance offered to create its own non-profit, bonded general contracting arm to bid on high priority, low revenue producing units that would not likely attract traditional contractors. The non-profit could offer below-market bids on service contracts, and then accept donations from local citizens and businesses to help cover expenses. These and other initiatives represented sincere desires by the Alliance and the local community to work with the Forest Service to implement widely supported hazard reduction activities in the watershed while avoiding the need for a controversial and potentially divisive commercial timber sale.

The Ranger decided to extend the initial 45 day comment period on the Draft EIS by an additional 30 days. She conducted a public "learning meeting" on September 1, 1999, to make herself and her staff available for questions about the Draft and address the public's interests and concerns. Even more impressively, she allowed citizens to check out keys to the Forest Service's locked gates in order to access portions of the Project area that were normally restricted. Of the 39 comment letters received (several of them contained the signatures of multiple individuals and / or groups), 21 letters expressed support for the Alliance's vision of proactive community involvement in all of the fire hazard reduction activities stages (planning, implementing, and monitoring) in the Watershed and WUI zone [Final EIS; App. I-3] (USDA FS 2001a).

Final Environmental Impact Statement

The Final EIS was issued on January 2, 2001, and included the important addition of a new alternative developed in response to public comments. Learning from the past mistake, the decisionmaker provided a 30 day comment period on the FEIS. The Stewardship Alliance was disappointed that the decisionmaker did not agree with them on some issues; nevertheless, the Final EIS represented a significant improvement over the Draft, with five times more acres to be treated with noncommercial methods, much less commercial thinning, and most importantly it included a new preferred alternative that was inspired by the Alliance's proposal.

Alternative Six, the agency's new preferred alternative, addressed the controversies surrounding overstory tree removal and shaded fuelbreak constructed by implementing a phased schedule of fuel treatments, beginning with surface fuel reduction in the WUI zone, prior to treatments to maintain existing ridgeline fuelbreaks in the interior of the watershed. The alternative also called for implementation of manual surface and understory fuels reduction for the first three years of the Project. For the next 3-5 years, mechanical thinning would occur, followed by prescribed underburning, all in the WUI zone. No new shaded fuelbreaks would be constructed, and although it would still remove overstory trees, it would not follow the same canopy spacing prescriptions (e.g., 20-60 feet spacing between individual tree crowns) proposed in the previous proposed actions in HazRed and the AWPP. The whole process of reducing fire hazards on 1,549 acres was estimated to take 8-12 years to complete, with implementation and effectiveness monitoring to be conducted at each stage. Alternative Six was analyzed as the most costly to implement, but the Alliance felt that the more costly, the better, since they believed that restoration activities should be funded through appropriated budgets such as the Hazardous Fuel Reduction fund within the National Fire Plan, and not have to depend on revenues derived from commodity timber extraction.

Record of Decision

The Record of Decision (ROD) for the AWPP was issued on May 25, 2001—fully five years after the initial HazRed Project was first proposed. The Ranger noted that it had been "a protracted agency process with stunning citizen involvement" [ROD; p.2] (USDA FS 2001b). She commended that "the Forest Service is fortunate to have membership with a community so committed to building its capacity to leverage diversity and resources to address difficult, and sometimes seemingly irresolvable, natural resource and social issues" [ROD; p.2] (USDA FS 2001b). It is significant that the Ranger considered herself and the District staff to be members of the community. A year earlier (June 2000) she was fired by the Rogue River Forest Supervisor in his last official act before retiring from the Forest Service. The alleged reason for her dismissal was her "incompetent management style." Community members believed, however, that she was fired because her management style emphasized active involvement and community collaboration, in contrast to the agency's traditional technocratic style. Dozens of local citizens called the Regional and Washington Offices of the Forest Service to complain. The result: two days after she had been dismissed she was fully reinstated as the Ashland District Ranger, and the community was elated. Clearly the Ranger considered herself, and was considered by local residents, to be a member of the community.

Acknowledging that the removal and sale of large trees was the most contentious aspect of the Project and provoked "vehement resistance" from citizens, the Ranger selected a modified Alternative Six that deferred for several years the commercial logging planned along ridgeline fuelbreaks, and imposed a diameter limit of 17 inches DBH for trees to be commercially thinned within the WUI zone. Citizens had long pushed for this diameter cap based on the Mt. Ashland LSR Assessment, which disclosed that trees larger than 17 inches DBH contribute to habitat quality for northern spotted owls. The ROD also withdrew the new road reconstruction and eliminated a proposed helispot that was the site of an old-growth ponderosa pine stand containing some of the biggest trees originally proposed for cutting. If and when commercial thinning for fuelbreaks might occur in the future, eliminating that new road and helispot will result in longer helicopter flights, ultimately making the logging more expensive to implement. Regardless, the Ranger justified her decision by saying that the citizen's and the City's environmental concerns far outweighed the agency's desires for least-cost project implementation and maximum fuels reduction effectiveness.

Finally, the Ranger acknowledged the benefits of community collaboration and outlined several opportunities for citizen volunteer participation and continued involvement in the Project's implementation. For example, the Ranger offered to help organize "volunteer days for community participation" in pre-treatment data collection and post-treatment monitoring to train citizen volunteers in manual fire hazard reduction techniques and to let them actually assume responsibility for treating selected units. The Ranger's appeal for citizen volunteers included helping to unmark all the previously marked trees within units dropped from mechanical treatments. In addition to helping to train and organize citizen volunteers, the Forest also promised to implement the Project with a variety of methods that utilize local labor resources, including the awarding of "best value" service contracts.

Although the ROD had successfully addressed the majority of the conservation community's concerns, the Klamath-Siskiyou Wildlands Center (KSWC) did appeal the portion of the Project that would have used commercial thinning in high risk landslide areas. In an informal disposition meeting, the Ranger agreed to drop all units in those landslide-prone areas, and thus reduced the total logging acreage down to 116 acres—fully one-fourth the acreage of the original HazRed proposal. The KSWC dropped their appeal, and the Project has been carried forward with the active support and involvement of the KSWC. The Ranger has gained the trust of the local conservation community who feels that under her leadership the Project has been transformed from essentially a pre-suppression timber sale into a genuine fuels reduction for fire restoration project.

Conclusion

The Ashland Ranger District has begun implementing the AWPP, and it has received priority National Fire Plan funding in Region 6 precisely because the Project had gained extensive community involvement and support. Both the District Ranger and the local conservation community should be credited with the willingness to communicate and collaborate in creating a management project that is both environmentally sound and socially acceptable. The fact that the Ranger did not come from a traditional forestry background, but instead, had an educational background in communication and leadership may account for her remarkable success in listening and learning from the community and leading her staff to develop a truly innovative Project. She is continuing to serve as a facilitator of community-based stewardship, working on a proposal with the City of Ashland and Southern Oregon University to create the

"Southern Oregon Institute for Watershed and Citizenship Studies" to help increase agency and community capacity for ecological restoration of public lands (Duffy 2001).

The Project's manual cutting and burning treatments will be expensive, but the National Fire Plan funding for Hazardous Fuels Reduction treatments comes at the perfect time for these noncommercial treatments. The Ranger, the Stewardship Alliance, and the City have also pledged to leverage the funding by organizing groups of volunteers to help implement the Project on the ground. A subgroup of Americorps called "REALcorps" (Regional Ecosystem Applied Learning) is organizing community work parties to do hand-piling of fuels, and the nonprofit Lomakatsi Restoration Project has received grants from the National Fire Plan fund to train local citizens in hazardous fuels reduction and restoration techniques. Local groups of students and Boy Scouts are also getting involved in cutting brush in the WUI zone. The Forest Service is experimenting with different kinds of service contracts to allow community groups to design their own manual hazardous fuels reduction prescriptions on three acre parcels of land. In sum, the community is taking several initiatives to implement the Project, and the Forest Service is playing a supportive role by providing technical assistance. This represents a most novel approach to agency-community collaboration: teaming up the expertise and authority of the agency with the volunteer labor power of the community to get the work done for mutual benefit.

The legacy of the Ashland Watershed Protection Project is a matter of perspective. To some Forest Service officials, the Project's prolonged analysis, successful appeals, and deferral of the timber sale represents a humbling erosion of managerial power. To other agency employees, the Project represents an important breakthrough in gaining the consensus of "hardcore" conservationists for active management in a forest reserve. To the local community, the Project symbolizes a new social solidarity and unprecedented opportunity to assume some civic responsibility for stewardship over the public lands they hold dear. The phased implementation of the Project, starting with noncommercial manual and prescribed burning treatments first, followed by implementation and effectiveness monitoring, and then later mechanical treatments possibly including commercial logging, created a successful solution to overcoming controversy and "analysis paralysis."

According to a local resident who actively participated in the NEPA process and helped develop the Stewardship Alliance's proposal, "Despite the prolonged and difficult process, the AWPP is evidence that NEPA actually works. Although it is time and energy consumptive, if followed faithfully the NEPA process produces projects that can unite instead of divide communities" (Lininger, personal communication). In this regard, the Ashland Watershed Protection Project offers a working model for agencies and communities to move forward together on implementing fuels reduction and forest restoration projects throughout the National Forest System.

References

Ashland Watershed Stewardship Alliance. 1999. Draft comment and proposal for the Ashland Ranger District and interested citizens in response to the Ashland watershed protection project draft environmental impact statement. November 1999.

Cohen, Jack. 1999. Reducing the wildland fire threat to homes: where and how much? Proceedings of the symposium on fire economics, planning, and policy:

bottom lines. Gen. Tech. Rep. PSW-GTR-173. U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 189-195.

Duffy, Linda. 2001. Transformation of a ranger district? The Stewardship Chronicle: the newsletter of the ecosystem workforce program. Volume 3, Number 1. Spring.

Lininger, Jay. 2002. Personal interview.

U.S. Department of Agriculture Forest Service. 1996. Late-successional reserve assessment, Mt. Ashland late-successional reserve (#RO-248). Pacific Northwest Region (R-6), Ashland Ranger District, Rogue River National Forest; Pacific Southwest Region (R-5), Oak Knoll Ranger District, Klamath National Forest. June 1996.

U.S. Department of Agriculture Forest Service. 1997. Draft environmental assessment, Ashland interface fire hazard reduction (HazRed) project. Ashland Ranger District, Rogue River National Forest. February 1997.

U.S. Department of Agriculture Forest Service. 1998. Final environmental assessment, decision notice, and finding of no significant impact, Ashland interface fire hazard reduction (HazRed) project. Ashland Ranger District, Rogue River National Forest. March 1998.

U.S. Department of Agriculture Forest Service. 1999. Draft environmental impact statement, Ashland watershed protection project. Ashland Ranger District, Rogue River National Forest. August 1999.

U.S. Department of Agriculture Forest Service. 2001a. Final environmental impact statement, Ashland watershed protection project. Ashland Ranger District, Rogue River National Forest. January 2001.

U.S. Department of Agriculture Forest Service. 2001b. Record of decision, Ashland watershed protection project. Ashland Ranger District, Rogue River National Forest. May 2001.

Hazardous Fuel Reduction in the Blue Mountains: Public Attitudes and Opinions

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Abstract—*Resource managers in the Blue Mountains region of eastern Oregon and Washington are utilizing prescribed fire and mechanized thinning treatments to reduce hazardous fuel loads and restore forest health. This paper uses panel data from a mail survey administered to the same individuals in 1996 and 2000 to measure change in public attitudes and opinions about fire management programs. Respondents are knowledgeable about, and supportive of, prescribed fire and thinning practices; prefer interactive over uni-directional education programs; and desire a role in management decision-making. While findings were generally similar throughout the study period, significant changes suggest a declining relationship between the Forest Service and Blue Mountains residents.*

Introduction

Forests in the Blue Mountains region of Oregon and Washington are threatened by drought, insect outbreaks, and the risk of catastrophic wildfire (Mutch et al. 1993, Tanaka et al. 1995). Resource managers on the National Forests within the region are utilizing multiple strategies to reduce hazardous fuels and restore forest health. Two of the most frequently used methods are prescribed fire and mechanized thinning treatments. Research demonstrates that a large-scale application of these treatments can address the principal causes of declining forest health and increased fire risks and serve to increase biological diversity, improve plant communities, reduce the number of invasive species, and ultimately create more natural forest conditions (Mutch et al. 1993).

The purpose of this study is to improve our understanding of the factors that contribute to public acceptance of prescribed fire and mechanized thinning in the Blue Mountains area. Specific study objectives were to a) compare the current research findings with the 1996 study to describe changes in public attitudes and behaviors, b) identify levels of support for fuel reduction activities, c) identify citizens' information needs, preferred forms of information exchange, and which delivery systems are most effective, and d) assess interactions between the public and the Forest Service.

Management Context

Declining forest health is not unique to the Blue Mountains but is evident throughout forests of the western United States. Poor forest conditions have greatly increased the risk of catastrophic wildfire and the fires of 2000 brought national attention to this situation. Interagency planning efforts have resulted

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in a national strategy to reduce wildfire risks while restoring ecosystem health and protecting communities through fuel reduction practices. Citizen support is a basic requirement to project implementation and long-term success. The public has legitimate concerns about these activities following decades of fire suppression activities, fire exclusion education from forest management agencies, and exposure to high-profile escaped prescribed fires such as the Cerro Grand Fire in Los Alamos, New Mexico, during spring 2000. Current agency educational programs nationwide, and within the Blue Mountains region particularly, attempt to address these concerns. Fuel reduction projects and information programs on the Wallowa-Whitman, Umatilla, and Malheur National Forests provide an opportunity to examine citizen perspectives on the legitimacy of these practices and the effectiveness of informational messages.

Although public awareness of prescribed fire is growing, this use of fire still seems contradictory to the practices that many have come to expect from forest agencies (Beebe and Omi 1993, Lee 1987). Public concerns with prescribed fire typically focus on risk (danger of escapes to public safety and private property), aesthetics (potential loss of scenic quality and recreation uses), health issues (the impact of smoke on air quality), ecological effects (impacts to wildlife, vegetation, water quality), and economic impacts (loss of valuable timber).

In spite of these concerns, public attitudes nationwide have been evolving toward a greater acceptance of the use of fire in forest ecosystems, particularly as agencies improve their communication strategies (e.g., Carpenter et al. 1986, Loomis et al. 2001, Stankey 1976). This is the case in the Blue Mountains where our 1996 study identified an association between increased support for prescribed burning activities with increased citizen knowledge about the role and uses of fire. Although other studies have also found associations between factors such as gender, income, age, backcountry experience, and support levels (Carpenter et al. 1986, McCool and Stankey 1986), knowledge and understanding of prescribed fire continue to be the most important contributors to public attitudes about the practice.

Little research has been conducted on public perspectives of mechanized thinning methods to reduce hazardous fuels. Concerns range from aesthetic impacts and potential ecological effects of harvesting practices (Brunson and Reiter 1996) to doubts about whether thinning treatments will result in a sufficient quantity of marketable timber to offset increased operation costs (Shindler and Collson 1998). Another important issue involves public trust in our forest agencies to effectively implement mechanized thinning programs on federal lands; in particular, citizens have reservations about how much license managers will take in thinning forest stands. Overall, timber harvesting is one of the most contentious issues in the highly charged sociopolitical environment in which our natural resource agencies operate. One of the dangers for fire managers is that many citizens believe using thinning treatments to reduce fuel loads is really just another way to continue harvesting or, in other words, conduct “business as usual” (Stankey 1995).

Methods

Methods regularly employed in social science research provide a “snapshot” of a cross-section of the population at one specific point in time; researchers then make inferences about existing conditions and circumstances.

Although careful analysis of cross-sectional data can provide considerable insight, there are limitations in our ability to understand ongoing processes with data collected from a single reference point (Babbie 1995). To overcome these limitations, longitudinal research designs provide for data collection and analysis over time.

A particularly beneficial type of longitudinal research is a panel study that involves evaluations of the same individuals using the same measures at different points in time. Panel data can provide a richer understanding of ongoing processes and be used to identify general trends within the population of interest. Responses from the individual study participants can be "paired," or linked, over the separate data collection points to allow identification of shifts in individual attitudes and beliefs. Paired data typically reduce the variability that could obscure small but significant differences between results (Devore & Peck 1986).

Data Collection

The current study is a panel study (designed to replicate research conducted in 1996 by Shindler and Reed). A new mail-back questionnaire was sent to all 1996 study participants in the summer and fall of 2000. The majority of questions were replicated to allow for comparison of responses between years; additional questions were included to address current concerns, notably specific Forest Service information programs and citizen-agency interactions. When appropriate, data from replicated questions (1996 and 2000) are compared utilizing paired t-tests and significant differences in responses are noted.

In 2000, 455 of the 533 original respondents were located. Of these, 32 were removed from the sample (29 were deceased or unable to complete the survey due to health reasons, and 3 had moved from the Blue Mountains region). From the useable sample of 423 names, 323 respondents completed questionnaires for a 76% adjusted response rate.

Findings

Public Knowledge, Information Sources, and Forest Health Conditions

To help gauge citizen understanding about the use of fire and thinning, a new line of inquiry was introduced in the 2000 questionnaire that engaged respondents in a 15-item true / false quiz. Quiz statements and responses are shown in table 1.

Overall, respondents were generally knowledgeable about both prescribed fire and thinning; the average correct score across all questions was 70%. However, some public misperceptions seem noteworthy. About one-third (35%) of the respondents did not know (answered incorrectly or not sure) about the important role fire has played in shaping natural forests in the Blue Mountains. Almost half were either misinformed or not sure about the effects of prescribed fire on small trees and understory vegetation, in promoting growth of ponderosa pine, and in controlling noxious weeds — all key objectives for the use of prescribed fire. Similarly, almost one-third (30%) did not know that thinning could be used to encourage growth of ponderosa pine as research has demonstrated (e.g., Cochran and Barrett 1993). Overall, participants appeared significantly more knowledgeable about the effects of thinning than about prescribed fire.

Table 1—Citizen knowledge about the effects of fire and thinning in Blue Mountain forests. Answer generally considered correct is in parentheses and percent answering correctly is indicated in bold.

	Generally True	Generally False	Not sure
- - - Fire - - -			
a. Fires have played a significant role in shaping natural forests in the Blue Mountains. (True)	**66%**	13%	22%
b. Prescribed fires make additional minerals and nutrients available for plants and trees. (True)	**67%**	12%	21%
c. Prescribed fires cause the immediate death of the majority of animals in the burned area. (False)	7%	**80%**	13%
d. Prescribed fires result in the death of the majority of large, established trees in the burned area. (False)	15%	**74%**	11%
e. Prescribed fires promote the growth of plants that serve as food for deer and elk. (True)	**84%**	9%	7%
f. Prescribed fires kill most of the small, young trees and vegetation beneath the forest canopy. (True)	**51%**	30%	19%
g. Prescribed fires encourage tree growth in ponderosa pine forests. (True)	**56%**	10%	34%
h. Prescribed fire is effective in controlling noxious weeds. (True)	**52%**	24%	24%
- - - Thinning - - -			
i. Selective thinning can be effective in controlling outbreaks of insects and disease. (True)	**78%**	7%	15%
j. Selective thinning reduces competition for minerals and nutrients on crowded sites. (True)	**80%**	7%	13%
k. Selective thinning mimics natural conditions by providing openings in the forest canopy. (True)	**74%**	9%	17%
l. Selective thinning causes the immediate death of the majority of animals in the thinned area. (False)	2%	**89%**	9%
m. Selective thinning encourages tree growth in ponderosa pine forests. (True)	**70%**	6%	24%
n. Selective thinning results in decreased habitat for deer and elk. (False)	13%	**77%**	11%
o. Selective thinning results in the death of the majority of the remaining trees on the site. (False)	3%	**90%**	8%

Knowledge and information play an important role in forming support for management practices. In peoples' everyday lives, there are many different places where citizens might obtain information about natural resource issues. Using a 4-point scale (none, slight, moderate, high) respondents were asked to rate the usefulness of nine likely sources of information about forest management in the Blue Mountains (figure 1).

Newspapers / magazines and friends or relatives continued to be the most useful sources and were the only ones to receive a moderate to high rating by a majority of respondents in both 1996 and 2000. Of particular interest are ratings for timber groups and the Forest Service; these are the only sources to receive significantly different ratings during the study period. The usefulness rating of timber groups rose (39% to 50%), while opinions of the Forest Service as a useful information source fell from 60% to 48%. This finding may suggest that the traditional communication formats utilized by the Forest Service (i.e. brochures, public meetings, exhibits) are less effective or that people doubt the credibility of government provided information (Shindler et al. 1996). Overall, the lowest ratings were for radio, environmental groups, and the internet.

To probe this area more thoroughly in 2000, additional questions focused on specific Forest Service information programs. Respondent familiarity as well as program components such as ease of understanding, convenience, and trustworthiness factored into usefulness ratings; however, only the percentage of moderately or highly useful ratings from respondents familiar with the program are displayed in figure 2. For a more complete analysis of the ratings of agency information programs see Shindler and Toman 2002.

Smokey Bear, elementary school educational programs, conversations with agency personnel, interpretive information, and guided field trips received the highest usefulness ratings. Television messages, newsletters, brochures, exhibits at fairs, and public meetings were moderately useful. Alternatively, few people

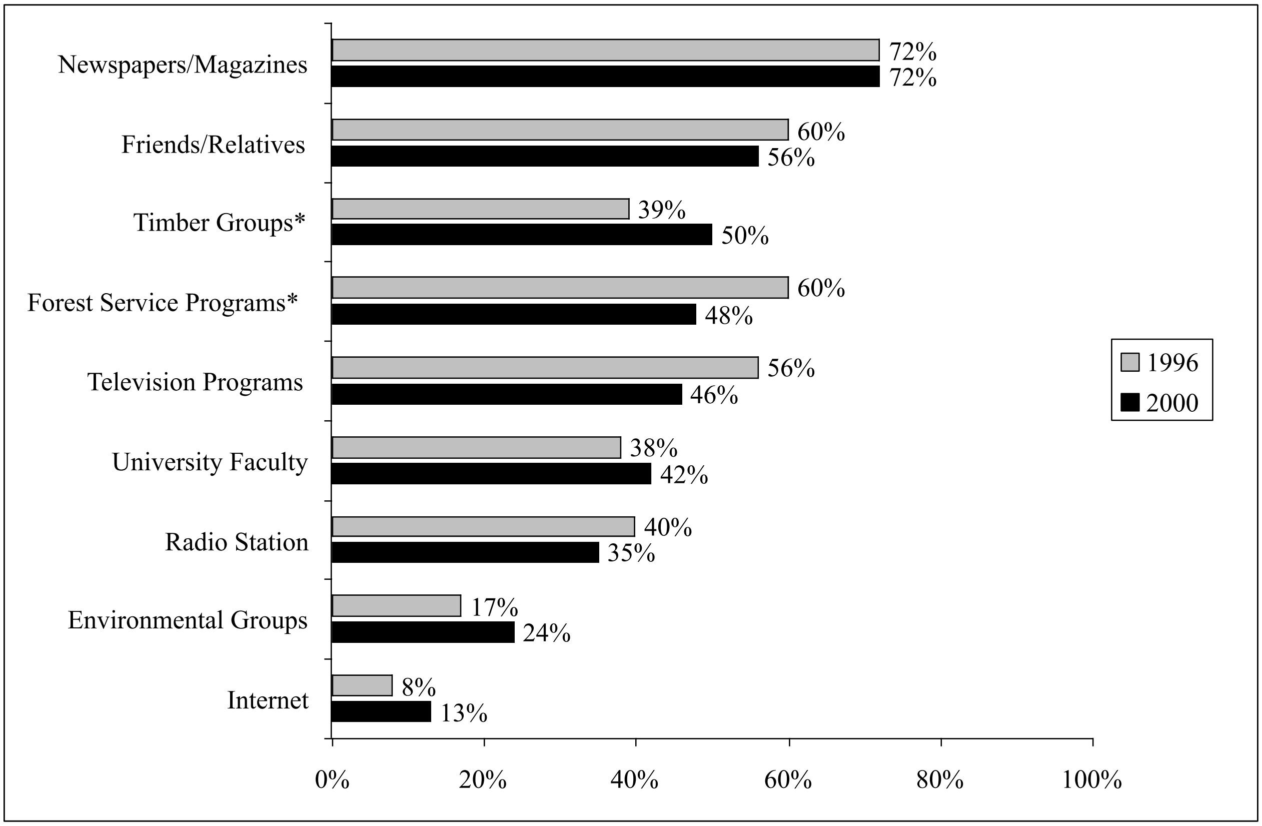

Figure 1—Useful information sources about forest management in the Blue Mountains. Data reflect percentage of citizens who rate usefulness as moderate or high on a 4-point scale (none, slight, moderate, high). *1996 and 2000 responses are significantly different at p £ .01

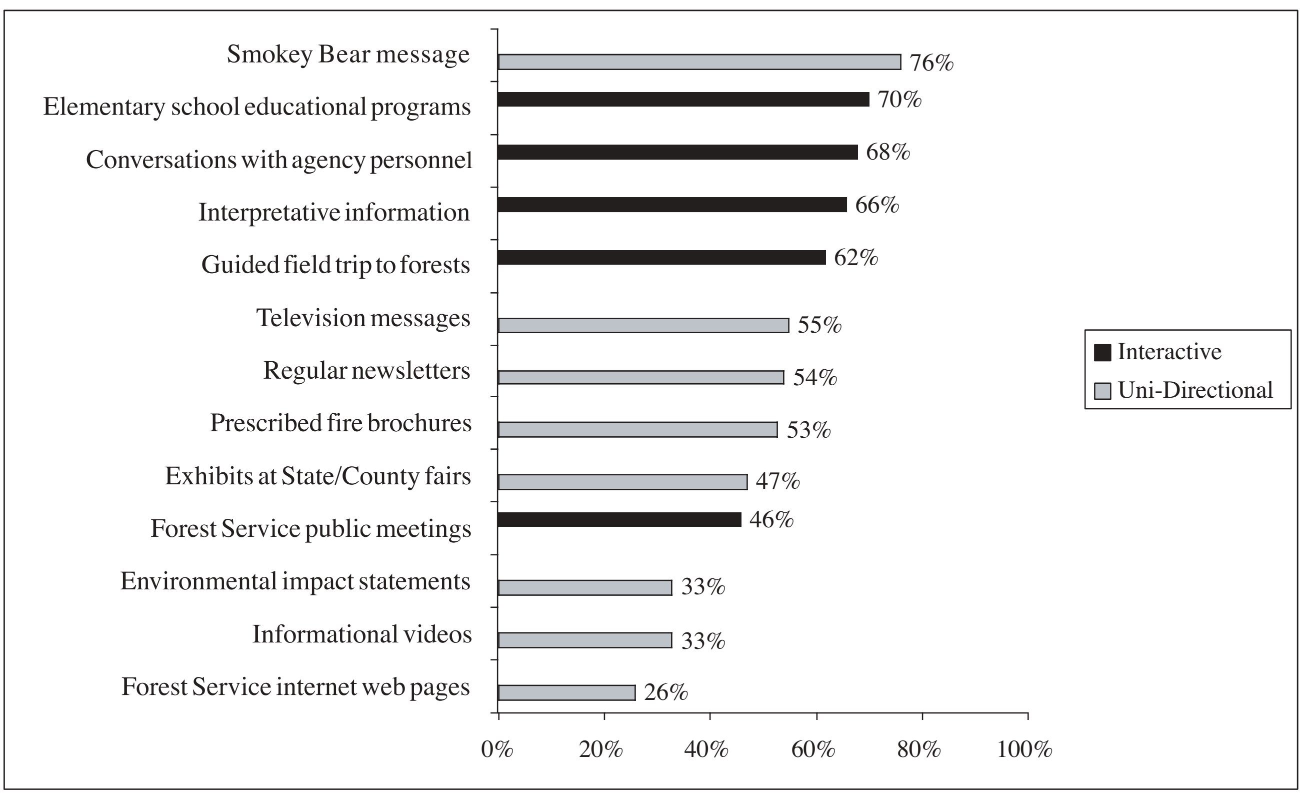

Figure 2—Usefulness of Forest Service information and outreach in the Blue Mountains. Data reflect percentage of citizens who rate usefulness as moderate or high on a 4-point scale (none, slight, moderate, high).

found Environmental Impact Statements, informational videos, and Forest Service internet web pages to be useful.

The outreach programs have been further divided into interactive and uni-directional formats. Interactive programs are those that provide for either personal contact with agency representatives or on-the-ground learning experiences. Uni-directional programs are those that typically involve a one-way flow of communication from the agency to the public. While web pages have the capability to provide a form of virtual interaction, they are included with the one-way messages because none of the Forest Service websites within the Blue Mountains offered an interactive option at the time of this study.

Four of the five most highly rated programs were interactive—elementary school programs, conversations with agency personnel, interpretative centers, and guided field trips—indicating greater dividends may be achieved from this form of outreach. Of the interactive programs, only Forest Service public meetings failed to resonate with a majority of the respondents. Of the uni-directional programs, four—Smokey Bear, television messages, newsletters, and prescribed fire brochures—were useful to a majority of respondents.

Given that one of the stated goals for the use of prescribed fire and mechanized thinning treatments is the restoration of healthy forest conditions, respondents were asked to indicate their perception of the condition of Blue Mountain forests. Opinions about forest health conditions have improved significantly since 1996 (figure 3). While only a few respondents considered conditions to be "very healthy" in either year, responses generally shifted from the lower end (unhealthy) toward the "somewhat healthy" category.

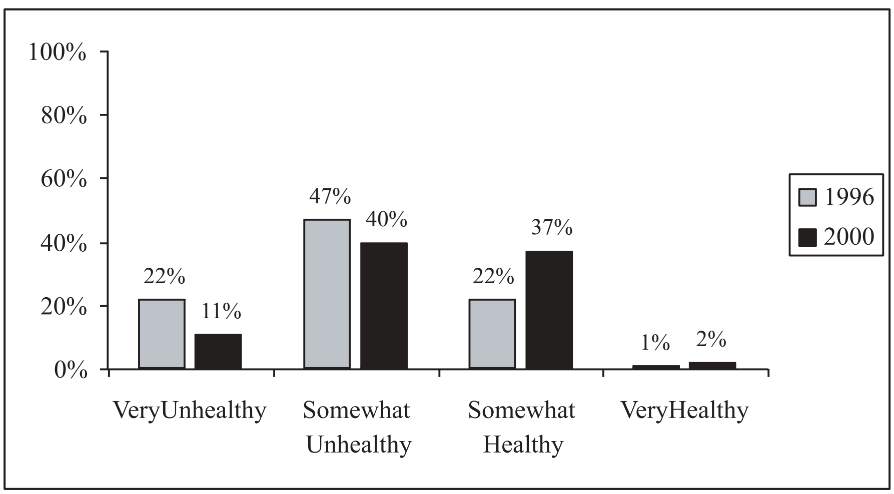

Figure 3—Overall condition of forests in the Blue Mountains. "Don't know" responses omitted. 1996 and 2000 responses significantly different at p £ .01.

Citizen-Agency Interactions

The ability for agencies to interact effectively with local publics is important to successful implementation of management activities (Shindler et al. 2002). Table 2 shows responses about citizens' experiences with the Forest Service from both surveys and responses to four new questions asked only in 2000. Opinions were mixed about these interactions; however, if viewed as a report card, the scores overall were not particularly good.

More specifically, the data show there were several changes since 1996 in respondent assessments of their interactions with the Forest Service. Although agreement was low to begin with, significantly fewer respondents agreed in 2000 that federal agencies use public input to shape management decisions.

Table 2—Experiences and interactions with forest management agencies.[a]

		Agree	Neutral	Disagree	Significance level[b]
Agencies like the Forest Service are open to public input and use it to shape forest management decisions.	**1996**	41%	25%	34%	£ .01
	2000	31%	27%	42%	
Forest managers usually create plans without input from local communities surrounding National Forests.	**1996**	55%	23%	22%	£ .01
	2000	46%	24%	30%	
The Forest Service does a good job of providing information about its management activities.	**1996**	33%	33%	35%	£ .05
	2000	27%	29%	43%	
Our federal forest management systems need major changes, not just minor adjustments.	**1996**	59%	24%	18%	NS
	2000	62%	23%	15%	
The Forest Service should provide a stronger leadership role.	**1996**	54%	32%	14%	NS
	2000	49%	35%	17%	
Federal forest managers build trust and cooperation with citizens so that people will feel that the agency is acting in their best interest.	**2000**	23%	25%	52%	N/A
I trust the local Blue Mountains Forest Service staff, but I don't trust government at the national level to let them do their job.	**2000**	65%	19%	16%	N/A
Local Forest Service staff are prohibited from doing their job because of national restrictions and regulations.	**2000**	68%	24%	8%	N/A
The Forest Service contributes to public knowledge by educating communities about potential benefits and costs of proposed plans.	**2000**	33%	37%	30%	N/A

[a] Responses on a 5-point scale from strongly agree to strongly disagree with a neutral midpoint.
[b] NS = Not Significant, N/A = question not asked in 1996

Given this level of response, it is curious that the number of respondents who agreed that managers usually create plans without input from local communities has decreased, although almost half still believe this to be the case. These two responses seem contradictory but could indicate that while more citizens recognize the Forest Service does solicit public comments, those same citizens do not believe these comments are reflected in management plans. Also, fewer respondents, just 27%, believed the Forest Service does a good job of providing information about its management activities. As in 1996, a majority of respondents agreed that our federal forest management systems need major changes and nearly 50% felt the Forest Service should provide a stronger leadership role.

Responses to new statements added on the 2000 questionnaire provide increased cause for concern. Few people viewed Forest Service actions as building trust and cooperation with citizens; in fact, a majority disagreed with this statement. The next two items seem to shed light on this finding. Frustration over national level politics and external influences on managing local forests runs high; about two-thirds indicated they trust local Forest Service personnel, but feel government at the national level hinders these individuals from doing their job. Participants were split in their agreement of whether the Forest Service educates communities about the benefits and costs of proposed plans.

A substantial number of people chose the neutral response for most statements. Since no "don't know" category was provided, it may be safe to assume that many of the neutral responses were from people who had no basis for judgment about these issues. This situation usually indicates an opportunity to reach out to a segment of the public and positively influence how these individuals come to view the agency.

Public Attitudes About Prescribed Fire and Mechanized Thinning

Given the many different terms and definitions associated with prescribed fire and thinning activities, care was taken to clearly explain the definition of each practice as used in the questionnaire. The use of the term prescribed fire was limited to management-ignited prescribed fire. Mechanized thinning treatments were defined to include the removal of down logs or standing dead and dying trees less than 15 inches in diameter. Overall support for the use of prescribed fire and mechanized thinning remained similar to 1996 (tables 3 and 4). Most respondents supported some use of prescribed fire in the Blue Mountains; 39% believed the Forest Service should have full discretion for its use, while an additional 50% felt the agency should use prescribed fire only in carefully selected areas—an apt description of current agency fire policy. Support was significantly higher for mechanized thinning; 97% of respondents supported some level of thinning with more than two-thirds giving the agency full discretion for its use.

Table 3—Prescribed fire policies.[a]

The use of prescribed fire in the Blue Mountains...	1996	2000
. . . is a legitimate management tool that the Forest Service should have the discretion to use for improving forest conditions.	44%	39%
. . . should be used sparingly by the Forest Service and only in carefully selected areas.	45%	50%
. . . creates too many impacts and should not be considered as a management alternative.	6%	7%
. . . is unnecessary and should not be utilized.	5%	4%

[a] No significant differences in responses between 1996 and 2000.

Table 4—Mechanized thinning policies.[a]

The use of mechanized selective thinning in the Blue Mountains. . .	1996	2000
. . . is a legitimate management tool that the Forest Service should have the discretion to use for improving forest conditions.	68%	69%
. . . should be used sparingly by the Forest Service and only in carefully selected areas.	28%	28%
. . . creates too many impacts and should not be considered as a management alternative.	2%	2%
. . . is unnecessary and should not be used.	1%	1%

[a] No significant differences in responses between 1996 and 2000.

When asked to indicate their preference for treating the build up of dead trees in the Blue Mountains, respondents clearly preferred a combined thinning and prescribed fire treatment (table 5). It is notable that very few respondents (4%) believed that doing nothing about this problem is a responsible option.

Despite high levels of support, citizen trust in the Forest Service to implement responsible and effective fuel reduction programs appears low (table 6). Regarding the use of prescribed fire, trust levels are not only low but decreased significantly during the study period (52% to 43%). While trust levels

Table 5—Preferred treatment of the existing build up of dead trees in the Blue Mountains.

	Agree
Selective thinning first, then follow with prescribed fire	75%
Use selective thinning only	20%
Use prescribed fire only	1%
Nothing, let nature take its course	4%

Table 6—Citizen trust in the Forest Service to implement fuel reduction programs.[a]

		Agree	Neutral	Disagree	Significance level
I trust the Forest Service to implement a responsible and effective prescribed fire program.	**1996**	52%	21%	25%	£ .05
	2000	43%	21%	34%	
I trust the Forest Service to implement a responsible and effective mechanized thinning program.	**1996**	59%	15%	24%	NS
	2000	52%	21%	26%	

[a] Responses on a 5-point scale from strongly agree to strongly disagree with a neutral midpoint.

were slightly higher for the use of mechanized thinning, only a slim majority (52%) expressed confidence in the agency. A substantial number of people were still neutral regarding both practices, most likely reserving judgment until they see the outcome of agency decisions.

Thus far, the most vocal opposition to the use of fire in the region has been over increased smoke levels. Although temporary in nature, smoke has long been recognized as a public concern associated with the use of prescribed fire, especially because of adverse effects on human health and visibility in communities or along transportation corridors. In 1996 a substantial majority of respondents indicated that smoke levels from prescribed fire were not a problem for their family, nor did they feel that prescribed fire use should be limited because of potential health problems from smoke (table 7). Although the majority of respondents still held similar views in 2000, significantly more people now view smoke as a problem, and, as a consequence, fewer support the use of prescribed fire. When dealing with smoke a simple majority in favor of management practices is probably an insufficient level of public approval. Smoke is a highly contentious issue that provides a rallying point for communities. Those adversely affected by smoke usually vocalize their concerns, a point made by fire managers within the study area.

Table 7—Assessments of smoke.[a]

		Agree	Neutral	Disagree	Significance level
In my area, smoke levels from fire are not a problem for me or my family.	**1996**	76%	10%	14%	£ .01
	2000	61%	15%	24%	
Prescribed fire should not be used because of potential health problems from smoke.	**1996**	12%	17%	71%	£ .01
	2000	14%	26%	61%	

[a] Responses on a 5-point scale from strongly agree to strongly disagree with a neutral midpoint.

Conclusions

Public acceptance is an essential element in virtually every resource management decision facing public agencies today. Problems such as fire management and forest health, given the attendant risk and uncertainties surrounding these issues, are particularly subject to public debate. Although we have pointed out numerous areas of public support for fuel management practices as well as several specific areas of concern, we recognize these are complex issues. Problems frequently are embedded within larger issues and connected to factors beyond the control of resource managers. Today, the roles that agency personnel are being asked to play are much different than in the past, when citizen participation was minimal and technical expertise was foremost. In this new role, greater public acceptance will be achieved by being aware of, and responsive to, the suite of intertwined ecological factors and community circumstances affecting fuel management. Thus, our concluding comments involve a set of three overarching themes, expressed as basic strategies that emerge from our analysis.

Capitalize on Existing Public Knowledge and Support for Fuel Reduction

Primary results from this study indicate the presence of a knowledgeable general public in the Blue Mountains, solid support for both prescribed fire and mechanized thinning to reduce forest fuels, and an overall stability of public attitudes throughout the study period. Collectively, these findings provide positive news for Forest Service programs. They also suggest that this existing base of well-informed, supportive stakeholders could be a central asset in building future management programs.

Study results also indicate that public understanding and support for treatments is not universal; trouble spots exist. Foremost is the provisional aspect of public support, as evidenced by a drop in the number of people who think the Forest Service can implement a responsible fuel management program. Trust in the agency is central to this issue, but other factors are also likely to apply. In the case of fire management, support is often dependent on the level of uncertainty about outcomes and public understanding of the risks involved. People want to know how serious conditions are, what will happen and when, and who will be affected. Answers to these questions are not simple, nor are they consistent across settings. But to be relevant to the public, fire management policies will need to be placed in a context that is important to them. To the extent possible, managers will need to provide scenarios that depict what changes in forest conditions will look like, how soon they could occur, and help citizens understand what the consequences of changes will mean for forest ecosystems and surrounding communities.

Our analysis suggests that citizens in the Blue Mountains are cautiously willing to allow these policies to proceed. As effective treatments are implemented and public awareness of these successes grows, so too will belief that the Forest Service can be trusted to handle the risks associated with fire and fuel management.

Focus on Relations With Citizens

The most troublesome finding from this study is the erosion of relations between citizens and the Forest Service. As elsewhere, many of these feelings are attributable to the tension between policies set at the national level and the need to create strategies for managing forests at the community level. But it is not likely this circumstance explains all aspects of the declining relationship with the public in the Blue Mountains. Thus, two approaches seem particularly useful. First, to the extent possible, agency personnel will need to filter out the national debate and focus on what can be accomplished in local forests. Programs will need to target local priorities. Second is the necessity to increase opportunities for ordinary citizens and organized groups to have a more meaningful role in planning and implementing fire management strategies. The common thread that runs through all aspects of forest management is the importance of trustworthy relations among stakeholders. It really makes little difference how meritorious a plan may be; nothing will be validated unless the people involved trust one another.

The initial requirement for improving relations and building public trust is an organizational commitment to multi-partner collaboration. This largely depends on whether the leadership on Forests and ranger districts is serious about genuine involvement of stakeholders and how well the actions of personnel reflect this philosophy. Currently, most collaborative efforts and the trust building process remain the job of individuals at the ranger district level where relationships are established and face-to-face interactions can make a difference for residents and their communities. The informal nature of these situations is perhaps the most productive form of relationship building. But this can only occur in a meaningful way when the agency promotes these ideas and supports personnel in their outreach efforts.

Develop a Comprehensive Communication Strategy

Natural resource agencies often think their job is to develop information and deliver it to the public. We still hear frustrated forestry professionals make statements like, "If we could just educate people about the forest and inform them about what we do, then they would understand and support us." But our tendency to confuse providing information with public understanding and eventual support is a mistake. Although information and knowledge are necessary elements of any public communication strategy, they are rarely sufficient to produce change in the way citizens respond to forest agencies. Public judgments, including judgments of forest conditions and fuel reduction practices, are formed by a suite of factors beyond technical knowledge (Stankey 1996). Thus, the *process* of how people come to understand forest conditions and support policies such as fuel reduction also needs to be an integral component of a communication strategy. A comprehensive public communication strategy will not only focus on the types and content of the information that is disseminated, but also on how and why it is communicated.

Regarding type and content, the data from this project can help organize an approach for communication activities. Two basic levels of communication exist, and each is useful depending on the purpose and intended coverage. One is general information dispersal; this usually involves broad messages that can be conveyed by traditional "bulk" formats such as newspapers, brochures and public service announcements. These are typically for general public consumption and, as such, provide few opportunities to target specific audiences. Because it is difficult to ensure that information is received and understood, their effectiveness as an educational tool is limited. Data in this study seem to reinforce this assessment; most forms of communication in this category are uni-directional formats that received moderate to low level usefulness ratings. Although the agency should continue to use these informational devices—they are often inexpensive, may be helpful in notifying large audiences about upcoming meetings or proposed projects, and provide some value collectively as a critical mass of information—outreach personnel should not rely on these as primary tools for communicating with local publics.

The second level of communication is more focused in scope and usually includes opportunities for interaction at the community or individual level. In this category are the more highly rated activities such as school programs, visitor centers, and guided field trips. Although these forms generally have widespread acceptance as effective outreach devices, they are also highly dependent on the communication abilities of agency personnel who plan and implement them. For example, public meetings fall into this category but our participants (as elsewhere) did not respond well to this format (Cortner et al. 1998). This seems to be a sign that the more traditional information-sharing or scoping meetings, typically convened to meet NEPA requirements, should be altered in favor of other meeting formats that provide for increased dialogue and two-way communication. But determining the most useful formats is a job to be shared. From a practical standpoint, findings from this study can be used by local personnel to engage communities about which level and forms of information exchange are preferable.

The substantive content of information sources is also a primary consideration. Most content ideas are simple and straightforward—almost intuitive—but neglecting them can be detrimental to communications. Research in forest communities by Shindler and Neburka (1997) as well as Winter et al. (2002) identified specific characteristics of good message content. These include:

- information where terms are defined for common use
- current, accurate, and understandable information that comes from a reliable source
- prescribed fire plans that specifically account for conditions such as weather, proximity to homes and timing of events
- fire and thinning plans that provide mitigation measures to reduce impacts on air quality, aesthetics, etc.
- contingency measures for escapes
- cost comparisons of various treatment alternatives
- specific details about who to contact for questions and concerns

Providing the opportunity for people to evaluate the range of information about fuel management and the choices involved brings them much closer to lending support for decisions. Most people like those in our study region are capable of assessing the tradeoffs, including positive and negative consequences, and welcome the chance to do so. When given a set of choices—even choices that are limited or imperfect—citizens will often choose the lesser of the two evils and accept it (Ehrenhalt 1994). The ability of fire management professionals to specify conditions and engage citizens in discussion about the nature of the options is just as essential as providing objective, unbiased information. Of course, this will mean that personnel must be forthcoming about the difficult decisions, including the uncertainty of outcomes associated with the use of fire and thinning treatments.

Useful forums for discussion about forest conditions and fuel management usually involve interactive exchanges, often in places where people can evaluate real-life scenarios prior to policy changes or broad scale implementation of treatments. This study shows that people prefer settings where they can have a more active, legitimate role—settings such as field visits to treatment sites to review alternatives or planning sessions where stakeholders are given consideration for their points of view and their suggestions are openly discussed and evaluated.

References

Babbie, E. 1995. The practice of social research. 7 ed. Belmont, CA: Wadsworth Publishing Company. 476 p.

Beebe, G.S., and P.N. Omi. 1993. Wildland burning: The perception of risk. Journal of Forestry 91 (9):19-24.

Brunson, M.W., and D.K. Reiter. 1996. Effects of ecological information on judgments about scenic impacts of timber harvest. Journal of Environmental Management 46 (1):31-41.

Carpenter, E.H., J.G. Taylor, H.J. Cortner, P.D. Gardner, M.J. Zwolinski, and T.C. Daniel. 1986. Targeting audiences and content for forest fire information programs. Journal of Environmental Education 17 (3):33-42.

Cochran, P.H., and J.W. Barrett. 1993. Long-term response of planted ponderosa pine to thinning in Oregon's Blue Mountains. Western Journal of Applied Forestry. 8 (4): 126-132

Cortner, H.J., M.G. Wallace, S. Burke, and M.A. Moote. 1998. Institutions matter: The need to address the institutional challenges of ecosystem management. Landscape and Urban Planning 40:159-166.

Devore, J., and R. Peck. 1986. Statistics: The exploration and analysis of data. Los Angeles, CA: West.

Ehrenhalt, A. 1994. Let the people decide between spinach and broccoli. Governing 7 (10):6-7.

Lee, R.G. 1987. Community fragmentation: Implications for future wildfire management. In Proceedings of the symposium on wildland fire 2000, 1987 April 27-30; South Lake Tahoe, CA. J.B. Davis and R.E. Martin, editors. Gen. Tech. Rep. PSW-100. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station.

Loomis, J.B., L.S. Bair, and A. Gonzalez-Caban. 2001. Prescribed fire and public support: Knowledge gained, attitudes changed in Florida. Journal of Forestry 99 (11):18-22.

McCool, S. F., and G. H. Stankey. 1986. Visitor attitudes toward wilderness fire management policy—1971-84. Research Paper INT-357. Odgen, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 7 p.

Mutch, R. W., S. F. Arno, J. K. Brown, C. E. Carlson, R. D. Ottmar, and J. L. Peterson. 1993. Forest health in the Blue Mountains: A management strategy for fire-adapted ecosystems. Gen. Tech. Rep. PNW-GTR-310. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 14 p.

Shindler, B., B. Steel, and P. List. 1996. Public judgments of adaptive management: A response from forest communities. Journal of Forestry 94 (6):4-12.

Shindler, B., and M. Reed. 1996. Forest management in the Blue Mountains: Public perspectives on prescribed fire and mechanical thinning. Corvallis, OR: Department of Forest Resources, Oregon State University. 57 p.

Shindler, B., and J. Neburka. 1997. Public participation in forest planning: Eight attributes of success. Journal of Forestry 95 (1):17-19.

Shindler, B., and P. Collson. 1998. Assessing public preferences for ecosystem management practices. In Ecosystem management: A social science perspective, D. Soden and B. L. Lamb, editors. Commack, NY: Nova Science.

Shindler, B., and E. Toman. 2002. A longitudinal analysis of fuel reduction in the Blue Mountains: Public perspectives on the use of prescribed fire and mechanized thinning. Corvallis, OR: Department of Forest Resources, Oregon State University. 76 p.

Shindler, B., M. Brunson, and G. H. Stankey. 2002. Social acceptability of forest conditions and management practices: A problem analysis. PNW-GTR-537, Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 68 p.

Stankey, G. H. 1976. Wilderness fire policy: An investigation of visitor knowledge and beliefs. Research Paper INT-180. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 17 p.

Stankey, G.H. 1995. The pursuit of sustainability: joining science and public choice. The George Wright Forum 12 (3):11-18.

Stankey, G.H. 1996. Defining the social acceptability of forest management practices and conditions: integrating science and social choice. In Defining social acceptability in ecosystem management: a workshop proceedings. M. Brunson, Kruger, L.E., Tyler, C.B., and S.A. Schroeder, editors. Gen. Tech. Rep. PNW-GTR-369. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.

Tanaka, J. A., G. L. Starr, and T. M. Quigley. 1995. Strategies and recommendations for addressing forest health issues in the Blue Mountains of Oregon and Washington. Gen. Tech. Rep. PNW-GTR-350, Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 18 p.

Winter, G.J., C. Vogt, and J.S. Fried. 2002. Fuel treatments at the wildland-urban interface: common concerns in diverse regions. Journal of Forestry 100 (1):15-21.

Fire, Fuel, and Restoration Priorities of the Forest Conservation Community

Gregory H. Aplet[1]

In February 2001, forest activists from all over the country gathered in Boulder, Colorado, to listen to presentations and engage in discussion about the science and practice of forest restoration. A year in the making, the result is a set of agreed-upon principles and criteria that form the foundation of the conservation community's stance on restoration and fuel treatment. The document is organized around three core principles reflecting ecological, economic, and community and workforce considerations. Paramount among the principles are planning and adaptive management. With regard to fire, the document acknowledges the importance of restoring natural fire regimes to fire-dependent forests. To achieve this important goal, The Wilderness Society advocates a four-step approach: first, protect communities by addressing home ignitability and adjacent fuels; second, complete fire management plans on federal lands to allow Wildland Fire Use for Resource Benefit; third, re-introduce prescribed fire where it is safe to do so; and fourth, where it is not safe to burn, mechanically treat fuels to create a structure into which fire can safely be reintroduced.

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Fire Regime Considerations

Key Issues in Fire Regime Research for Fuels Management and Ecological Restoration

Thomas T. Veblen[1]

Abstract—*The premise behind many projects aimed at wildfire hazard reduction and ecological restoration in forests of the western United States is the idea that unnatural fuel buildup has resulted from suppression of formerly frequent fires. This premise and its implications need to be critically evaluated by conducting area-specific research in the forest ecosystems targeted for fuels or ecological restoration projects. Fire regime researchers need to acknowledge the limitations of fire history methodology and avoid over-reliance on summary fire statistics such as mean fire interval and rotation period. While fire regime research is vitally important for informing decisions in the areas of wildfire hazard mitigation and ecological restoration, there is much need for improving the way researchers communicate their results to managers and the way managers use this information.*

Introduction

The two major management themes of this conference are: 1) fuel treatments for the purpose of reducing fire hazard, and 2) ecological restoration through a variety of management practices including prescribed fire. The title and content of the conference might lead to the impression that fire hazard reduction through fuel treatments and ecological restoration have convergent objectives in all forest ecosystems in the western United States. However, this implication needs to be explored on the basis of existing knowledge of historical fire regimes and forest conditions on a case by case basis for different forest cover types and different locations. In some forest ecosystems fire hazard reduction through fuels management may be achieved by restoring historic fire regimes of frequent surface fires. However, in other forest ecosystems, historic fire regimes included widespread stand-replacing fires at long intervals. In those systems, restoration of the historic fire regime will not reduce the hazard to property and humans. This essay introduces a series of papers on fire regimes by identifying some of the key issues and research challenges for fire regime research.

Political leaders and many resource management professionals often stress the convergence of the goals and strategies of fire hazard reduction and ecological restoration in the forests of the western United States. For example, the official position of the Society of American Foresters in response to the 2000 fire season included the statement that.

> "The buildup of combustible materials (fuels) in the forests of the West is at an all-time high. Much of this can be attributed to the decades of fire suppression that allowed the fuels to build up so fires will now burn bigger and hotter than ever" — Society of American Foresters, August 11, 2000, press release.

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There is a widespread belief among resource managers, reflected by many of the papers presented in the current conference, that fuel accumulation during nearly a century of fire suppression in western forests was the major cause of the widespread wildfires of the 2000 season. Likewise, there is a consensus that a perceived decline in "forest health" (tree diseases, mistletoe infection, and forest insect pests) is the result of fire exclusion. One of the leading experts on ecological restoration in western U.S. forests has written:

> "The dry forest ecosystems of the American West, especially those once dominated by open ponderosa pine forest, are in widespread collapse. We are now witnessing sudden leaps in aberrant ecosystem behaviour long predicted by ecologists and conservation professionals (see Nature 407, 5; 2000). Trends over the past half-century show that the frequency, intensity and size of wildfires will increase – by orders of magnitude – the loss of biological diversity, property and human lives for many generations to come." — Covington 2000, p. 135.

The view that current fire hazard is largely attributable to fuel buildup under decades of fire exclusion is strongly reflected in the following passage from the National Fire Plan:

> "While the policy of aggressive fire suppression appeared to be successful, it set the stage for the intense fires that we see today. ...after many years of suppressing fires, thus disrupting normal ecological cycles, changes in the structure and make-up of forests began to occur. Species of trees that ordinarily would have been eliminated from forests by periodic, low-intensity fires began to become a dominant part of the forest canopy. Over time, these trees became susceptible to insects and disease. Standing dead and dying trees in conjunction with other brush and downed material began to fill the forest floor. The resulting accumulation of these materials, when dried by extended periods of drought, created the fuels that promote the type of wildfires that we have seen this year.
>
> "In short, decades of aggressive fire suppression have drastically changed the look and fire behavior of Western forests and rangelands. Forests a century ago were less dense and had larger, more fire-resistant trees. For example, in northern Arizona, some lower elevation stands of ponderosa pine that once held 50 trees per acre, now contain 200 or more trees per acre. In addition, the composition of our forests have changed from more fire-resistant tree species to non-fire resistant species such as grand fir, Douglas-fir, and subalpine fir. As a result, studies show that today's wildfires typically burn hotter, faster, and higher than those of the past."— National Fire Plan 2001.

While the National Fire Plan also recognizes the importance of other contributing factors to our current wildfire management crisis, including weather influences (i.e., the effects of the El Niño-Southern Oscillation) and the land use policies that have permitted uncontrolled growth at the wildland-urban interface, this quotation represents the dominant view of the current wildfire management problem among political leaders, resource managers, and the general public. For convenience, I will refer to this view as the "fire exclusion / fuel buildup" perspective on current fire hazard in western U.S. forests.

An important theme of this essay is that assessment of fire hazard, and especially ecological restoration, requires a sound understanding of historic fire regimes in the ecosystem of interest. "Fire regime" is used here to refer to the spatial and temporal variation of fires and their effects in a given area over a given time period. The parameters used for describing fire regimes are briefly discussed below under Methodological Issues. One major goal of fire regime research is to discover under what historic and present fire regimes and ecosystem conditions do the goals and methods of fuels reduction and ecological restoration converge.

The idea that current fuel levels are unnaturally high due to suppression of formerly frequent surface fires originated to a large extent from studies of ponderosa pine ecosystems. This viewpoint is best supported by multiple lines of research on Southwest ponderosa pine ecosystems showing that frequent-fire disturbance regimes were disrupted after Euro-American settlement throughout the Southwest resulting in major increases in stand densities and in larger and more intense wildfires (Moore et al. 1999). Supporting evidence comes from numerous retrospective studies of fire-scar reconstruction of fire regimes (Swetnam and Baisan 1996, Fulé et al. 1997), tree-ring based reconstructions of past stand structures (Covington and Moore 1994, Fulé et al. 1997, Mast et al. 1999), and historical evidence from photographs and early 20th century forest inventories (Moore et al. 1999). The frequent surface fires that had maintained open-canopy conditions declined dramatically in the late 19th century due to grass fuels reduction by introduced livestock and subsequently due to organized fire suppression activities (Swetnam and Baisan 1996, Moore et al. 1999). Detailed studies of past fire and forest conditions support a series of carefully planned and executed restoration projects in Southwest ponderosa pine ecosystems (Covington et al. 1997; Moore et al. 1999). This overall approach has become a model for a step-by-step process of conducting careful historical ecological research followed by experimentation and monitoring of restoration treatments. Similar step-by-step approaches to ecological restoration based on area-specific research of fire regimes and past forest conditions are in earlier phases of development in ponderosa pine forests in other regions, including Colorado (e.g., City of Boulder 1999; Brown et al. 1999, 2001; Kaufmann et al. 2000, 2001; Huckaby et al. 2001; Mackes and Lynch 2003; Romme et al. 2003).

How applicable is the Southwest ponderosa pine model of fire exclusion and subsequent changes in forest conditions (e.g., Covington et al. 1997, Moore et al. 1999) to other forest ecosystem types and to ponderosa pine in other regions? For example, there is abundant documentation of pre-1900 stand-replacing fires occurring in apparently denser stands of ponderosa pine in the Colorado Rocky Mountains (Veblen and Lorenz 1986, 1991; Brown et al. 1999; Kaufmann et al. 2000; Mast et al. 1998; Ehle 2001). The occurrence of stand-replacing fires in some ponderosa pine forests prior to c. 1900 raises the issue of geographical variability in fire regimes for ponderosa pine-dominated forests. This theme will be explored further in the next section on ponderosa pine forests in the northern Front Range of Colorado. For other ecosystem types, such as California shrublands and boreal and subalpine forests in Canada the validity of the fire exclusion / fuel buildup argument has been directly challenged (Keeley and Fotheringham 2001; Johnson et al. 2001). Thus, there is a need to conduct unique fire regime research for each particular area in order to evaluate the general applicability of the fire exclusion / fuel buildup viewpoint.

The goal of this essay is to show the need for conducting area-specific fire regime research to test the applicability of the fire exclusion / fuel buildup viewpoint to particular ecosystems and potential management areas. My intent is not to evaluate the validity of the fire exclusion / fuel buildup generalization for all the forests of the western U.S. Nor is it my intent to suggest that fuels treatments are not appropriate for particular ecosystem types and land use classes, or that we return to a Smokey Bear type of fire suppression policy. Instead, I propose that the broad generalizations of the fire exclusion / fuel buildup viewpoint be used to generate specific research questions and hypotheses that can be critically evaluated for particular cover types and locations. This essay will identify ways in which fire regime research can support resource planning and management decisions in both the contexts of fire hazard management and ecological restoration. I will first draw on several examples from northern Colorado. Then I will suggest ways in which fire regime research can better support resource planning and management decisions.

Assessing the Fire Exclusion/Fuel Buildup Perspective in Some Northern Colorado Forests

The fire exclusion / fuel buildup perspective is based on several general premises that logically generate specific questions or hypotheses for particular forested landscapes (table 1). Each question needs to be examined across a range of scales from individual stands (e.g., a few hectares to 100s of hectares) to landscape scales (e.g., 10s to 100s of square kilometers). Examples from northern Colorado illustrate major variations in fire regimes of different forest ecosystem types and allow comparison with similar ecosystem types in other regions.

Table 1—Examples of some of the premises of the fire exclusion/fuel buildup viewpoint and possible area-specific research questions.

Major premises	Possible research questions to be examined for particular areas
Fire exclusion has created unnatural fuel buildup.	Do modern fire regimes differ greatly from historic fire regimes? Is there clear evidence of disruption of frequent fires that occurred before EuroAmerican settlement?
Severe, widespread fires are due to unnatural fuel buildup after decades of fire exclusion.	Did large, crown fire events occur prior to any effects of fire exclusion?
Elimination of formerly frequent surface fires has created dense stands in the modern landscape.	What was the historic range of tree densities prior to effects of fire exclusion? What other explanations might account for dense stands today, such as stand responses to logging or abundant burning in the late 19th century or the effects of changes in grazing pressure?
Recent years of widespread, severe fires (e.g., the 2000 fire season) are due primarily to the effects of fire exclusion rather than climatic variation.	Did historic fire regimes include fire events similar to those of the 2000 fire season? Has recent climatic variation contributed to any recent increases in fire frequency or severity? Has climatic variation in the past resulted in fires of similar extent and severity to recent fires?
Current levels of pathogen and insect outbreaks are unnatural and are the consequence of fire exclusion.	What was the historical variability of pathogen and insect outbreaks prior to fire exclusion?

Spruce-Fir Forests: Long Fire Intervals

To evaluate the premise that fire exclusion has resulted in unnatural fuel buildup, it is logical to ask: How different are modern fire regimes from historic fire regimes in spruce-fir forests in northern Colorado? Spruce-fir forests in northern Colorado have been shaped primarily by stand-replacing (crown) fires that recur to the same point or stand at relatively long intervals, usually much greater than a century in length (Veblen 2000, Sibold 2001, Kulakowski 2002). Surface fires occasionally occur but to date there is no documented occurrence of frequent (i.e., at repeated intervals of <50 years) and widespread (i.e., affecting >8 ha) surface fires. Although crown fires in the spruce-fir type typically kill most (>90%) of the canopy trees over large areas (100s to over 1000s of hectares), some fires have apparently been less intensive or less continuous resulting in younger post-fire cohorts intermixed with older trees that survived the most recent fire (Sibold 2001, Kulakowski 2002). It is noteworthy that large percentages (i.e., >25%) of spruce-fir forests mapped in areas of >4000 hectares and at minimum map units of c. 8 hectares do not record any stand-replacing fires in the past c. 400 years. In other words, a large part of the spruce-fir cover type has not been significantly affected by fire for more than 400 years.

Clearly, the fire regime of the spruce-fir cover type in northern Colorado is characterized by infrequent, crown fires that burn large areas. High severity fires resulting in spruce-fir stands of high tree densities are part of the natural fire regimes of this ecosystem type (Veblen 2000). Due to the long intervals between fires in the pre-1900 period, it is unlikely that fire exclusion has created forest conditions that are outside the historic range of variation. Fire history mapping in large areas (i.e., >4000 ha) at multiple sites in northern Colorado show that the post-1900 fire regime is not unique in comparison with time periods of similar length during the past c. 400 years (Sibold 2001, Kulakowski 2002). Periods of 80 to well over 100 years of no widespread (i.e., >100 ha) fires in study areas of 4000 or more hectares are typical of the historic fire regimes of the spruce-fir cover type. Given these long intervals between widespread fires in these spruce-fir forests, the fire-free interval that began with fire suppression after c. 1910 is not outside the historic range of variability for this cover type.

This conclusion is specific to the c. 4000 hectare scale at which these studies were conducted and to the spruce-fir cover type. Future research at broader spatial scales potentially may alter these research findings, but the current state of knowledge indicates that fire occurrence in these spruce-fir forests during the past 100 years is not outside the historic range of variability of the past c. 400 years. Thus, the premise that fire exclusion has created unnatural fuel buildup in these spruce-fir forests is not supported. Likewise, apparent forest health problems should not be attributed to unnaturally long fire intervals resulting from fire exclusion in the spruce-fir cover type. Indeed, widespread outbreaks of the major lethal forest insect in this cover type, the spruce beetle (*Dendroctonus rufipennis*), caused massive mortality of spruce during a well documented 19th century outbreak in northwestern Colorado long before any significant influences of EuroAmericans on these forests (Baker and Veblen 1990, Veblen et al. 1991).

Ponderosa Pine-Dominated Montane Forests: Spatial Variability

The long fire intervals typical of spruce-fir forests make it a relatively clear example of where the fire exclusion / fuel buildup viewpoint is not valid, but the situation is more complex at lower elevations in the montane zone of ponderosa pine and Douglas-fir forests. In the northern Front Range there are areas primarily at lower elevations and near ecotones with grasslands where fire-scars indicate relatively frequent occurrence of non-lethal surface fires in ponderosa pine stands prior to the early 1900s (i.e., many fire intervals <20 years at a spatial scale of c. 100 ha; Veblen et al. 2000). Historical photographs and tree ages indicate that since the early 1900s there has been a substantial increase in tree densities in these ponderosa pine ecosystems (Veblen and Lorenz 1986, 1991; Mast et al. 1998). Thus, at lower elevations (and at mid-montane sites adjacent to grasslands) there are sites in the northern Front Range where conversion of formerly open woodlands to relatively dense stands of ponderosa pine are qualitatively similar to the pattern widely documented in Arizona (Moore et al. 1999). Likewise, this pattern of increased tree density under reduced fire frequency is documented for some sites in the southern Front Range (Brown et al. 1999, Kaufmann et al. 2000). In such areas the fire exclusion / fuel buildup viewpoint is supported.

The challenge is to determine the spatial limits of this pattern of substantial increase in tree density following exclusion of formerly frequent surface fires. Toward higher elevation and at more mesic sites in the northern Front Range, a variety of evidence indicates that the historic fire regime was a mixed-severity regime including both stand-replacing and surface fires. In the northern Front Range in ponderosa pine forests, the pre-1900 frequency of fire events inferred from fire scars declines dramatically with increasing elevation (Veblen et al. 2000). At the spatial scale of c. 50 to 200 ha at elevations above c. 2100 m, most fire intervals are well over 50 years in length, and there is no evidence of frequent (i.e., repeated intervals <20 years), widespread surface fires. In these stands with relatively long fire intervals and in the surrounding areas, the predominant age structure type is even-aged with most stands originating between the mid-1800s and early 1900s, but with remnants of older cohorts as well (Veblen and Lorenz 1986; Sherriff and Veblen unpublished data).

Historical photographs of the upper montane zone taken in the late 1800s to early 1900s show that large areas of ponderosa pine-dominated forests (typically with some component of Douglas-fir and other species) had burned in stand-replacing crown fires in the mid- to late-1800s prior to any significant fire exclusion or unnatural fuel buildup (Veblen and Lorenz 1991). Research is currently underway to spatially define habitats according to the relative importance of past stand-replacing versus surface fires in shaping the current structure of ponderosa pine-dominated stands across their full elevational range in Boulder County (Sherriff and Veblen in progress). Our preliminary evidence indicates that except for a small area at lower elevations, on drier aspects, and near grassland ecotones, the structure of existing ponderosa pine forests was shaped primarily by stand-replacing fires. Over most of the surface area of the ponderosa pine cover type in the areas where we have collected data or done reconnaissance, the pattern of dense stands due to recovery following 19th century burning is much more common than the pattern of dense stands resulting from tree encroachment following cessation of frequent surface fires.

Although current understanding of changes of fire regimes and forest conditions in the low elevation ponderosa pine forests is consistent with restoration of more frequent surface fires (e.g., on City of Boulder Open Space lands; City of Boulder 1999), it does not support that prescription for the larger part of the distribution of the ponderosa pine cover type. For much of the montane zone in Boulder County, restoration of the historic fire regime would require a significant (probably dominant) component of stand-replacing fires. Due to the high density of residences in this area, it is unlikely that restoration of the historic fire regime is feasible. Prior land-use decisions have severely limited the current opportunities for ecological restoration. Stand thinning and prescribed burning may be appropriate prescriptions if priority is given to reducing fire hazard, but in this case the goals of fire hazard reduction and ecological restoration do not converge.

In addition to this lack of convergence with the goal of ecological restoration for a large part of the montane zone of the Front Range, fire hazard mitigation through extensive thinning and subsequent prescribed fire raises numerous contentious issues. It is uncertain how effective different intensities of thinning will be in mitigating crown fires and protecting structures, especially under extreme weather conditions. Observations of fire behavior in areas with and without fuels treatments are helpful, but are unlikely to be conclusive because of the other variables affecting fire spread (such as fire suppression activities and weather). There is a consensus that thinning to create “defensible space” around structures is effective and socially desirable, but the extent to which neighboring or remote tracts of forests lacking residences should be thinned is disputed (Stein 2002). Likewise, there is a potential conflict between the cutting of larger diameter trees to support mitigation costs and the desire to retain old trees for wildlife and other values. Operationally, long-term fuels management to maintain a desirable level of fuels, even if agreement can be reached on what that desirable level is, is a formidable and presently unresolved problem for management scientists (Hof and Omi 2002). Furthermore, economic and environmental costs of fire hazard mitigation are extremely high as are the costs of fire suppression and of catastrophic fires. There is much need for an informed debate over the nature of these costs under different management scenarios and of who should pay these costs in the context of private landowners’ decisions to locate in such hazardous environments.

An important caveat to the above discussion is that it is based on observations and interpretations largely at the scale of individual stands of a few ha to 200 ha in extent. Although cessation of frequent surface fires does not appear to account for currently dense stands over most of the range of this cover type in Boulder County, it is possible that the post-1900 reduction in fire occurrence has resulted in fewer young, post-fire stands than what would have occurred without fire suppression. However, relatively young stands have originated after extensive logging at the end of the 19th and beginning of the 20th centuries (Veblen and Lorenz 1991). Furthermore, widespread burning in the late 19th century and subsequent development of even-aged post-fire stands has undoubtedly contributed to the homogeneous age structure of the montane zone (Veblen and Lorenz 1986, Veblen et al. 2000). If fuel continuity is contributing to increased fire hazard, the effects of logging and past increases in fire occurrence in the late 19th century play at least as great a role as does fire exclusion.

Fire regime research in the upper montane zone of ponderosa pine-dominated forests in Boulder County indicates that prior to 1900, infrequent years of exceptionally favorable fire weather are associated with evidence of extremely widespread fire, including a major component of stand-replacing fires (Veblen et al. 2000). That retrospective perspective, in combination with dense residential development, implies that fire hazard reduction is likely to take precedence over ecological restoration in this area. Tree-ring evidence indicating that large areas of the montane zone burned during extreme droughts in the past supports management that gives priority to fire hazard mitigation. Yet, at the same time the scale and severity of pre-1900 fires in this ecosystem type raises doubts about the long-term effectiveness of fire hazard mitigation activities. The complexities of developing sustainable strategies of balancing concerns over fire hazard mitigation and ecological restoration in the montane zone of Boulder County are greater than most observers have recognized. These complexities include the tendency to apply a single thinning prescription indiscriminately without regard to the cause of high stand densities, an under appreciation of the importance of extreme weather in creating extreme fire hazard, and fundamental conflicts in the values of the stakeholders in resource management. There is an urgent need to make much greater use of existing research results on the range of variability of these ecosystems as well as to conduct new research to inform the debate over resource management.

Major Research Challenges for Ecological Restoration

In the context of restoration of fire to western forest ecosystems, the first objective is to have a sound understanding of the historic fire regime and the potential effects of EuroAmericans on the fire regime and forest conditions. This requires area-specific research for the ecosystem of interest. Once the general nature of trends in the fire regime and especially the possible effects of fire exclusion are known, there remain a number of research challenges applicable to many ecosystems targeted for restoration. The following are examples of the most common of these research challenges.

1. What was the temporal variability of the fire regime over multi-century reference periods?

Reference conditions should not be defined by a snapshot in time, such as the conditions for a few years or decades at the time of extensive EuroAmerican settlement which for most of the West is between c.1850 and 1880. Use of reference conditions should not stress a reconstruction of static conditions at a particular point in time. Instead, the goal should be to understand the recent evolutionary environment of an ecosystem, which, at a minimum requires knowledge of temporal variability over periods of several centuries. Most importantly, the historical context should be as complete as possible to identify temporal trends that may be related to climatic variation for one or two centuries just prior to and during intensive EuroAmerican settlement. For example, for the Southwest and Colorado there is abundant tree-ring evidence showing that the second half of the 19th century was climatically more conducive to fire occurrence than the period from c. 1790 to 1830 (Swetnam and Betancourt 1998, Grissino-Mayer and Swetnam 2000, Veblen et al. 2000, Donnegan et al. 2001). The 1790 to 1830 period of reduced fire occurrence in this large region has been linked to variation in the El Niño-Southern Oscillation which has significant teleconnections to the weather of the Southwest and the southern Rocky Mountains (Swetnam and Betancourt 1998, Kitzberger et al. 2001, Veblen and Kitzberger 2002). These decadal to centennial scale variations in climatic influences on fire regimes during the reference periods need to be recognized and considered in our understanding of current and future ecosystem fluxes.

2. How was the fire regime influenced by Native Americans?

There are strong and contrasting opinions about the influence of Native Americans on historic fire regimes in the western United States (e.g., see the regional reviews of this theme in Vale 2002). Broad generalizations about the pervasive influence of Native Americans (e.g., Denevan 1992) are not testable and lead to sterile debates unlikely to resolve the issue. Instead, the question of Native American influence needs to be re-framed into tractable research questions. The roles of burning by Native Americans need to be assessed for particular ecosystem types and locations. For example, in northern Colorado in spruce-fir forests the dependence of years of widespread fire on exceptionally dry conditions that may only occur a few times per century reduces the likelihood that fires set by Native Americans could have had a major influence on the structure of this landscape. In contrast, at the ecotone of ponderosa pine forests and the Plains-grasslands, fuel dessication is sufficient in most summers so that anthropogenic ignitions are more likely to have spread and to have burned significant areas. Potentially, comparative studies of areas with and without archeological evidence of human occupation can detect past effects of Native Americans on fire frequency and seasonality through fire-scar studies (e.g., Kaye and Swetnam 1999). However, determination of a detectable human influence on fire frequency does not directly address the larger question of how significantly landscapes were modified by burning by Native Americans. Multiple lines of evidence, including reconstructions of vegetation from fossil pollen and historical observations of early explorers, may improve our understanding of this issue, but it is likely to remain highly controversial.

3. How did native and introduced herbivores affect fuels and fire regimes?

It is widely recognized that in some ecosystems, such as in Southwest ponderosa pine ecosystems, fire occurrence declined with the introduction of sheep and cattle, which must have reduced grass fuels (Swetnam and Baisan 1996). However, variations in the populations of native herbivores such as bison, deer and elk due to Native American hunting or natural causes potentially had significant impacts on quantity and type of fuels in some ecosystem types. Browsing and grazing by large herbivores can either increase or decrease the success of tree establishment, depending on the tree species and competing shrub and herbaceous species. Early predator control efforts in some areas may have resulted in irruptions of populations of large herbivores that changed vegetation composition and structure early in the 20th century. These potential influences of fluctuating populations of large herbivores on fine fuels and stand structures have received relatively little research attention (but see Fulé et al. 2002).

4. How have invasive plant species altered fire regimes?

Exotic plant species potentially can change the fire regime by changing fuel continuity. Cheatgrass (*Bromus tectorum*) is an introduced grass that has increased the fire hazard over millions of acres in the western United States (Menakis et al. 2003). Thus, restoration of fire to some ecosystem types needs to take into account that the potential for fire spread and intensity has been significantly altered by such fuel changes.

5. What was the spatial variability of the fire regime within a particular ecosystem type?

As stated previously, different locations of the same forest ecosystem type have had different historic fire regimes for a variety of reasons: subtle differences in climatic seasonality, lightning patterns, understory characterisitics, site productivity (related to geology, soils, and / or climate), and potentially use by Native Americans. Such factors constitute the geographic context for particular ecosystems of a given cover type, such as the ponderosa pine cover type. Geographical context is likely to account for some of the differences reported for fire parameters like frequency and severity within the ponderosa pine cover type. Forest cover types determined solely by the physiognomic dominant (such as ponderosa pine) are too broadly defined to expect them to have uniform fire regimes. Forest cover types with broad geographical distributions are likely to exhibit significant differences in historic fire regimes due primarily to regional scale climatic variation. At a local scale, spatial variability is also important within the same cover type as previously illustrated by the spatial variability within the ponderosa pine cover of the northern Front Range (Veblen et al. 2000). Spatial variation at both local and broad scales needs to be better understood to avoid over generalizations about fire regimes at the level of forest cover types.

Methodological Issues

Most methodological discussions of fire history techniques have focused on the description of fire regimes from the basic descriptors of fire frequency and area burned or their analytical derivatives such as mean fire interval or fire rotation (e.g., Arno and Peterson 1983; Johnson and Gutsell 1994; Baker and Ehle 2001). However, ecological understanding of the effects of past fires requires a much more comprehensive description of a fire regime including spatial pattern, severity, effects on tree demography, and interactions with other disturbances (table 2). For modern fires a wide variety of techniques and data sources (e.g., maps of the pre-burn vegetation, field sampling, remote sensing and monitoring) can be used to obtain comprehensive descriptions of the basic descriptors of fire events and their ecological effects. In contrast, in retrospective studies of fire regimes, quantification of the descriptors is unlikely to be completely accurate.

Table 2—Some basic descriptors of fire regimes of potential use in historic fire regime studies.

Descriptor	Definition and comments
Fire frequency	This is the number of fires per unit time in some designated area (Romme 1980).
Fire area	The surface area burned by each fire. Spatial variability of severity within the burn perimeter is often difficult to determine in retrospective studies, especially those based primarily on fire scars. Often perimeters are assigned to fire areas even though it is known that some undetermined amount of the area within the perimeter did not burn.
Fire spatial pattern	This is a description of the spatial pattern of the area burned in relation to the spatial heterogeneity of the abiotic (slope, aspect, elevation) and biotic environment (species composition, stand structure, stand age).
Severity	Severity is usually measured as the amount of damage done by the fire (e.g., tree basal area killed, height of scorching) but in some situations responses to the post-fire environment may indicate severity (e.g., tree -growth releases, amount of post-fire tree establishment, sedimentary records of hydrological and depositional changes).
Fire effects on tree demography	This includes changes in tree establishment or mortality rates that can be linked to the fire.
Interactions with other disturbances	This includes the timing and severity of other disturbance events such as insect outbreaks, pathogen attacks, and wind throw that may either influence or be influenced by the fire event.

Fire history in forested areas can be described quantitatively on the basis of two principal types of tree-ring evidence: dates of fire scars (fire-interval approach) or age of stands that presumably regenerated following stand-replacing fires (stand-origin approach). The fire-scar based approach usually provides annual (or even seasonal) resolution of the dating of past fire events but is limited in its ability to determine the spatial extent of past fires. Two important limitations of the fire-scar based method that have long been recognized are the lack of scar evidence of some fires and the disappearance of fire-scar evidence due to tree death and gradual decay or the consumption of fire-scarred trees in more recent stand-replacing fires (Arno and Sneck 1977; McBride 1983; Agee 1993). Because not all fires leave scars, fire-scar evidence should be regarded as an *index* of past fire occurrence rather than as a complete record of past fire. Absence of the evidence (the fire scar) is not necessarily evidence of absence of the event (the fire). Fire scars as an index of past fire occurrence may not record fires under certain circumstances (e.g., when trees are not the appropriate species or size to record a scar or when subsequent fires destroy the evidence of earlier fires). Furthermore, the locations of fire-scarred trees may not be representative of the unsampled landscape, and subjective (targeted) selection of sample trees may be a source of bias. Opinions vary widely about the importance of these limitations, how to best sample the landscape for fire-scarred trees, and how to compute and interpret the quantitative information from fire-scar dates (Johnson and Gutsell 1994; Swetnam and Baisan 1996; Lertzman et al. 1998; Baker and Ehle 2001).

In more mesic forests where crown fires are common and major episodes of tree establishment typically follow fire, fire history studies are based on the dating and mapping of stand origins (Johnson and Gutsell 1994). There are numerous potential sources of error in this approach as well. These include the difficulty of identifying the oldest trees in post-fire cohorts and of precisely determining tree germination dates (Kipfmueller and Baker 1998). The occurrence of time lags of variable duration between the fire event and tree establishment may be a major source of error in dating fires, particularly if fire scars do not clearly narrow down the range of possible dates. Recognition of post-fire cohorts is sometimes difficult when the same patch has been affected by multiple burns that each kill only part of the tree population and create several post-fire cohorts. In some cases it may be difficult to distinguish between post-fire cohorts and tree establishment following other disturbances (blowdown, insect outbreaks) or the influences of climatic variation on tree demography. One of the most intractable problems is the "overburn problem." The stand-origin method is based on the observation that fires are stand-replacing, which means that part or all of the evidence of previous fires may be destroyed by more recent burns. The determination of areas burned by previous fires is imprecise because decisions must be made about how to draw the perimeters of earlier fires based on often extremely fragmentary evidence or subjective estimates of past fire spread. The difficulty of determining past fire perimeters varies widely from event to event. For example, the perimeter of the most recent crown fire usually can be reliably estimated but the former perimeter of a centuries-old burn which has been partially burned over by several subsequent burns may be impossible to determine accurately. The most common summary statistic used in stand-origin studies is fire rotation, which is the time required to burn the entire study area (Romme 1980). Since fire rotation requires accurate measurement of past fire areas, the rotation statistic may be seriously inaccurate.

Fire regime researchers take many measures to assure that they properly recognize the field evidence of past fires, sample them effectively, and interpret them appropriately. Nevertheless, in most fire regime studies there is uncertainty about the accuracy and completeness of the fire regime reconstruction. Furthermore, there is substantial difference of opinion about the appropriateness and utility of different summary statistics of fire regimes (Johnson and Gutsell 1994; Swetnam and Baisan 1996; Huggard and Arsenault 1999; Minnich et al. 2000; Baker and Ehle 2001). In the current volume, Falk and Swetnam (2003) explore the spatial scaling dependency of fire frequency distributions with the aim of developing scaling rules for high frequency fire regimes. Baker and Ehle (2003) examine some of the uncertainties of estimating mean fire intervals in ponderosa pine ecosystems, and conclude that past studies have over-estimated fire occurrence in these systems. MacLean and Cleland (2003) demonstrate the application of geostatistical procedures to historical land survey data to better estimate the spatial extent of fires from surveyors' notes. Although such approaches are important to pursue, they are unlikely to completely remove the uncertainties in reconstructions of historic fire regimes and their effects on forest conditions.

Given the limitations of both fire-scar and stand-origin methods of describing fire regimes, and the uncertainty that either improved sampling procedures or analytical techniques can remedy these problems, I propose some recommendations to fire regime researchers:

1. *Clarify objectives and assess reliability of methods.* For example, a fire-history study based solely on fire-scar data can produce valid and useful analyses of temporal trends of fire occurrence in relation to land use or climatic variation, but will usually not yield a comprehensive description of the fire regime. If the goal is to assess past fire severity and effects on tree demography, then evidence of tree age population structures, tree mortality, and / or tree growth releases are also necessary.
2. *Use multiple lines of evidence to interpret past fire regimes.* Evidence of past fires should be collected from as many different sources as possible. Whenever feasible, tree-ring evidence should be complemented by written sources (e.g., General Land Office surveys and other landscape descriptions) and historical photographs.
3. *Researchers should present their reconstructions of past fire regimes and stand conditions as estimates.* Particularly due to the problem of disappearing evidence it is unlikely that all fires will be recorded over time periods of many centuries. This is less of a problem in fire regimes of exclusively non-lethal surface fires, but is a major problem in mixed-severity and crown fire systems due to destruction of evidence by the more recent stand-replacing fires. Ranges of parameters should be given based on alternative interpretations of the accuracy and precision of the data.
4. *Researchers should not overemphasize summary statistics such as mean fire intervals or fire rotation.* Mean fire intervals (both composite and individual tree intervals) have an uncertain ecological meaning. To place too much emphasis on the statistical significance of differences in mean fire intervals is dangerous because of the probable inaccuracy of recording all fire events. Likewise, fire rotation is unlikely to be measured accurately because of the difficulty of measuring the perimeters of past fires from either fire scars or from the fragments of post-fire cohorts that may have survived more recent fires.
5. *Researchers need to report full descriptive data and ranges of estimated parameters to the resource management community.* Restrictions on publication space in peer-reviewed journals often allow presentation of only concise summary statistics to describe fire regimes. Researchers should make

available to managers full descriptions of the unreduced data sets. For example, reports to management agencies should include full fire history charts including all sample trees rather than composite lines that summarize the data from numerous trees. Likewise, reports should include complete stand-origin maps and detailed accounts of the procedures used for reconstructing the perimeters of past fires.

For managers concerned with ecological restoration I make the following recommendations about using the results of fire regime studies:

1. *Do not adopt summary statistics such as the mean fire interval as management goals.* These are rarely accurate enough to justify mimicking them, and in addition, changes in other variables (e.g., climate, herbivores, invasive plant species) may make them inappropriate as management prescriptions. Instead of attempting to mimic a potentially inaccurate summary statistic of a fire regime (i.e., mean fire interval or rotation period), managers should consider multiple lines of evidence (e.g., tree population age structures, historical photographs, repeated twentieth century inventories or surveys) that help the manager to identify trends or trajectories in ecosystem conditions that may be related to changes in fire regimes. Quantitative fire history data are vitally important to management decisions but they need to be presented in their entirety (e.g., as complete charts of fire-scar chronologies for individual trees and as stand-origin maps) rather than as mean fire intervals and rotation periods. The manager should be able to see the trends and management implications in the more basic data compilations instead of relying on summary statistics that often have inherent limitations due to incomplete preservation of the evidence of past fires.
2. *Define goals in terms of ranges of desired vegetation conditions.* Precise and accurate descriptors of the vegetation conditions are unlikely to be obtained. More broadly defined ranges of desired management conditions is consistent both with the uncertainties of reconstructing past conditions from fragmentary evidence and with the notion that ecological heterogeneity is often more desirable than homogeneity. Managing for a broader range of conditions builds some buffering into the management plan to account for surprises such as mortality events caused by insect outbreaks. In most cases, greater heterogeneity resulting from a range of management prescriptions is likely to contribute to management success.
3. *Require area-specific data and analyses to support management decisions.* Studies conducted elsewhere rarely yield results or a model of fire regime and past stand conditions that can be uncritically applied to an unstudied situation. This is true even for the same forest cover type. Thus, the findings from studies done off site may at best be used as insights into the formulation of hypotheses to be tested by data collection and analysis in new study areas or management units.
4. *Use adaptive management and monitoring to assess management success.* Current knowledge of ecosystem dynamics is incomplete and may change in ways that are important for the goals of ecological restoration. Management goals and strategies should be regarded as hypotheses to be tested by future research and monitoring (Christensen et al. 1996). This requires continued communication between managers and researchers. Managers need to be able to adapt to inevitable surprises and trends, such as unpredicted diseases and forest pest outbreaks as well as climatic variation. Adaptability and accountability require that a high funding priority be given to monitoring programs that compare expected outcomes with objective measures of results.

Conclusions

This essay was written to introduce some of the key limitations and issues in fire regime research in the context of wildfire hazard management and ecological restoration. One of the principal messages is that broad generalizations and premises need to be carefully examined for particular ecosystems and management objectives. The goal of the essay has not been to challenge the widespread consensus that fire exclusion has had undesirable consequences in many western forest ecosystems. Rather, it is hoped that critical evaluation of the premises of the fire exclusion / fuel buildup viewpoint for particular ecosystem types and locations will help to avoid inappropriate or ineffective management strategies. Forest ecosystem types with demonstrated historic fire regimes of frequent surface fires and fuel buildup during the fire exclusion period should be targeted for ecological restoration, which may also converge with reduction of fire hazard to property and humans. In contrast, in forest ecosystems characterized by historic fire regimes with long intervals between stand-replacing fires, attempts to create new fire regimes of frequent surface fires are inconsistent with ecological restoration and likely to be futile.

As implied by the subtitle of the Conference, fuels management and ecological restoration need to be attentive to “proper place” and “appropriate time.” Fire regime research can inform management decisions about the proper place and time for fuels management and restoration of fire to ecosystems. Fire regime research continues to inform management decisions in useful and important ways, but the quantity and quality of this research needs to be improved. Clearly there is a need for greater involvement of fire regime researchers in the early phases of project planning and continued communication between researchers and managers in the monitoring phases of restoration and fire mitigation projects.

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Literature Cited

Agee, J.K. 1993. Fire ecology of Pacific Northwest forests. Washington, DC: Island Press.

Arno, S.F., and K.M. Sneck. 1977. A method for determining fire history in coniferous forests of the mountain west. Gen. Tech. Rep. INT-42. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 28 p.

Arno, S.F. and T.D. Peterson. 1983. Variation in estimates of fire intervals: a closer look at fire history on the Bitterroot National Forest. Res. Pap. INT-301. U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 8 p.

Baker, W.L. and D. Ehle. 2001. Uncertainty in surface-fire history: The case of ponderosa pine forests in the Western United States. Canadian Journal of Forest Research. 31: 1205-1226.

Baker, W.L. and D. Ehle. 2003. Uncertainty in surface-fire history and restoration of ponderosa pine forests in the western United States. This volume.

Baker, W.L. and T.T. Veblen. 1990. Spruce beetles and fires in the nineteenth century subalpine forests of western Colorado. Arctic and Alpine Research. 22: 65-80.

Brown, P.M., D.R. D'Amico, A.T. Carpenter, and D.M. Andrews. 2001. Restoration of montane ponderosa pine forests in the Colorado Front Range: A forest ecosystem management plan for the City of Boulder. Ecological Restoration. 19: 19-26.

Brown, P.M., M.R. Kaufmann, and W.D. Shepperd. 1999. Long-term landscape patterns of past fire events in a montane ponderosa pine forest of central Colorado. Landscape Ecology. 14: 513-532.

Christensen, N.L., A.M. Bartuska, J.H. Brown, S. Carpenter, C. D'Antonio, R. Francis, J.F. Franklin, J.A. MacMahon, R.F. Noss, D.J. Parsons, C.H. Peterson, M.G. Turner, and R.G. Woodmansee. 1996. The report of the Ecological Society of America committee on the scientific basis for ecosystem management. Ecological Applications. 6: 665-691.

City of Boulder. 1999. Forest ecosystem management plan, Open Space and Mountain Parks. City of Boulder, Colorado.

Covington, W.W. 2000. Helping western forests heal: the prognosis is poor for US forest ecosystems. Nature. 408: 135-136.

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

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

Denevan, W.M. 1992. The native population of the Americas in 1492. 2nd edition. Madison, WI: University of Wisconsin Press.

Donnegan, J., T.T. Veblen, and J.S. Sibold. 2001. Climatic and human influences on fire history in Pike National Forest, central Colorado. Canadian Journal of Forest Research. 31: 1526-1539.

Ehle, D.S. 2001. Spatial and temporal patterns of disturbance and ponderosa pine forest structure in Rocky Mountain National Park. Laramie, WY: University of Wyoming. Thesis.

Falk, D.A. and T. W. Swetnam. 2003. Scaling rules and probability models for ponderosa pine fire regimes. This volume.

Fulé, P.Z., W.W. Covington, and M.M. Moore. 1997. Determining reference conditions for ecosystem management of Southwestern pondersosa pine forests. Ecological Applications. 7: 895-908.

Fulé, P.Z., W.W. Covington, M.M. Moore, T.A. Heinlein, and A.E.M. Waltz. 2002. Natural variability in forests of the Grand Canyon, USA. Journal of Biogeography. 29: 31-47.

Grissino-Mayer, H.D., and T.W. Swetnam. 2000. Century-scale climate forcing of fire regimes in the American Southwest. Holocene. 10: 213-220.

Hof, J. and P. Omi. 2003. Operations research approaches to scheduling removals for fuels management. This volume.

Huckaby, L.S., M.R. Kaufmann, J.M. Stoker, and P.J. Fornwalt. 2001. Landscape patterns of montane forest age structure relative to fire history at Cheesman Lake in the Colorado Front Range. In: Vance, R.K.; Covington, W.W.; Edminster, C.B.; Blake, J.A., tech. coords. Ponderosa pine ecosystems restoration and conservation:

steps toward stewardship; 2000 April 25-27; Flagstaff, AZ. Proc. RMRS-P-22. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 19-27.

Huggard, D.J. and A. Arsenault. 1999. Comment—Reverse cumulative standing age distributions in fire-frequency analysis. Canadian Journal for Forest Research. 2: 1449-1456.

Johnson, E. A. and S. L. Gutsell. 1994. Fire frequency models, methods and interpretations. Advances in Ecological Research. 25: 239-287.

Johnson, E.A., K. Miyanishi, and S.R.J. Bridge. 2001. Wildfire regime in the boreal forest and the idea of suppression and fuel buildup. Conservation Biology. 15: 1554-1557.

Kaufmann, M.R., C.M. Regan, and P.M. Brown. 2000. Heterogeneity in ponderosa pine / Douglas-fir forests: Age and size structure in unlogged and logged landscapes of central Colorado. Canadian Journal of Forest Research. 30: 698-711.

Kaufmann, M.R., P.J. Fornwalt, L.S. Huckaby, and J.M. Stoker. 2001. Cheesman Lake– A historical ponderosa pine landscape guiding restoration in the South Platte watershed of the Colorado Front Range. In: Vance, R.K.; Covington, W.W.; Edminster, C.B.; Blake, J.A., tech. coords. Ponderosa pine ecosystems restoration and conservation: steps toward stewardship; 2000 April 25-27; Flagstaff, AZ. Proc. RMRS-P-22. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 9-18.

Kaye, M. and T.W. Swetnam. 1999. An assessment of fire, climate and Apache history in the Sacramento Mountains, New Mexico. Physical Geography. 20: 305-330.

Keeley, J.E. and C.J. Fotheringham. 2001. Historic fire regime in southern California shrublands. Conservation Biology. 15: 1536-1548.

Kipfmueller, K.F. and W.L. Baker. 1998. A comparison of three techniques to date stand-replacing fires in lodgepole pine forests. Forest Ecology and Management. 104: 171-177.

Kitzberger, T., T.W. Swetnam, and T.T. Veblen. 2001. Inter-hemispheric synchrony of forest fires and the El Niño-Southern Oscillation. Global Ecology and Biogeography. 10: 315-326.

Kulakowski, D. 2002. Interactions among natural disturbances in subalpine forests in northwestern Colorado. Boulder, CO: University of Colorado. Ph.D. Dissertation.

Lertzman, K., J. Fall and B. Dorner. 1998. Three kinds of heterogeneity in fire regimes: At the crossroads of fire history and landscape ecology. Northwest Science. 72: 4-23.

Maclean, A.L., and D.T. Cleland. 2003. The use of geostatistics to determine the spatial extent of historical fires as an aid in understanding fire regimes for northern lower Michigan. This volume.

Mackes, K.H. and D.L. Lynch. 2003. Evaluating costs associated with fuel hazard reduction and forest restoration project in Colorado. This volume.

Mast, J.N., P.Z. Fule, M.M. Moore, W.W. Covington, and A. Waltz. 1999. Restoration of presettlement age structure of an Arizona ponderosa pine forest. Ecological Applications. 9: 228-239.

Mast, J.N., T.T. Veblen and Y.B. Linhart. 1998. Disturbance and climatic influences on age structure of ponderosa pine at the pine / grassland ecotone, Colorado Front Range. Journal of Biogeography. 25: 743-767.

McBride, J.R. 1983. Analysis of tree rings and fire scars to establish fire history. Tree Ring Bulletin. 43: 51-67.

Menakis, J.P., D. Osborne, and M. Miller. 2003. Mapping departure from historic natural fire regimes caused by cheatgrass in the Great Basin. This volume.

Minnich, R.A., M.G. Barbour, J.H. Bruk, and J. Sosa-Ramírez. 2000. Californian mixed-conifer forests under unmanaged fire regimes in the Sierra San Pedro Mártir, Baja California, Mexico. Journal of Biogeography. 27: 105-129.

Moore, M.M., W.W. Covington, and P.Z. Fulé. 1999. Reference conditions and ecological restoration: a southwestern ponderosa pine perspective. Ecological Applications. 9: 1266-1277.

Romme, W.H. 1980. Fire history terminology: report of the Ad Hoc Committee. In: Stokes, M.A.; Dieterich, J.H., eds. Proceedings of the fire history workshop, 1980 October 20-24; Tucson, AZ. Gen. Tech. Rep. RM-81. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 13-137.

Romme, W.H., M.L. Floyd-Hanna, D.D. Hanna, and P. Kemp. 2003. Experimental thinning and burning of ponderosa pine forests in southwestern Colorado: effects on canopy structure, understory composition, and fuels. This volume.

Sibold, J. 2001. The forest fire regime of an upper montane and subalpine forest, Wild Basin, Rocky Mountain National Park. Boulder, CO: University of Colorado. Masters Thesis.

Stein, T. 2002. Blazes ignite calls for forest thinning. Denver Post, June 17: 14A.

Swetnam, T.W. and C.H. Baisan. 1996. Historical fire regime patterns in the southwestern United States since AD 1700. In: Allen, C.D., ed. Fire effects in southwestern forests; Proceedings of the second La Mesa fire symposium; 1994 March 29-31; Los Alamos, NM. Gen. Tech. Rep. RM-286. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 11-32

Swetnam, T.W. and J.L. Betancourt.1998. Mesoscale disturbance and ecological response to decadal variability in the American Southwest. Journal of Climate. 11: 3128-3147.

Swetnam, T.W., C.D. Allen, and J.L. Betancourt. 1999. Applied historical ecology: Using the past to manage for the future. Ecological Applications. 9: 1189-1206.

Vale, T.R. 2002. Fire, native peoples, and the natural landscape. Washington, DC: Island Press.

Veblen, T.T. 2000. Disturbance patterns in southern Rocky Mountain forests. In: Knight, R.L.; Smith, F.W.; Buskirk, S.W.; Romme, W.H.; Baker, W.L., eds. Forest fragmentation in the Southern Rocky Mountains. Boulder, CO: Colorado University Press: 31-54

Veblen, T.T., K.S. Hadley, M.S. Reid, and A.J. Rebertus. 1991. Stand response to spruce beetle outbreak in Colorado subalpine forests. Ecology. 72: 213-231.

Veblen, T.T. and T. Kitzberger. 2002. Inter-hemispheric comparison of fire history: The Colorado Front Range, U.S.A. and the Northern Patagonian Andes, Argentina. Plant Ecology. 163: 187-207.

Veblen, T.T., T. Kitzberger and J. Donnegan. 2000. Climatic and human influences on fire regimes in ponderosa pine forests in the Colorado Front Range. Ecological Applications. 10: 1178-1195.

Veblen, T.T. and D.C. Lorenz. 1986. Anthropogenic disturbance and recovery patterns in montane forests, Colorado Front Range. Physical Geography. 7: 1-24.

Veblen, T.T. and D.C. Lorenz. 1991. The Colorado Front Range: a century of ecological change. Salt Lake City, UT: University of Utah Press.

Lessons From the Fires of 2000: Post-Fire Heterogeneity in Ponderosa Pine Forests

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Abstract—*We evaluate burn-severity patterns for six burns that occurred in the southern Rocky Mountains and the Colorado Plateau in 2000. We compare the results of two data sources: Burned Area Rehabilitations Teams (BAER) and a spatial burn-severity model derived from satellite imagery (the Normalized Burn Ratio; NBR). BAER maps tended to overestimate area of severe burns and underestimate area of moderate-severity burns relative to NBR maps. Low elevation and more southern ponderosa pine burns were predominantly understory burns, whereas burns at higher elevations and farther north had a greater component of high-severity burns. Thus, much, if not most, of the area covered by these burns appears to be consistent with historic burns and contributes to healthy functioning ecosystems.*

Concern that the size and severity of the 2000 fires were "beyond the range of natural variability" in ponderosa pine (*Pinus ponderosa*) and mixed conifer systems has provided justification for ecological restoration programs. However, little is known about the spatial heterogeneity resulting from recent or pre-historic fires. Here we evaluate the effects of burn-severity patterns on landscape heterogeneity for burns that occurred in the southern Rocky Mountains and the Colorado Plateau in 2000. We compare the results of two data sources: Burned Area Rehabilitation Teams (BAER) and a spatial burn-severity model derived from satellite imagery.

Burn-severity maps were developed using the Normalized Burn Ratio (NBR). NBR is derived from comparisons of pre- and post-fire Thematic Mapper imagery (30 m resolution). Band 4 (near infrared) reflects changes in vegetation greenness and soil moisture, whereas band 7 (mid infrared) reflects soil type and moisture levels. Band 4 tends to decrease post-fire, whereas band 7 tends to increase post fire. NBR is based on the inverse relationship between bands 4 and 7: NBR = (band 4 – band 7)/(band 4 + 7). Delta NBR values are derived from differences in pre- and post-fire NBR scores, which in turn can be used as an index of burn severity (higher delta NBR indicates higher burn severity). The NBR methodology was developed to be repeatable and quantifiable, and it offers several advantages for quantifying burn severity compared to BAER maps, which are designed for rapid assessment and targeting of high-severity burns for rehabilitation (table 1).

We created burn severity maps using delta NBR for six burns (figure 1): Bobcat Gulch and Hi Meadow in Colorado, Viveash and Cerro Grande in New Mexico, and Outlet and Pumpkin in Arizona. Burns in general were dominated by ponderosa pine (*Pinus ponderosa*), but Douglas-fir (*Pseudotsuga menziesii*), aspen (*Populus tremuloides*), subalpine fir (*Abies lasiocarpa*), and Engelmann spruce (*Picea engelmannii*) were also present at some burns. Subalpine fir and Engelmann spruce were dominant at Viveash and at higher elevations at Outlet.

Preliminary analyses indicate that in general, BAER maps corresponded fairly well to NBR maps at these burns, but there were several discrepancies (table 2). At the southernmost burns (Pumpkin and Cerro Grande), which

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Table 1—Comparison of attributes of six burn-severity maps created by Burned Area Rehabilitation Teams (BAER) and the general Normalized Burn Ratio (NBR) method.

BAER	NBR
Targets areas of high-severity burns for rehabilitation	All burn severities addressed equally
Categorical data (unburned/low, moderate, and high severity)	Continuous data
Burn-severity polygons	Grid of 30 m^2 pixels
Subjectivity involved in boundary definitions and severity classification; variation among burns in methodologies and data sources	Standardized methodology

were dominated by ponderosa pine, at least half of the area was classified as a low-severity burn by NBR. At these burns, BAER maps tended to underestimate areas of moderate-severity burns and overestimate high-severity burns compared to NBR. Outlet (large areas of aspen / spruce / fir forests) and Hi Meadow (upper montane ponderosa pine / Douglas-fir) were still predominated by low-severity burns, but moderate- and high-severity burns comprised a greater proportion of the burns than Pumpkin and Cerro Grande. BAER and NBR were similar at Outlet, but BAER maps indicated greater high-severity and less low-severity burns than NBR at Hi Meadow. Bobcat Gulch was similar to Hi Meadow, but had a fairly even distribution of area across all burn severities. BAER maps indicated greater high-severity and less moderate-severity area burned than NBR maps. Viveash (predominantly spruce / fir) was the most severe of all six burns and was largely a high-severity burn. The area of moderate-severity burn at Viveash was lowest in BAER maps. Visual comparisons of BAER vs. NBR maps indicate that small-scale patchiness is missed by BAER maps, and may reflect the differences observed in the two mapping techniques. The BAER mapping process may underestimate moderate-severity patches in particular, which tend to be relatively small.

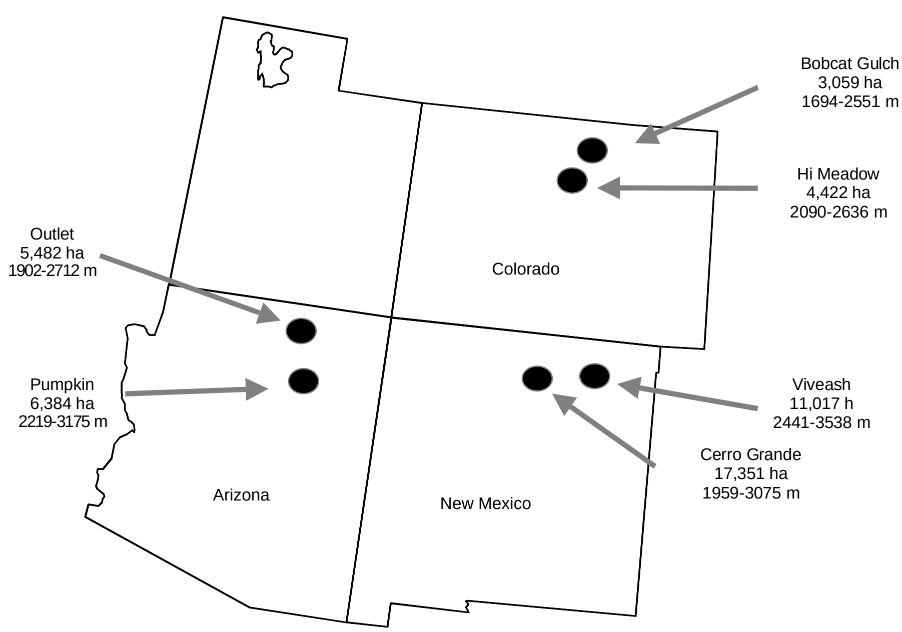

Figure 1—Burns of 2000. Approximate locations and area of study sites.

Table 2—Area (%) of burn severity classes using BAER and NBR methodologies.

		BAER burn severity			NBR burn severity[1]		
Burn	**Dominant cover types**	Low	Moderate	High	Low	Moderate	High
Pumpkin	Ponderosa pine	71%	2%	26%	65%	15%	20%
Cerro Grande	Ponderosa pine	57%	8%	34%	60%	20%	20%
Outlet	Ponderosa pine Spruce/fir	39%	34%	23%	40%	30%	25%
Hi Meadow	Ponderosa pine/Douglas-fir	2%	53%	45%	45%	35%	20%
Bobcat Gulch	Ponderosa pine/ Douglas-fir	30%	25%	45%	33%	33%	33%
Viveash	Spruce/fir	41%	11%	48%	15%	20%	65%

[1] Because NBR burn-severity values are continuous, we estimated the approximate area in each burn-severity category to compare to BAER categorical data.

Pronounced overall differences were observed among the six burns, which likely reflects variation in cover types, fuels, historic fire regimes, elevation, and topography. Lower elevation and southern sites (e.g., Pumpkin and Cerro Grande) had the greatest area of low-severity burns compared to higher elevation (e.g., Viveash) and northern sites (e.g., Bobcat Gulch and Hi Meadow). Although it has repeatedly been suggested that recent burns are beyond the range of natural variation, much, if not most, of the area covered by the burns are consistent with our current understanding of historic fire regimes. This is especially true of the burns occurring in more northern latitudes and higher elevation (Brown et al. 1999; Veblen et al. 2001).

It is important to recognize that although portions of some burns (e.g., Cerro Grande and Pumpkin) may have burned hotter than most fires in southwestern ponderosa pine forests prior to Euro-American settlement (Allen et al. 2002), large portions of these burns did not. Indeed, the area of these burns in low and moderate severity classes performed the desired functions of many prescribed fires. Furthermore, the broad spectrum of burn severity observed across the six burns is important for the integrity of ecological communities (Kotliar et al. 2002). Thus, taken overall, these burns were not as extreme or destructive as has been frequently suggested. Although restoration of historic forest structure to reduce the risk of wildfire may be justified in the wildland-urban interface, the premise that current wildland fires are beyond the range of natural variability and need to be controlled may not always be valid.

References

Allen, C.D.; Savage, M.; Falk, D.A.; Suckling, K.F.; Swetnam, T.W.; Sculke, T.; Stacey, P.B.; Morgan, P.; Hoffman, M.; Klingel, J.T. 2002. Ecological restoration of southwestern ponderosa pine ecosystems: a broad perspective. Ecological Applications. 12: 1418-1433.

Brown, P.M.; Kaufmann, M.R.; Shepperd, W.D. 1999. Long-term, landscape patterns of past fire events in a montane ponderosa pine forest of central Colorado. Landscape Ecology. 14: 513-532.

Kotliar, N.B.; Hejl, S.; Hutto, R.L.; Saab, V.A.; Melcher, C.P.; McFadzen, M.E. 2002. Effects of fire and post-fire salvage logging on avian communities in conifer-dominated forests of the western United States. Studies in Avian Biology. 25:49-64.

Veblen, T.T.; Kitzberger, T.; Donnegan, J. 2000. Climatic and human influence on fire regimes in ponderosa pine forests in the Colorado Front Range. Ecological Applications. 10: 1178-1195.

Mapping the Cheatgrass-Caused Departure From Historical Natural Fire Regimes in the Great Basin, USA

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Abstract—*Cheatgrass (*Bromus tectorum*) is an exotic grass that has increased fire hazard on millions of square kilometers of semi-arid rangelands in the western United States. Cheatgrass aggressively outcompetes native vegetation after fire and significantly enhances fire size and frequency. To evaluate the effect of cheatgrass on historical natural fire regimes, we combined cheatgrass data mapped from Advanced Very High Resolution Radiometer images of the Great Basin with Fire Regime Condition Class (FRCC) data mapped from plant succession data incorporated with several spatial data layers for the conterminous United States. These FRCCs depict the degree of departure from historical fire regimes resulting in alterations of key ecosystem components such as species composition, structural stage, stand age, and canopy closure. While the FRCC data layer adequately depicted forest communities, it insufficiently depicted grassland and shrubland communities. By adding cheatgrass, FRCC 3 (areas that have been significantly altered from their historical range) increased by 20 percent on Federal lands to almost 60,522 square kilometers for the conterminous United States.*

Introduction

In April 1999, the General Accounting Office (GAO) published a report recommending that the Secretary of Agriculture direct the Chief of the Forest Service to develop a cohesive strategy for reducing accumulated vegetation and maintaining it at acceptable levels on National Forests of the Interior West (US GAO 1999). In October 2000, the United States Department of Agriculture (USDA) Forest Service (FS) responded with the report, "Protecting People and Sustaining Resources in Fire-adapted Ecosystems: a Cohesive Strategy" (USDA FS 2000). This FS report establishes a framework that restores and maintains ecosystem health in fire adaptive ecosystems by directing the agency to:

- improve the resilience and sustainability of forests and grasslands at risk,
- conserve priority watersheds, species, and biodiversity,
- reduce wildland fire costs, losses, and damages, and
- better ensure public and firefighter safety.

To assist in the FS response to the GAO report, the Fire Modeling Institute (FMI) at the USDA FS Fire Science Laboratory, Rocky Mountain Research Station, Missoula, Montana, created spatial data layers to provide managers with national-level data on current conditions of vegetation and fuels (Schmidt and others 2002). FMI developed these spatial data layers, hereafter referred to as layers, to address the following questions (Schmidt and others 2002):

- How do current vegetation and fuels differ from those that existed historically?

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- Where on the landscape do vegetation and fuels differ from historical levels? In particular, where are high fuel accumulations?
- When considered at a coarse scale, which areas estimated to have high fuel accumulations represent the highest priorities for treatment?

They created these layers from ecologically based methods to map vegetation changes resulting from the departure of historical natural fire regimes (Hardy and others 2001). These layers have been subsequently used in a joint cohesive fuels management strategy developed by the FS and those Department of Interior (DOI) agencies with wildland fire management responsibilities. In this document, we will refer to these layers as the "Coarse Scale."

One of the key Coarse Scale layers used in both cohesive strategies was Fire Regime Condition Class (FRCC). FRCC depicts the degree of departure from historical fire regimes resulting in alterations of key ecosystem components such as species composition, structural stage, stand age, and canopy closure (table 1) (Schmidt and others 2002). During the development of the joint Cohesive Strategy, it appeared to rangeland managers that the Coarse Scale layers potentially underestimated the amount of area departed from historical fire regimes in grassland and shrubland communities. This appeared to be especially true in the Great Basin, where the exotic species cheatgrass (*Bromus tectorum*) has become widespread.

Cheatgrass is an exotic annual grass that has increased fire hazard on millions of square kilometers in the western United States (USDA FS 2002). Cheatgrass aggressively outcompetes seedlings of native vegetation after fire, particularly in semi-arid rangelands of the Interior West. The fuel bed created by cheatgrass results in significantly increased fire size and frequency, compared to the native shrub / grass vegetation that was historically present on

Table 1—Fire Regime Condition Class[a] descriptions.

Fire Regime Condition Class	Fire regime	Example management options
FRCC 1	Fire regimes are within an historical range and the risk of losing key ecosystem components is low. Vegetation attributes (species composition and structure) are intact and functioning within an historical range.	Where appropriate, these areas can be maintained within the historical fire regime by treatments such as fire use.
FRCC 2	Fire regimes have been moderately altered from their historical range. The risk of losing key ecosystem components is moderate. Fire frequencies have departed from historical frequencies by one or more return intervals (either increased or decreased). This results in moderate changes to one or more of the following: fire size, intensity and severity, and landscape patterns. Vegetation attributes have been moderately altered from their historical range.	Where appropriate, these areas may need moderate levels of restoration treatments, such as fire use and hand or mechanical treatments, to be restored to the historical fire regime.
FRCC 3	Fire regimes have been significantly altered from their historical range. The risk of losing key ecosystem components is high. Fire frequencies have departed from historical frequencies by multiple return intervals. This results in dramatic changes to one or more of the following: fire size, intensity, severity, and landscape patterns. Vegetation attributes have been significantly altered from their historical range.	Where appropriate, these areas may need high levels of restoration treatments, such as hand or mechanical treatments, before fire can be used to restore the historical fire regime.

[a]Fire Regime Condition Classes (FRCC) are a qualitative measure describing the degree of departure from historical fire regimes, possibly resulting in alterations of key ecosystem components such as species composition, structural stage, stand age, canopy closure, and fuel loadings. One or more of the following activities may have caused this departure: fire suppression, timber harvesting, livestock grazing, introduction and establishment of exotic plant species, introduced insects or disease, or other management activities.

these sites (USDA FS 2002). These repeated fires kill the remaining native plants (Monsen 1994). Since FRCC depicts departure from historical fire regimes, a change in vegetation from shrub / grass to one dominated by cheatgrass would change the assignment from FRCC 1 (fire regimes are within their historical range) to FRCC 3 (fire regimes have been significantly altered from their historical range, and the risk of losing key ecosystem components from fire is high). See table 1 for a full definition of FRCC.

In this paper, we evaluate whether the extent of FRCC 3 was underestimated in the cheatgrass type. We conducted the evaluation by combining a cheatgrass layer developed after the Coarse Scale project to several of the original Coarse Scale layers.

Methods

To map the effect of cheatgrass on historical natural fire regimes, we integrated several layers from two projects. We obtained a Cheatgrass layer classified from 2000 satellite imagery by the National Science and Technology Center (NSTC) at the Bureau of Land Management, Denver, Colorado, for the Great Basin. NSTC developed this layer as part of their Cheatgrass Community Mapping and Change Detection project. From the Coarse Scale mapping project, we used the Fire Regime Condition Classes, Potential Natural Vegetation Groups, and Current Cover Types version 2000 layers. Since Hardy and others (2000) and Schmidt and others (2002) explained the methods used in developing the Coarse Scale layers, we will not describe them here.

NSTC developed the Cheatgrass layer from a study of 26 scenes of Advanced Very High Resolution Radiometer (AVHRR) satellite images collected from March 3 to June 15, 2000. NSTC examined each image for quality, accuracy, and cloud cover, then created a Normalized Difference Vegetation Index (NDVI) for each scene. Next, NSTC selected two of the NDVI scenes based on comparisons to cheatgrass phenology data. The early spring scene (March 3, 2000) represented the period when cheatgrass greens-up, and the late spring (April 26, 2000) scene represented the period when cheatgrass cures (browns-out). Only areas that mapped both green-up and curing were mapped by NSTC as cheatgrass to create the Cheatgrass layer.

In the GIS, we combined the Cheatgrass layer with the Potential Natural Vegetation Groups and Current Cover Types version 2000 layers. When combined, spatial inconsistencies occurred because the layers came from different sources. We resolved these inconsistencies by excluding the areas mapped in the Cheatgrass layer that did not ecologically match classes in the Potential Natural Vegetation Groups or Current Cover Types layers. Since cheatgrass cannot ecologically occur in high elevations or wet grasslands, we excluded cheatgrass areas that occurred in the following Potential Natural Vegetation Groups classes: Spruce – Fir – Douglas-fir; Western spruce – fir; Lodgepole pine – Subalpine; Wet Grassland; and Alpine Meadows – Barren. We also excluded areas mapped as cheatgrass in the following Current Cover Type classes: Agriculture; Urban / Development / Agriculture; Water; and Barren. These cover types were also not mapped in the FRCC layer.

Finally, we combined the edited version of the Cheatgrass layer with the FRCC layer. Areas where cheatgrass occurred in FRCC layer classes FRCC 1 and FRCC 2 were assigned a new FRCC class called FRCC 3 – Cheatgrass. Areas where cheatgrass occurred in the FRCC layer class FRCC 3 stayed FRCC 3.

Results

The original Cheatgrass layer provided by NSTC mapped 127,396 square kilometers of cheatgrass in the Great Basin. From this, 22 percent (28,067 square kilometers) of the total cheatgrass area was excluded because of inconsistencies with Potential Natural Vegetation Groups and Current Cover Types version 2000 layers. Of this, only 4 percent (1,106 square kilometers) of the excluded cheatgrass area was in FRCC 1, 2, and 3, and the rest was in non-vegetative areas (like agriculture, urban, water, and barren). The final Cheatgrass layer (created for this projects) mapped 99,329 square kilometers of cheatgrass in the Great Basin.

Of the 99,329 square kilometers in the final Cheatgrass layer, 48,247 square kilometers (49 percent) was in the original FRCC 1 and 31,672 square kilometers (32 percent) was in the original FRCC 2. These areas, totaling 79,919 square kilometers (81 percent of the final Cheatgrass layer), were re-assigned to FRCC 3 – Cheatgrass (table 2) from the original coarse scale analysis. Since the rest of the area in the final Cheatgrass layer, 19,410 square kilometers (19 percent), was already in the original FRCC 3, they were kept as FRCC 3 and not reassigned to FRCC 3 – Cheatgrass.

Across the conterminous United States, incorporating cheatgrass into the original FRCC layer increased FRCC 3 by almost 11 percent (from 735,630 square kilometers to 815,549 square kilometers) (table 2). Figure 1 compares the difference in FRCC 3, before and after incorporating cheatgrass into the original FRCC layer for the western United States. Of the total reassigned area of FRCC 3 – Cheatgrass, 76 percent (60,522 square kilometers) occurred on federal ownership, increasing the original FRCC 3 by 20 percent (from 301,892 square kilometers to 362,414 square kilometers) (table 3).

When compared to the coarse scale fire regimes data, 53,516 square kilometers (67 percent) of the reassigned FRCC 3 – Cheatgrass was in the Historical Natural Fire Regime class III (35 – 100+ years; Mixed Severity) (table 2). In a historical natural fire regime with more frequent fires, 19,836 square kilometers (25 percent) of the reassigned FRCC 3 – Cheatgrass was in Historical Natural Fire Regime class I (0 – 35; Low Severity) (table 2).

Table 2—A summary of **all land ownerships** for the conterminous United States of historical natural fire regimes by fire regime condition classes with cheatgrass added. Summary does not include the following cover types: agriculture, barren, water, and urban/development/agriculture.

	Fire Regime Condition Class (FRCC)				
	FRCC 1	FRCC 2	FRCC 3 (without cheatgrass)	FRCC 3 – cheatgrass	
Historical natural fire regime	**Km²**	**Km²**	**Km²**	**Km²**	**Total km²**
I. 0-35 years; low severity	705,430	695,976	313,605	19,836	1,734,847
II. 0-35 years; stand replacement	778,245	537,541	41,870	2,393	1,360,049
III. 35-100+ years; mixed severity	480,779	436,558	218,545	53,516	1,189,398
IV. 35-100+ years; stand replacement	210,708	142,847	141,757	4,174	499,486
V. 200+ years; stand replacement	196,511	55,470	19,853	0	271,834
Total	2,371,673	1,868,392	735,630	79,919	5,055,614

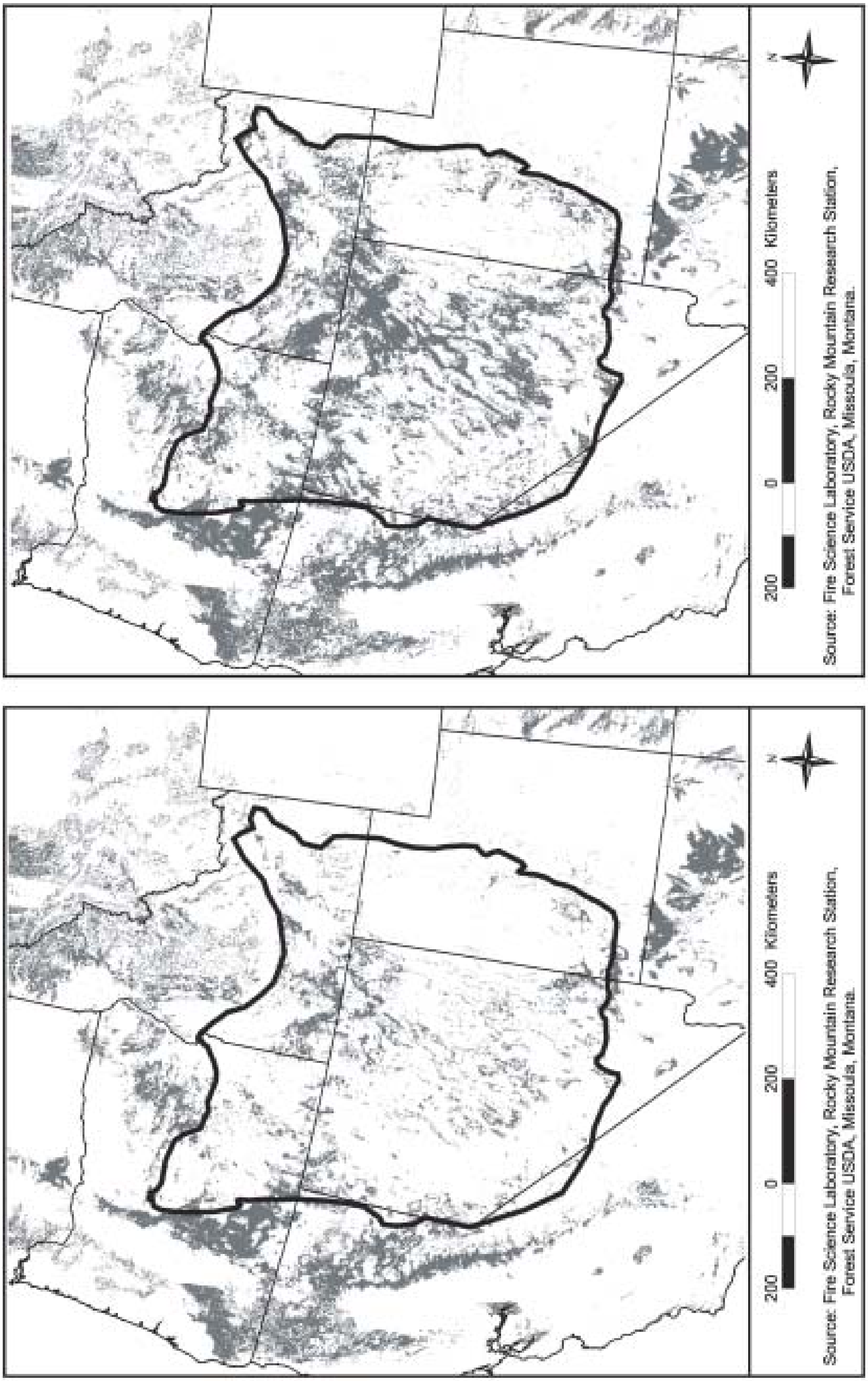

Figure 1—Comparison of Fire Regime Condition Class 3 without Cheatgrass added (left map) to Fire Regime Condition Class 3 with Cheatgrass added (right map). The boundary for the Great Basin is represented by the bold line.

Table 3—A summary of **federal land ownerships** for the conterminous United States of historical natural fire regimes by fire regime condition classes with cheatgrass added. Summary does not include the following cover types: agriculture, barren, water, and urban/development/agriculture.

	Fire Regime Condition Class (FRCC)				
	FRCC 1	FRCC 2	FRCC 3 (without cheatgrass)	FRCC 3 – cheatgrass	
Historical natural fire regime	Km^2	Km^2	Km^2	Km^2	Total km^2
I. 0-35 years; low severity	150,366	227,174	142,835	16,498	536,873
II. 0-35 years; stand replacement	96,230	126,701	2,938	1,475	227,344
III. 35-100+ years; mixed severity	290,685	198,692	85,341	39,284	614,002
IV. 35-100+ years; stand replacement	115,175	41,470	69,196	3,265	229,106
V. 200+ years; stand replacement	95,257	12,710	1,582	0	109,549
Total	747,713	606,747	301,892	60,522	1,716,874

Discussion

By incorporating cheatgrass spatial data for the Great Basin with the Coarse Scale FRCC layer for the nation, areas mapped as FRCC 3 increased by 11 percent. This would strongly suggest that the Coarse Scale FRCC layer underestimated FRCC 3 for rangelands and shrublands in the Great Basin. However, it would be difficult to extrapolate these numbers to the rest of the Continental United States. One could not use this Cheatgrass FRCC layer as a substitute for the Coarse Scale Version 2000 layers, because cheatgrass was only mapped for the Great Basin. The Coarse Scale layers were derived from national level data developed with the same methods throughout the conterminous United States (Schmidt and others 2002). This approach allows for uniform analysis and interpretation.

Unfortunately, we do not know of any national level spatial data that adequately maps exotic grasses. This might be the result of attempting to map these cover types at the wrong scale. Much of the departure from historical fire regimes in rangelands and shrublands involve the encroachment of exotic species into native communities resulting in changes in fire frequency and severity. This encroachment can be difficult to map with coarse scale spatial data (1 square kilometer pixels), because these species rarely dominate a pixel. Mid or fine scale spatial data (30 square meters or less) would probably be more appropriate.

Lastly, no accuracy assessment or field verification of the layers used in this project was conducted. Many authors have documented the difficulty in providing an accuracy assessment for coarse scale projects of 1 square kilometer pixel size or greater (Kloditz and others 1998; Loveland and others 1991; Schmidt and others 2002). This is because ground truth data is difficult and expensive to collect, and would only represent a very small portion of the study area (Schmidt and others 2002).

Management Implications

Cheatgrass has replaced native vegetation and increased fire hazard on millions of square kilometers in the western United States (USDA FS 2002). It has been most successful in invading disturbed Wyoming big sagebrush and salt desert shrub communities, and its density and distribution have increased significantly in many ponderosa pine, pinyon juniper, antelope bitterbrush, and mountain brush communities (Monsen 1994). Its competitive and flammable nature makes it difficult to restore shrub / grasslands to their natural conditions. We have not previously had a consistent assessment of the distribution and extent of cheatgrass dominated areas in the Great Basin, the area with the greatest acreage of vegetation change caused by this species. This study provides the Department of Interior a much-needed perspective on the scale of the restoration effort required to convert these lands into healthy productive rangelands.

References

Hardy, Colin C.; Schmidt, Kirsten M.; Menakis, James P.; Sampson, Neil R. 2001. Spatial data for national fire planning and fuel management. International Journal of Wildland Fire. 10: 353-372.

Kloditz, Christiane; Boxtel, Angelien; Carfagna, Elisabetta; van Deursen, William. 1998. Estimating the accuracy of coarse scale classification using high scale information. Photogrammetric Engineering and Remote Sensing. 64(2): 127-133.

Loveland, T.R.; Merchant, J.M.; Ohlen, D.O.; Brown, J.F. 1991. Development of a landcover characteristics database for the conterminous U.S. Photogrammetric Engineering and Remote Sensing. 57(11): 1453-1463.

Monsen, Stephen B. 1994. The competitive influences of cheatgrass (Bromus tectorum) on site restoration. In: Monsen, Stephen; Kitchen, Stanley G., eds. Proceedings – ecology and management of annual rangelands. Gen. Tech. Rep. INT-GTR-313. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 43-50.

Schmidt, Kirsten M.; Menakis, James P.; Hardy, Colin C.; Hann, Wendall J.; Bunnell, David L. 2002. Development of coarse-scale spatial data for wildland fire and fuel management. Gen. Tech. Rep. RMRS-GTR-87. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 46 p.

U.S. Department of Agriculture, Forest Service. 2000. Protecting people and sustaining resources in fire-adapted ecosystems: a cohesive strategy. The Forest Service management response to the General Accounting Office Report GAO / RCED-99-65, April 13, 2000. 89 p.

U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (2002 April). Fire Effects Information System, [Online]. Available: http: / / www.fs.fed.us / database / feis / [3 / 20 / 2002].

United States General Accounting Office. 1999. Western National Forests: A cohesive strategy is needed to address catastrophic wildfire threats. Report to the Subcommittee on Forests and Forest Health, Committee on Resources, House of Representatives. GAO / RCED-99-65. 60 p.

Determining the Spatial Extent of Historical Fires With Geostatistics in Northern Lower Michigan

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***Abstract**—Interpolated General Land Office fire occurrence notes were used to determine the spatial extent of pre-European settlement fires for 26 counties in northern lower Michigan using ordinary kriging with probability output. Best fit of a surface was achieved using a spherical model with a lag distance of 860 meters, an angular tolerance of 45 degrees, and consideration of anistropy. The interpolated data were associated with Land Type Associations to determine fire rotation and occurrence intervals for pre-European and modern day fires. The results show that modern day fire suppression efforts have curbed the size and frequency of fires.*

Introduction

Many natural resource conservation and production issues stem from concerns regarding the effects of natural and human-caused disturbance both at the landscape and local level. These issues are international in scope and range from loss of species diversity to threats to human safety and property from wildfire. The need to improve our understanding of natural disturbance and apply that knowledge to forest resource management practices is well documented (Watt 1947; Heinselman 1963, 1973, 1981; Wright and Heinselman 1973; Borman and Likens 1979; Canham and Loucks 1984; Sousa 1984; Botkin 1990; Forman and Godron 1991; Christensen 1993; and Tillman 1996). In the past, fire and wind disturbance have interacted with biological and physical components of the ecosystem to regulate patterns in the composition, structure, and age of forested landscapes in Michigan (Whitney 1986, 1987). Today humans are also disturbing forests through resource extraction, fire suppression, recreational use, and rural development. Understanding the beneficial or adverse effects of disturbance, such as fire risk, is essential to conflict resolution and ultimately sustainable forest management.

Large modern day fires in the Lake States are rare due to effective fire suppression, though they do occur with devastating results. In the late summer of 1976, a fire near Seney in Michigan's upper peninsula burned approximately 74,000 acres. The fire, started by lightning, resulted in fire suppression and damage costs of more than $8,000,000. The Mack Lake fire, which occurred in northern lower Michigan in May, 1980, burned more than 20,000 acres in 6 hours. It eventually burned 24,000 acres, destroyed 44 homes and buildings, and caused one fatality. Simard and Blank (1982) reported that there have been 5 other fires in excess of 10,000 acres since 1820 within the area burned by the 1980 Mack Lake fire. The average return interval for these fires is 28 years. Simard et al. (1983) noted, "Given that fires will continue to

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occur, and that critical weather conditions will occasionally prevail, there is every reason to believe that some future jack pine fires will escape initial attack." There have been 215 fires in Oscoda County in the Mack Lake area within the past 16 years. Five were larger than 100 acres with 1 fire larger than 5,000 acres. The potential for a major conflagration exists within this area and in many other Michigan counties due to the extensive acreages of xeric outwash plains supporting pyrophilic jack and red pine ecosystems.

Simard et al. (1983) also stated, "Each fire is only a single observation of a complex process, and many observations are needed before patterns are observed." Clark (1987) noted that fire regimes are inherently difficult to assess because "the high variance associated with any low-probability event requires large sample sizes to determine expected values." While there are large numbers of modern fire records available for developing predictive models of fire ignition, modeling the potential of fires of varying size may be difficult due to the low number of observations of larger fires due to effective fire suppression. For example, of the 65,535 fires reported by state and federal agencies in the Lake States between 1985 and 2000, only 1,104 were larger than 100 acres and 122 larger than 1,000 acres. Thus, fires larger than 100 acres represent only 1.6 percent, or 1,104 of the 65,535 fires reported during this period. Therefore determining fire locations and extents during the pre-suppression era may provide information essential to estimating where large fires could occur today if undetected, or under circumstances where several fires occur concurrently, exceeding fire fighting capacity.

A number of approaches have been taken to estimate historical fire regimes in terms of frequency and extent or rotation. Clements (1910), Heinselman (1973), Arno and Sneck (1977), Simard and Blank (1982), Loope (1991), and Brown et al. (2001) used dendrochronological methods to examine fire scars for dating fire events at particular points. They then extrapolated the point data to represent the area under investigation. Van Wagner (1977) introduced the use of current age-class data fitted to a negative exponential curve to calculate fire rotations such that reconstructions of past fire events was not needed. Clark (1988a, 1988b) used stratigraphic charcoal analysis on petrographic thin sections to reconstruct a 750 year fire history in Itasca State Park, Minnesota. Each of these methods has advantages and disadvantages (Agee 1993) related to adequately assessing fire regimes at appropriate or relevant spatial and temporal scales. Area effects on estimates of fire return intervals or fire rotations (Arno and Petersen 1983), assumptions regarding flammability of fuels and fire behavior across heterogeneous landscapes (Gosz 1992, Brown et al. 2001), and adequacy of approaches for understanding long-term burn patterns (Clark 1987, 1988, 1990) are among the many challenges associated with meaningfully assessing fire regimes in space and time.

For this research, the approach to estimating fire locations and extent and subsequent interpretations involves the use of spatial statistics, specifically kriging, to interpolate fire observations made by General Land Office (GLO) surveyors. The original land survey by the GLO was initiated in Michigan in 1826, providing the earliest systematically recorded information on forest conditions in the Lake States. GLO surveyors noted fire locations along section lines and at section and quarter section corners. This provides a grid of observations along transects approximately one mile apart (Almendinger 1997). GLO records have been used to provide information on tree species composition, diameter size distribution, and disturbance patches in the pre-European settlement forests of the Lake States (Cottam 1949; Stearns 1949; Bourdo 1956, 1983; Cottam and Curtis 1956; Curtis 1959; Loucks 1983; Whitney 1986, 1987; Frelich 1995; and Owens 2001). Our use of GLO fire observations enables us to develop spatially explicit estimates of fire frequency and rotation intervals over an extensive geographic area.

Methodology

GLO Data Set

Microfilmed GLO notes for the section and quarter-section corners of 26 counties in Michigan's Northern Lower Peninsula (figure 1) were converted to ArcInfo point coverages. The coverages were rectified and georeferenced to a Modified Albers Conical Equal Area projection. Projection parameters on the modified projection are as follows: false easting and false northing 0 degrees, central meridian -89.50 degrees, first standard parallel 42.33 degrees, second standard parallel 47.66 degrees, latitude of origin 41.00 degrees, datum NAD 27, and spheroid Clark 1866. Attribute information associated

Figure 1—Michigan counties (shaded) included in fire occurrence study.

with the point coverages includes: corner record number; X and Y geographic coordinates of the corner; corner number; county of location, witness tree species, diameters, azimuths and directions; noted landscape disturbances; ecosystem classification; timber composition; other observations; surveyor's name and year of survey.

Noted fire occurrences were utilized to determine fire point locations. Designations of visible burn area (VB), entering burned area (EB), leaving burned areas (LB),visible burn area and fallen timber (VU), entering burn area and fallen timber (EU), and leaving burn area and fallen timber (LU) were included in the analysis. An addition field was created in the attribute table and labeled fire_code. A fire_code of 1 indicates a point of noted fire occurrence, and a value of 0 indicates no notation of fire damage.

Analysis Procedures

The ESRI ArcGIS (Version 8) Geostatistical Analyst extension was used to perform the analysis. This extension provides advanced surface modeling using deterministic and geostatistical methods. It also bridges the gap between geostatistics and GIS through integration of interpolation procedures into the ArcGIS software. The software provides the capability for analyzing data sets using different kriging approaches (simple, ordinary, probability, co-kriging, indicator and disjunctive), evaluating variogram or covariance plots utilizing different surface models, and calculation and evaluation of the effects of anistrophy.

A preliminary assessment of the data showed that the fire locations were naturally grouped into neighborhoods across the region (figure 2). This grouping can be explained in part by the fact that certain vegetation types, such as jack pine, are more susceptible to fire. The data were subset into these neighborhoods, and each neighborhood independently interpolated. The subsetting also facilitated the evaluation of directional autocorrelation within each neighborhood.

Ordinary kriging was used for the interpolation of the fire occurrence data points with output in the form of a probability map. It was chosen over simple kriging since it requires neither knowledge nor stationarity of the mean over the entire study area. Goovaerts (1997) noted that ordinary kriging with local search neighborhoods amounts to estimating the local mean at each location with data specific to the neighborhood, then applying the simple kriging estimator using that estimate of the mean rather than the stationary mean. Use of probability of occurrence not only provided predictions of the spatial extent of the fires, but also provided a level of confidence for the prediction.

Omni-directional variograms were generated to explore the structure of each neighborhood. Best fit of a surface for all of the neighborhoods was achieved using a spherical model. It is recommended by Isaaks and Srivastava (1989) that if the sample points are located on a grid, that the grid spacing is usually a good lag spacing. The distance between the points on a perfectly surveyed GLO grid would be 804.67 meters. However the greatest distance was found to be 858 meters, and distance of 860 meters was chosen to provide distance tolerance.

Glacial landforms strongly influence the location of the various vegetation types. The direction of the landform is influenced by the direction of advancement and retreat of the glaciers. Hence the data is expected to exhibit directional autocorrelation. Direction of anistrophy was calculated for each data set and used in the interpolation. A directional angular tolerance of 45 degrees was specified to account for directional variation in the north-south and east-west section lines. This information was then used to define the shape of the search neighborhood. The search neighborhood was divided into 4 sectors with the

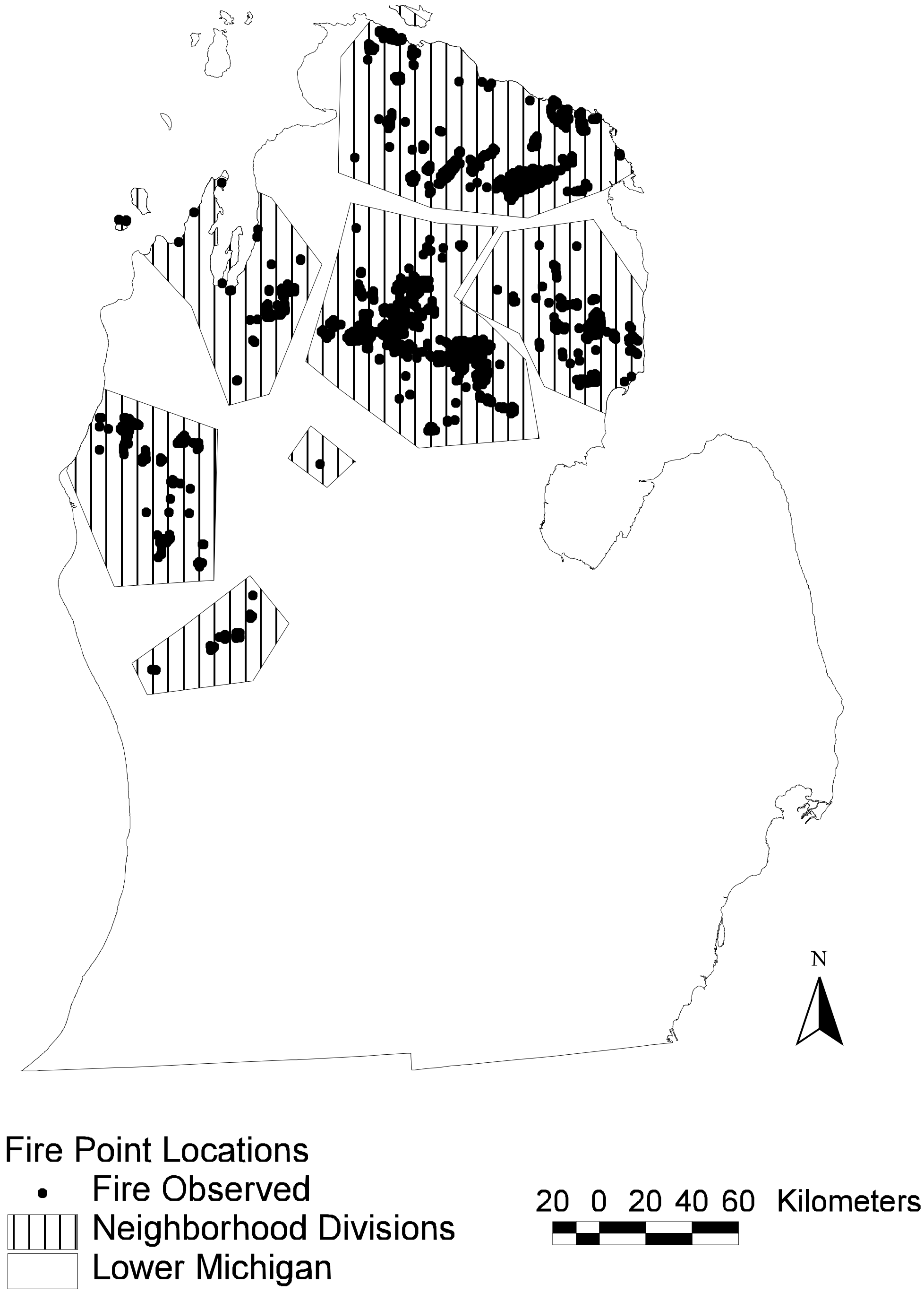

Figure 2—Locations of GLO fire points and neighborhood groupings.

sector axes running NW-SE and NE-SW. This reduced the directional influence of the GLO grid on the interpolation.

The probability of occurrence interpolations was converted from ArcGIS layer (.lyr) files to ARC grid files with a 100 meter spatial resolution. This spatial resolution provided the same scale as the original GLO point files.

Discussion

Results of Kriging

Probabilities of fire occurrence ranged from 30 to 100%. Low probabilities tend to be found in those ecosystems supporting long-lived, fire resistant northern hardwood and hardwood-hemlock forests including sugar maple, basswood, and white ash, or wetland hardwoods and mixed hardwood-conifer forests including black and green ash, silver maple, elm, and cedar. As we were interested in looking at probability of fire occurrence greater than 70%, the continuous probability classes were reclassified into three discrete classes with probability ranges of 70-79%, 80-89%, and 90-100%. Probabilities < 70% were not utilized in our analysis. The division into discrete classes facilitated acreage calculations and further analysis of the data set. However, it is important to note that forest managers could treat the output as a continuous data set and utilize the full range of output across the entire study area.

Applications for Research, Management, and Fire Risk

The interpolated probabilities of fire occurrence thematic layers are being used in several research and management applications in Michigan. Through a Joint Fire Science Program funded research project, the Great Lakes Ecological Assessment (Cleland et al. 2000; is characterizing historical and modern fire disturbance regimes of the Lake States. This research is conducting a comprehensive literature review and documenting how fire regimes have changed since European settlement. Spatially explicit estimates of historical and modern fire frequencies and rotations are being developed for landscape ecosystems mapped by interagency teams. Maps are being revised where necessary based on associations of ecological factors known to influence fire regimes. The assessment of changes in fire regimes since European settlement involves the comparison of historical fire frequencies and rotation intervals to those occurring between 1985 and 2000. We are using a hierarchical approach to assess interactions and spatial relationships among fire-dependent and fire-sensitive forest ecosystems and their associated disturbance regimes at three spatial scales. Results of these analyses are being incorporated into planning and management activities on the Hiawatha, Huron-Manistee, and Ottawa National Forests.

Use of the landscape ecosystem approach (Rowe 1980, 1984, 1992; Spies and Barnes 1985) is premised upon the assumption that fire behavior and risk are related to the conditions, processes, and spatial dimensions of particular ecosystems defined by integrating important physical and biological factors (Cleland et al. 1997). We are testing the hypothesis that historical and modern fire frequencies and rotation intervals are significantly different among multi-scaled ecological units *a posteriori*. Two principal measures of fire regimes, fire frequencies and fire rotations, provide critical information on fire risk (Agee 1993). Fire frequency is simply the number of fires per unit time and area. Fire rotation is the length of time necessary for an area equal to the entire area of interest to burn (fire cycle). This definition does not imply that the entire area will burn during a cycle; some sites may burn several times and others not at all. Meaningful estimates of these measures require clearly specifying the size of the area of interest. Thus identifying ecologically homogenous areas within which fire regimes can be analyzed is an essential step in the assessment of this process. Furthermore, mapping the location of modern and historical fires over large areas accommodates the random distribution of fires within smaller areas, improving estimates of fire regimes within ecologically similar spatial units.

Figure 3 displays a preliminary natural disturbance regime map based on aggregations or subdivisions of Land Type Associations (LTAS) for northern lower Michigan (Albert et al. 1996, Corner et al. 1999). Each polygon was evaluated using a number of GIS data sets, including Natural Resource

Conservation Service digital soil surveys, GLO notes on tree species and diameter, a 30-meter digital elevation model, hydrography, and current vegetation. Interpretations based on associations of ecological factors known to influence fire regimes were made, and each polygon was assigned to one of six fire rotation categories. The definitions for each category were based on a synthesis of the literature.

Fire rotations usually are determined by calculating the average stand age of a forest whose age distribution fits a negative exponential or a Weibull function (Van Wagner 1978). For this research, fire rotations were determined by calculating the area burned for each fire rotation category, and dividing this area by 15 to estimate area burned per annum while assuming this to be a conservative burned area recognition window (Canham and Loucks 1984). Table 1 displays the historical and modern fire rotations calculated for the draft natural disturbance regime categories in northern lower Michigan. These results are an example of the application of interpolated GLO fire points in landscape ecosystem analyses. The following briefly describes the landscape ecosystem fire regime based on fire rotation forest rotation (FR) classes.

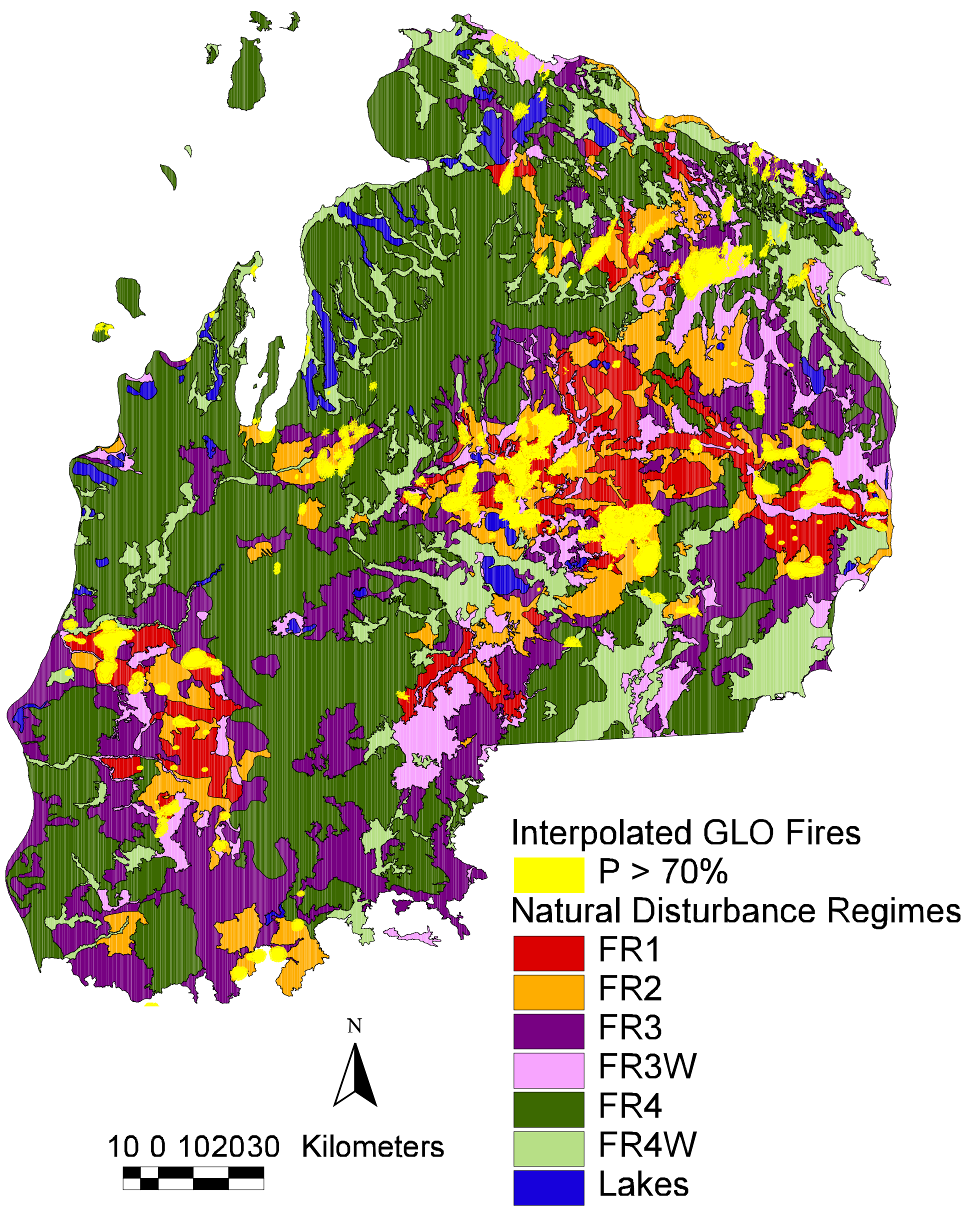

Figure 3—Natural disturbance regimes overlaid with interpolated historical fire locations with a probability of occurrence > 70%. FR1 sites experience frequent large catastrophic, stand-replacing fires. FR2 experiences less frequent large catastrophic, stand-replacing fires. FR3 and FR3W (wetlands) experience relatively infrequent stand-replacing fires. FR4 and FR4W (wetlands) experience very infrequent stand-replacing fires.

Table 1—Historic and modern fire rotations in Northern Lower Michigan.

Historical (1800s) fires **LTA grouping**	**Fire regime**	**Unit size**	**P>70% Acres burned**	**% burn/yr**	**Fire rotation**
Xeric LTAs dominated by jack pine and barrens	FR1	836,192	211,075	1.683	60
Less xeric LTAs dominated by white-red pine	FR2	1,029,138	144,850	0.938	107
Dry-mesic LTAs dominated by hemlock-white pine	FR3	1,652,410	52,396	0.211	473
Wetland LTAs adjacent to fire-prone LTAs	FR3W	494,638	61,618	0.830	120
Mesic LTAs dominated by northern hardwoods	FR4	3,771,745	40,862	0.072	1,385
Wetland LTAs adjacent to mesic hardwood LTAs	FR4W	958,232	21,012	0.146	684
Average fire rotation- 247 years	Total	8,742,355	531,813	0.406	

Modern (1985-2000) fires **LTA grouping**	**Fire regime**	**Unit size**	**Acres burned**	**% burn/yr**	**Fire rotation**
Xeric LTAs dominated by jack pine and barrens	FR1	902,052	15,552	0.115	870
Less xeric LTAs dominated by white-red pine	FR2	1,066,009	13,766	0.086	1,162
Dry-mesic LTAs dominated by hemlock-white pine	FR3	2,052,353	7,219	0.023	4,264
Wetland LTAs adjacent to fire-prone LTAs	FR3W	845,278	1,763	0.014	7,192
Mesic LTAs dominated by northern hardwoods	FR4	4,340,305	3,402	0.005	19,137
Wetland LTAs adjacent to mesic hardwood LTAs	FR4W	1,325,801	2,103	0.011	9,456
	Total	10,531,798	43,805	0.042	
Average fire rotation- 2,381 years					

FR1 represents landscape ecosystems historically experiencing frequent, large catastrophic stand-replacing fires. These ecosystems typically occur within very dry, flat outwash plains underlain by coarse-textured sandy soils. The pre-European settlement dominant forest types were short-lived jack pine forests and pine barrens.

FR2 represents landscape ecosystems historically experiencing large, catastrophic stand-replacing fires at lower frequencies, hence longer fire rotations, than the FR1 category. These ecosystems typically occur within dry outwash plains and ice-contact landforms underlain by sandy and loamy sand soils. The dominant pre-European forest types were white-red pine and mixed red-white-jack pine forests.

FR3 represents landscape ecosystems historically experiencing relatively infrequent stand-replacing fires at much longer fire rotations than the FR1 or FR2 categories. These ecosystems typically occur within dry-mesic ice-contact, glacial lakebed, and morainal landforms underlain by loamy sand to sandy loam soils, and commonly occur within close proximity to fire-prone ecosystems. The dominant pre-European forest type was long-lived mixed hemlock-white pine forests with minor elements of northern hardwood forests. Frequent ground-fires prevented succession to fire-sensitive hardwoods.

FR3W represents landscape ecosystems historically experiencing relatively infrequent stand-replacing fires. These ecosystems typically occur within wetlands embedded within or adjacent to fire-prone landscapes. The dominant pre-European forest types were wetland conifers including spruce, fir, and tamarack. Fire regimes and fuel formation were likely caused by interactions of insect and disease and large-scale blow-downs, as well as periods of drought.

FR4 represents landscape ecosystems historically experiencing very infrequent stand-replacing or community maintenance (ground) fires. These ecosystems typically occur within mesic (moist) moraines and glacial lakebeds underlain by fine-textured sandy loam to heavy clay and silt loams soils. The dominant pre-European forest types were long-lived, fire-sensitive northern hardwood and hardwood-hemlock forests including sugar maple, basswood, and white ash.

FR4W represents landscape ecosystems historically experiencing very infrequent stand-replacing or community maintenance (ground) fires. These ecosystems typically occur within wetlands embedded within or adjacent to fire-sensitive, hence fire protected landscape ecosystems (FR4). The dominant pre-European forest types were wetland hardwoods and mixed hardwood-conifer forests including black and green ash, silver maple, elm, and cedar.

Results of this research are also being applied by the North Central Research Station and cooperating universities as part of a fire risk assessment of the Lake States. This effort is assessing both historical and modern fire frequencies and rotation intervals, current vegetative conditions, and human population densities. We are investigating historical fire regimes in addition to modern regimes because preliminary analyses of a 1985-2000 modern fire database suggest that areas with the potential for large fires may not be adequately identified through regression due to the low number of large fires and the overwhelming influence of humans on fire ignition and spread. For example, Cardille and Ventura (2001) reported more than 97% of all fires occurring in the Lake States are due to human ignition, and 58% of all fires larger than 100 acres are due to arson. All fires reported were suppressed by fire fighting crews. These anthropogenic influences may obscure the elucidation of ecological factors associated with fires of different sizes. We believe the use of data on both historical and modern fire regimes will improve estimates of where the risk of large fires is greatest. Results will also provide insight into the effectiveness of fire suppression activities by allowing comparisons of pre-suppression fire regimes to those occurring today.

In summary, integrating information on historical disturbance regimes with other information such as ecological units, potential natural vegetation, and current vegetation will aid in understanding natural disturbance regimes and fire risk. This knowledge will also be useful in addressing a larger goal of improving our understanding of the characteristic rate of change, technically termed the dynamics of homeorhetic stability (Reice 1994, O'Neill et al. 1986), that formerly distinguished and maintained landscape and local ecosystems of the Lake States.

References

Agee, J.K. 1996. Fire regimes and approaches for determining fire history. Gen. Tech. Rep. INT-GTR-341. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 86 p.

Albert, D.A.; Corner, R.A.; Raab, J.B.; Delain, C.J. 1996. The landtype associations of northern lower Michigan. Michigan Natural Features Inventory. Report to the Bureau of Land Management.

Almendinger, J.C. 1997. Minnesota bearing tree data base. Biol. Rep. No. 56. Minnesota Department of Natural Resources.

Arno, S.F.; Petersen, T.D. 1983. Variation in estimates of fire intervals: a closer look at fire history on the Bitterroot National Forest. Res. Pap. INT-301. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 8 p.

Arno, S.F.; Sneck, K.M.. 1977. A method for determining fire history in coniferous forests of the mountain west. Gen. Tech. Rep. INT-42. Ogden, UT: U.S.

Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 28 p.

Bormann, F.H.; Likens, G.E. 1979. Catastrophic disturbance and the steady state in northern hardwood forests. American Scientist. 67: 660-669.

Botkin, D.B. 1990. Discordant harmonies. New York: Oxford University Press.

Bourdo, E.A, Jr. 1956. A review of the general land office survey and of its use in quantitative studies of former forests. Ecology. 37: 754-768.

Bourdo, E.A., Jr. 1983. The forest the settlers saw. In: Flader, S.L., ed. The Great Lakes Forest: an environmental and social history. Minneapolis: University of Minnesota Press: 3-16.

Brown, P.M.; Margot, K.W.; Laurie, H.S.; Christopher, B.H. 2001. Fire history along environmental gradients in the Sacramento Mountains, New Mexico: influences of local patterns, and regional processes. Ecoscience. 8(1): 115-126.

Canham, C.D.; Loucks, O.L. 1984. Catastrophic windthrow in the presettlement forests of Wisconsin. Ecology. 65: 803-809.

Cardille, J.A.; Ventura, S. J. 2001. Occurrence of wildfire in the northern Great Lakes region: effects of land cover and land ownership assessed at multiple scales. International Journal of Wildland Fire. 10: 145-154.

Christensen, N.L. 1993. Fire regimes and ecosystem dynamics. In: Crutzen, P.J.; Goldammer, J.G., eds. Fire and the environment: the ecological, atmospheric, and climatic importance of vegetation fires. New York: John Wiley & Sons: 233-244.

Clark, J.S. 1988a. Effect of climate change on fire regimes in northwestern Minnesota. Nature. 334: 233-235.

Clark, J.S. 1988b. Stratigraphic charcoal analysis on petrographic thin sections: applications to fire history in northwestern Minnesota. Quaternary Research. 30: 81-91.

Clark, J.S. 1990. Twentieth-century climate change, fire suppression, and forest production and decomposition in northwestern Minnesota. Canadian Journal of Forest Research. 20: 219-232.

Cleland, D.T.; Leefers, L.A.; Vasievich, M.J.; Crow, T.R.; Padley, E.A. 2000. The Great Lakes ecological assessment. In: Jensen, M.; Bourgeron, P., eds. A guidebook for integrated ecological assessments. New York: Springer-Verlag.

Clements, F.E. 1910. The life history of lodgepole burn forests. Bull. 79. Washington, DC: U.S. Department of Agriculture, Forest Service.

Corner, R.A.; Albert, D.A.; Delain, C.J.; Austin, M.B. 1999. Landtype associations of northern lower Michigan. Michigan Natural Features Inventory. Report number 1999-02. Northern Lower Michigan Ecosystem Management Project.

Cottam, G. 1949. The phytosociology of an oak woods in southwestern Wisconsin. Ecology. 30: 271-287.

Cottam, G.; Curtis, J.T. 1956. The use of distance measures in phytosociological sampling. Ecology. 37: 451-60.

Curtis, J.T. 1959. The vegetation of Wisconsin—an ordination of plant communities. Madison: University of Wisconsin Press.

Forman, R.T.; Godron, M. 1981. Patches and structural components for landscape ecology. Bioscience. 31: 733-740.

Frelich, L.E. 1995. Old forests in the Lake States today and before European settlement. Natural Areas Journal. 15: 157-167.

Goovaerts, P. 1997. Geostatistics for natural resources evaluation. New York: Oxford University Press. 481 p.

Gosz, J.R. 1992. Gradient analysis of ecological change in space and time: implications for forest management. Ecological Applications. 2(3): 248-261.

Heinselman, M.L. 1963. Forest sites, bog processes, and peatland types in the glacial Lake Agassiz region, Minnesota. Ecological Monographs. 33: 327-374.

Heinselman, M.L. 1973. Fire in the virgin forests of the Boundary Waters Canoe Area, Minnesota. Quaternary Research. 3: 329-382.

Heinselman, M.L. 1981. Fire intensity and frequency as factors in the distribution and structure of northern ecosystems. In: Fire regimes and ecosystem properties. Gen. Tech. Rep. WO-26. Washington, DC: U.S. Department of Agriculture, Forest Service: 7-57.

Isaaks, E.H.; Srivastava, R.M. 1989. An introduction to applied geostatistics. New York: Oxford University Press. 561 p.

Loucks, O.L. 1983. New light on the changing forest. In: Flader, S.L., ed. The Great Lakes Forest: an environmental and social history. Minneapolis: University of Minnesota Press: 17-32.

Owens, K.E. 2001. Development and analysis of current and presettlement forest cover mapping methods for the eastern upper peninsula of Michigan, Luce District. On file at: Michigan Technological University, Houghton, MI. 215 p. Dissertation.

Rowe, J.S. 1980. The common denominator in land classification in Canada: an ecological approach to mapping. Forest Chronicle. 56: 19-20.

Rowe, J.S. 1984. Forest land classification: limitations of the use of vegetation. Proceedings of the symposium on forest land classification; 1984, March 18-20; Madison, WI: 132-147.

Rowe, J.S. 1992. The ecosystem approach to forest management. Forest Chronicle. 68: 222-224.

Simard, A.J.; Blank, R.W. 1982. Fire history of a Michigan jack pine forest. Michigan Academy: 59-71.

Simard, A.J.; Haines, D.A.; Blank, R.W.; Frost, J.S. 1983. The Mack Lake Fire. Gen. Tech. Rep. NC-83. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Research Station. 36 p.

Sousa, W.P. 1984. The role of disturbance in natural communities. Annual Review Ecological Systems. 15: 353–391.

Spies, T.A.; Barnes, B.V. 1985. A multifactor ecological classification of the northern hardwood and conifer ecosystems of Sylvania Recreation Area, Upper Peninsula, Michigan. Canadian Journal of Forest Research. 15: 949-960.

Tilman; D. 1996. The benefits of natural disasters. Science. 273: 1518.

Watt, A.S. 1947. Pattern and process in the plant community. Journal of Ecology. 35: 1-22.

Whitney, G.G. 1986. Relation of Michigan's presettlement pine forests to substrate and disturbance history. Ecology. 67(6): 1548-1559.

Whitney, G.G. 1987. An ecological history of the Great Lakes Forest of Michigan. Journal of Ecology. 75: 667-684.

Wright, H.E., Jr.; Heinselman, M.L. 1973. The ecological role of fire in conifer forests of western and northern North America: introduction. Quaternary Research. 3: 319-328.

Scaling Rules and Probability Models for Surface Fire Regimes in Ponderosa Pine Forests

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Abstract—*Statistical descriptors of the fire regime in ponderosa pine forests of the Jemez Mountains, New Mexico, are spatially scale-dependent. Thus, quantification of fire regimes must be undertaken in a spatially explicit framework. We apply a variety of analytical tests adapted from species-area relationships to demonstrate an analytical framework for understanding scaling of disturbance regimes. A new spatio-temporal scaling index, the slope of the event-area function, can provide a useful measure of the synchrony of events within watersheds (where fire spread regulates the distribution of events) as well as among mountain ranges. We propose two alternative mathematical models of fire interval distributions based on inherent properties of the fire record and the ecology of frequent-fire disturbance regimes; a discrete probability model, and a probabilistic application of the lognormal distribution. Because they involve distribution of energy and matter, these spatial and temporal scaling rules indicate more general disturbance event-area relationships that can facilitate the analysis of disturbance regimes in a broader ecological framework.*

Introduction

Many ecological processes scale in time and space in ways that are determined by underlying mechanistic or stochastic processes. For example, at the level of the individual organism, body size, growth and metabolic rates, and a variety of life history traits are related systematically and can be expressed as allometric or bioenergetic scaling rules (Wiens 1989; West, Brown, and Enquist 1997; Enquist et al. 1999). Recent work (Enquist and Niklas 2001; Niklas and Enquist 2001) has shown that these scaling proprieties can be extended to the structure, composition, mass, and productivity of complex ecological communities. The unifying force across these levels of biological organization is the efficiency of energy flow, which is a strong selective force in organismal evolution. Ostensibly emergent properties of communities and ecosystems can thus be related to fundamental biophysical processes.

Because they involve distribution and flows of energy and materials, disturbance processes similarly can be expected to follow scaling rules in space and time (Holling 1992; West, Brown, and Enquist 1997; Enquist, Brown, and West 1998; Ritchie and Olff 1999). In general, we may predict that ecosystem process will scale both spatially and temporally with the factors that regulate such events, and not simply as a function of geometry. Before we can assess scaling patterns in disturbances, however, we must be able to characterize disturbance events quantitatively and measurably.

Disturbances of a particular type are often grouped together under a "regime" (Pickett and White 1985; Agee 1993). While "regimes" are often discriminated qualitatively (e.g., "stand-replacing" *vs.* "surface fire" regimes), they are usefully defined by a set of quantitative descriptors (table 1). Because

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Table 1—Dimensions of fire regimes. Adapted and expanded from Agee (1993), Johnson (1992), Whelan (1995), White and Pickett (1985).

Dimension	Typical units or metrics
Temporal distribution:	
Frequency (f)	Number of events time^{-1} .
Interval (i)	Number of yrs between events; yr event^{-1} (= 1/f).
Interval variability (temporal heterogeneity)	Statistics of central tendency (mean, median, mode) and higher moments of the frequency distribution (variance, kurtosis, skewness).
Duration	Elapsed time of a single event over a defined area (from a point to entire extent).
Fire cycle	In a stand-replacing regime, mean age of a stand with a modeled age distribution.
Fire rotation	Mean number of years required for fires to burn a specified amount of area.
Seasonality	Intra-annual occurrence for a single event or group of events.
Spatial distribution:	
Extent	Total area covered (km^2).
Spatial heterogeneity	Proportion of burn area by intensity or severity class (%); patch size (ha) and aggregation, fractal dimension (D).
Intensity	Physical properties, e.g.: flame length (m), fireline intensity (kW m^{-1}), rate of spread (m hr^{-1}), energy output per unit time (BTU hr^{-1} m^{-2} or kJ m^2), peak temperature (˚C.), residence time (min).
Severity	Effects on biotic and abiotic elements of the community, e.g.: mortality by species (%).

a disturbance regime involves multiple dimensions, disturbance regimes comprise multivariate space, not a single dimension (Agee 1993). By using multivariate space, we open the way for quantitative analyses, rather than verbal descriptors, to characterize the regime at a given place and time. Moreover, by describing the regime with a set of quantitative variables, each with associated measures and statistical characterization, we can evaluate the natural range of variability across locations and spatial scales and across time for any given spatial extent.

To illustrate these principles, we examine a 450-yr record of fire events in an old-growth New Mexico (USA) ponderosa pine forest, using tools of dendrochronology. Because fires in this ecosystem generally have low to moderate intensity of the flaming front with variable duration at the tree scale (Weaver 1951; Kilgore and Taylor 1979; Agee 1993; Allen 2001), mortality is size-dependent: most mature trees survive most fires, while most smaller and younger trees do not (i.e., mortality decreases with size). For fires above a threshold of temperature and exposure time, the cambium of mature trees will be killed locally at the locus of highest exposure (Keane et al. 2001). This dead cambium, and the tree growth response around it, creates a lesion that persists in the wood long after the event has passed. In tree species with growth rings that can be dated with annual or sub-annual precision, the year (and, in many cases, season) of individual fire events can be determined exactly (Arno and Sneck 1977; Kilgore and Taylor 1979; Dieterich 1980a; Romme 1980; McBride 1983; Swetnam and Dieterich 1985; Veblen et al. 1999). The spatial distribution of fire events in any given year can be assessed if samples are georeferenced and suitably distributed across the landscape (Niklasson and Granström 2000; Heyerdahl, Brubaker, and Agee 2002). With these tools a record of disturbance events is created, and their distribution in space and time becomes available for analysis.

Disturbance regimes are ultimately composed of events. Where these events are discrete, they can be mapped in space and time, and their distribution and scaling properties analyzed. In this paper we apply an analytical framework adapted from species-area relations and macroecology to study scaling effects in the disturbance regime. In effect, we substitute fire dates for species and then evaluate spatial and temporal scaling properties of the disturbance regime. We ask: Do quantitative descriptors of the fire regime scale in space and time? Is there an underlying probability distribution that can describe intervals between fire events? And finally, do these scaling relationships reflect governing biophysical processes, such as entrainment of the fire regime by climate?

Methods

Study Site

Monument Canyon Research Natural Area (MCN) is located in the western Jemez Mountains of north-central New Mexico, USA (figure 1). MCN was among the first Research Natural Areas in the United States, and includes stands of *Pinus ponderosa* var. *scopulorum* (ponderosa pine) and other conifers more than 420 years old (Touchan and Swetnam 1995; Touchan, Allen, and Swetnam 1996). At elevations of 2,438-2,560 m (8,000-8,400 ft), MCN is near the upper elevation limit for ponderosa pine dominance (Regional forest type 122.3, Petran Montane Conifer Forest) on mixed topography in northern New Mexico (Brown and Lowe 1980). As elevation increases above this level, forest communities transition to mixed-conifer types (121.3, Petran Subalpine Conifer Forest) regardless of aspect. In MCN, mixed conifer stands are found mostly on protected north-facing slopes and some small drainage bottoms. Soils are derived largely from tuff that formed the caldera of a large (10 km radius) Pleistocene volcano. The study area encompasses the entire 256 ha (1 mi^2) RNA, of which approximately 80% is relatively level mesa-top and 20% steep north-northeast slopes dominated by shade-tolerant conifers, including *Pseudotsuga menziesii* var. *glauca* (Rocky Mountain Douglas-fir), *Abies concolor* (white fir), and *Pinus flexilis* (limber pine).

Field Methods

We established a sampling grid with 200 m spacing across the study area, using GPS field-accurate to ± 7 m (figure 2). Each point was permanently marked with metal rebar and tagged for future relocation. At alternate points (400 m lateral spacing; N = 40), we located and sampled an average of four fire-scarred trees (min = 2, max = 12) showing evidence of the largest number of fire scars within a 40 m radius (approximately 0.5 ha) (Niklasson and Granström 2000; Heyerdahl, Brubaker, and Agee 2001). Further details of field sampling protocols are provided elsewhere (Falk 1999).

Specimen Preparation and Data Reduction

Fire-scar specimens (partial or full cross-sections) collected in the field were prepared and analyzed at the University of Arizona Laboratory of Tree-Ring Research (LTRR), using standard techniques in dendrochronology (Dieterich and Swetnam 1984; Fritts and Swetnam 1989). Each section was crossdated to a local master chronology, to ensure accuracy of dating trees with locally absent or missing rings. Once the entire ring sequence for each specimen was dated, we recorded the year of each visible fire lesion.

Data Analysis

Fire dates for each tree were entered into FHX2, a software program designed specifically for fire history analysis (Grissino-Mayer 2001). The resulting

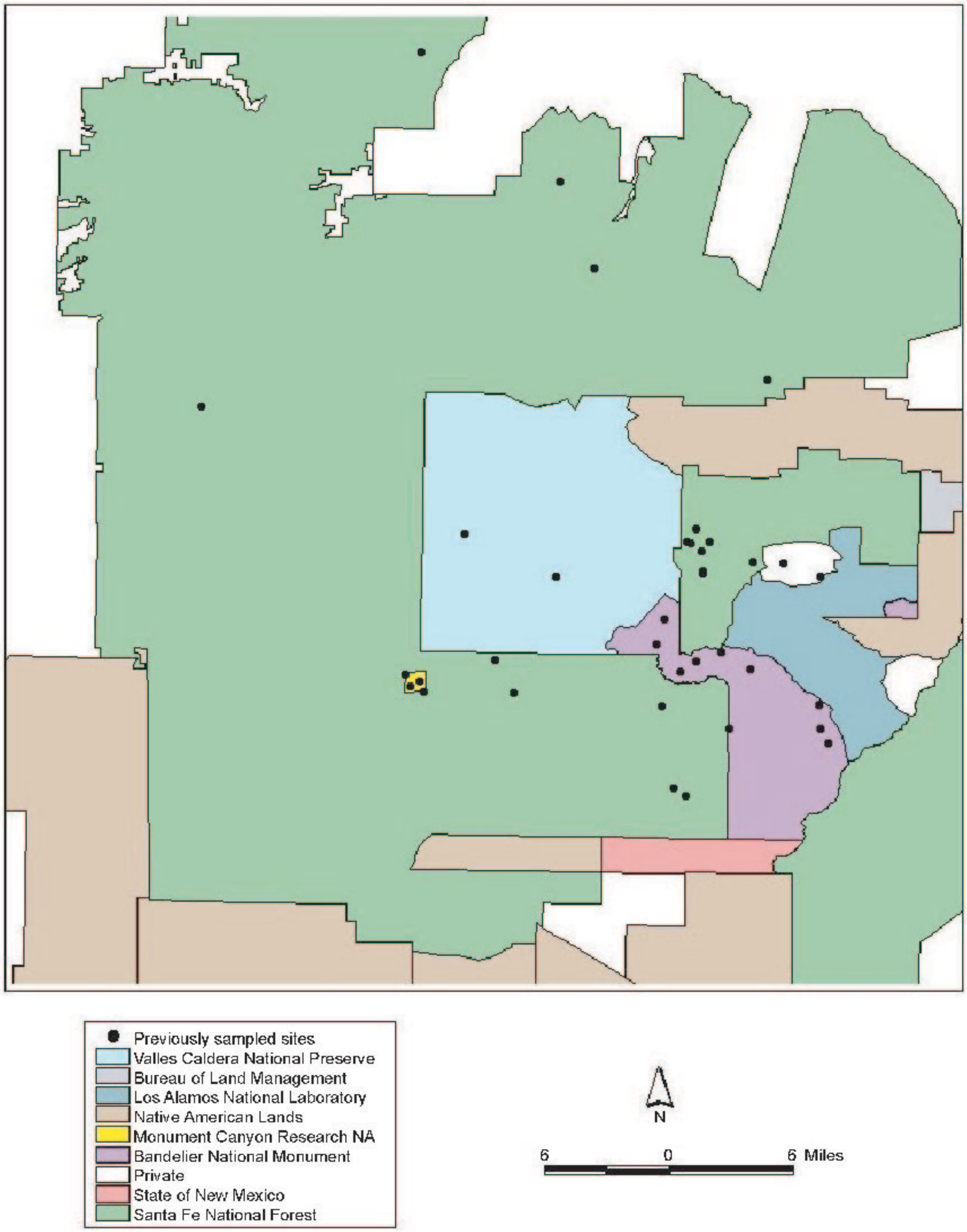

Figure 1—Site map of Jemez Mountains and Monument Canyon study area. Figure courtesy of K.L. Beeley (NPS) and C.D. Allen (USGS).

data set is a years × trees matrix, in which each cell is valued as 1 (fire recorded for that year) or 0 (no fire recorded). Because individual trees do not always record every fire event in their vicinity, we made a composite record for each grid point consisting of all fire dates recorded by any tree at that location (Dieterich 1980b). Thus, the grid-based 0.5 ha search area constituted our minimum resolution or Minimum Map Area (MMA) for reliable reconstruction of the fire record. For the same reason, fire occurrence is recorded only as presence (1) or absence (0), not relative abundance (e.g., proportion of trees recording a fire).

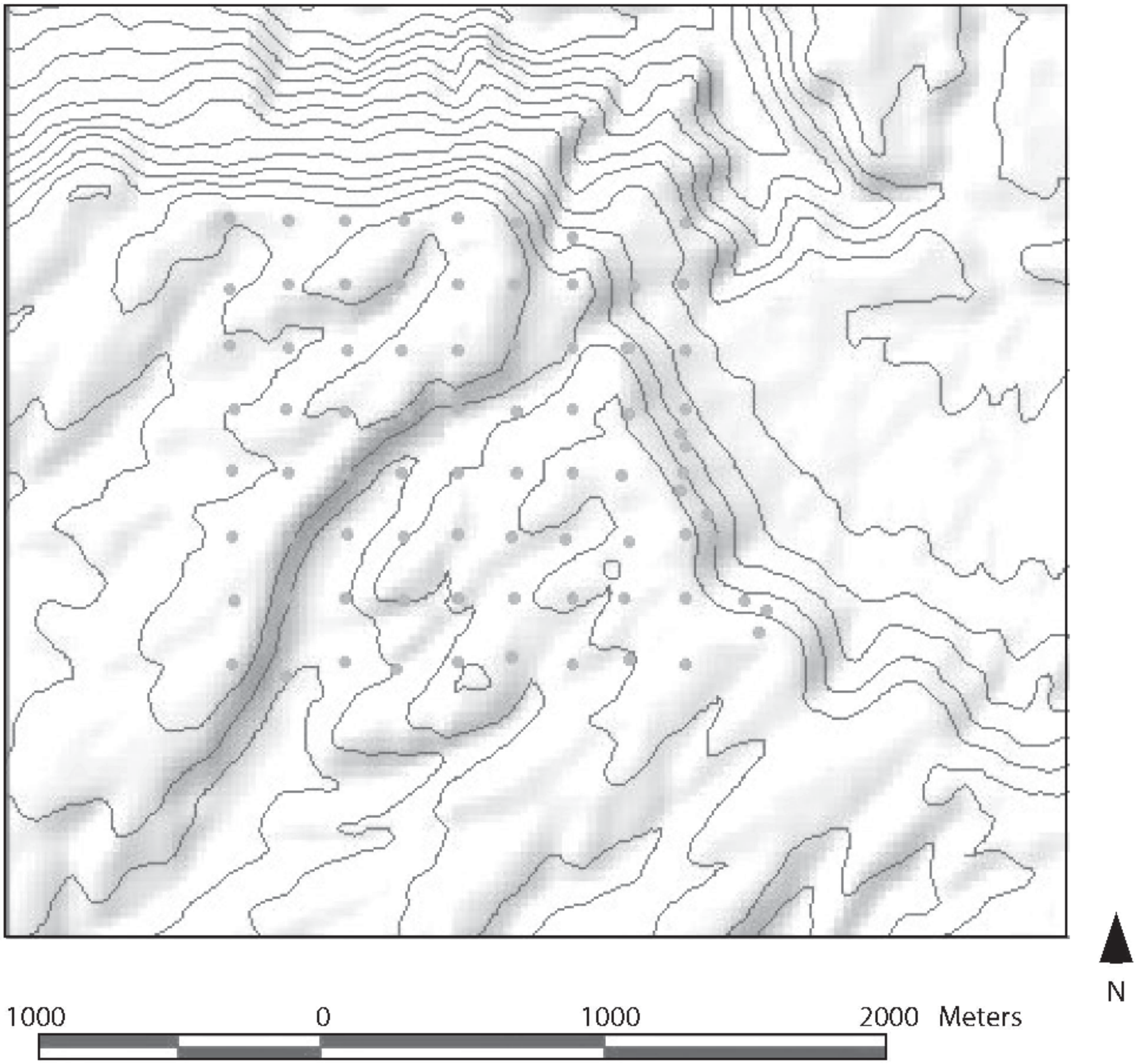

Figure 2—Sampling grid at Monument Canyon RNA.

Fire history metrics. For each grid-point composite, we calculated statistical measures of the fire regime (table 1), including total number of events detected, measures of central tendency (arithmetic mean, median, and mode) for fire intervals (yr event^{-1}), and higher moments (skewness and kurtosis).

Analytical tests. We applied a series of procedures commonly used in species-area relations to test for spatial and temporal scaling relationships in the statistical descriptors of the fire regime. We did so by substituting "fire years" for "species" in the 450-yr record of events. In a surface fire regime such as the one studied here, subsequent fires often do not eliminate the evidence of prior events. Thus, a given site can retain a record of individual fire events hundreds of years long, and individual trees with >20 scars are not uncommon. This contrasts with high-intensity, stand-replacing regimes where large, catastrophic fires destroy the tree record of prior fires on a given location (Heinselman 1981; Romme 1982). The retention of multiple fire records at a single point in space allows mapping the spatial extent of individual fires, given that not all trees near the perimeter of a fire may record, and that multiple independent fires within a single year could not be discriminated. Tests used included:

Accumulation functions ("collector's curves") for fire events. We tested for the effect of sample size (number of trees) on the number of fire events detected and mean fire interval, using a bootstrap resampling program (SSIZ) developed at the LTRR (Holmes 1995). SSIZ complies a list of fire dates and calculates the mean fire interval with confidence limits for randomly selected subsets of the original data set from 1… N trees. We ran 1,000 iterations of the resampling procedure without replacement.

Event-area relationships. The relationship between species richness and area of a sample or ecosystem is among the most widely studied patterns in species biogeography (Arrhenius 1921; MacArthur and Wilson 1967; Connor and McCoy 1979; Palmer and White 1994; Rosenzweig 1995). At small scales the accumulation ("collector's") curve dominates (Pielou 1977; Magurran 1988), but once sample size is sufficiently large, the number of species encountered increases as a power function of area, $s = cA^z$, where s is species richness, c a scaling constant that varies among organism groups, A is sample area, and z a rate constant that varies with ecosystem type and biogeographic scale (MacArthur and Wilson 1967; Rosenzweig 1995). With logarithmic transformation, the relationship is a linear function, $\log s = \log c + z \log A$. The value of s at the y-intercept (c or $\log c$) reflects "point" (alpha) diversity. Steeper values of the slope term z indicate faster accumulation of new species with increasing area. Because $z < 1$ (most published values are in the range 0.15 - .35), the rule says in effect that small areas are more species-rich per unit area than large areas.

In the current context, we can assess event-area relationships in two ways. First, we can count the number of fires (f) detected in a sample; this will stand as equivalent to species richness (S). Because fires are events in both space and time, we must define the sample temporally as well as spatially; this is conventionally done by calculating fire frequency, which is the number of fires per unit time, giving f units of fires time^{-1}. A related measure, fire interval, is the number of years between fire events, or $1/f$, with units of yr fire^{-1}. Both frequency and interval can be expressed statistically as mean, median, or modal values, and their higher moments calculated.

We tested for the effect of area by making spatially explicit subsets of the data set. Because the data are from known locations, we can create composite fire chronologies for any defined area within the study site. This can be accomplished by creating either nested series of samples beginning at any point, or non-nested samples of varying area centered on random points in the study site up to the full extent of the study (figure 3) (Palmer and White 1994). Although not reported here, sample size and sample area can be varied independently in our study design (Falk 2003 in prep.).

The interval-area relationship. The time interval between fires is of direct ecological interest because it represents the time between fire events. This interval is potentially important in forest demography because of size-dependent mortality: seedlings and saplings are unlikely to survive fires (Ryan and Reinhardt 1988; Peterson et al. 1994), so if the time interval between events is short, the probability increases that they will be exposed to a lethal event. By contrast, longer fire-free intervals allow young trees to attain size and morphology that makes their survival more likely (although fires occurring after very long intervals may be more intense, due to accumulation in the larger fuel sizes and changes in vertical fuel structure).

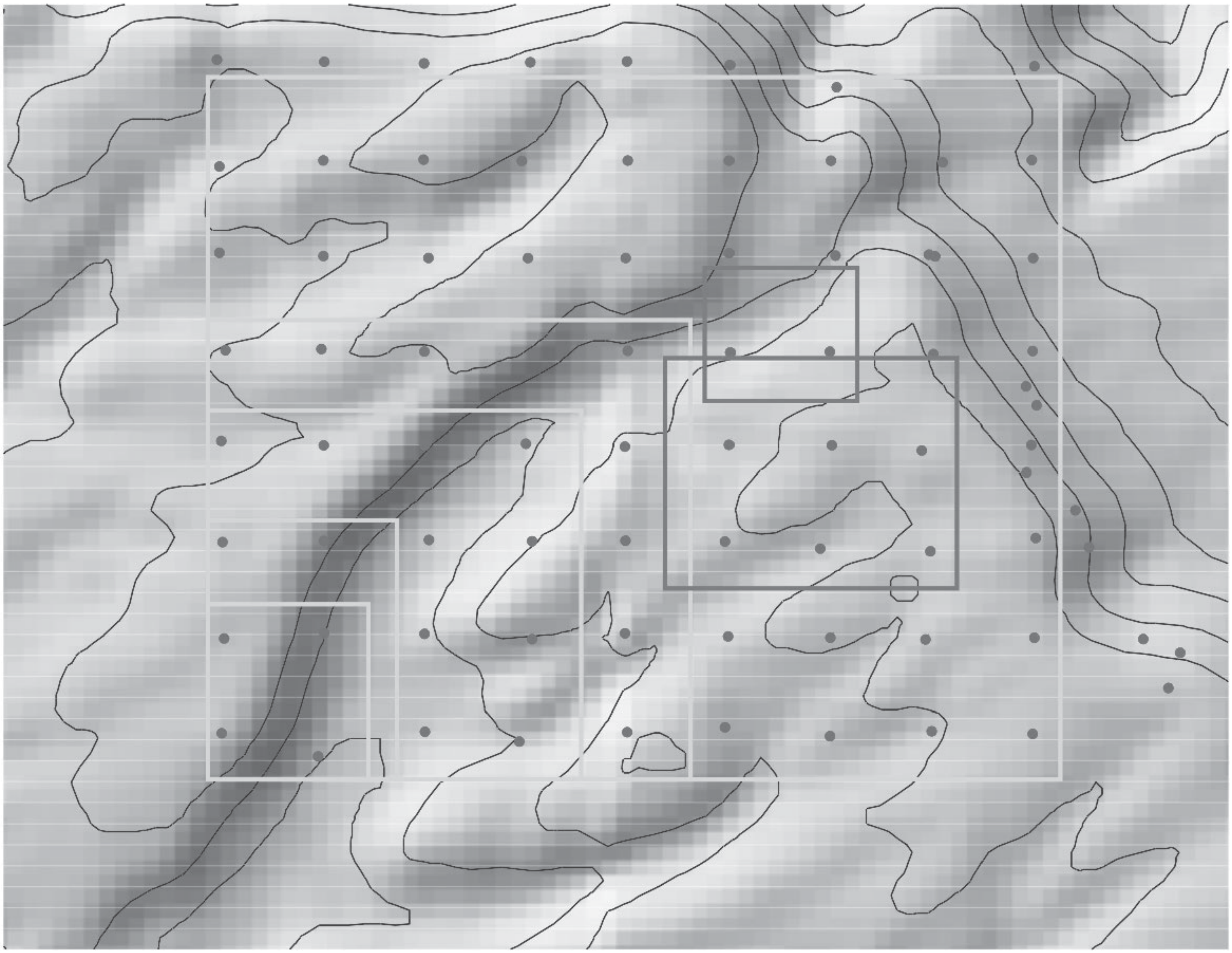

Figure 3—Nested and non-nested methods for creating simulated search areas for calculation of composite fire statistics. Fire dates are composited within each box to test for sample area effects.

Both f (fire frequency) and MFI (mean fire interval = $1/\bar{f}$) can be tested for area-dependence in a fashion similar to species richness. The predicted event-area function for frequency is a power law,

$$f = eA^{y}, \text{ hence } \log f = \log e + y \log A,\ y \geq 0$$

where e and y are scaling constants analogous to c and z in the species-area relationship. Similarly, the predicted interval-area relationship for mean fire interval over any defined period t is

$$MFI_t = pA^{-y}, \text{ and } \log MFI_t = \log p - y \log A,\ p = e^{-1}\ .$$

We expect $f(A)$ to have a positive slope (i.e., $y > 0$), since sampling larger areas across the landscape should encounter more fires in a patchy fire regime (Arno and Peterson 1983). Likewise, $MFI_t(A)$ should be negatively sloped, because if larger area samples detect more fires, the denominator of the interval statistic increases. The decreasing fire interval statistic can also be interpreted as follows: as larger areas are sampled, the likelihood increases of a fire occurring somewhere in any given year.

Probability models for interval distributions. Fire interval probabilities have most commonly been modeled using 2- and 3-parameter versions of the Weibull distribution (Johnson and VanWagner 1985; Johnson 1992; Johnson and Gutsell 1994). Although alternative models have been inadequately explored, use of the Weibull to model fire interval probabilities has become widespread (Clark 1989; Johnson 1992; Agee 1993; Swetnam and Baisan 1996; Gardner, Romme, and Turner 1999; Grissino-Mayer 1999).

The Weibull distribution is a continuous probability model that describes the effects of stress accumulation, and hence is often used to model time to failure (e.g., metal fatigue, breaking points of materials, etc.) (Bain and Engelhart 1987; Johnson, Kotz, and Balakrishnan 1994). However, it is not clear that continuity is an appropriate assumption for fire regimes in the Southwest. Fire data in the dendroecological record are inherently annual in their resolution, as reflected in the record of annual fire dates derived from analysis of fire scars or establishment dates. One way of making this evident is by attempting to increase the resolution of the record, which should be possible in a truly continuous distribution. When we decrease the time interval to subannual patterns of fire occurrence, we are asking a different question (i.e., seasonality), not simply improving the precision (i.e., smaller units) of a fire interval estimate. Thus, the fire scar record is not infinitely divisible as is typically assumed for continuous data.

A related implication of a continuous distribution is that observations are unitary. For example, we do not interpret a temperature of 32° as the sum of two temperatures of 16°, nor a blood pressure of 120 mg as the sum of 80 mg and 40 mg. In fire history studies, the parallel assumption is that fire intervals of *t* years are a single event, which permits the use of a continuous frequency distribution.

Forest fires in ponderosa pine ecosystems may not conform to this unitary assumption. Both the fire record and fire events are composed inherently of a series of discrete, binary events. In southwestern forests, fire occurrence has a finite probability each year, with an outcome of fire or no fire. Thus, each year can be defined as a Bernoulli trial, where the outcome is one of two possible states (0,1) (Bain and Engelhart 1987). A fire interval of 10 yr is thus the accumulation of 10 separate no-fire years. In this respect, discrete probability models such as the negative binomial (years before the first success) may be more appropriate null model. An assumption of Bernoulli probability is that each trial is independent, whereas fire (or its absence) in year *t*-1 may have some effect on the probability of fire in year *t*.

Here we propose an alternative approach based on first principles in fire ecology and mathematical statistics. The capture of a fire scar in a sample is the result of a series of contingent events, each with its respective probability distribution (sufficient fuel, proper fuel moisture and wind, ignition source, tree species, age and size, prior scarring survival of the fire, capture in a sample). Thus, the eventual probability P_{tot} that a fire event will occur, be recorded by the tree, and sampled by a researcher is the contingent product of the probabilities of *n* constituent factors:

$$P_{tot} = p_1 \times p_2 \times p_3 \times p_4 \times \ldots p_n = \prod_{i=1}^{n} p_i \,.$$

Taking the log of both sides gives:

$$\log P_{tot} = p_1 + p_2 + p_3 + p_4 + \ldots p_n = \sum_{i=1}^{n} p_i \,.$$

Sums of random variables approach normality under the central limit theorem, provided there are a sufficient number of factors (Montroll and Shlesinger 1982). The resulting distribution of log transformed variates is thus expected to approach normality. We therefore propose that fire intervals are lognormally distributed.

Results

Collections

For this analysis we used records from a preliminary sample of 53 fire-scarred trees at 16 grid points in the study area (figure 2). Most trees in the sample were *Pinus ponderosa*; other species included *Pseudotsuga menziesii* (Douglas-fir), *Abies concolor* (white fir), and *Pinus flexilis* (limber pine). We restricted our analyses to the period 1600-2000, for which there is sufficient sample size at all grid points.

Analytical Tests

Sample size accumulation function. The total number of events increased asymptotically with sample size (figure 4-a). The function is a classic "collector's curve," reflecting the capture of more fire events as sample size increases. Notably, very small samples appear unlikely to capture the full set of fire dates (although they probably record the most widespread events). The collector's curve is a saturating function, with a positive first and negative second derivative, and an asymptotic frequency of 9-10 events per century. The collector's curve in figure 4-a is for the full set of 53 trees across the sampled area. In a nested multi-scale analysis, each scale would have its own collector's curve reaching a characteristic asymptote (Palmer & White, 1994).

Event-area and interval-area relationships. Fire dates also accumulated in simulated samples of increasing area (figure 4-b). The lack of downward concavity suggests high patchiness in the fire regime: with increasing area, new events continue to be encountered at a high rate, although many of these events were small. This area effect appears to be independent of the accumulation of fire dates with increasing sample size (figure 4-a).

Mean fire interval was strongly scale-dependent (figure 5). Following the predicted power rule, the function is linear in log-linear space MFI $\approx$ 71.1 $A^{-0.20}$, $r^2 = 0.68$. Individual plots (points in figure 5 along the y-axis) recorded fires at mean intervals ranging from 7-23 yr (mean = 12). Extrapolation to the tree scale (0.01-0.05 ha) suggests common intervals of 9-35 yr, although we consider this scale below the minimum reliable spatial resolution for field verification. As data from adjacent grid points were added together to form larger spatial composites, more fires were encountered and MFI decreased to 7 yr for 10-ha composite samples and 4 yr for sample areas of 100 ha.

Probability model. Lognormal functions provided as close a fit to the observed distribution of fire intervals as did the Weibull distribution for spatial composites of 1-16 grid points (figure 6). This suggests that the more

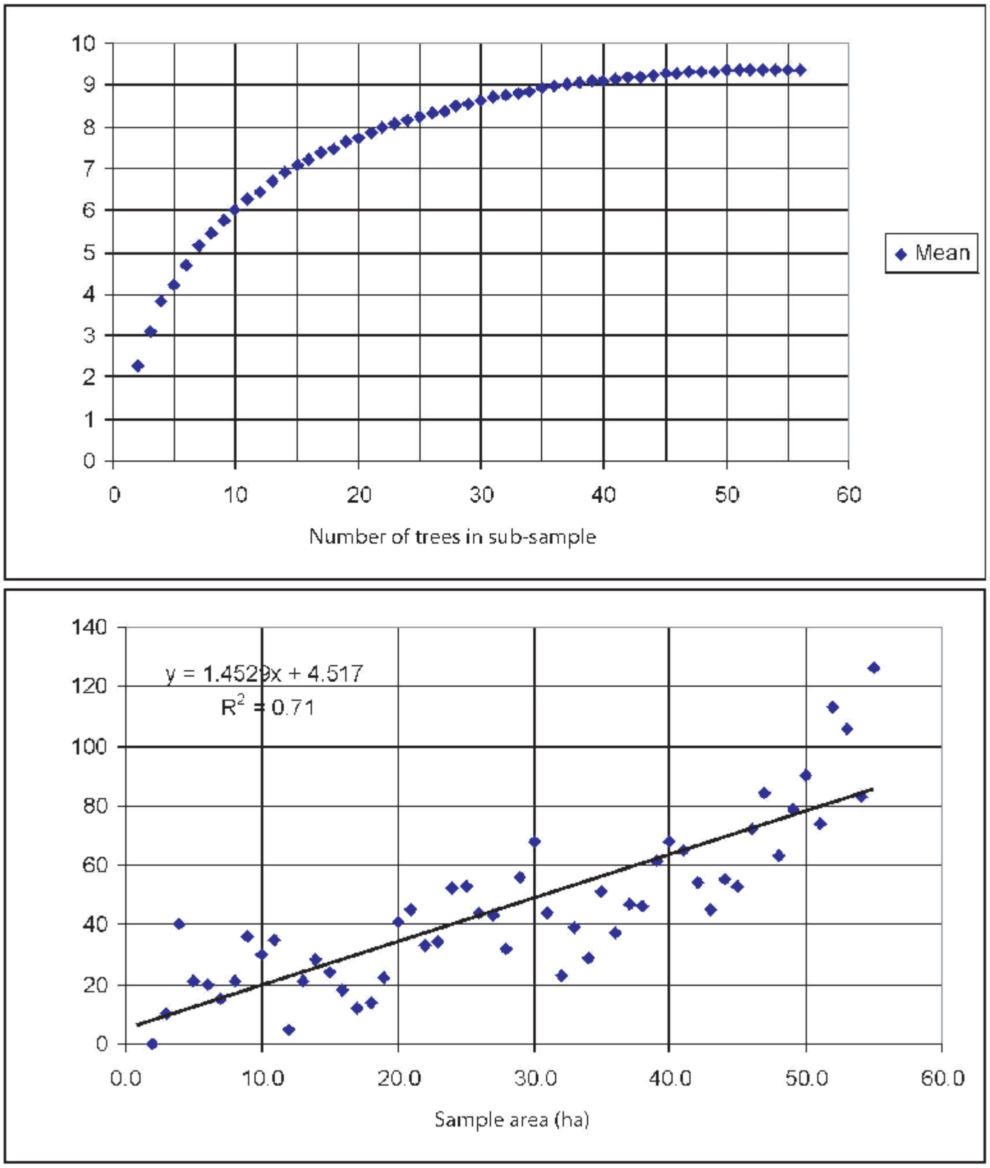

Figure 4—Accumulation functions ("collector's curves") for the number of fire events recorded as a function of (a) number of trees sampled and (b) sample area.

parsimonious and theoretically grounded lognormal distribution has potential application for modeling surface fire regimes. Interestingly, a comparison of the computed Weibull median probability interval (WMPI) with the simple arithmetic mean was highly correlated (i.e., little added information in the more complex model) (figure 7).

Conclusions, Discussion, and Future Research Directions

Scale dependence is demonstrated in the fire regime of an old-growth ponderosa pine forest. All measures of the fire regime tested appear sensitive to sample area; number of fires and mean interval were also found to be sensitive to sample size (number of trees). Thus, the notion of a unitary fire regime independent of scale is untenable; instead, we see that the fire regime is a scale-dependent characterization of an ecosystem.

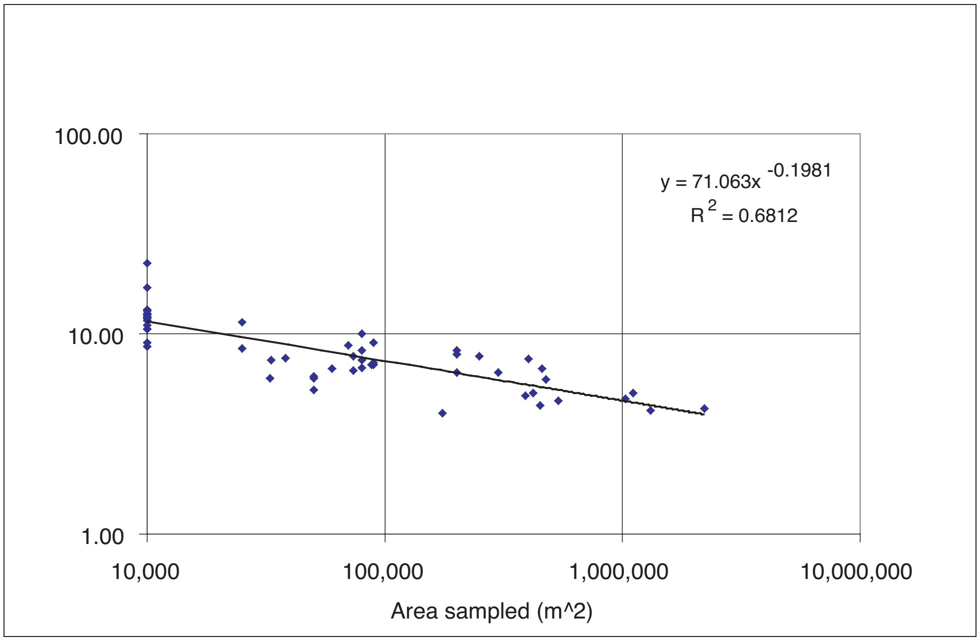

Figure 5—Dependence of mean fire interval on number of trees sampled. MFI (ordinate) decreases with increasing area sampled (abscissa, log scale). Data points are MFI for simulated sample areas of differing sizes.

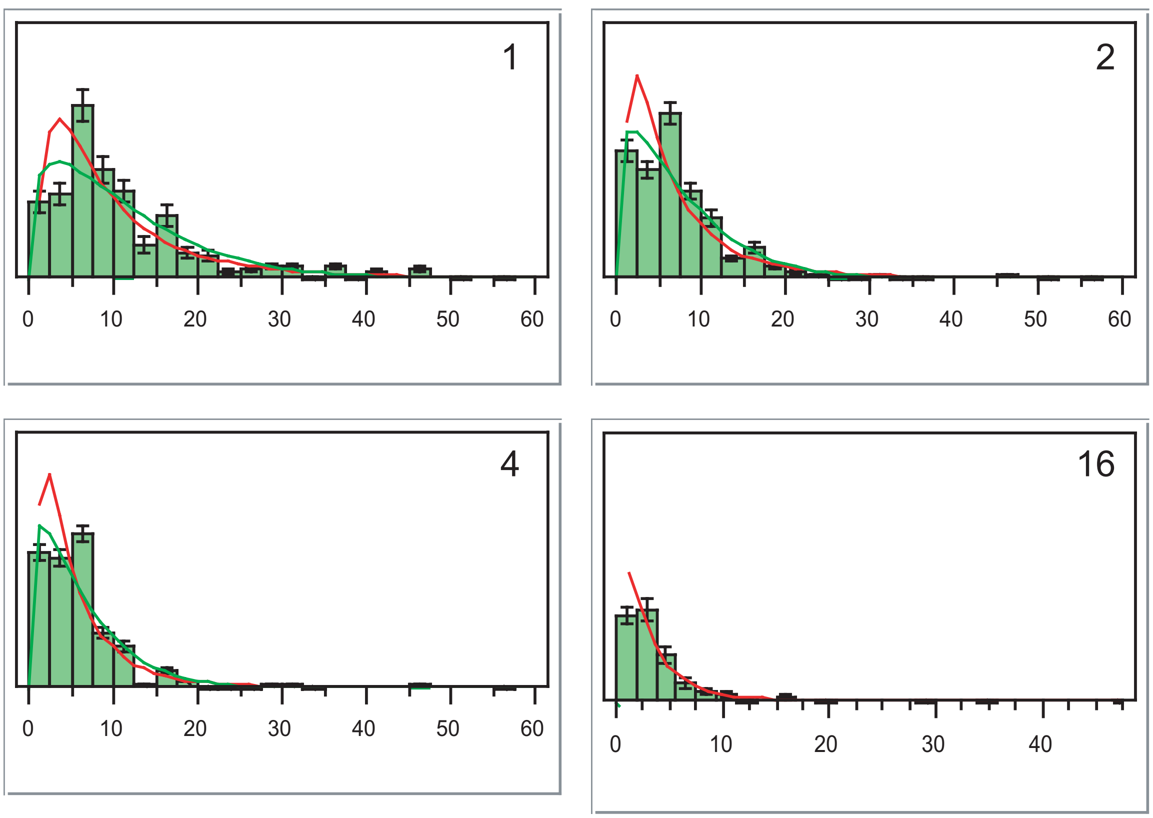

Figure 6—Scale dependence of the frequency distribution of fire intervals for nested spatial samples of 1-16 grid points, with fitted lognormal (green line) and Weibull (red line) distributions.

This preliminary analysis used approximately 20% of the eventual full data set of ≈275 trees at MCN. In addition to adding grid points (sampling locations), the full data set includes some sample points at closer intervals (200 m compared to 400 m in the present analysis), providing finer spatial resolution in the analysis. The present data set also covers approximately 150 ha; the full MCN data set will cover approximately 250 ha. Inclusion of samples from previous research efforts in adjacent areas of the Jemez Mountains (Morino,

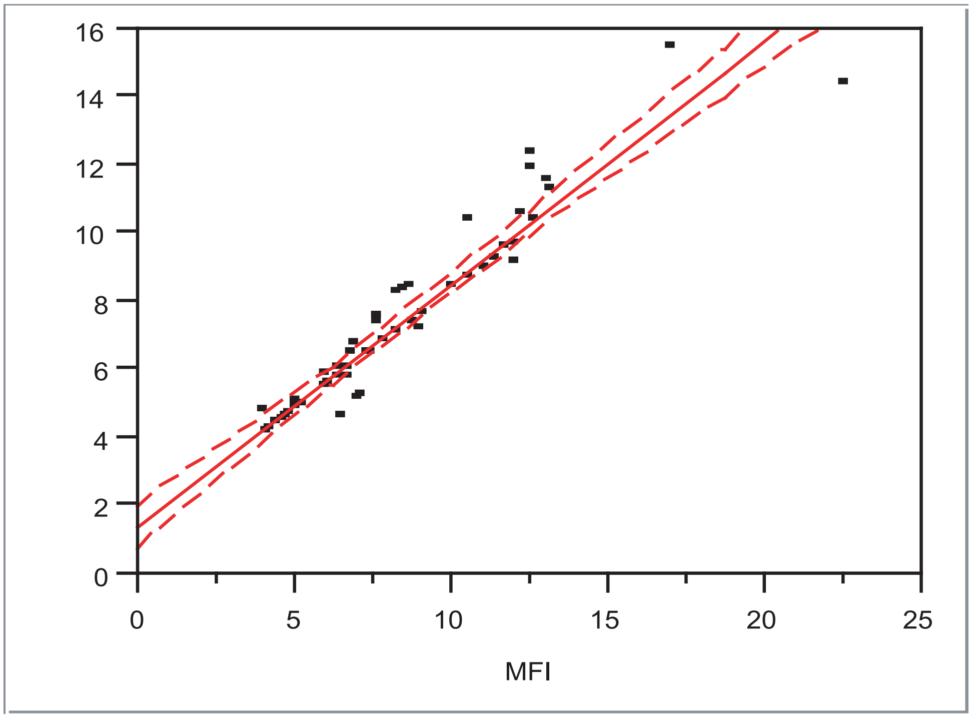

Figure 7—Correlation of mean and Weibull median probability interval (WMPI) for trees and composites at Monument Canyon (N = 53). WMPI = 1.37 + 0.71 MFI (r^2 adj. = 0.90).

Baisan, and Swetnam 1998; Allen 2002) will allow the tests used here to be applied across five orders of magnitude in spatial scale (5×10^{-1} - 10^4 ha). Several other recent fire history studies have used a spatial array of sampling points (Fulé, Covington, and Moore 1997; Brown, Kaufmann, and Shepperd 1999; Niklasson and Granström 2000; Skinner 2000; Veblen, Kitzberger, and Donnegan 2000; Heyerdahl, Brubaker, and Agee 2001). A meta-analysis of these studies may reveal even more general scaling rules.

Sample size and sample area are partly confounded in the results presented here. Distinguishing these two factors is important, for the same reasons as in species biogeography: the collector's curve can dominate the species- (event-) area function at small spatial scales (Palmer and White 1994). In the MCN case, the grid-based design allows sample size and sample area to be decomposed using a factorial procedure, assembling a composite data set for increasing sample size (say, 5-30 trees) selected at random from a series of increasing sample areas (5-250 ha). The resulting data can be tested by MANOVA, or used to generate Fisher's a statistic for increasing species (fire date) richness with increasing sample size. Stability in Fisher's α indicates that increased richness is attributable only to sampling more individuals, while an increasing Fisher's α indicates an increase in richness independent of sample size.

A filtering approach can also be used with fire history data to identify widespread fire years. For example, Swetnam & Baisan (1996) analyzed fire data from southwestern North America, selecting fire dates found respectively on $\geq$25%, $\geq$10%, or any recording tree within each site (the same approach can be applied at the site level, filtering out fire years recorded by only a few sites). Filtering removes fire dates found on only a few trees (or sites), and is thus useful for identifying widespread fire years. One difference with the present approach is that filtering is non-spatial. For example, in a sample of 100 trees, the filter does not discriminate between a fire year recorded by nine widely dispersed trees from a date recorded by nine trees in a cluster. Filtering would tend to reduce the number of fire years in a population, and thus decrease the upper asymptote of a collector's curve.

The event-area relationship has many potential applications in fire and forest management (Baker 1989). In ecology, scaling relationships have important implications for forest demography and stand dynamics. In southwestern pine forests, both mortality and scarring of survivors from fire events is strongly age- and size-dependent. Individual trees are affected only by fires that are proximate enough to generate the threshold values of exposure time and cambial temperature; fires that are too far away would have little effect. Thus, to understand the regulatory influence of fires of forest demography, we must evaluate fire occurrence (as well as fire-free intervals) at the "tree-scale." While much further research is required to define the radius of effect for fires on seedlings and saplings, the spatial domain of demographically effective surface fires is undoubtedly closer to 0.05 ha than to 500 ha. A fire occurring 1,500 m away probably has no demonstrable effect on the survival, growth, or reproduction of a target tree, whereas a fire within 50 m is likely to affect all three demographic parameters. Scale dependence in the fire regime is the key to understanding the spatial aspects of forest demography.

The event-area relationship can also be used as the basis for a measure of spatio-temporal synchrony of events. Independent fire events can be synchronized regionally by climate, particularly periodic events such as El Niño Southern Oscillation (ENSO) and the Pacific Decadal Oscillation (PDO). (Swetnam and Betancourt 1990; Swetnam and Betancourt 1992; Grissino-Mayer and Swetnam 2000; Heyerdahl, Brubaker, and Agee 2002). When events are synchronous across the landscape, fires that occur anywhere occur everywhere, leading to a collector's curve that immediately reaches the asymptote. The corresponding MFI-area function will have a flat (approaching 0) slope, because as sample area increases, few events are encountered that have not already been detected. By contrast, during periods when fires are not synchronized regionally, the collector's curve rises more gradually, and the MFI-area function declines at a faster rate because the landscape is dotted with small, un-correlated events. We propose that the slope of the interval-area relationship may provide a statistical measure of regional entrainment of the fire regime by climate, and a potential indicator of regime shifts.

In terms of forest management, understanding the scaling relationships of the natural fire regime can be a powerful tool for restoring natural or prescribed fire intervals in fire management programs (Allen et al. 2002). In a simplistic example, fire intervals from data collected at the 100 ha scale (a common extent for fire history samples) might be applied uniformly across the landscape down to the level of individual tree clusters. The negative slope of the MFI-area relationship shows, however, that smaller areas experience fires less frequently than do larger areas. Scaling rules can help to make prescribed natural fire programs more realistic in their application.

The non-zero slope of the MFI-area function also suggests the importance of reporting search area as an integral element of fire interval statistics. Because the mean fire interval is non-stationary over area, sample area should be reported explicitly for a particular sample area (e.g., "7.5 yr for 100 ha"). The use of area-corrected units should become standard practice to avoid confusion in interpretation of interval data in research and management alike.

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Literature Cited

Agee, J. K. 1993. Fire ecology of the Pacific Northwest. Washington, DC: Island Press.

Allen, C. D., M. Savage, D. A. Falk, K. F. Suckling, T. W. Swetnam, T. Schulke, P. B. Stacey, P. Morgan, M. Hoffman and J. T. Klingel. 2002. Ecological restoration of southwestern Ponderosa pine ecosystems: A broad perspective. Ecological Applications. 12(5): 1418-1433.

Allen, C. D. 2002. Lots of lightning and plenty of people: An ecological history of fire in the upland Southwest. Chapter 5 in: Vale, T. R., ed. Fire, native peoples, and the natural landscape. Covelo, CA: Island Press.

Arno, S. F., and T. D. Peterson. 1983. Variation in estimates of fire return intervals: a closer look at fire history on the Bitterroot National Forest. Gen. Tech. Rep. INT-301. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station.. 8 pp.

Arno, S. F., and K. M. D. Sneck. 1977. A method for determining forest fire history in coniferous forests in the Mountain West. Gen. Tech. Rep. GTR INT-42. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest & Range Experiment Station. 28 pp.

Arrhenius, O. 1921. Species and area. Journal of Ecology. 9: 95-99.

Bain, L. J., and M. Engelhart. 1987. Introduction to probability and mathematical statistics. Boston, MA: PWS-Kent Publishing Co.

Baker, W. L. 1989. Effects of scale and spatial heterogeneity on fire-interval distributions. Canadian Journal of Forest Research. 19: 700-706.

Brown, D., and C. Lowe. 1980. Biotic communities of the Southwest. Tucson, AZ: University of Arizona Press.

Brown, P. M., M. R. Kaufmann, and W. D. Shepperd. 1999. Long-term, landscape-level patterns of past fire events in a montane ponderosa pine forest of central Colorado. Landscape Ecology. 14: 513-532.

Clark, J. S. 1989. Ecological disturbance as a renewal process: theory and application to fire history. Oikos. 56: 17-30.

Connor, E. F., and E. D. McCoy. 1979. The statistics and biology of the species-area relationship. American Naturalist. 113: 791-833.

Dieterich, J. H. 1980a. Chimney Springs fire history. Gen. Tech. Rep. RM-220. Ft. Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station.

Dieterich, J. H. 1980b. The composite fire interval — a tool for more accurate interpretation of fire history. In: Proceedings of the Fire History Workshop; 1980 October 20-24; Tucson, AZ. Gen. Tech. Rep. RM-81 Ft. Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 8-14.

Dieterich, J. H., and T. W. Swetnam. 1984. Dendrochronology of a fire-scarred ponderosa pine. Forest Science. 30 (1):238-247.

Enquist, B. J., J. H. Brown, and G. B. West. 1998. Allometric scaling of plant energetics and population density. Nature. 395:163-165.

Enquist, B. J., and K. J. Niklas. 2001. Invariant scaling relations across tree-dominated communities. Nature. 410:655-660.

Enquist, B. J., G. B. West, E. L. Charnov, and J. H. Brown. 1999. Allometric scaling of production and life-history variation in vascular plants. Nature. 401:907-911.

Falk, D. A. 1999. Monument Canyon Research Project: Manual of field sampling protocols. Tucson, AZ: Laboratory of Tree-Ring Research.

Falk, D. A. 2003 in prep. Scaling rules and probability models for surface fire regimes in Ponderosa pine forests. Tucson, AZ: University of Arizona. Ph. D. dissertation.

Fritts, H. C., and T. W. Swetnam. 1989. Dendroecology: a tool for evaluating variations in past and present forest management. Advances in Ecological Research. 19: 111-189.

Fulé, P. Z., W. W. Covington, and M. Moore. 1997. Determining reference conditions for ecosystem management of Southwestern Ponderosa pine forests. Ecological Applications. 7(3): 895-908.

Gardner, R. H., W. H. Romme, and M. G. Turner. 1999. Predicting forest fire effects at landscape scales. In: Mladenoff, D.J.; Baker, W.L., eds. Spatial modeling of forest landscape change. Cambridge, UK: Cambridge University Press: 163-185.

Grissino-Mayer, H. D. 1995. Tree-ring reconstructions of climate and fire history at El Malpais National Monument, New Mexico. Tucson, AZ: University of Arizona. Ph.D. dissertation,.

Grissino-Mayer, H. D. 1999. Modeling fire interval data from the American Southwest with the Weibull distribution. International Journal of Wildland Fire. 9 (1): 37-50.

Grissino-Mayer, H. D. 2001. FHX2 — Software for analyzing temporal and spatial patterns in fire regimes from tree rings. Tree-ring Research. 57 (1): 115-124.

Grissino-Mayer, H. D., and T. W. Swetnam. 2000. Century-scale climate forcing of fire regimes in the American Southwest. The Holocene. 10 (2): 213-220.

Heinselman, M. L. 1981. Fire and succession in the conifer forests of northern North America. In: West, D.C.; Shugart, H.H.; Botkin, D.B., eds. Forest succession: concepts and application. New York: Springer Verlag: 374-405.

Heyerdahl, E. K., L. B. Brubaker, and J. K. Agee. 2001. Spatial controls of historical fire regimes: a multiscale example from the interior west, USA. Ecology. 82 (3): 660-678.

Heyerdahl, E. K., L. B. Brubaker, and J. K. Agee. 2002. Annual and decadal climate forcing of historical fire regimes in the interior Pacific Northwest, USA. The Holocene. 12(5): 597-604.

Holling, C. S. 1992. Cross-scale morphology, geometry, and dynamics of ecosystems. Ecological Monographs. 62 (4): 447-502.

Holmes, R. L. 1995. SSIZ: Sample Size Estimation Program for Fire Frequency. Laboratory of Tree-Ring Research, University of Arizona.

Johnson, E. A. 1992. Fire and vegetation dynamics. Cambridge Studies in Ecology. Cambridge, U.K.: Cambridge University Press.

Johnson, E. A., and S. L. Gutsell. 1994. Fire frequency models, methods, and interpretations. Advances in Ecological Research. 35: 239-287.

Johnson, E. A., and C. E. VanWagner. 1985. The theory and use of two fire history models. Canadian Journal of Forest Research. 15: 214-220.

Johnson, N. L., S. Kotz, and N. Balakrishnan. 1994. Univariate continuous distributions. New York: John Wiley & Sons.

Keane, R. E., M. Austin, C. Field, A. Huth, M. J. Lexer, D. Peters, A. Solomon, and P. Wyckoff. 2001. Tree mortality in gap models: Application to climate change. Climate Change. 51: 509-540.

Kilgore, B. M., and D. Taylor. 1979. Fire history of a sequoia-mixed conifer forest. Ecology. 60: 129-142.

MacArthur, R. H., and E. O. Wilson. 1967. The theory of island biogeography. Monographs in Population Biology No. 2. Princeton, NJ: Princeton University Press.

Magurran, A. E. 1988. Ecological diversity and its measurement. Princeton, N.J.: Princeton University Press.

McBride, J. R. 1983. Analysis of tree rings and fire scars to establish fire history. Tree-Ring Bulletin. 43: 51-67.

Montroll, E. W., and M. F. Shlesinger. 1982. On $1/f$ noise and other distributions with long tails. Proceedings of the National Academy of Science. 79: 3380-3383.

Morino, K. A., C. H. Baisan, and T. W. Swetnam. 1998. Expanded fire regime studies of the Jemez Mountains, New Mexico. Tucson, AZ. National Biological Service, Bandelier National Monument. Final Report. 94 p.

Niklas, K. J., and B. J. Enquist. 2001. Invariant scaling relationships for interspecific plant biomass production rates and body size. Proceedings of the National Academy of Science (USA). 98 (5): 2922-2927.

Niklasson, M., and A. Granström. 2000. Numbers and sizes of fires: long-term spatially explicit fire history in a Swedish boreal landscape. Ecology. 81(6): 1484-1499.

Palmer, M. W., and P. S. White. 1994. Scale dependence and the species-area relationship. American Naturalist. 144(5): 717-740.

Peterson, D. L., S. S. Sackett, L. J. Robinson, and S. M. Haase. 1994. The effects of repeated prescribed burning on *Pinus ponderosa* growth. International Journal of Wildland Fire. 4(4): 239-247.

Pickett, S. T. A., and P. S. White, eds. 1985. The ecology of natural disturbance and patch dynamics. New York: Academic Press.

Pielou, E. C. 1977. Mathematical ecology. New York: John Wiley & Sons.

Ritchie, M. E., and H. Olff. 1999. Spatial scaling laws yield a synthetic theory of biodiversity. Nature. 400: 557-560.

Romme, W. H. 1980. Fire frequency in subalpine forests of Yellowstone National Park. In: Proceedings of the Fire History Workshop; 1980 October 20-24; Tucson, AZ. Gen. Tech. Rep. RM-81 Ft. Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 27-30.

Romme, W. H. 1982. Fire and landscape diversity in subalpine forests of Yellowstone National Park. Ecological Monographs. 52: 199-221.

Rosenzweig, M. L. 1995. Species diversity in space and time. Cambridge, UK.: Cambridge University Press.

Ryan, K. C., and E. D. Reinhardt. 1988. Predicting postfire mortality of seven western conifers. Canadian Journal of Forest Research. 18: 1291-1297.

Skinner, C. N. 2000. Fire history of upper montane and subalpine lake basins in the Klamath Mountains of northern California. Paper read at Fire Conference 2000: National Congress on Fire Ecology, Prevention, and Management, at San Diego, CA.

Swetnam, T. W., and C. H. Baisan. 1996. Historical fire regime patterns in the southwestern United States since AD 1700. In: Allen, C.D., ed. Fire effects in southwestern forests: The Second La Mesa Fire Symposium; Los Alamos, NM; 1994 March 29-31. Gen. Tech. Rep. RM-286. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 11-32.

Swetnam, T. W., and J. L. Betancourt. 1990. Fire-Southern Oscillation relations in the southwestern United States. Science. 218: 165-173.

Swetnam, T. W., and J. L. Betancourt. 1992. Temporal patterns of El Niño/Southern Oscillation — wildfire teleconnections in the southwestern United States. In: Diaz, H.D.; Markgraf, V., eds. El Niño: Historical and paleoclimatic aspects of the Southern Oscillation. Cambridge, UK: Cambridge University Press: 259-270.

Swetnam, T. W., and J. H. Dieterich. 1985. Fire history of ponderosa pine forest in the Gila Wilderness, New Mexico. In: Lotan, J. E.; Kilgore, B. M.; Fischer, W. C.; Mutch, R. W., tech. coords. Proceedings—symposium and workshop on wilderness fire. Gen. Tech. Rep. INT-182. Missoula, MT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station: 390-397.

Touchan, R., C. D. Allen, and T. W. Swetnam. 1996. Fire history and climatic patterns in Ponderosa pine and mixed-conifer forests of the Jemez Mountains, northern New Mexico. In: Allen, C.D., tech. ed. Fire effects in southwestern forests: Proceedings of the second La Mesa fire symposium. Gen. Tech. Rep. RM-GTR-286. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 33-46.

Touchan, R., and T. W. Swetnam. 1995. Fire history in Ponderosa pine and mixed-conifer forests of the Jemez Mountains, northern New Mexico. Tucson, AZ. Report to the Santa Fe National Forest (USDA Forest Service) and Bandelier National Monument (USDI Park Service).

Veblen, T. T., T. Kitzberger, and J. Donnegan. 2000. Climatic and human influences on fire regimes in Ponderosa pine forests in the Colorado Front Range. Ecological Applications. 10 (4): 1178-1195.

Veblen, T. T., T. Kitzberger, R. Villalba, and J. Donnegan. 1999. Fire history in northern Patagonia: the roles of humans and climatic variation. Ecological Monographs. 69: 47-67.

Weaver, H. 1951. Fire as an ecological factor in the southwestern ponderosa pine forests. Journal of Forestry. 49: 93-98.

West, G. B., J. H. Brown, and B. J. Enquist. 1997. A general model for the origin of allometric scaling laws in biology. Science. 276: 122-126.

Whelan, R. J. 1995. The ecology of fire. Editors H. J. B. Birks and J. A. Weins. Cambridge studies in ecology. Cambridge, UK: Cambridge University Press.

White, P. S., and S. T. A. Pickett. 1985. Natural disturbance and patch dynamics: An introduction. In: White, P.S.; Pickett, T.A., eds. The ecology of natural disturbance and patch dynamics. New York: Academic Press: 3-13.

Wiens, J. A. 1989. Spatial scaling in ecology. Functional Ecology. 3: 385-397.

Uncertainty in Fire History and Restoration of Ponderosa Pine Forests in the Western United States

William L. Baker[1] and Donna S. Ehle[1]

Abstract—*Fire-history data for ponderosa pine forests in the western U.S. have uncertainties and biases. Targeting multiple-scarred trees and using recorder trees when sampling for fire history may lead to incomplete records. For most of the western U.S., research is insufficient to conclude that high-severity fires did or did not occur in these forests prior to EuroAmerican settlement, because the needed data are not commonly collected. The composite fire interval is shown here to be misleading, but this can be remedied in part with interval estimates by fire size class. These problems mean that an assumption—that high surface-fire frequencies will restore and maintain the structure of these forests—lacks a foundation in reliable fire-history research.*

Introduction

Restoration of fire in ponderosa pine forests depends upon fire-history data that are potentially biased and more uncertain than generally recognized (Minnich et al. 2000, Baker and Ehle 2001). Problems include a lack of modern calibration, inappropriate measures, targeted sampling, absence of fire-severity evidence, and insufficient treatment of variability and uncertainty (table 1). Some of these problems may be resolved quickly, while others will require longer study or may never be resolved. Here we highlight a few of the problems, suggest some remedies, and provide some thoughts regarding restoration of fire, given these problems.

No Modern Calibration

A significant problem plaguing fire-history research is a lack of modern calibration. Pollen studies, fire-history studies, and other paleo-ecological studies require calibration to determine whether evidence is preferentially preserved or lost and how it can be interpreted. Little is known about how fires leave evidence in the landscape over time. There is no way of knowing, without observing actual fires over time, whether it is possible to accurately reconstruct parameters (e.g., mean fire interval) of the fire regime from fire scars, and, if so, how to sample to best accomplish this. Calibration may allow corrections to be derived that enable reasonably accurate reconstructions.

One calibration approach might be to use fire boundaries reconstructed using aerial photographs (e.g., Minnich et al. 2000) or use other historical records, such as atlases of past fires. This would be particularly valuable if multiple approaches to sampling on the ground were compared to aerial-photo

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Table 1—Some limitations, potential biases, and uncertainties in fire-history studies in ponderosa pine forests.

<u>No modern calibration</u>		
	Only know that some historical fires can be detected	
<u>Biases</u>		
	Targeted sampling	
		Trees with multiple fire scars
		Places with high fire-scar densities
		Old trees or forests with long fire records; avoid young trees and forests
		Trees with open scars
	Fire severity unstudied, but assumed to be low	
		Necessary age-structure data not collected
	Analysis and treatment of fire-scar data	
		Recorder trees-do they work?
		Only scar-to-scar intervals included
		Compositing is biased toward smaller fires
<u>Uncertainties</u>		
	Fire perimeters unknown	
	Fire record is uncertain due to unrecorded fires and unburned area within fire perimeters	
	Variability in fire-intervals is large and seldom explicitly treated	
	Large variability means sample sizes provide insufficient power for comparisons	
	Bracketing and confidence intervals are warranted	

or map estimates. However, photographs and historical sources also have limitations and biases. Small fires may be undetectable in typical aerial photographs, and dating to single years is usually not possible (Minnich et al. 2000). There is no research program at the present time to actually undertake this calibration work, but it is surely needed.

In lieu of calibration, all that can be done is to work with sampling designs, sample sizes, and analysis techniques to see how the sampling estimates vary relative to a more complete sample. Some of this relative comparison work has been underway (Baker and Ehle 2001), but even this work is in its infancy. New sampling designs are being proposed and studied (e.g., Arno et al. 1993, Heyerdahl et al. 2001). There are promising signs that in a few years we will know how to sample in the most efficient, unbiased manner.

Potential Biases and Uncertainties

Targeting Multiple-Scarred Trees

Fire-history researchers have seldom sampled randomly or in an unbiased manner. Instead, they typically and purposely seek trees containing multiple scars and places that contain high scar densities (table 1). These are assumed to increase the length of the record and maximize identification of the fires that burned a stand. However, no study has actually compared the fires identified through targeting with those on non-targeted trees, or examined the effects of targeting on estimates of fire intervals in ponderosa pine forests.

To compare how targeted and non-targeted trees record fires and fire intervals, we sampled all visible scars on trees in nine plots randomly placed within the ponderosa pine zone in Rocky Mountain National Park (Ehle and Baker, in press). A total of 137 scarred trees was sampled. All fire scars were visually crossdated using a master chronology. Most trees had a single fire scar, but six trees had four or more scars per tree (figure 1). Trees with four or more scars are those that typically would have been selected for sampling using a targeting approach, based on a review of ponderosa pine fire histories (Baker and Ehle 2001). These six trees contained a total of 35 fire scars. We randomly selected an equal sample of 35 scars from trees that would not have been

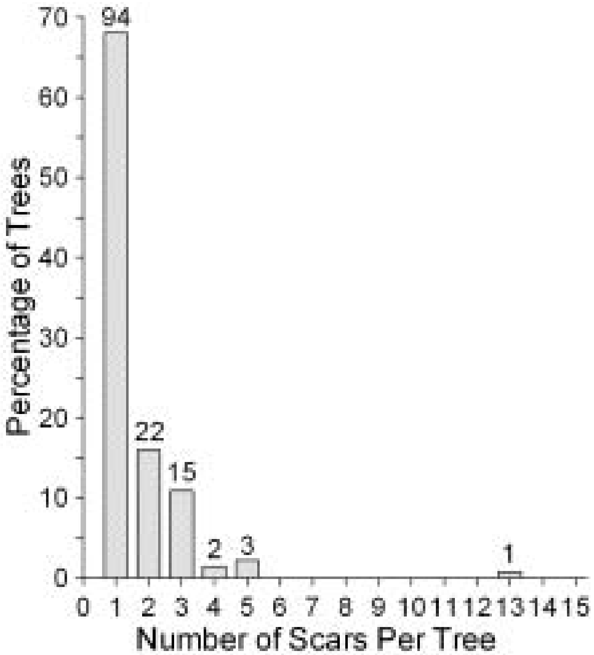

Figure 1–Percentage of sampled, fire-scarred trees (n=137) that have one or more than one scar per tree. The number of trees is listed above each bar.

targeted (trees containing £3 scars). A third sample of 35 scars was obtained from single-scarred trees. Individual trees did not occur in more than one of these samples.

Then, we separated the fires that were identified by these scars into five combined size and severity classes (figure 2; see also Ehle and Baker, in press). Low-severity fires leave numerous surviving trees, while mixed-severity fires leave only a few survivors in a plot, or are high-severity fires in part of a landscape and low-severity elsewhere (Ehle and Baker, in press). Small fires in this study scar more than one tree, and are not known to have spread beyond a 50 m X 50 m plot, but could have been as large as 1.2 km^2. Large fires burned >1.2 km^2.

The targeted sample identified more fires (n = 29) than did the single-scarred trees (n = 20) or the non-targeted sample (n = 16) even though the

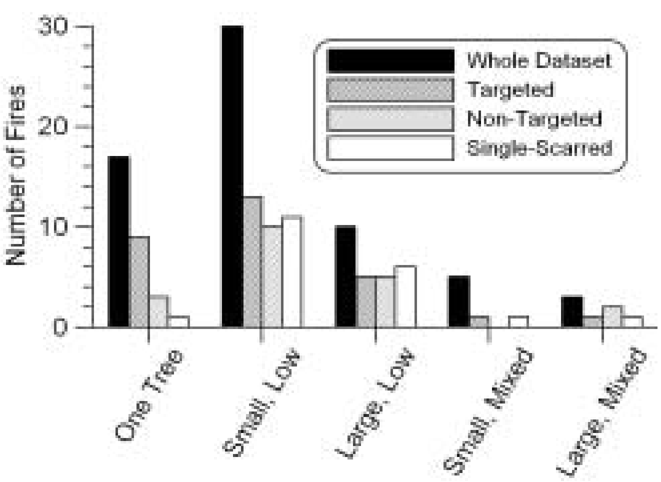

Figure 2–Effects of targeted sampling on the number of detected fires for fires of different sizes and severities. Small fires likely do not exceed the area of a sampling plot (50 m X 50 m), while large fires burn > 1.2 km^2.

number of scars was 35 in all cases. The fires identified by the samples can be compared to the total set of 60 fires identified by the 137 scarred trees in the nine sampled plots ("Whole Dataset" in figure 2). The targeted sample generally identified more of the small fires affecting only one tree and the small, low-severity fires, while the non-targeted sample and single-scarred trees identified few one-tree fires, but did as well or slightly better at identifying large, low-severity fires and mixed-severity fires (figure 2). Seventeen one-tree fires occurred in the nine plots (each of 0.25 ha) over a period of about 300 years, which is a rate of about one tree / ha scarred by fire every 40 years, an insignificant amount. If one-tree fires are ignored, there is not much difference among the samples in ability to detect fires of different size and severity.

However, an important difference is that the targeted sample comes from only six trees, while the single-scarred sample comes from 35 trees. Less effort is required to obtain the 35 scars from only six trees than from 35 single-scarred trees. However, 35 trees provide a much better spatial sample of where the fires burned, thus making it possible to more correctly identify fire size and severity (if age-structure data are also collected). If 35 trees can be sampled in either case, many more fires will be detected with a targeted sample of trees containing >4 scars than with a sample of single-scarred trees.

In our review (Baker and Ehle 2001), we expressed concern that fire intervals identified in a targeted sample might be much shorter on average than in a non-targeted sample. To test this, we used the same sets of samples from targeted, non-targeted, and single-scarred trees, each sample containing 35 fire scars. Then, we listed all the fires and fire intervals identified by each sample of 35 fire scars, and used an ANOVA (done using Minitab 12.1; Minitab, Inc. 1998) to test the null hypothesis that the mean fire interval for small, low-severity fires is equal regardless of sampling technique. While fire-interval data can have non-normal distributions, parametric statistical tests remain valid (Johnson 1995). We repeated the ANOVA for large, low-severity fires. Comparisons for mixed-severity fires are not possible due to small sample sizes (figure 2). The null hypothesis cannot be rejected for small, low-severity fires ($F = 0.21$, $p = 0.810$) or large, low-severity fires ($F = 0.00$, $p = 0.997$).

While the sample from multiple-scarred trees may not be biased in this regard, multiple-scarred trees alone will not identify all the fires in a stand. Three of the 60 fires were only found on single-scarred trees, five were only on double-scarred trees, and three were only on triple-scarred trees, all of which would be missed if trees containing four or more scars were targeted. Of these 11 fires (18% of the 60 fires), two were one-tree fires (figure 2), but eight were small, low-severity fires, while one was a significant high-severity fire. Three of these 11 fires occurred near or before AD 1700 and documented 30% of the 10 ancient fires found in the study area. Researchers seeking complete fire histories or long fire histories will miss important fires and ancient fires if only multiple-scarred trees are sampled, at least in this study area.

We conclude that targeting multiple-scarred trees in this case study does not produce a biased estimate of the fires that occurred in a larger sample or a biased estimate of the mean fire interval relative to that found with other samples. However, fire histories derived from targeted sampling may be incomplete, particularly missing some important fires and ancient fires.

However, this one small study is insufficient to draw strong conclusions about targeting. Fire intervals in this case study are quite variable, and the test, as a result, may not have much statistical power. Further testing is needed before these results are applied elsewhere. The other potentially significant targeting biases (Baker and Ehle 2001) also need testing. Moreover, until there is a modern calibration, the possibility remains that these sampling approaches simply produce equally biased estimates of fire intervals and other parameters of fire regimes.

Crown Fires and Mixed-Severity Fires Not Sampled in Ponderosa Pine Forests

If restoration of fire in ponderosa pine forests is to be successful, historical variability in fire severity must also be known. The evidence needed to determine fire severity is a combination of fire-scar data and age-structure data near each scar. Low-severity fires generally lead to low mortality of larger, established trees. High-severity fires can lead to pulses or a cohort of post-fire regeneration (Ehle and Baker, in press). A mixed-severity fire has a high-severity, crown-fire component and an associated low-severity component.

A fire scar alone, or even multiple fire scars across a landscape, reveal little about the severity of the fire. Fire scars indicate only that a fire was on the surface at the scarred tree itself. This tree could be a lone survivor of a fire that was in the crown of every other tree in the surrounding landscape. Scattered surviving trees are not uncommon in crown-fire landscapes (e.g., Kipfmueller and Baker 1998). The fire may have also have been mixed-severity, burning on the surface over a part of the landscape where the scar was found, and then crowning out in patches (e.g., Huckaby et al. 2001).

The idea that surface fires predominate in ponderosa pine forests has been so pervasive that fire-history researchers commonly study fires in these forests without collecting age-structure data, then erroneously conclude that it is known that surface fires predominate or that crown fires did not occur. Some researchers have even implied that, if fire-scars are present and ponderosa pine is present, this indicates that the fire regime sustained only low-severity surface fires (Heyerdahl et al. 2001). This is false, as crown fires in ponderosa pine forests can be followed within a few decades by surface fires as the stand develops (Ehle and Baker, in press).

Thirty-nine studies constitute nearly all the published scar-based fire-history research on pure ponderosa pine forests in the western United States (Baker and Ehle 2001). Only nine of the 39 collected the age-structure data needed to determine whether fire severity was low, medium, or high (table 2). Four other studies collected age structure, but not fire-scar data. These 13 studies with age-structure data reveal three general patterns. First, some studies of small areas or plots reveal an uneven age structure, often with apparent pulses of regeneration separated by gaps in regeneration, suggesting an absence of crown fires. Regeneration pulses in these plots are sometimes linked to variations in surface-fire frequency (Arno et al. 1995, 1997; Morrow 1986) or a combination of fire and climate (Cooper 1960), or they cannot presently be explained (Mast et al. 1999, White 1985). Second, some plots contain an even age structure, characterized by large pulses of regeneration commencing after a date identified on a nearby fire scar, suggesting a crown fire at the level of the plot (Arno et al. 1995, 1997; Mast et al. 1998). Brown and Sieg (1996) thought that ages of scarred trees in one plot were roughly synchronous, suggesting a possible crown fire or a climatic event. Age data (apparently collected but not presented) suggest that infrequent stand-replacing fires occurred in some parts of two study areas prior to EuroAmerican settlement (Barrett 1988, Swetnam and Baisan 1996b).

Third, more extensive landscape-scale studies that include multiple plots across an area of a few thousand hectares have revealed a mixed- or high-severity fire regime in the pre-EuroAmerican era. This was found in pure ponderosa pine landscapes of Rocky Mountain National Park, Colorado (Ehle

Table 2–Evidence of mixed-severity and high-severity (crown) fires in the pre-EuroAmerican period from studies of ponderosa pine fire history and age structure in the western United States.

	Age data[a]	Historical data[b]	Fire scar data	Comments on crown fires
Northwestern U.S.				
Bork 1984	No	No	Yes	No
Heyerdahl 1997	No	No	Yes	They did not occur because surface fires did occur.
Morrow 1986	Yes	No	Yes	No, uneven age structure with pulses of regeneration linked to low fire frequency
Sherman 1969	No	No	Yes	No
Soeriaatmadja 1966	No	No	Yes	Yes, they probably occurred on higher elevation, more moist sites
Weaver 1943	No	Yes	No	Yes, direct observation of even-aged stands suggesting past crown fires.
Northern Rockies				
Arno 1976	No	No	Yes	No
Arno and Petersen 1983	No	No	Yes	No
Arno et al. 1995	Yes	No	Yes	Yes, one stand of six dry-site stands and some wet-site stands
Arno et al. 1997	Yes	No	Yes	Some dry-site ponderosa pine forests must have experienced occasional stand replacement fires
Barrett 1988	Yes	No	Yes	Yes, infrequent stand-replacing fires are possible in upper elevations
Freedman and Habeck 1985	No	Yes	Yes	Yes, early historical observations suggest they occurred
Steele et al. 1986	No	No	Yes	Yes, hypothesizes that they occurred in the past during periods of drought and high winds.
Black Hills				
Brown and Sieg 1996	Scars	No	Yes	Yes, they were possible, but not verified; climate an alternative cause of regeneration events
Brown and Sieg 1999	No	No	Yes	No
Brown et al. 2000	No	No	Yes	No
Shinneman and Baker 1997	No	Yes	No	Historical records document large stand-replacing fires, particularly in the moister northern Black Hills
Southern Rockies				
Brown et al. 1999; Kaufmann et al. 2000; Huckaby et al. 2001	Yes	No	Yes	Yes, 71% of sampled polygons had stand-replacing fires
Brown et al. 2000	No	No	Yes	No
Ehle 2001; Ehle and Baker, in press	Yes	No	Yes	Yes, in 6 of 9 plots
Goldblum and Veblen 1992	No	No	Yes	Yes, but only in post-settlement
Laven et al. 1980	No	No	Yes	No
Mast et al. 1998	Yes	No	Yes	Even-aged cohorts and post-fire pulses of establishment, but linked to gaps or spot fires (crown fires)
Rowdabaugh 1978	No	No	Yes	No
Skinner and Laven 1982	No	No	Yes	No
Veblen and Lorenz 1986, 1991	Yes	Yes	No	Age structures and early photographs that show crown fires that occurred near or before EuroAmerican settlement
Veblen et al. 2000	No	Review	Yes	Yes, early photographs show them, and fire intervals are long enough to allow them at higher elevations
Southwestern U.S.				
Cooper 1960	Yes	Yes	No	No evidence of crown fires except possibly on a part of the Prescott National Forest
Dieterich 1980a	No	No	Yes	No
Dieterich 1980b	No	No	Yes	No
Dieterich and Hibbert 1990	No	No	Yes	No
Fule et al. 1997	No	No	Yes	No
Grissino-Mayer 1995	No	No	Yes	No
Madany and West 1980	No	No	Yes	No
Mast et al. 1999	Yes	No	No	Same site studied by White (1985); uneven age structure with pulses of regeneration not clearly linked to either climate or fire.
McBride and Jacobs 1980	No	No	Yes	No
McBride and Laven 1976	No	No	Yes	No
Morino 1996	No	No	Yes	No
Savage 1989; Savage and Swetnam 1990	No	No	Yes	No
Stein 1988	No	No	Yes	No
Swetnam and Baisan 1996a	No	No	Yes	No
Swetnam and Baisan 1996b	Yes	No	Yes	Yes, some evidence in dates of tree mortality and tree recruitment relative to fires synchronous over large areas
Swetnam and Dieterich 1985	No	No	Yes	No
Touchan et al. 1995	No	No	Yes	No
Touchan et al. 1996	No	No	Yes	No
White 1985	Yes	No	No	No, uneven age structure with pulses of regeneration

[a]Sufficient tree age data to be able to identify a crown fire in the pre-EuroAmerican period.
[b]Early photographs or historical observations from near or before settlement by EuroAmericans.

and Baker, in press) and in mixed-conifer landscapes with considerable ponderosa pine dominance at Cheesman Lake, Colorado (Brown et al. 1999, Kaufmann et al. 2000, Huckaby et al. 2001). In the Rocky Mountain National Park study, six of nine plots had stand-replacing fires and another plot had a stand-replacing event caused by an unidentified agent (Ehle and Baker, in press). In the Cheesman Lake study, 71% of sampled polygons had stand-replacing fires (Huckaby et al. 2001). Fires in both landscapes often were mixed-severity at the landscape scale, burning as surface fires in some areas and then crowning over other areas. Both studies reported that smaller parts of these landscapes contained uneven-aged stands with no evidence of crown fires for the past few hundred years.

Studies that use historical records or early photographs also found that crown fires occurred in some ponderosa pine forests, but not others, prior to EuroAmerican settlement (table 2). Shinneman and Baker (1997) reviewed historical evidence of extensive crown fires in the moister parts of the Black Hills, and Freedman and Habeck (1985) also noted historical evidence of crown fires prior to EuroAmerican settlement in a valley in western Montana. In early historical photographs Veblen and Lorenz (1991) could see ponderosa pine landscapes that were burned in stand-replacing fires some time before EuroAmerican settlement. Cooper (1960) reported that a review of early literature failed to find evidence of crown fires in ponderosa pine forests in Arizona before 1900, except on part of the Prescott National Forest. There is no further explanation of the Prescott case. Weaver (1943 p. 9), describing a broad region in the Pacific Northwest, simply stated that "extensive even-aged stands of ponderosa pine can probably be accounted for by the past occurrence of severe crown fires, by severe epidemics of tree-killing insects...or by the occurrence of extensive windthrows..." A more extensive review of early historical reports and photographs might reveal where stand-replacing fires had or had not occurred prior to EuroAmerican settlement.

For most of the ponderosa pine forests of the western United States there are no data at all that would allow a determination of whether crown fires or mixed-severity fires were present or absent before EuroAmerican settlement, or have increased or decreased. Where studies have been done or historical data were examined, crown fires or mixed-severity fires were sometimes found and sometimes not (table 2). There is a hint in these data that crown- or mixed-severity fires may occur on moister sites in the ponderosa pine zone.

No one, in any study anywhere in the West, has yet estimated how frequent crown- or mixed-severity fires were in ponderosa pine forests, how large these fires may have been, or what the fire rotation for these fires might have been prior to EuroAmerican settlement. The data are perhaps there to allow this estimation for study sites at Cheesman Lake, Colorado (Huckaby et al. 2001) and in Rocky Mountain National Park (Ehle and Baker, in press). These study areas, however, are small relative to the size of some recent fires (e.g., Hayman Fire, 2002). Larger areas have been logged or burned, destroying the evidence of past fires. It may be difficult or impossible to determine whether large, high-severity fires did or did not occur in ponderosa pine landscapes prior to EuroAmerican settlement.

Given the lack of data, there is little basis for the general perception that high- or mixed-severity fires, such as the 2000 fire that burned into Los Alamos, New Mexico, are not natural in ponderosa pine forests (Allen 2002). The conclusion that a particular fire is unnaturally severe is premature given the absence of the necessary data. For nearly all the ponderosa pine forests in the western United States it would also be premature to suggest that treatments that lower the probability of crown fire or high-severity fire or lower fire risk are "restoration." For most of the range of ponderosa pine in the West it is not yet known whether these kinds of fires were or were not a part of the pre-EuroAmerican fire regime. Where crown fires occurred, thinning may be an inappropriate restoration technique, just as it is inappropriate in some pinyon-juniper woodlands (Romme et al., this volume). In some cases, restoration might even require reintroduction of high-severity fires, if they were unnaturally suppressed.

Analysis and Treatment of Fire-Scar Data

Recorder Trees—Do They Work?

It has long been thought that until a tree receives a fire scar, it is a poor recorder of fires. Thus, fire historians often do not consider a stand to be generally capable of recording the fires that occur in a stand until after some number of trees has received a first scar (e.g., 3; Grissino-Mayer 1995). The idea of a previously scarred "recorder tree" is that if there is an open scar, subsequent fires should be more effectively recorded than if fires must produce the first scar. If recorders work, fires should show up more often on recorder trees than as a first scar.

In our complete sample from 137 scarred trees, we found 60 fires. Nineteen of these fires (31.7%) show up only as first scars, while 17 fires (28.3%) show up only as scars after the first scar (i.e., on recorder trees). This result could occur if previously scarred trees are actually no better recorders or if different fires affected the recorder trees and the trees with first scars. However, 24 fires (40%) show up as a mixture of first scars and scars on recorder trees. Ninety-six of the 154 total scars (62%) documenting these 24 fires are first scars while only 58 of the 154 scars (38%) occur on recorder trees. A chi-square test leads to rejection of the null hypothesis that recorder trees and trees without scars are equal recorders of fires when fires show up on both ($\chi^2 = 4.761$, $p = 0.029$). In our study area, previously scarred trees are poorer recorders of fire than are unscarred trees. Previously scarred trees do not perform as commonly expected, perhaps because multiple factors influence whether a fire produces a scar. Smaller trees, for example, typically have thinner bark, which offers less resistance to scarring, perhaps making them better recorders than are larger trees. Our results suggest that if a complete history is desired, fire-history data should be collected and used whether a tree is or is not previously scarred. Fire-history studies that only use recorder trees may miss a significant part of the fire history.

Which Intervals Should Be Used?

Fire historians nearly always have focused on scar-to-scar (SS) intervals recorded on trees, omitting the interval between tree origin and the first scar (OS interval; Baker and Ehle 2001) as well as the interval between the last scar and tree death or the present. Yet, the OS interval estimates the real fire-free interval needed for trees to reach a size sufficient to survive surface fires (Baker and Ehle 2001). Since the OS interval does not necessarily begin with a fire, the real fire-free interval may be underestimated by the OS interval.

The OS interval is typically longer than the SS interval (Baker and Ehle 2001). In our sample of 137 fire-scarred trees from Rocky Mountain National Park's ponderosa pine zone (figure 3), the pre-EuroAmerican OS interval on individual trees (n = 71) has a mean of 55.4 years and an estimated median of 51.5 years. The pre-EuroAmerican SS intervals on individual trees (n = 40), in

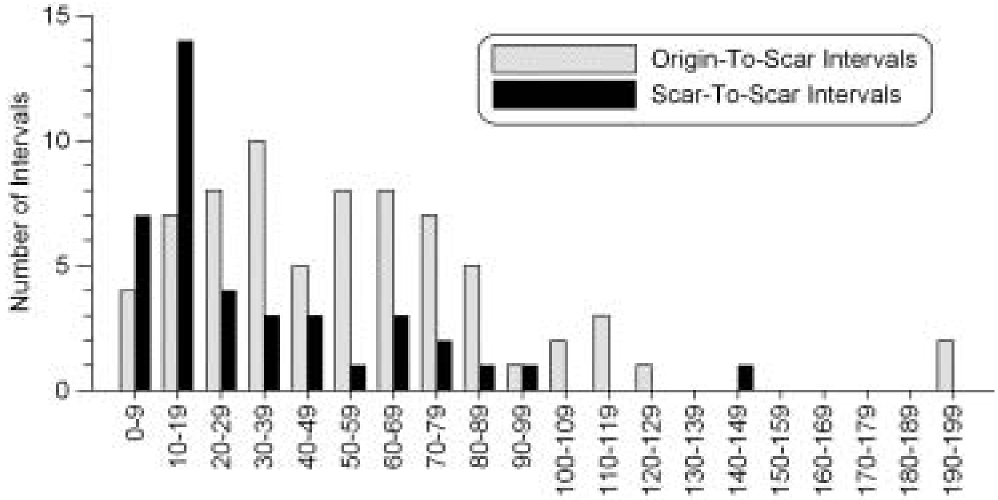

Figure 3–Distribution of pre-EuroAmerican fire-scar intervals for individual trees from a sample of 137 fire-scarred trees in ponderosa pine forests of Rocky Mountain National Park.

contrast, have a mean of 33.3 years and an estimated median of 28.5 years. The estimated difference in means is 22.1 ±13.2 years (95% confidence interval). The regression equation in Baker and Ehle (2001) for estimating the OS interval, if only the SS interval is known, suggests that the mean OS interval would be 53.5 years for a mean SS interval of 33.3 years, reasonably close to the 55.4 years actually found. The OS interval should be included as a real fire interval, and including it generally lengthens the estimated mean fire interval by about 1.6 times (Baker and Ehle 2001).

Compositing Biased Toward Small Fires

The mean "Composite Fire Interval" or CFI (Dieterich 1980a) is the traditional measure of central tendency in fire intervals, but this measure is flawed as a general measure of the fire regime (Baker and Ehle 2001). One problem is that the CFI pools fires of different extent and frequency. Regardless of the real mean fire interval in a landscape, the mean CFI decreases as the number of sampled fire-scarred trees and sampled area increase (Arno and Petersen 1983). The reason is that the numerous fires that scar only one tree (e.g., figure 2) are counted the same as an infrequent fire that scars many trees (Minnich et al. 2000, Baker and Ehle 2001). By adding sampling area or sampled trees, one quickly adds these apparently small fires. As a result, a CFI can be interpreted as mostly reflecting the frequency of small fires that affect little of the landscape.

A remedy for this shortcoming of a CFI is to analyze and report fire intervals separately for individual classes of fire size. Laven et al. (1980) may have been the first to use this approach for ponderosa pine forests when they reported separate intervals for small fires and large fires. Bork (1984) showed means and standard errors for fires varying in size from 1 plot to 5 plots (figure 4). Morino (1996) calculated separate fire-interval distributions and descriptive parameters (e.g., mean) for small fires, medium fires, and large fires. Mean fire intervals for larger fires in ponderosa pine forests are 41.7 years (Laven et al. 1980), 60-150 years (Bork 1985 and figure 4), and 24.4 years (Morino 1996), while the corresponding mean fire intervals for small fires in these studies are 20.9 years, 5.25 years, and 2.7 years, respectively. Thus, larger fires in these cases have mean intervals that are 2-10 times as long as are mean intervals for smaller fires. These estimates are imprecise, but illustrate that the mean fire interval for the fires that do most of the work in ponderosa pine forests is much longer than suggested by typical CFIs.

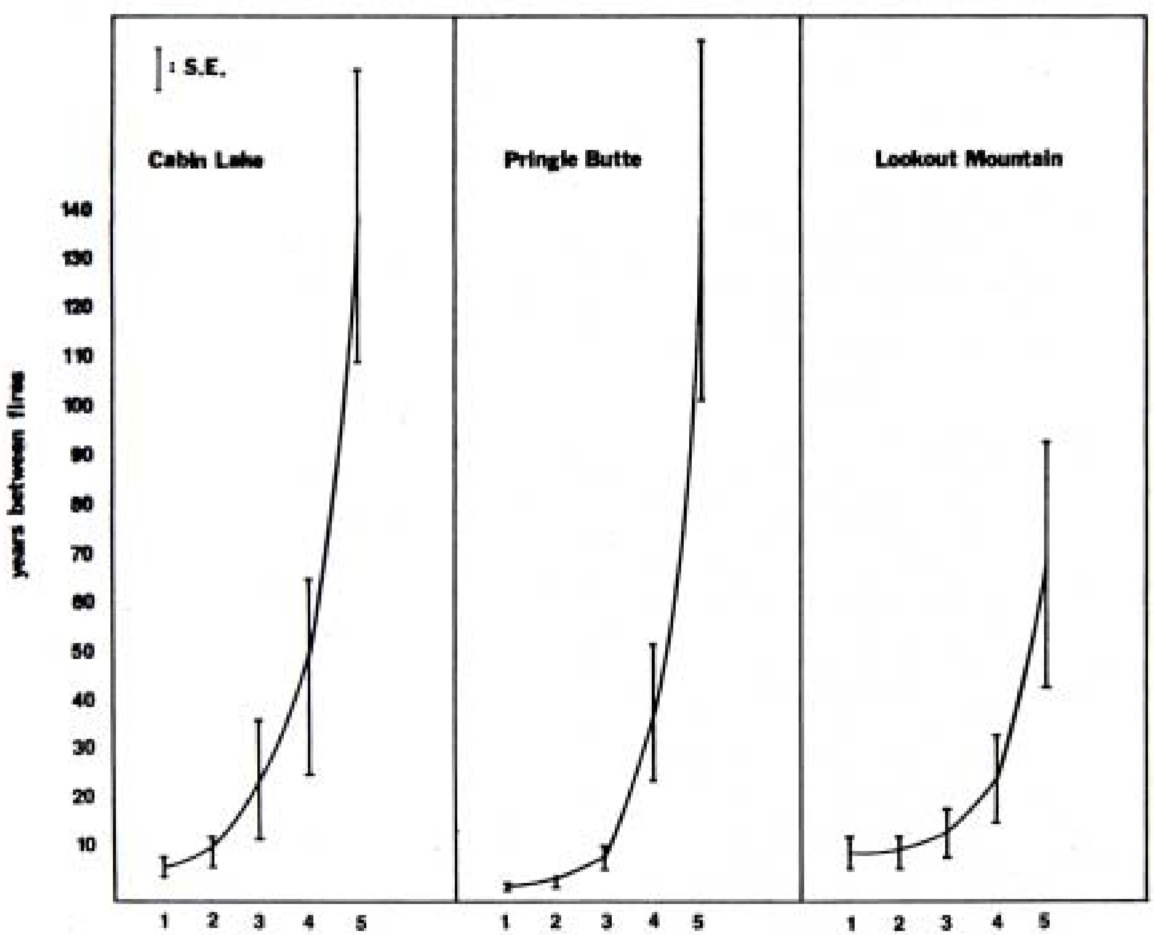

Figure 4–Mean return interval for fires of different size from three sites in eastern Oregon, estimated by proportion of plots having at least two fire-scarred trees. Reproduced from Bork (1985) Figure I-24 with permission from Joyce L. Bork.

Our review of 11 studies in the western United States show that about 50% of known fires are documented by a scar on only one tree (Baker and Ehle 2001). Given that these fires affect little land area, but dominate the CFI, we particularly suggest that the frequency of one-tree fires should be reported separately.

The idea that there is value in reporting intervals for fires of different sizes underlies the now-popular reporting of interval data for all fires compared to those that scar >10% or >25% of recorder trees (Grissino-Mayer 1995). However, it is not progressive restriction (sizes exceeding a certain size) that is needed, but separate reporting of intervals for each size class. Reporting a CFI for study areas of increasing size (e.g., Brown et al. 1999) is also not what is needed, as it is well known that CFIs decrease as study area size increases, even if the fire regime is the same across scales (Arno and Petersen 1983).

Separating fire-intervals by fire size also allows estimation of the fire rotation, a fundamental measure of the fire regime (Minnich et al. 2000, Baker and Ehle 2001). Data on the relative frequency and importance of fires of different sizes are invaluable for fire managers, as this information can be used directly in prescribed burning plans, regardless of the size of the management area. This is not the case for the traditional CFI, which is heavily dependent on the size of the study area in which the CFI was calculated (Arno and Petersen 1983, Baker and Ehle 2001).

What are appropriate fire size classes to use? Even reporting intervals based on number of affected plots (figure 4), or linear distances between plots, would be an improvement over a traditional CFI. Size classes used by the U.S. Forest Service and other agencies would be advantageous, as data from fire-history studies could then be compared to contemporary data from monitoring programs. Where fire-history data are insufficient to make fine distinctions in fire size, pooling of adjacent categories would still allow useful comparisons with modern data, particularly if small fires are segregated from large fires.

Unfortunately, it is difficult to estimate the size of surface fires using fire scars. Grid-based or random sampling methods are increasingly making it possible to approximate fire extent (Arno et al. 1993, Heyerdahl et al. 2001, Morino 1996). However, there is not yet a calibration to guide correction of size estimates from spatial sampling networks or sufficient study of appropriate spatial sampling designs for detecting fire sizes. Until this calibration and sampling design work is done, a method to bracket the potential uncertainty associated with assigning fires to size classes is needed.

Uncertainties

Fire intervals vary, and this variability is often large within a single tree and among all the intervals within a stand (e.g., figure 3). This variability suggests that fire intervals are not predictable results of the time for fuel to build up after a fire; fire intervals are shaped by the timing of weather that promotes fine-fuel accumulations and the timing of droughts (Veblen et al. 2000). This variability in fire intervals makes comparison of sets of fire intervals from different periods or different sites difficult, as sample sizes must be large to be able to detect even 50% or 100% differences in mean with adequate statistical power (Baker and Ehle 2001). However, few researchers have actually used statistical inference, instead simply presenting the sample data. Previous evidence that fire intervals have changed over time or differ among sites may not bear up under statistical analysis, except where the change is obvious, as when fires appear to virtually stop near or after settlement (e.g., Savage and Swetnam 1990).

Fire-interval data also have uncertainty that comes from at least two sources– unrecorded fires and unburned area within fire perimeters. There is presently no method to estimate the magnitude of these sources of uncertainty in a particular stand or area. Baker and Ehle (2001) thus suggest that all estimates of mean or median fire intervals should be bracketed using the restricted (>10% scarred) CFI and individual-tree mean fire intervals. However, if fire intervals are reported separately by fire size, as we recommend here, then the appropriate brackets for the estimate of mean fire interval for a stand are the unrestricted composite and individual-tree fire intervals.

Implications for Restoration

Fire-history research methods are in need of reassessment, as traditional measures are misleading or in error as sources of information useful for designing a program for restoring fire in ponderosa pine forests. The time that it took for fire to burn through these forests prior to EuroAmerican settlement is much longer than is implied by typical composite fire intervals, which have been reported to be between 2-25 years (Baker and Ehle 2001). The large fires, that actually account for most burned area, occur at intervals that are several times longer than reported composite fire intervals. Baker and Ehle argued that the population mean fire interval in western ponderosa pine forests is instead more likely to lie between 22-308 years. However, until there is a modern calibration and further testing of the potential biases and uncertainties we have identified, it would be premature to draw strong conclusions about what the fire intervals were in pre-EuroAmerican ponderosa pine forests.

Our analysis suggests that repeated prescribed burning of large areas of ponderosa pine forests at short intervals (e.g., less than 20 years) lacks a sound basis in science, and should not be done at the present time if the goal is restoration. In most parts of the western United States there is also insufficient evidence to support the idea that mixed- or high-severity fires were or were not absent or rare in the pre-EuroAmerican fire regime. Thus, programs to lower the risk of mixed- or high-severity fires in ponderosa pine forests (e.g., the National Fire Plan, Laverty and Williams 2000) have insufficient scientific basis if the goal is restoration.

Fire practitioners interested in restoration can certainly proceed with reintroducing fire into these forests on a limited basis, however. In many areas, fire has been excluded by livestock grazing or intentional suppression for a long period. We suggest that prescribed burning a large area once is not likely to push the ecosystem outside its historical range of variability. Reintroduction of small prescribed fires that burn a single tree or a few trees in a landscape is also appropriate, at least in our study area. However, prescribed burning of large land areas after short intervals (e.g., <20 years) has little scientific basis at the present time, if the goal is to restore the natural variability of the pre-EuroAmerican fire regime.

References

Allen, C. D. 2002. Lots of lightning and plenty of people: An ecological history of fire in the upland Southwest. In: Vale, T. R., ed. Fire, native peoples, and the natural landscape. Washington, DC: Island Press: 143-193.

Arno, S. F. 1976. The historical role of fire on the Bitterroot National Forest. Res. Pap. INT-187. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 29 p.

Arno, S. F.; Petersen, T. D. 1983. Variation in estimates of fire intervals: a closer look at fire history on the Bitterroot National Forest. Res. Pap. INT-301. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 8 p.

Arno, S. F.; Reinhardt, E. D.; Scott, J. H. 1993. Forest structure and landscape patterns in the subalpine lodgepole pine type: a procedure for quantifying past and present conditions. Gen. Tech. Rep. INT-294. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 17 p.

Arno, S. F.; Scott, S. F.; Hartwell, M. G. 1995. Age-class structure of old growth ponderosa pine / Douglas-fir stands and its relationship to fire history. Res. Pap. INT-RP-481. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 25 p.

Arno, S. F.; Smith, H. Y.; Krebs, M. A. 1997. Old growth ponderosa pine and western larch stand structures: Influences of pre-1900 fires and fire exclusion. Res. Pap. INT-RP-495, Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 20 p.

Baker, William L.; Ehle, Donna. 2001. Uncertainty in surface-fire history: the case of ponderosa pine forests in the western United States. Canadian Journal of Forest Research. 31: 1205-1226.

Barrett, S. W. 1988. Fire suppression's effects on forest succession within a central Idaho wilderness. Western Journal of Applied Forestry. 3: 76-80.

Bork, J. L. 1985. Fire history in three vegetation types on the eastern side of the Oregon Cascades. On file at: Oregon State University, Corvallis, OR. Dissertation.

Brown, P. M.; Kaufmann, M. R.; Shepperd, W. D. 1999. Long-term, landscape patterns of past fire events in a montane ponderosa pine forest of central Colorado. Landscape Ecology. 14: 513-532.

Brown, P. M.; Ryan, M. G.; Andrews, T. G. 2000. Historical surface fire frequency in ponderosa pine stands in Research Natural Areas, central Rocky Mountains and Black Hills, USA. Natural Areas Journal. 20: 133-139.

Brown, P. M.; Sieg, C. H. 1996. Fire history in interior ponderosa pine communities of the Black Hills, South Dakota, USA. International Journal of Wildland Fire. 6: 97-105.

Brown, P. M.; Sieg, C. H. 1999. Historical variability in fire at the ponderosa pine—Northern Great Plains prairie ecotone, southeastern Black Hills, South Dakota. Ecoscience. 6: 539-547.

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

Dieterich, J. H. 1980a. The composite fire interval—a tool for more accurate interpretation of fire history. In: Stokes, M. A.; Dieterich, J. H., tech. coords. Proceedings of the fire history workshop. Gen. Tech. Rep. RM-81. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 8-14.

Dieterich, J. H. 1980b. Chimney Spring forest fire history. Res. Pap. RM-220. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 8 p.

Dieterich, J. H.; Hibbert, A. R. 1990. Fire history in a small ponderosa pine stand surrounded by chaparral. In: Krammes, J. S., tech. coord. Effects of fire management of southwestern natural resources: Proceedings of the symposium. Gen. Tech. Rep. RM-191. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 168-173.

Ehle, D. S. 2001. Spatial and temporal patterns of disturbance and ponderosa pine forest structure in Rocky Mountain National Park. On file at: University of Wyoming, Laramie, Wyoming. 100 p. Thesis.

Ehle, D. S.; Baker, W. L. [In press]. Disturbance and stand dynamics in ponderosa pine forests in Rocky Mountain National Park. Ecological Monographs.

Freedman, J. D.; Habeck, J. R. 1985. Fire, logging, and white-tailed deer interrelationships in the Swan Valley, northwestern Montana. In: Lotan, J. E.; Brown, J. K., eds. Fire's effects on wildlife habitat—Symposium proceedings. Gen. Tech. Rep. INT-186, Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 23-35.

Fulé, P. Z., Covington, W. W.; Moore, M. M. 1997. Determining reference conditions for ecosystem management of southwestern ponderosa pine forests. Ecological Applications. 7: 895-908.

Goldblum, D.; Veblen, T. T. 1992. Fire history of a ponderosa pine / Douglas fir forest in the Colorado Front Range. Physical Geography. 13: 133-148.

Grissino-Mayer, Henri. 1995. Tree-ring reconstructions of climate and fire history at El Malpais National Monument, New Mexico. On file at: University of Arizona, Tucson, AZ. 407 p. Dissertation.

Heyerdahl, Emily K.; Brubaker, L. B.; Agee, James K. 2001. Spatial controls of historical fire regimes: a multiscale example from the Interior West, USA. Ecology. 82: 660-678.

Huckaby, L. S.; Kaufmann, M. R.; Stoker, J. M.; Fornwalt, P. J. 2001. Landscape patterns of montane forest age structure relative to fire history at Cheesman Lake in the Colorado Front Range. In: Vance, R. K.; Edminster, C. B.; Covington, W. W.; Blake, J. A., comps. Ponderosa pine ecosystems restoration and conservation: steps toward stewardship. Proceedings RMRS-P-22. Fort Collins, CO: U.S.

Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 19-27.

Johnson, D. H. 1995. Statistical sirens: The allure of nonparametrics. Ecology 76: 1998-2000.

Kaufmann, M. R.; Regan, C. M.; Brown, P. M. 2000. Heterogeneity in ponderosa pine / Douglas-fir forests: age and size structure in unlogged and logged landscapes of central Colorado. Canadian Journal of Forest Research. 30: 698-711.

Kipfmueller, K. F.; Baker, W. L. 1998. Fires and dwarf mistletoe in a Rocky Mountain lodgepole pine ecosystem. Forest Ecology and Management. 108: 77-84.

Laven, R. D.; Omi, P. N.; Wyant, J. G.; Pinkerton, A. S. 1980. Interpretation of fire scar data from a ponderosa pine ecosystem in the central Rocky Mountains, Colorado. In: Stokes, M. A.; Dieterich, J. H., tech. coords. Proceedings of the fire history workshop. Gen. Tech. Rep. RM-81. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 46-49.

Laverty, L.; Williams, J. 2000. Protecting people and sustaining resources in fire-adapted ecosystems: a cohesive strategy. Washington, DC: U.S. Department of Agriculture, Forest Service.

Lehmann, E. L. 1975. Nonparametrics: statistical methods based on ranks. San Francisco, CA: Holden-Day, Inc. 457 p.

Madany, M. H.; West, N. E. 1980. Fire history of two montane forest areas of Zion National Park. In: Stokes, M. A.; Dieterich, J. H., tech. coords. Proceedings of the fire history workshop. Gen. Tech. Rep. RM-81. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 50-56.

Mast, J. N.; Fulé, P. Z.; Moore, M. M.; Covington, W. W.; Waltz, A. E. M. 1999. Restoration of presettlement age structure of an Arizona ponderosa pine forest. Ecological Applications. 9: 228-239.

Mast, J. N.; Veblen, T. T.; Linhart, Y. B. 1998. Disturbance and climatic influences on age structure of ponderosa pine at the pine / grassland ecotone, Colorado Front Range. Journal of Biogeography. 25: 743-755.

McBride, J. R.; Jacobs, D. F. 1980. Land use and fire history in the mountains of southern California. In: Stokes, M. A.; Dieterich, J. H., tech. coords. Proceedings of the fire history workshop. Gen. Tech. Rep. RM-81. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 85-88.

McBride, J. R.; Laven, R. D. 1976. Scars as an indicator of fire frequency in the San Bernardino Mountains, California. Journal of Forestry. 74: 439-442.

Minitab, Inc. 1998. Minitab Reference Manual, Version 12.1. Chicago: Minitab, Inc.

Minnich, R. A.; Barbour, M. G.; Burk, J. H.; Sosa-Ramírez, J. 2000. California mixed-conifer forests under unmanaged fire regimes in the Sierra San Pedro Mártir, Baja California, Mexico. Journal of Biogeography. 27: 105-129.

Morino, K. A. 1996. Reconstruction and interpretation of historical patterns of fire occurrence in the Organ Mountains, New Mexico. On file at: University of Arizona, Tucson, AZ. 144 p. Thesis.

Morrow, R. J. 1986. Age structure and spatial patterns of old-growth ponderosa pine in Pringle Falls Experimental Forest, central Oregon. On file at: Oregon State University, Corvallis, OR. 80 p. Thesis.

Rowdabaugh, K. M. 1978. The role of fire in the ponderosa pine-mixed conifer ecosystems. On file at: Colorado State University, Fort Collins, CO. 121 p. Thesis.

Savage, M. 1989. Structural dynamics of a pine forest in the American southwest under chronic human disturbance. On file at: University of Colorado, Boulder, CO. 185 p. Dissertation.

Savage, M.; Swetnam, T. W. 1990. Early 19th -century fire decline following sheep pasturing in a Navajo ponderosa pine forest. Ecology. 71: 2374-2378.

Sherman, R. J. 1969. Spatial and developmental patterns of the vegetation of Black Butte, Oregon. On file at: Oregon State University, Corvallis, OR. 80 p. Dissertation.

Shinneman, D. J.; Baker, W. L. 1997. Nonequilibrium dynamics between catastrophic disturbances and old-growth forests in ponderosa pine landscapes of the Black Hills. Conservation Biology. 11: 1276-1288.

Skinner, T.; Laven, R. D. 1982. Background data for natural fire management in Rocky Mountain National Park. Final Report. Fort Collins, CO: Colorado State University, Department of Forest and Wood Sciences. 16 p.

Soeriaatmadja, R. E. 1966. Fire history of the ponderosa pine forests of the Warm Springs Indian Reservation, Oregon. On file at: Oregon State University, Corvallis, OR. 123 p. Dissertation.

Steele, R.; Arno, S. F.; Geier-Hayes, K. 1986. Wildfire patterns change in central Idaho's ponderosa pine-Douglas-fir forest. Western Journal of Applied Forestry. 1: 16-18.

Stein, S. J. 1988. Fire history of the Paunsaugunt Plateau in southern Utah. Great Basin Naturalist. 48: 58-63.

Swetnam, T. W.; Baisan, C. H. 1996a. Historical fire regime patterns in the southwestern United States since AD 1700. In: Allen, C. D., tech. ed. Fire effects in southwestern forests: Proceedings of the second La Mesa fire symposium. Gen. Tech. Rep. RM-GTR-286. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 11-32.

Swetnam, T. W.; Baisan, C. H. 1996b. Fire histories of montane forests in the Madrean borderlands. In: Ffolliott, P. F.; [and others], tech. coords. Effects of fire on Madrean Province ecosystems: a symposium proceedings. Gen. Tech. Rep. RM-GTR-289, Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 15-36.

Swetnam, T. W.; Dieterich, J. H. 1985. Fire history of ponderosa pine forests in the Gila Wilderness, New Mexico. In: Lotan, J. E.; Kilgore, B. M.; Fischer, W. C.; Mutch, R. W., tech. coords. Proceedings—symposium and workshop on wilderness fire. Gen. Tech. Rep. INT-182, Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 390-397.

Touchan, R.; Swetnam, T. W.; Grissino-Mayer, H. D. 1995. Effects of livestock grazing on pre-settlement fire regimes in New Mexico. In: Brown, J. K.; Mutch, R. W.; Spoon, C. W.; Wakimoto, R. H., tech. coords. Gen. Tech. Rep. INT-GTR-320, Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 268-272.

Touchan, R.; Allen, C. D.; Swetnam, T. W. 1996. Fire history and climatic patterns in ponderosa pine and mixed-conifer forests of the Jemez Mountains, northern New Mexico. In: Allen, C. D., tech. ed. Fire effects in southwestern forests: Proceedings of the second La Mesa fire symposium. Gen. Tech. Rep. RM-GTR-286. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 33-46.

Veblen, T. T.; Kitzberger, T.; Donnegan, J. 2000. Climatic and human influences on fire regimes in ponderosa pine forests in the Colorado Front Range. Ecological Applications. 10: 1178-1195.

Veblen, T. T.; Lorenz, D. C. 1986. Anthropogenic disturbance and recovery patterns in montane forests, Colorado Front Range. Physical Geography. 7: 1-24.

Veblen, T. T.; Lorenz, D. C. 1991. The Colorado Front Range: a century of ecological change. Salt Lake City: University of Utah Press.

Weaver, H. 1943. Fire as an ecological and silvicultural factor in the ponderosa-pine region of the Pacific slope. Journal of Forestry. 41: 7-15.

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

Ancient Piñon-Juniper Forests of Mesa Verde and the West: A Cautionary Note for Forest Restoration Programs

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Abstract—*Fuel reduction and fire mitigation activities may be linked to restoration of overall forest health, but the two goals do not always coincide. We illustrate the importance of understanding both historic and contemporary fire regimes by evaluating the piñon-juniper forests of Mesa Verde National Park in southwestern Colorado. These dense forests are characterized by infrequent, severe fires occurring at intervals of many centuries. Stand structure, composition, and fire behavior have not been substantially altered by 20th century fire suppression, in contrast to other piñon-juniper systems. We hypothesize that three qualitatively different disturbance regimes characterize piñon-juniper ecosystems throughout the West.*

Introduction

A major effort is under way to reduce the fire hazard in western forests by thinning tree canopies, removing shrubs and coarse woody debris, and conducting low-intensity prescribed burning. Often, these fuel reduction or fire mitigation activities are linked to restoration of overall forest health (e.g., Covington et al. 1997). This combination of pragmatic objectives (reducing fire hazard at the wildland-urban interface) and ecological objectives (restoring ecological integrity and resilience) is quite sensible in forests that were formerly characterized by frequent, low-intensity fires, and where twentieth century activities like logging, grazing, and fire suppression have dramatically changed forest structure and function — e.g., in many western ponderosa pine forests (Hardy and Arno 1996, Covington et al. 1997). Indeed, the authors of this paper are currently involved in just such a program of thinning and burning degraded ponderosa pine forests in southwestern Colorado, both to achieve the pragmatic objectives of reducing fire hazard and providing material for the local logging industry, and to achieve the ecological objectives of stimulating the suppressed herbaceous stratum and providing habitat for cavity nesting birds and species that prefer open forest conditions (Lynch et al. 2000).

We are concerned, however, that the distinction between fire hazard mitigation and ecological restoration may become blurred, with sometimes unfortunate consequences for ecological integrity and resilience. In particular, because of pressures to meet treatment targets under the National Fire Plan (targets that may be defined primarily in terms of acreage treated per year), managers may be tempted to apply thinning and burning in forest systems where such treatments are unnecessary or even inappropriate from an ecological standpoint (Cole and Landres 1996). To illustrate this concern, we present a case study of the piñon-juniper forests of Mesa Verde National Park in southwestern Colorado.

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Piñon-juniper vegetation covers a vast area in western North America, and exhibits a wide range of stand structures and dynamics (Wangler and Minnich 1996, Miller et al. 1999). Recent research in the Great Basin has demonstrated clearly that piñon and juniper are increasing in density throughout much of this region. Many areas that were formerly dominated by shrubs are now being taken over by piñon and juniper, a trend that began in the late 1800s and is continuing. The causes of piñon and juniper expansion include livestock grazing, fire exclusion, and climatic changes (Miller and Wigand 1994, Miller and Tausch 2001). Some piñon-juniper and juniper savannas in New Mexico and Arizona also appear to have developed denser tree canopies in the twentieth century, or the trees have invaded grassland areas (Jameson 1962, Dwyer and Pieper 1967). Some former grasslands in Bandelier National Monument in northern New Mexico are now largely dominated by juniper and piñon trees <150 years old (C. Allen, personal communication). Because of these well-documented examples of piñon-juniper expansion that were clearly caused by fire exclusion and other human activities, many managers throughout the West have tended to view the entire piñon-juniper vegetation type as being degraded and in need of intensive restoration – usually by means of chaining, roller-chopping, hydro-mulching, and / or burning.

What is often missing in plans for restoration of piñon-juniper vegetation is recognition of the fact that throughout the West there are also ancient piñon-juniper stands with trees >400 years old (examples below). These stands were already well developed even before the late 1800s, and should not be regarded as abnormal consequences of grazing, fire exclusion, and climate change. Unfortunately, thinning and burning programs that would be appropriate and effective in piñon-juniper stands that have developed abnormal tree density during the last century, are being proposed or implemented in old-growth piñon-juniper stands that probably need no restoration. The result of such a well intentioned but misguided restoration effort is likely to be degradation of a relatively rare vegetation type (i.e., old-growth piñon-juniper forest) that contributes much to the biological diversity of western North America. As the pace of fuel reduction and restoration programs accelerates under the National Fire Plan, we have an urgent need to provide criteria for distinguishing between piñon-juniper stands that need treatment and those that do not. We believe that the situation we describe for piñon-juniper also applies to several other important forest types where mitigation and restoration plans may be under way.

This paper has two objectives. First, we describe the fire history, stand structure, and natural fire regime in old-growth piñon-juniper forests of Mesa Verde National Park (MVNP), as a case study to demonstrate that some piñon-juniper vegetation has not changed substantially in the last century, and therefore is not in need of broad-scale thinning or burning to achieve ecological objectives. Second, we develop a set of general hypotheses for predicting where in western North America we are most likely to find piñon-juniper vegetation that has undergone abnormal successional changes in the last century as a result of grazing, fire exclusion, and climate change, and where we are likely to find old-growth piñon-juniper stands that have not been degraded and do not require treatment to restore ecological integrity.

Fire History in Piñon-Juniper Forests of Mesa Verde National Park

Mesa Verde is a large, prominent cuesta in southwestern Colorado composed of uplifted and deeply eroded Cretaceous sedimentary rocks (Griffits

1990). Mesa Verde National Park (MVNP) occupies the northeastern portion of the cuesta. The remainder of the cuesta is mostly in the Ute Mountain reservation and is environmentally similar to the park. The two most extensive vegetation types are piñon-juniper forests (*Pinus edulis* and *Juniperus osteosperma*), found mostly on the southern end of the cuesta at slightly lower elevations, and Petran chaparral or oak-serviceberry shrubland (*Quercus gambellii* and *Amelanchier utahensis*), found mostly on the northern end of the cuesta at higher elevations. Floyd et al. (2000) mapped the major fires since the 1840s in the shrubland portion of MVNP, based on post-fire cohorts of re-sprouting oak, and determined that the average fire return interval is about 100 years. A graph of cumulative time-since-fire indicates little difference between late-nineteenth century and late-twentieth century area burned per decade. Thus, the fire exclusion policy in effect since the Park's establishment in 1906 probably has prevented many fires that would have remained small if not suppressed, but had little impact on the large fires that ignited under conditions of extreme drought and high wind. These latter kinds of fires, uncontrollable even with modern fire suppression technology, are infrequent but account for most of the area burned in a century — a situation similar to many boreal forests (Johnson 1992), subalpine forests (Romme 1982), and chaparral vegetation (Minnich and Chou 1997, Moritz 1997).

Floyd et al. (2000) presented preliminary data on fire history in the piñon-juniper forests of MVNP. We summarize those data here and augment them with additional, unpublished information to provide a fuller picture of the fire regime in the piñon-juniper forests. We structure our analysis around four questions: (1) Are low-severity surface fires an important component of the natural fire regime? (2) How frequent were fires of any kind before 1900? (3) Have piñon and juniper invaded other vegetation types since 1900? (4) Has tree density increased since 1900 in piñon-juniper stands?

Are low-severity surface fires an important component of the natural fire regime?

The large fires of the twentieth century in MVNP have all been stand-replacing crown fires, in both piñon-juniper and shrubland vegetation. Low-severity surface fires have occurred only at the margins of the crown fires as those fires were dying out. But were low-severity surface fires more common prior to Park establishment in 1906? We think not. Low-severity fires often scar susceptible trees, and indeed, the resulting fire scars are the basis of reconstructions of fire history in ponderosa pine and other forests characterized by frequent, low-severity fires. Despite explicitly searching for fire scars throughout MVNP during our 10 years of field work, we have not found any – aside from two trees having single basal scars that could be from fire but probably are from other causes. Piñon and juniper do not form the best scars for dendrochronological dating purposes, but they can be scarred by low-intensity fires (Tausch and West 1988). Moreover, numerous small patches of ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*) exist within and adjacent to the piñon-juniper forests, but these excellent recorders of low-severity fire also lack fire scars in MVNP. Given the paucity of fire scars, we conclude that low-severity surface fires simply were never extensive in piñon-juniper forests of MVNP.

However, we find ample evidence that stand-replacing crown fires occurred in the piñon-juniper forests even before 1900. Figure 1 is a photograph taken in 1934 (from MVNP archives) of an area that burned at some unknown time prior to Park establishment in 1906. The standing snags and the sharp border

Figure 1—Photograph taken in 1934 in the western portion of Mesa Verde National Park, near an area that burned in that year on Wetherill Mesa. The photo is not of the 1934 burn, but shows an area that was burned at an unknown time prior to Park establishment in 1906. Fire history reconstructions (Floyd et al. 2000) suggest that the area in this photo probably burned in the 1880s. Note the edge of dense, unburned piñon-juniper forest in the background.

of the unburned forest indicate clearly that this was a stand-replacing crown fire. From our fire history map (Floyd et al. 2000) and the general location of the photograph, we suspect that the fire occurred in the 1880s – some 50 years before the photo. Indeed, the scene looks much like areas today that were burned in the middle part of the twentieth century.

We also find patches of charred juniper snags within the piñon-juniper forests of MVNP. We took increment cores from piñon trees growing beneath or adjacent to the charred snags in one such patch on Chapin Mesa, known locally as "the glades." The living trees that we sampled were so close to the snags that they could not possibly have survived the fire that created the snags. Some of the sampled piñons were >200 years old, indicating that the fire occurred in the 1700s or earlier (Floyd et al. 2000). This finding also indicates that evidence of past fires (in the form of charred snags) can persist for a very long time in this semi-arid environment.

How frequent were fires before 1900?

By measuring the extent of piñon-juniper forest burned during the twentieth century, and considering that twentieth century fire suppression probably did not have much effect on the few large fires that accounted for most of this burned area, we determined that the mean fire interval in piñon-juniper forests of MVNP is approximately 400 years (Floyd et al. 2000). In a landscape characterized by such long fire intervals, we would expect to find numerous stands that had remained unburned for even longer periods. Indeed, much of the piñon-juniper forest in MVNP contains no evidence of past fire whatsoever – no fire scarred trees, no charred snags, no charred logs. Tree ages (see below) also indicate that many of the stands are very old. Based on a convergence of several lines of evidence (paucity of fire-scars and charred wood, very old trees, and historic photographs – see below), we believe that much of the piñon-juniper forest in MVNP has not been subjected to any major disturbance since the ancestral Puebloan people (who built the famous cliff dwellings) abandoned the area 700 years ago.

Have piñon and juniper invaded other vegetation types since 1900?

Young piñon and juniper appear to be slowly increasing in some of the Park's shrublands. The stands so affected are developing after fires that occurred in the mid to late 1800s (Floyd et al. 2000). Given the naturally long intervals between fires in Mesa Verde's shrublands (mean of ca 100 years), this trend probably represents the natural course of post-fire succession rather than a true "invasion" by trees (Erdman 1970).

We also see young piñon and juniper establishing in some stands of big sagebrush (*Artemisia tridentata*) growing on small patches of deeper soils within the piñon-juniper forests. This may represent actual "invasion," since the trees are mostly young (<100 years old). However, it does not appear to be an anomaly caused by twentieth-century fire exclusion, because these areas apparently burned rarely even before 1900 (above). The extent of the area so affected within MVNP is very small.

Has tree density increased in piñon-juniper stands since 1900?

The piñon-juniper forests of MVNP are very dense in many places. Piñon generally has more stems per hectare, but juniper has greater basal area (unpublished data). Small and presumably young piñon are especially noticeable and give a visual impression of an ongoing increase in stand density.

However, old photographs from MVNP indicate that the piñon-juniper forests have been very dense since at least the early part of the twentieth century. Figures 2 and 3, taken in 1929 and 1934, respectively (from MVNP archives) show dense stands — much like what we see today. The stand depicted in figure 3 (from 1934) gives the impression of a two-tiered structure, with a sparse canopy of older trees and a thick understory of younger trees. However, without information on mortality rates in the understory, it is difficult to interpret long-term stand dynamics. We need detailed age structure data to confidently interpret trends in stand structure. We currently have age structure data derived from age-dbh regressions in one stand that appears representative of the old forests at the southern end of MVNP (Floyd et al. 2000), and we are in the process of directly determining age structure via ring counts in several additional stands. A plot of log(stems / ha) vs log(tree age) in this one stand is approximately linear for both piñon and juniper (figure 4) — a pattern associated with stable, old-growth forests that are not undergoing any major successional trends (Leak 1975). The largest piñon in figure 4 are >400 years old and the largest juniper are >600 years old.

We need additional detailed age distribution data to resolve the question of increasing stand density in piñon-juniper forests of MVNP during the last

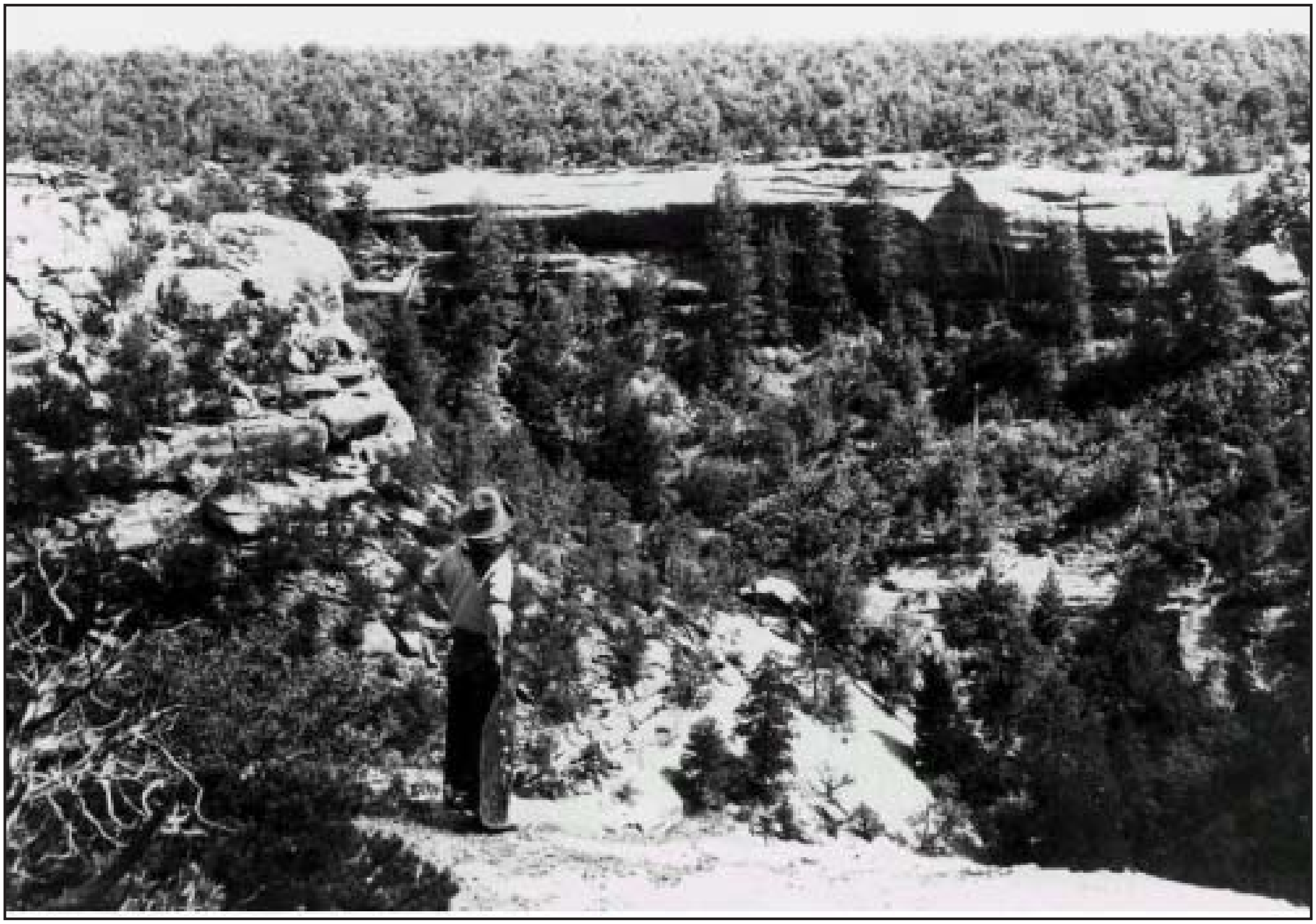

Figure 2—Photograph taken in 1929 of cliff dwellings in the southern portion of Mesa Verde National Park. Note the dense piñon-juniper forest on the rim above the ruins, a forest that does not look much different from the dense forests of today.

century. The abundance of small and young trees in the forests today does suggest that density has increased somewhat in the last 100 years. However, the old photos indicate that the magnitude of increase has not been great. And even if the forests have become more dense, the mechanism probably is related to climatic changes, direct effects of increasing atmospheric CO_2, or other causes (Miller and Wigand 1994)—but *not* to fire exclusion. Given that these forests rarely burned before 1900 (as argued above), post-1900 fire exclusion cannot be the primary reason for increasing tree density after 1900.

Reasons for the Distinctive Fire Regime in Mesa Verde

The evidence just presented indicates that the natural fire regime in Mesa Verde's piñon-juniper forests is characterized by occasional tiny fires (often a single lightning-ignited tree) but few fires of any significant extent. Most ignitions fail to spread, and much of the forest has not burned for at least 700 years. Large fires do occur periodically, however, under extreme weather conditions, and these rare fires are severe and stand-replacing. This fire regime probably has been altered only slightly by twentieth century fire suppression or other activities.

Figure 3—Photograph taken in 1934 in the western portion of Mesa Verde National Park, near an area that burned in that year on Wetherill Mesa. The photo was taken to show the kind of forest that burned in that year. Note the high density of the stand in 1934, similar to the dense stands in this area today.

Why do piñon-juniper forests of MVNP have such infrequent large fires? The reason is not a lack of fire starts: lightning ignites numerous fires every summer in the Park's piñon-juniper zone (Omi and Emrick 1980). However, the vast majority of these fires burn no more than an individual tree or snag before going out – with or without the intervention of a fire suppression crew. Even though the piñon-juniper forests in MVNP support enormous fuel loads of both dead and live material (Omi and Emrick 1980), the fuels are very patchily distributed. Patches of heavy fuels (a few square meters in extent) are separated by comparably sized patches of rock, bare ground, or sparse cover of herbs that do not carry fire readily. Because of the lack of horizontal fuel continuity, fire can spread from fuel patch to fuel patch only under conditions of strong winds and extremely low fuel moisture. In these circumstances, fire spreads through the crowns of live trees, generating extremely high energy release and displaying great rates of spread (personal observations). The necessary weather conditions for such fire behavior occur in only a few years of each decade, and coincide with local ignition only once every decade or two. Once such a fire is started, however, it may cover a large area, going out only when the wind dies down, rain falls, or the fire reaches an extensive area of non-flammable terrain. The deep canyons of MVNP, bordered by tall cliffs with poorly vegetated shale slopes at their bases, probably are significant

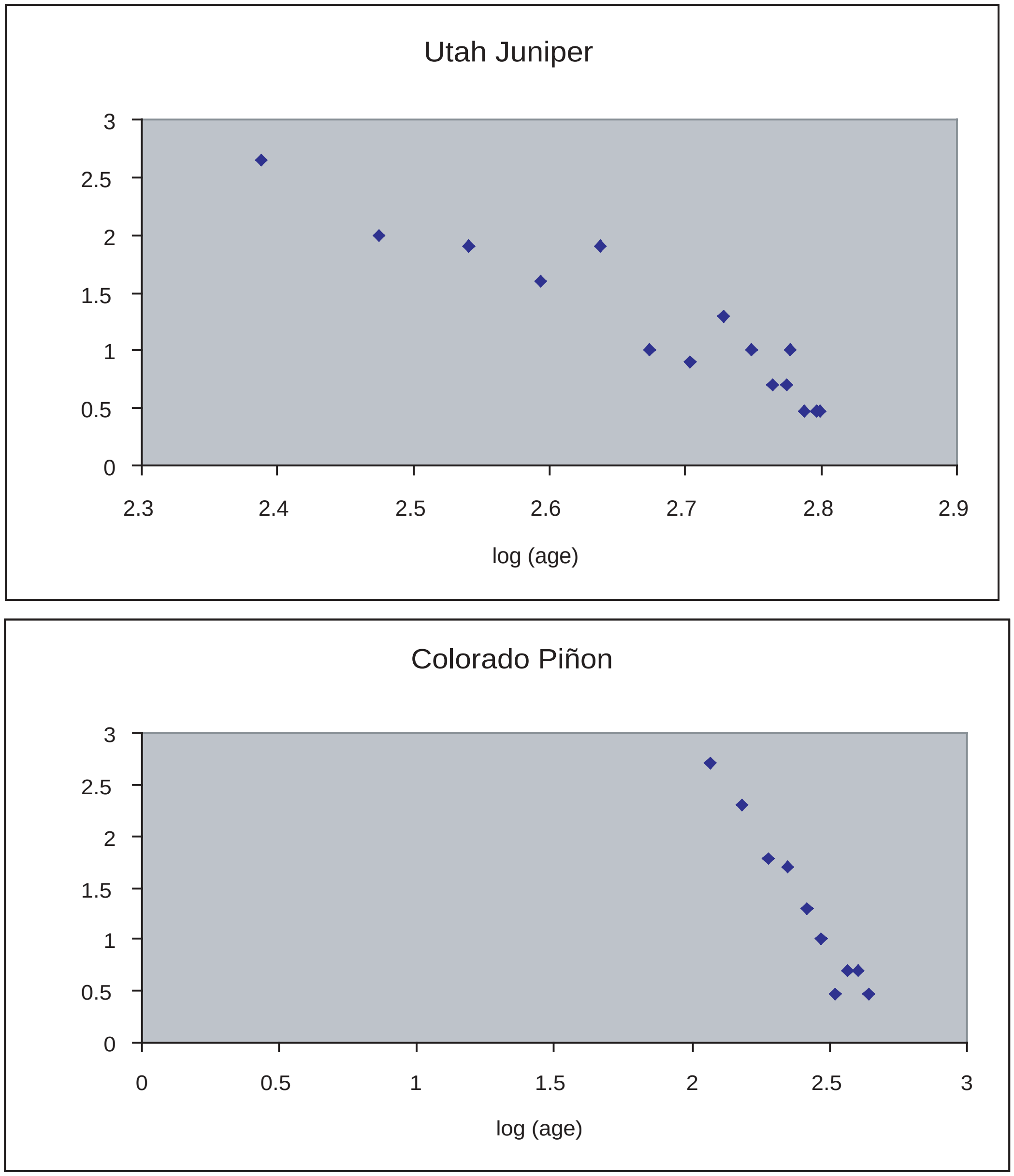

Figure 4—A plot of log(stems/ha) vs. log (tree age) for a stand of dense piñon-juniper forest at the southern end of Mesa Verde National Park. The linear relationship for *Juniperus osteosperma* (a) and for *Pinus edulis* (b) indicates that both tree species have an approximately stationary age structure of the kind that is associated with stable, old-growth forests (Leak 1975).

natural barriers to fire spread (figure 5). As one park ranger put it, "Fire in MVNP tends to burn either one tree or one mesa" (S. Budd-Jack, personal communication).

Heavy livestock grazing in the late 1800s and early 1900s altered fire regimes in many western forests by removing the grass and other fine fuels that formerly carried frequent, low-intensity fires (e.g., Miller and Wigand 1994, Covington et al. 1997). Heavy grazing also occurred in MVNP from around 1900 through the early 1930s (superintendents' annual reports and photos in MVNP archives). The reports are quite vague as to specific locations where cattle were grazed. However, we think that nearly all of the heavy livestock grazing occurred in the meadows and shrublands at the north end of the Park—not in the piñon-juniper forests at the south end. The homesteads, windmills, and stock tanks were all located in the northern area, and all of the old photos that we have located showing heavily grazed lands in MVNP depict meadows and adjacent shrublands. Heavy grazing in the piñon-juniper forests would have been problematic in any event because of lack of water on the dry

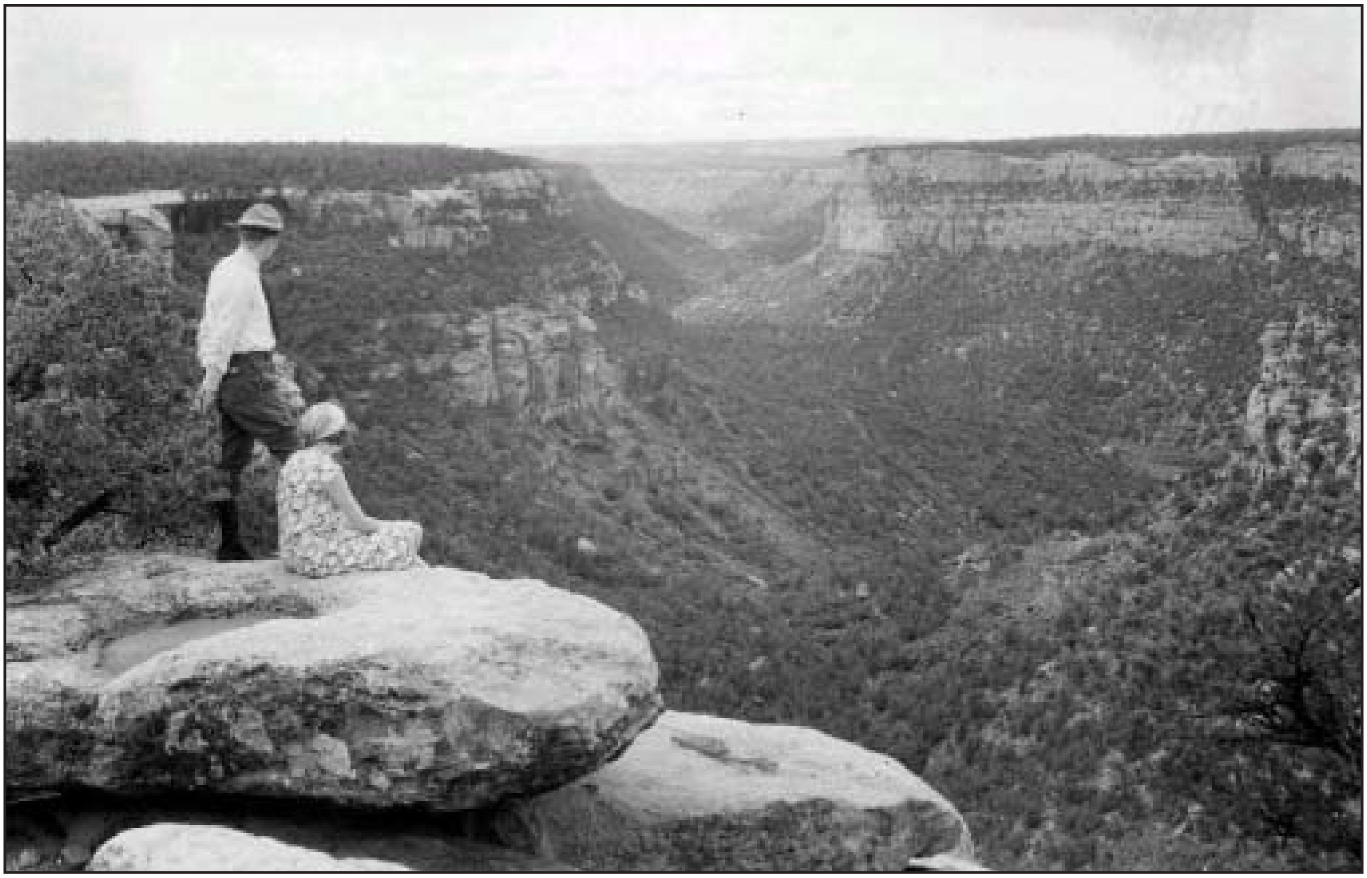

Figure 5—Photo taken in the southern portion of Mesa Verde National Park in the 1930s. Note the dense piñon-juniper forests on the mesa tops, which resemble the dense stands of today, and the cliffs and sparsely vegetated slopes that probably tend to prevent fire spread from one mesa to another.

mesa-tops and lack of forage. The heavily grazed meadows and shrublands have regained their grass cover in the 65 years since grazing was terminated in MVNP, but the old piñon-juniper forests still have only a sparse cover of *Poa fendleriana* and a few other native bunchgrasses. We doubt that grass cover was ever very great in the dry, shallow soils at the south end of Mesa Verde. Even if there was somewhat more grass prior to 1900, it apparently did not carry surface fires, because we find almost no fire scars (above).

Implications for Fire Management in Mesa Verde National Park

MVNP contains one of the greatest concentrations of prehistoric cultural sites and artifacts in North America. The Park was established in 1906 to protect these archaeological treasures, and protection of cultural resources remains the central mission of MVNP. Many of the most outstanding cultural resources are located in the piñon-juniper forests at the southern end of the Park and are vulnerable to damage or destruction by fire (Romme et al. 1993). Recognizing this dangerous situation, MVNP has implemented a fire mitigation program that emphasizes mechanical thinning of dense forests in the immediate vicinity of sensitive cultural resources.

We laud the proactive steps being taken by Park managers to protect the Park's special cultural resources and to provide for the safety of visitors and staff. However, we believe it is important to stress that fuel reduction in MVNP is strictly a program of fire mitigation—it is *not* ecological restoration. To remove >50 % of stand density and basal area from forests that have not been subjected to any major disturbance for 700 years is to create a stand structure that probably never existed previously—at least not within the last several centuries.

Because of the distinctive fire history of piñon-juniper forests in MVNP, especially the paucity of past fire occurrences in this system, we recommend against thinning and burning in areas remote from sensitive cultural sites. Not only would such a program create stand structures and disturbance regimes far outside the historical range of variability, but the treatments themselves would create at least two new problems for managers to deal with. First, where piñon-juniper forests in MVNP have burned or been cleared in the last century, they have not yet begun to recover their tree component. We have sampled all of the large burns of the twentieth century (1934, 1959, 1972, 1989, and 1996), and have found almost no re-establishment of piñon or juniper even in the oldest burns (unpublished data). The trees are thought to re-establish only after shrubs have recovered adequately to provide sheltered micro-sites for tree seedlings (Floyd 1982, Miller and Rose 1995, Wangler and Minnich 1996, Chambers 2001). However, we find no trees even in 50-year-old burns in MVNP, where the shrubs are well established, suggesting that additional mechanisms are involved in this area. We are initiating studies to elucidate the reasons for this poor regeneration of trees after fire, but at present we do not fully understand why piñon and juniper are so slow to re-establish after fire. Considering that these are some of the oldest forests in the southwest, and that they support a variety of old-growth fauna and flora (e.g., black-throated gray warbler and *Pedicularis centranthera*), their conservation value should be considered before subjecting them to what may be essentially irreversible changes.

Our second big concern about extensive thinning and burning of Mesa Verde's piñon-juniper forests is related to what we may get in place of young trees. Forests that burned in the 1930s through 1950s were replaced by shrublands, through what appear to be normal successional processes. Indeed, a major reason for the extensive shrublands at the northern end of MVNP is the extensive fires that have occurred periodically during the last two centuries and possibly earlier (Erdman 1970, Floyd et al. 2000). However, the piñon-juniper forests that have burned in the past two decades are being replaced by invasive non-native species, including muskthistle (*Carduus nutans*), Canada thistle (*Cirsium arvense*), wild lettuce (*Lactuca serriola*), and the annual mustard *Alyssum minor*. Various post-fire mitigation actions (aerial seeding of native grasses, mechanical removal, herbicides, and bio-control) have been effective in reducing the density of weeds after fire, but none of these techniques has prevented the weeds from becoming major components of the post-fire plant community (unpublished data). The burned areas also appear vulnerable to invasion by cheatgrass (*Bromus tectorum*), which readily carries surface fire. Cheatgrass invasion has caused drastic alteration of fire regimes in much of the northern Great Basin (Miller and Tausch 2001). Cheatgrass expansion in Mesa Verde may be limited by the area's cooler temperatures and higher summer precipitation, compared with the northern Great Basin (R. Tausch, personal communication). However, cheatgrass already dominates some overgrazed rangelands at lower elevations near Mesa Verde, and it is present, though not abundant, in the burned forests of MVNP. If it should increase in the park, it could carry ever more frequent fires into the Park's ancient forests.

Thus, even though mechanical thinning is a very appropriate technique for localized, small-scale protection of sensitive cultural resources, even in this context it will require careful monitoring and weed control. And a more ambitious program of extensive thinning and burning of piñon-juniper forests in MVNP could create an ecological disaster: weeds and more frequent fires could potentially destroy nearly all of the ancient forests within a few decades. Such a loss would be especially tragic if it resulted from well-intentioned but misguided efforts to "restore" the piñon-juniper ecosystem.

Implications for Piñon-Juniper Vegetation Throughout the West

The piñon-juniper "type" covers a vast area in western North America, but has received surprisingly little research attention given its ecological, economic, and aesthetic importance. In particular, the natural range of variability in disturbance regimes and post-disturbance recovery processes is poorly understood. We urgently need a more comprehensive understanding of piñon-juniper dynamics so that we can direct our restoration and fire hazard mitigation efforts towards those stands and landscapes where such efforts are most appropriate. As a first step, we suggest that it is useful to distinguish among three fundamentally different kinds of piñon-juniper stand structures and fire regimes, and to identify the environmental conditions with which each kind is associated (table 1). We regard table 1 as a set of *hypotheses*, and hope that this compilation will stimulate research to critically test these suggested relationships across the full geographic range of piñon-juniper vegetation in western North America.

The first type in table 1 is what we call **piñon-juniper grass savanna**. Vegetation of this kind has been described in northern Mexico (Segura and Snook 1992), Arizona (Jameson 1962, Dwyer and Pieper 1967), and New Mexico (Dick-Peddie 1993:87-93 (cited in Scurlock 1998: 206), C. Allen personal communication), in places where soils are fine-textured, topography is gentle, and summer moisture is relatively abundant. The well-developed grass component formerly carried frequent low-intensity fires that killed or thinned encroaching trees and maintained an open woodland structure. Many of these stands have been altered profoundly in the last century by grazing and fire exclusion, which led to loss of the grass component, abnormal tree densities, and abnormally severe fire behavior when the stands burn today. In such stands, fire hazard mitigation via mechanical thinning and prescribed burning can be linked to a broader goal of ecological restoration, and aggressive treatment of this kind is an urgent need in many places.

The second type in table 1 is what we call **piñon-juniper shrub woodland**. This kind of vegetation has been described in northern and central portions of the Great Basin (Koniak 1985, Tausch and West 1988, Miller et al. 1995, Miller and Tausch 2001) and probably also occurs in many portions of the Colorado Plateau, where precipitation occurs mostly in winter, on deep soils that support an abundant shrub layer (e.g., sagebrush). Shrublands generally support more intense fires than grasslands, and fires in this vegetation type probably always have tended to be stand-replacing. Prior to Euro-American settlement in the mid-1800s, trees would become established during the fire-free intervals that lasted from several years to a few decades, but nearly all of

Table 1—Structure and disturbance dynamics, distribution patterns, and current status of three contrasting types of piñon-juniper communities in western North America: a synthesis and set of hypotheses for further research. HRV = historic range of variability.

	Piñon-Juniper Grass Savanna	Piñon-Juniper Shrub Woodland	Piñon-Juniper Forest
Pre-1900 fire regime	frequent, low-severity, surface fires … carried by grasses	moderately frequent, high-severity, crown fires … carried by shrubs & trees	very infrequent, very high-severity, crown fires … carried by tree crowns
Pre-1900 stand structure	sparse trees, few shrubs, dense grass and other herbs	sparse to moderately dense trees, sparse to very dense shrubs, moderately dense to sparse herbs … all depending on time since last fire	dense trees, sparse to moderately dense shrubs, sparse herbs
Pre-1900 stand dynamics	low tree density and high herbaceous biomass maintained in part by recurrent fire	seral trend from herb to shrub to tree dominance, interrupted periodically by fire which returns a stand to early seral herb dominance	stable/stationary tree age structure and little change in shrub or herbaceous layers during the long intervals without fire … very slow recovery after fire
Post-1900 changes in disturbance regime	reduced fire frequency, great increase in fire severity	reduced fire frequency, small increase in fire severity	little change in fire frequency or fire severity
Post-1900 changes in structure	increasing tree density, decreasing herbaceous biomass	increasing tree density, decreasing shrubs and herbs	little change in tree density or in shrubs and herbs
Overall current status	outside HRV for disturbance regime, structure, & composition	outside HRV for disturbance regime, structure, & composition	still within HRV for disturbance regime, structure, & composition
Implications for restoration	urgent need for active restoration	urgent need for active restoration	no need for restoration … protect instead
Current stand age structure	very old trees (> 300 years) present, but not numerous … young trees (< 150 years) dominate stands	very old trees (> 300 years) absent or rare … young trees (< 150 years) dominate stands	very old trees (> 300 years) numerous … stands with all-aged structure, including old & young trees
Distribution: soil characteristics	deep, fine-textured soils	deep, fine-textured soils	shallow, rocky, or coarse-textured soils
Distribution: precipitation regime	summer peak in precipitation	winter peak in precipitation	variable
Distribution: topographic characteristics	gentle plains and broad valley bottoms, with few barriers to fire spread	gentle plains and broad valley bottoms, with few barriers to fire spread	rugged slopes, canyons, and mesa tops, with many barriers to fire spread
Distribution: adjacent vegetation types	grasslands, ponderosa pine, or other types that burn frequently	grasslands, big sagebrush, or other types that burn frequently	desert scrub, “slickrock,” or other types with sparse herbaceous vegetation that rarely burn
Geographic distribution	most common in northern Mexico, southern New Mexico & Arizona, northern NM, and possibly SE Colorado	most common in the northern and central Great Basin, and the Colorado Plateau	scattered throughout the Colorado Plateau, Great Basin, central Oregon, southern Rocky Mountains, and southern California mountains
Examples	Jameson 1962, Dwyer and Pieper 1967, Segura and Snook 1992, Dick-Peddie 1993, C. Allen personal communication	Tausch et al. 1981, Koniak 1985, Tausch and West 1988, Miller et al. 1995, Miller and Tausch 2001	Tausch et al. 1981, Tress and Klopatek 1987, Kruse and Perry 1995, Wangler and Minnich 1996, Miller et al. 1999, Tausch and Nowak 1999, Floyd et al. 2000, Waichler et al. 2001

the trees would be killed by the next fire. Fire exclusion during the last century has allowed this normal successional process to proceed to the point of tree dominance across large areas where trees were formerly sparse. Fuel loads have become very high and continuous, and some recent fires probably have been larger and more severe than would have occurred before the late 1800s (Miller and Tausch 2001). As with the piñon-juniper-grass savanna, fire hazard mitigation via mechanical thinning and prescribed burning (some of it designed to be stand-replacing) can be linked to a broader goal of ecological restoration in the piñon-juniper shrub woodland, and aggressive treatment of this kind is an urgent need in many places.

The third type in table 1 is what we call **piñon-juniper forest**. This kind of vegetation has been described in scattered locations throughout the Colorado Plateau (Tress and Klopatek 1987, Floyd et al. 2000), the Great Basin (Tausch et al. 1981, Miller et al. 1999, Tausch and Nowak 1999), central Oregon (Waichler et al. 2001), the mountains of southern California (Wangler and Minnich 1996, Minnich and Everett 2001), and in central Arizona (Kruse and Perry 1995). Rather than being associated with a particular soil type and climatic regime, piñon-juniper forest appears to be restricted to an unusual combination of soils and topographic conditions that protect the stands from frequent fires. Soils are too shallow or too coarse-textured to support a continuous cover of grass or shrubs, such that fires tend to spread through a stand only under conditions of extreme drought and wind. The topography is rugged and broken, with cliffs, bare slopes, or other natural barriers that tend to prevent fires from spreading into a stand except under conditions of extreme drought and wind. Thus, this kind of vegetation may escape fire for many centuries and develop striking old-growth characteristics, including a dense, multi-storied canopy with ancient living and dead trees. The old forests of Mesa Verde have these characteristics, and Waichler et al. (2001) describe western juniper forests in central Oregon that contain living trees >1000 years old and dead trees nearly 2000 years old. When fire does occur in old piñon-juniper forest stands, however, it tends to be very severe and stand-replacing. In dramatic contrast to the other two kinds of piñon-juniper vegetation (piñon-juniper grass savanna and piñon-juniper shrub woodland), most of the piñon-juniper forest type probably has *not* been substantially altered by fire exclusion in the last century, and probably is *not* outside its historic range of variability in stand structure, fire frequency, and fire behavior. Thus, these forests generally should *not* be subjected to extensive mechanical thinning or prescribed burning, although fuel reduction may be appropriate in localized areas to protect human lives, property, or other sensitive resources. Such localized fire mitigation treatments should be called just that – they should *not* be called "restoration" and they should acknowledge that relatively rare and ecologically significant old-growth characteristics are being sacrificed to protect other resources and values.

The *piñon-juniper grass savanna* (table 1) differs qualitatively from the other two piñon-juniper types, in that its pre-1900 fire regime was characterized by frequent, low-severity fires, whereas the piñon-juniper shrub woodland and the piñon-juniper forest were dominated by infrequent, high-severity fires. The piñon-juniper shrub woodland had shorter pre-1900 fire intervals than the piñon-juniper forest (mean fire intervals perhaps <100 vs >100 years, respectively), but both types probably followed similar successional trajectories after fire. However, the *piñon-juniper forest* type is distinct from the other two types today, in that its natural fire regime and stand dynamics have been disrupted to a far lesser extent by human activities of the past century.

We emphasize that all of the hypotheses in table 1 need further critical testing. We also point out that, although this paper has focused on piñon-juniper vegetation, its conceptual framework probably is broadly applicable to several other vegetation types. Despite the well-supported need for urgent restoration in many areas (e.g., Covington et al. 1997), we believe that there also are many ecosystems throughout the West in which Twentieth century alterations of community structure and function have been minimal. Systems that probably still remain within their historic range of variability include high-elevation spruce-fir and lodgepole pine forests, which are naturally characterized by infrequent but sometimes large, intense fires (Romme 1982, Veblen this volume). Richard Minnich (personal communication) also suggests that the fire regimes of numerous semi-desert and woodland vegetation types (including blackbrush scrub, Joshua tree woodland, and closed-cone conifer woodlands) have been altered very little in the past century – at least in places where non-native species invasions have not shortened fire intervals substantially. Even some ponderosa pine forests in northern Colorado and the Black Hills may have been altered less by twentieth century fire exclusion than is commonly believed (e.g., Shinneman and Baker 1997). These examples help to underscore the primary message that we hope to convey in this paper: that restoration treatments must be based on a good understanding of local stand history and historic range of variability. Above all, we must avoid the temptation to apply "one-size-fits-all" prescriptions for management.

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Literature Cited

Chambers, J.C. 2001. *Pinus monophylla* establishment in an expanding pinyon-juniper woodland: environmental conditions, facilitation, and interacting factors. Journal of Vegetation Science. 12: 27-40.

Cole, D.N., and Landres, P.B. 1996. Threats to wilderness ecosystems: impacts and research needs. Ecological Applications. 6: 168-184.

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

Dick-Peddie, W.A. 1993. New Mexico vegetation: past, present, and future. Albuquerque, NM: University of New Mexico Press. 244 p.

Dwyer, D.D.; Pieper, R.D. 1967. Fire effects on blue grama – pinyon – juniper rangeland in New Mexico. Journal of Range Management. 20: 359-362.

Erdman, J.A. 1970. Piñon juniper succession after natural fires on residual soils of Mesa Verde, Colorado. Provo, UT: Brigham Young University. Biological Series Vol. XI (2). 58 p.

Floyd, M.E. 1982. Interaction of piñon pine and *Quercus gambelii* in succession near Dolores, Colorado. Southwestern Naturalist. 27: 143-147.

Floyd, M.L.; Romme, W.H.; Hanna, D.D. 2000. Fire history and vegetation pattern in Mesa Verde National Park, Colorado, USA. Ecological Applications. 10: 1666-1680.

Griffits, M.O. 1990. Guide to the geology of Mesa Verde National Park. Mesa Verde Museum Association, Lorraine Press, Utah.

Hardy, C.C.; Arno, S.F. 1996. The use of fire in forest restoration. Gen. Tech. Rep. INT-GTR-341. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 86 p.

Jameson, D.A. 1962. Effects of burning on a galleta – black grama range invaded by juniper. Ecology. 43: 760-763.

Johnson, E.A. 1992. Fire and vegetation dynamics: Studies from the North American boreal forest. Cambridge: Cambridge University Press. 129 p.

Koniak, S. 1985. Succession in pinyon-juniper woodlands following wildfire in the Great Basin. Great Basin Naturalist. 45: 556-566.

Kruse, W.H.; Perry, H.M. 1995. Ecosystem management research in an “old-growth” piñon-juniper woodland. In: Shaw, D.W.; Aldon, E.F.; LoSapio, C., tech. coords. Desired future conditions for piñon-juniper ecosystems; 1994 August 8-12; Flagstaff, AZ. Gen. Tech. Rep. RM-258. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 219-224.

Leak, W.B. 1975. Age distribution in virgin red spruce and northern hardwoods. Ecology. 56: 1451-1454.

Lynch, D.L.; Romme, W.H.; Floyd, M.L. 2000. Forest restoration in southwestern ponderosa pine. Journal of Forestry. 98: 17-24.

Miller, R.F.; Wigand, P.E. 1994. Holocene changes in semiarid piñon-juniper woodlands. BioScience. 44: 465-474.

Miller, R.F.; Rose, J.A. 1995. Historic expansion of *Juniperus occidentalis* (western juniper) in southeastern Oregon. Great Basin Naturalist. 55: 37-45.

Miller, R.; Rose, J.; Svejcar, T.; Bates, J.; Paintner, K. 1995. Western juniper woodlands: 100 years of plant succession. In: Shaw, D.W.; Aldon, E.F.; LoSapio, C., tech. coords. Desired future conditions for piñon-juniper ecosystems; 1994 August 8-12; Flagstaff, AZ. Gen. Tech. Rep. RM-258. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 5-8.

Miller, R.; Tausch, R.; Waichler, W. 1999. Old-growth juniper and pinyon woodlands. In: Monsen, S.B.; Stevens, R., comps. Proceedings: ecology and management of pinyon-juniper communities within the Interior West; 1997 September 15-18; Provo, UT. Proc. RMRS-P-9. Ogden, UT: U. S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 375-384.

Miller, R.F.; Tausch, R.J. 2001. The role of fire in juniper and pinyon woodlands: a descriptive analysis. In: Galley, K.E.M.; Wilson, T.P., eds. Proceedings of the invasive species workshop: the role of fire in the control and spread of invasive species. Fire conference 2000. Miscellaneous Publication Number 11. Tallahassee, FL: Tall Timbers Research Station: 15-30.

Minnich, R.A.; Chou, Y.H. 1997. Wildland fire patch dynamics in the chaparral of southern California and northern Baja California. International Journal of Wildland Fire. 7: 221-248.

Minnich, R.A.; Everett, R.G. 2001. Conifer tree distributions in southern California. Madroño. 48: 177-197.

Moritz, M.A. 1997. Analyzing extreme disturbance events: fire in Los Padres National Forest. Ecological Applications. 7: 1252-1262.

Omi, P.; Emrick, L. 1980. Fire and resource management in Mesa Verde National Park. Contract CS-1200-9-B015. Unpublished report, on file at Mesa Verde National Park.

Romme, W.H. 1982. Fire and landscape diversity in subalpine forests of Yellow-stone National Park. Ecological Monographs. 52: 199-221.

Romme, W.H.; Floyd Hanna, L.; Conner, M. 1993. Effects of fire on cultural resources at Mesa Verde National Park. Park Science. 13(3): 28

Scurlock, D. 1998. From the Rio to the Sierra: an environmental history of the Middle Rio Grande Basin. Gen. Tech. Rep. RMRS-GTR-5. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 440 p.

Segura, G.; SnookL.C. 1992. Stand dynamics and regeneration patterns of a piñon pine forest in east central Mexico. Forest Ecology and Management. 47: 175-194.

Shinneman, D.J.; Baker, W.L. 1997. Nonequilibrium dynamics between catastrophic disturbances and old-growth forests in ponderosa pine landscapes of the Black Hills. Conservation Biology. 11: 1276-1288.

Tausch, R.J.; West, N.E.; Nabi, A.A. 1981. Tree age and dominance patterns in Great Basin pinyon-juniper woodlands. Journal of Range Management. 34: 259-264.

Tausch, R.J.; West, N.E. 1988. Differential establishment of pinyon and juniper following fire. The American Midland Naturalist. 119: 174-184.

Tausch, R.J.; Nowak, R.S. 1999. Fifty years of ecotone change between shrub and tree dominance in the Jack Springs Pinyon Research Natural Area. In: McArthur, E.D.; Ostler, W.K.; Wambolt, C.L., comps. Proceedings: shrub ecotones; 1998 August 12-18; Ephraim, UT. Proc. RMRS-P-11. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 1-7.

Tress, J.A.; Klopatek, J.M. 1987. Successional changes in community structure of pinyon-juniper woodlands of north-central Arizona. In: Everett, R.L. comp. Proceedings: pinyon-juniper conference; 1986 January 13-16; Reno, NV. Gen. Tech. Rep. INT-215. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 80-85.

Veblen, T.T. An introduction to key issues in fire regime research for fuels management and ecological restoration. This volume.

Waichler, W.S.; Miller, R.F.; Doescher, P.S. 2001. Community characteristics of old-growth western juniper woodlands. Journal of Range Management. 54: 518-527.

Wangler, M.J.; Minnich, R.A. 1996. Fire and succession in piñon-juniper woodlands of the San Bernardino Mountains, California. Madroño. 43: 493-514.

Landscape Planning

Expectation and Evaluation of Fuel Management Objectives

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Abstract—*The success of fuel management in helping achieve wildland fire management goals is dependent first upon having realistic expectations. Second, the benefits of fuel management can be realized only when treatments are applied at the appropriate scale to the appropriate source of the problem(s). Scales range from the site- or stand-level to landscape-level, but apply differently for purposes of benefiting wildland values than for increasing home survivability. Lastly, accomplishing the broad goals for fuel management requires understanding how proposed treatments directly contribute to solving specific problems. This process of finding solutions to fire problems is framed in terms of "fire risk management" or reduction of "expected loss." This conceptually depicts the way that treatments can influence fire behavior and thus produce benefits by reducing losses and it avoids the unrealistic expectations that fuel management will stop wildfires and prevent homes from burning.*

Introduction

Fuel management is receiving increasing attention as a means of modifying wildland fire behavior and mitigating threats to the urban interface (National Fire Plan http://www.fireplan.gov/), including the Cohesive Strategy (http://www.fireplan.gov/cohesive_strategy_1_28_02.cfm) and 10-year Comprehensive Plan (http://www.fireplan.gov 10_yr_strat_pg_1.html). The rationale for treating fuels follows from:

1) recent and well publicized failures of fire suppression to protect wildlands and developed areas under extreme fire conditions (Colorado/Arizona/Oregon 2002, Montana/Idaho/Colorado 2000, California 1987), and
2) the realization that the extreme nature of these fires has sometimes been exacerbated by human modification of fuel conditions.

Large fires burning under extreme conditions of high winds and low humidity are difficult, if not impossible, to suppress. These extreme weather conditions are expected regularly during the fire seasons of the western United States. The prevalence of extreme fire behavior in low-elevation forests is, however, partly a consequence of effective fire suppression during the past century. Exclusion of historically frequent fire from these ecosystems has resulted in dramatic changes to vegetation structure and fuels compared to conditions in the 19th century (Wilson and Dell 1971; Arno and Brown 1989, 1991). These alterations of the fuel structure, specifically the in-growth of trees and accumulation of dead woody fuels, tend to readily support extreme fire behavior (crown fire, spotting). This reduces the effectiveness of fire suppression and creates uncharacteristically severe effects in those ecosystems compared to pre-existing ecological disturbance regimes. Management of these fuels directly is, therefore, seen as a proactive means to change fire behavior

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and effects (Brackebusch 1973; Davis and Cooper 1963; Kallender 1969; Koehler 1992; Martin et al. 1989; Wood 1982). The need for fuel management solutions has recently been made especially acute in these low-elevation areas because of human encroachment and development of areas formerly classified as wildlands.

Although the conceptual basis of fuel management is well supported by ecological and fire behavior research in some vegetation types, the promise of fuel management has lately become loaded with the expectation of a diffuse array of benefits. Presumed benefits range from restoring forest structure and function, bringing fire behavior closer to ecological precedents, reducing suppression costs and acres burned, and preventing losses of ecological and urban values. For any of these benefits to be realized from fuel management, a supporting analysis must be developed to physically relate cause and effect, essentially evaluating how the benefit is physically derived from the management action (i.e. fuel management). Without such an analysis, the results of fuel management can fail to yield the expected return, potentially leading to recriminations and abandonment of a legitimate and generally useful approach to wildland fire management.

In this paper, we seek to improve the performance and acceptance of fuel management by examining:

1) common expectations of fuel management effects and performance compared to reality,
2) some implications of recent research on fuel treatment scale, prescriptions, and locations, and
3) goals, objectives, and using the concept of “fire risk” to support and direct fuel treatment projects.

Expectations for Fuel Management

A number of false or exaggerated expectations are endemic to the general public and fire management organizations alike. The persistence of these expectations serves to hinder the proper use of fuel management because:

1) they suggest excessively high standards for the success of the fuel treatment,
2) the scope of application or benefits is too broad for a single fuel treatment technique, and
3) they transfer responsibility for fire losses or fire protection to the wrong people or place.

Some of these perceptions or expectations are listed in table 1 along with clarifications as to more realistic views.

Local and Lanscape Scales of Fuel Management

The process of developing specific objectives for fuel treatments and evaluating how treatments might perform necessarily requires an explicit consideration of spatial scale. Two basic scales can be identified with respect to the way that fuel management affects fire behavior. The *local scale* applies to fuel management efforts within a forest stand, a treatment unit, and next to and including a house or structure. Surface fuels removed by prescribed burning or canopy fuels removed by thinning change fire behavior within the local domain of the treatment unit. Many studies have shown that fire behavior

Table 1—Expectations of fire and fuel management compared to more realistic performance.

Expectation	Reality
Adding more firefighting resources will reduce acres burned	The reality is that fire suppression works except when it doesn't. Most fires are already successfully attacked (~96%) leaving the rest to burn under conditions too extreme for suppression success. More firefighting resources can be expected to change wildfire acreage very little because only a slim fraction (~4%) of fires currently escape. Furthermore, it makes little sense to increase fire suppression efforts to solve a fire behavior problem that is widely recognized as having been exacerbated by fire suppression effects on fuels.
Structures and homes will be protected by firefighting resources	Urban interface fires typically overwhelm resources because of the extreme conditions under which they occur (i.e., when fire suppression fails). Thus, exposure of dozens of structures simultaneously to fire brands and fire encroachment exceeds the capacity of existing suppression forces to protect and extinguish them. The problem is compounded in dense neighborhoods when structures start to burn or become fully involved because of their tendency to ignite adjacent structures.
Wildland fuel management prevents structure loss	Wildland fuel management changes wildland fire behavior. Structure loss (i.e., homes burning) is dependent on local properties of the structure and its immediate surroundings. This means that the proximate responsibility for structure loss from fire primarily resides with the private owners of the structure and immediate property, not with public land management agencies.
Fuel treatments will stop wildland fires	Fuel treatments change fire behavior within limitations of their prescription. That is, the design criteria or prescription of fuel treatments (see below) allows them to perform alterations in fire behavior up to a limit of weather conditions (primarily fuel moisture and winds). This change in behavior includes reduced intensities and spread rates, but does not prevent combustion. The changes in fire behavior and fuel conditions may enhance the effectiveness of fire suppression tactics, but it is impossible for fuel treatments alone to stop fires from burning or spreading.
Fuel management can be equally successful for all vegetation and fire regimes.	Fuel management can alter fire behavior but the longevity of these alterations and the ecological appropriateness of the treatment are specific to a given vegetation type. The most common fuel treatments today are concomitant with forest restoration of low-elevation pine and mixed-conifer forests. The same ecological justification and desired changes in fire behavior are inappropriate models for fuel hazard reduction in grasslands that recover following a single growing season or to high-elevation forest characterized by stand-destroying fire regimes. Fuel management strategies and ecological rationale are required for each fire regime and vegetation community.

responds at this local scale to fuel management measures (Helms 1982; Martin et al. 1989; Deeming 1988; Pollet and Omi 2002). This scale, and only this scale, corresponds to the physics of home ignition, whether from firebrands or flames impinging upon home construction materials (Cohen 2000b). The physical properties of the home and its immediate surroundings determine ignition potential and are restricted to the structure and material in very close proximity as determined by principles of radiation and convective heat transfer.

The other scale is described here as the *landscape scale* when concerning wildland areas, or the *community scale* with respect to urbanized environments. This broad scale is a collection of elements from the local scale. That is, wildland landscapes are composed of many stands and treatment units whereas communities are composed of various combinations of structures and undeveloped lots. Many wildland fires are almost an archetypal landscape process because they are larger than a single stand or structure and they move over, across, and through the collection of smaller scale elements like forest stands and homes. Thus, the fire behavior at these broader scales involves the topology or spatial arrangement of stands and homes, each affecting the fire at its own local scale. This spatial arrangement of stands and homes is crucial to determining the success of fuel management activities in changing effects of large fires either at the local or landscape scale. Individual treatment units, regardless of their shape or position, will be irrelevant to the progress and behavior of the fire at the landscape scale unless the spatial nature (topology) of treatment arrangement is considered.

Stand Level Prescriptions for Fuel Management

Fire behavior responds to fuels, weather, and topography. Changes to fuels, for example from prescribed burning or thinning, are related to potential fire behavior at that site and have resulted in reduced severity of wildfires where fuel treatments have occurred (Martin et al. 1989; Helms 1979; Agee 1998). For many fuel management objectives, the goal is to limit surface fires from becoming crown fires. To design a fuel management prescription within a treatment unit, prescription elements must specify changes to specific fuel attributes. These fuel attributes must be connected to a desired change in fire behavior through some physical mechanism. Such a physical mechanism relating surface and crown fires was described by Van Wagner (1977, 1993). His formulation identifies two thresholds that define crown fire activity. Crowns are ignited after the surface fire reaches a critical fireline intensity relative to the height of the base of the aerial fuels in the crown. This crown ignition can become an "active" crown fire that spreads much more rapidly through the crowns, if its spread rate is high enough to surpass the second threshold based on the crown bulk density (kg m^{-3}). Thus, Van Wagner's (1977) relationships suggest that fuel management prescriptions can limit crown fire activity by first reducing surface fuels to limit fireline intensity, then thinning the smallest trees or pruning to elevate the base of aerial fuels from the ground surface. A final measure may involve crown thinning (removal of some canopy level trees) to make difficult the transition to active crowning. This linkage between surface and crown fire has been described by Scott and Reinhardt (2001) and provides a method for determining stand-level prescriptions for fuel management.

Landscape Level Treatment Planning

Fire and fuel managers are familiar and generally comfortable with developing prescriptions for individual stands, whether for silvicultural purposes, forest restoration, or wildland fuels treatment as described above. However, an individual stand treated to a given prescription will probably be irrelevant to fire behavior and effects at the landscape scale because wildfires are often larger than individual treatment units (Salazar and Gonzalez-Caban 1987; Dunn 1989). Thus, some means of spatially organizing treatment units must be considered in order to accomplish the landscape level goals for fuel management. Brackebusch (1973) suggested large-scale frequent mosaic burning. Another landscape strategy described by Finney (2001) seeks specifically to disrupt fire growth and modify fire behavior rather than to stop fires since the latter is not realistic (see Expectations above). Strategic area treatments (Finney 2001; Hirsch et al. 2001) create landscape fuel patterns that slow fire growth and modify behavior while minimizing the amount of treated area required (figure 1). Similar ideas in forest management have been developed to achieve spatial harvest objectives (Baskent and Jordan 1996; Baskent 1999). The impetus follows from limitations on the amount and placement of fuel treatments because of land ownership, endangered species, riparian buffers, etc. It has precedence in the way that natural fire patterns serve to fragment fuels across landscapes to produce self-limiting fire growth and behavior as shown in Yosemite National Park (van Wagtendonk 1995), Sequoia National Park

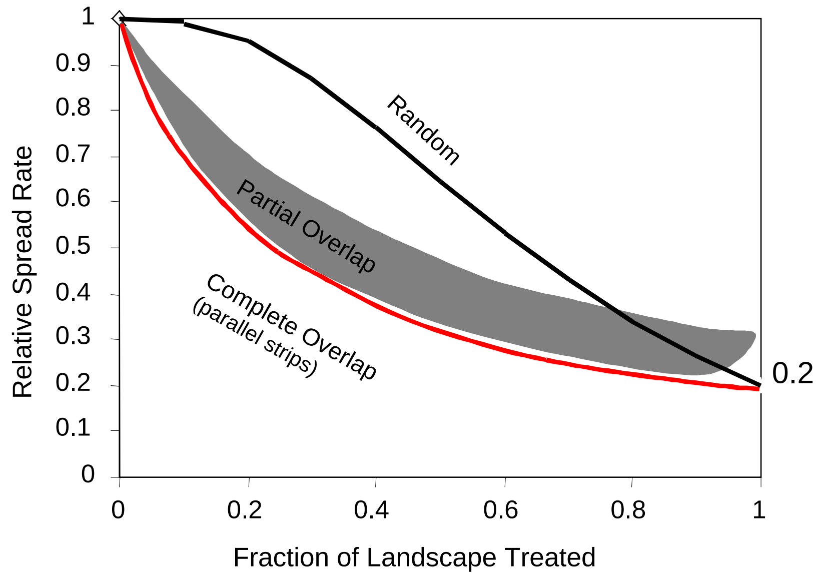

Figure 1—Overall fire spread rate as a function of treatment fraction for different spatial patterns of treatment units (from Finney 2001 and Finney 2003, treatment assumed to reduce spread rate to 0.2 of the untreated fuels). Compared to patterns that require overlap among treatments, the random treatment pattern produces little reduction in overall fire spread rate until relatively large proportions of the landscape are treated (because fire goes around the treated patches).

(Parson and van Wagtendonk 1996), and Baja California (Minnich and Chou 1997). Landscape analysis of fire behavior and spread patterns is now prompting research into computer software for optimally locating fuel treatments for slowing fire growth and limiting effects (figure 2).

A frequently proposed alternative to this strategic landscape approach involves the fuel break concept (Weatherspoon and Skinner 1996; Agee et al. 2000). The stated purpose of fuel breaks is to reinforce an existing defensible location for use by fire suppression forces in stopping fire spread (Green 1977).

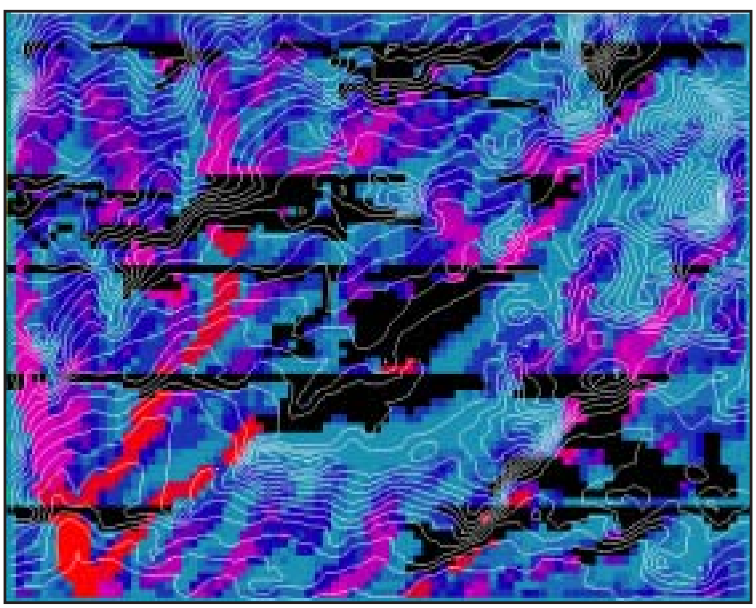

Figure 2—Routes resulting in the most burned area can be identified using graph theory (Finney 2002). These routes reflect the greatest opportunities for disrupting the simulated fire growth using fuel management. Red indicates high influence and blue little or none. Heuristic algorithms can then optimize fuel treatment locations (shown in black) that result in efficient reductions in fire spread rate per unit area treated. The treatments shown in black reduce fire spread rate by 40% with less than 16% of the area treated because treatments are located to block the fastest and most influential routes.

The putative benefits of fuel breaks are achieved when undesirable fire effects are avoided by holding fires to smaller sizes. No change in behavior or effects is achieved away from the fuel break or if fuel breaks fail to stop fires. Thus, fuel break performance and benefit is based on the questionable expectation that fire suppression will be capable of “stopping” fires after initial attack fails (see Expectations above). Large fires escape initial attack for many reasons that include resource scarcity due to high numbers of ignitions, and spotting and crown fire behaviors that make holding a pre-defined position by firefighters untenable and perhaps dangerous. Furthermore, the only firefighting tactics supported by fuel breaks are categorized as "indirect" (Brown and Davis 1973). This means that the rest of firefighting tactics (direct attack and parallel attack) are not enhanced regardless of the current fire behavior or fire position on the landscape relative to the location of the fuel break. A large fire that slows before reaching a fuel break (because of a change in weather conditions, nighttime, etc.) must be attacked (by direct or parallel tactics) with no benefit of the fuel break. Utilizing fuel breaks involves a large burnout operation, which may be of a size equal to the original wildfire, take place regardless of the fire behavior at its current location, and produce negative effects on wildland vegetation greater than the original wildfire. Maintenance costs of fuel breaks are often ignored by proponents but maintenance is a perpetual burden that is likely to divert efforts from managing fuels and vegetation on the remaining majority of the landscape.

Structure Ignition

Research findings indicate that a home's characteristics and the characteristics of a home's immediate surroundings within 30 meters principally determine the potential for wildland-urban fire destruction. This area, which includes the home and its immediate surroundings, is termed the *home ignition zone*. The home ignition zone implies that activities to reduce the potential for wildland-urban fire destruction can address the *necessary* factors that determine ignitions and can be done *sufficiently* to reduce the likelihood of ignition. Wildland fuel reduction outside and adjacent to a home ignition zone might reduce the potential flame and firebrand exposure to the home ignition zone (i.e., within 30 m of the home). However, the factors contributing to home ignition within this zone have not been mitigated. Given a wildfire, wildland fuel management alone (i.e., outside the home ignition zone) is not sufficient nor does it substitute for mitigations within the home ignition zone.

The home ignition zone applies to a single home as well as a neighborhood of homes in proximity. The fire physics and the requirements for ignition do not change with increasing housing density, but the social response changes. Homes in areas where the home ignition zone falls within property boundaries can largely be addressed as individual homes without interaction with other homes. As densities increase and home ignition zones extend across property lines, the ignition potential of the home ignition zones depend on the activities of more than one property owner. At higher housing densities more than one house can fall within a home ignition zone. In such cases a neighboring house burning may become a flame and firebrand source for igniting adjoining homes. When homes share home ignition zones, the wildland-urban fire problem must become a collective, community effort. Wildland-urban fires such as Panorama (San Bernardino, CA 1980), Baldwin Hills (Los Angeles, CA 1985), Oakland (Oakland, CA 1991), and Laguna (Laguna, CA 1993) indicate that communities that do not act collectively to reduce their home ignitability have a high potential to burn collectively.

These findings were based on a diverse research approach utilizing modeling, experiments, case studies, and wildland-urban fire investigations. The model calculations were made on the assumptions of intense fire conditions (e.g., crown fire) and ideal heating characteristics (flames radiating as an ideal radiator, a black body). Model estimates of direct flame heating indicated that wood ignition would not occur during the burning duration of crown fire flames at distances greater than 30 meters (Cohen 2000a). Experiments were conducted to check the results of the modeling. The experimental crown fires provided radiation and convection heating as well as firebrands capable of numerous spot fire ignitions. Home ignition studies during the International Crown Fire Modeling Experiment (Alexander and others 1998) showed that wood walls only ignited at distances from the crown fires closer than 20 meters. Wood walls at 20 and 30 meters did not ignite or significantly scorch. The home ignition experiments indicated that model calculations over-estimated the distance at which a wood wall would ignite (Cohen 2000a). Two former case studies analyzed home survival during severe wildland-urban fires. The case studies found that 85 to 95 percent home survival largely depended on two factors—a nonflammable roof and vegetation cleared within 10 meters of a home (Howard and others 1973; Foote 1994). Investigations of severe wildland-urban fires indicated that home destruction was not necessarily caused by the nearby flames of the intense crown fires; less intense surface fires spreading to the home or direct ignition from firebrands ignited the homes. Investigations also revealed severe wildland-urban fire destruction associated with nearby low intensity surface fires (Cohen 2000b) as well as surviving homes (without protection) surrounded by intense crown fires (Cohen in process). The possible associations between wildland fire behavior and home survival can be displayed in an association matrix (figure 3). Because homes survive high intensity fires and are destroyed in low intensity fires (Cohen 1995; Cohen and Butler 1998; Cohen 2000a; Cohen 2000b; Cohen 2001) it is questionable whether wildland fuel reduction activities are necessary and sufficient for mitigating structure loss in wildland urban fires.

		HOME SURVIVAL	
		yes	no
LOW WILDLAND FIRE INTENSITY	yes	y,y expected	y,n unexpected
	no	n,y unexpected	n,n expected

Figure 3—An association matrix depicts the assumed relationship between wildland fire intensity and home survival. The frequency with which combinations (y,n) and (n,y) have been observed supports the results of physical modeling in questioning the dependency of home survival on wildland fire behavior and fuel management conducted in wildlands.

Goals and Objectives of Fuel Management

The purposes of national fuel management activities in the United States are described by the broad goals stated in the National Fire Plan. These policy documents identify general goals for fuel management activities as:

1. Reduce risk of catastrophic fire
2. Protect communities
3. Reduce fuel hazards
4. Reduce wildfire acres and costs
5. Restore fire-adapted ecosystems

While adequate to express general directions for national fuel management policy, these statements are not intended to provide specific guidance to field-level fuel treatment projects. The "field-level," for this paper, describes the organizational level (e.g., USFS district) where specific fuel treatment units are identified and landscape planning is performed. This is the critical level that determines the success or failure of fuel management where the "rubber meets the road" and the fire meets the fuels. In other words, the success of an entire national policy hinges on the success of fuel treatments accomplishing the field-level benefits promised and expected.

If the broad policy goals are to be used to guide field-level projects, a set of specific objectives must be developed to justify field-level fuel treatment plans. These objectives must be based on a local problem analysis and have standards for evaluating the success or failure of the project. The following are steps that can help bring these broad policy-oriented goals down to specific objectives that permit treatments to be designed, evaluated, and justified:

1. Identification of the specific problems to be addressed by fire / fuel management.
2. Identification of the cause of these problems as relating to fuels or fire behavior.
3. Description of the desired outcome of the treatment measure (i.e., how much change is needed).
4. Identification of the appropriate scale of treatment needed to effectuate the desired outcome.
5. Description of the specific cause and effect relationship between the desired outcome and the proposed treatment(s).

Despite the apparent differences among the general policy goals, the field-level problems associated with them are almost identical. All deal with fuels and the dynamics of the local ecosystems, potential fire behavior, and the likelihood of undesirable effects of fire on urban and wildland areas (costs, losses, expenses). This suggests that the broad policy goals are so closely linked that a unified means of describing this linkage would facilitate understanding how to accomplish all of them. We suggest that all of these goals can be collapsed into the single broad category of "fire risk management." As the term suggests, fire risk is managed, not eliminated. That is, we don't eliminate natural disturbances, we mitigate the associated human disasters.

"Fire Risk Management" and Expected Loss

Risk is a word commonly used to describe threats from fire but it suffers from ambiguous meaning. The absence of a consistent and precise definition of "risk" hinders communication and, more importantly, the possibility of actually achieving a reduction in fire "risk" through fuel management. In other words, "*You can't do what you can't say.*" Historically, risk was used for fire prevention and was equated to the probability of a fire starting (Brown and Davis 1973). These data could be obtained from historical records for a local jurisdiction and partitioned according to location and cause. Although relatively easy to measure, this component has little to do with more critical questions concerning whether the fire once started would achieve a given size, burn a particular area, or cause a particular effect. The probability of burning and consequence of burning are far more relevant to the business of fuel and fire management than the probability of fire starts, but are completely different with respect to the method of calculating or estimating them.

Outside the realm of wildland fire management, risk is often expressed in terms of *expected loss*. Insurance companies calculate the expected loss of a home ($ / year) so that the owner's premium can be determined. Despite the vagueness of its colloquial usage, "risk" defined as an expected loss has an exact mathematical formula involving the product of two numbers: 1) the probability of the event, and 2) the value change in the property because of the event. Wildland fires don't necessarily result in loss or negative consequences, so a more appropriate term would be *expected net value change*. Wildland fires have many different behaviors (e.g., intensity, spotting) that can produce value changes (e.g., fuel reduction, tree mortality, sedimentation of watersheds, structure damage). Since fire behaviors vary in place and time, there would be a distribution of behaviors and a distribution of corresponding changes in value (benefits and losses). In other words a theoretical expected net value change (E_{nvc}) from a wildland fire at a given location on the ground could be obtained for all N categories of a given fire behavior:

$$E_{nvc} = \sum_{i=1}^{N} p(F_i)[B_i - L_i]$$

where F_i is a given fire behavior (e.g., fireline intensity, firebrand density) and B_i and L_i are the respective *benefits* and *losses* resulting from that fire behavior (e.g., dollars from structure loss or tree mortality). Benefits and losses can be combined into a single net value change, but separating the terms in this equation emphasizes the importance accounting for potential benefits of some wildland fire behaviors to some wildland values in addition to losses. Note that $p(F_i)$is the probability of the i^{th} category of the fire behavior occurring and B_i and L_i are the respective benefits and losses for the i^{th} fire behavior category. This kind of equation would apply separately to each value of concern and related fire behavior(s).

To apply the expected value change to wildland fires, research is required to find ways to estimate the parts on the right hand side of the equation. The first part $p(F_i)$ (probability distribution of fire behavior) is particularly challenging because wildland fires are spatial and dynamic, occurring at different places and times and burning over space and time. A brute-force approach to calculating wildland fire probabilities would entail estimating fire growth across the landscape from every ignition point on a landscape, for every ignition date and sequence of weather, for all possible fire seasons and suppression responses. A given cell or node on the landscape would burn differently by backing, flanking, and heading fires depending on the relative location of the ignition and ensuing spread. Each cell or node on the landscape would thus have a probability distribution of fire behaviors represented by $p(F_i)$ in the equation above. The second parts B_i and L_i of E_{nvc} would then need to be determined for each fire behavior in the distribution (e.g., dollars lost for a specified level of fireline intensity). Some fire behaviors cause benefits to some values (e.g., fuel reduction, ecosystem health) but others can result in a total loss. That is the reason that fire expected loss would be the product of two probability distributions, one for the fire behavior and one for the net change in value (benefits minus losses) resulting from that fire behavior. If E_{nvc} could be calculated for all values, then their sum at a given site and over an entire landscape would provide maps that spatially ordinate "risk." These would be spatially sensitive to all scales of fuel management, from the local properties of structures to landscape-level fuel treatments. If ecological modeling and forecasts can be made of future landscape conditions, then cumulative E_{nvc} up to a given future date can be estimated, permitting tradeoffs and opportunity costs to be compared for different action plans. Only then will it be possible to examine the long-term ramifications of today's action or inaction, which very likely will be different from the short-term effects.

Although complicated and difficult to calculate, this mathematical definition of fire risk as an expected value change clearly demonstrates the ways that wildland and urban fuel management activities influence the components of fire risk. Fuel management in wildlands changes the probability that wildland fires move across the landscape, and whether they ultimately impinge on urban areas containing structures, or result in fires of different sizes and ecological effects. Thus, wildland fuel management changes the first part of the equation in terms of probability of a fire reaching a given location. It also changes the distribution of fire behaviors and ecological effects experienced at each location because of the way fuel treatments alter local and spatial fire behaviors (Finney 2001). The probability that a structure burns, however, has been shown to depend exclusively on the properties of the structure and its immediate surroundings (Cohen 2000a). This means that construction materials and their condition at the time of fire exposure that abate ignition from firebrands or flames change the second part of the risk equation only (e.g., replacing wood shingle roof with asphalt shingles changes the structure response to fire behavior). Changes to the flammable materials immediately surrounding the house affects the fire behavior distribution in the first part of the risk equation. Thus, fire risk as E_{nvc} can be improved for wildland and urban areas by:

1. Changing wildland fuels for a "fireshed" involving a wide area around the community (for many miles that include areas that fires can come from). This changes probability of fire movement and skews the fire behavior distribution by increasing the relative frequency of milder behaviors.
2. Treating fuels and reducing fire behavior immediately adjacent to the structures. This changes the fire behavior relevant locally to the ignition of structures.
3. Changing the properties of the structure. This improves its response when exposed to a given fire behavior.

The formulation of E_{nvc} also implies that risk is completely eliminated (goes to zero) when values vanish (total value and value change). This means that human systems of valuation are really at the heart of the idea of risk. There would be no expected loss if humans didn't exist, humans placed no values on wildlands or developed property, or the values didn't change as consequence of fire. More importantly, this shows that both the causes of risk and the solutions to risk reduction lie with human beings, not wildlands or natural dynamics. That is, a change of human perspective can make problems appear or disappear without changes in biophysical reality. The importance of human values in E_{nvc} also suggest the possibility that the necessarily long-term (multidecade) management solutions based on E_{nvc} (if it could even be calculated) could be outdated by changing social and political values during those time periods.

The concept of expected net value change in managing fire risk encompasses all of the broad policy goals detailed above. Community protection as a goal expresses the desired reduction in expected losses and maximizing expected benefits (maximize E_{nvc}). Reducing fuel hazards involves local- and landscape-scale fuel modifications that limit fire behavior and thereby diminish the losses ($p(F_i)*L_i$) and increased ecological benefits ($p(F_i)*B_i$). Reducing fire suppression costs and wildfire acres is produced by changes in the probability distribution of fire sizes brought about by landscape fuel management and reduced duration of extended attack and difficulty with suppressing smaller and less extreme fires. Ecological restoration may also be addressed as diminished fire behavior, reduced losses of ecological values, and increased sustainability of ecosystems that are properly managed. Lastly, it is likely that quantitative assessment of E_{nvc} would lead to more realistic perspectives and expectations for the effects of fire and fuel management activities because alternative management scenarios could be compared and interpreted based on a common methodology.

Community Protection

As an example of how the components of risk management apply to the above policy goals, we can look at the term "community protection." The term community protection is one of the most widely and prominently stated goals for fire and fuel management. A community is really a collection of many tangible and intangible parts that are held in common, including both developed areas and wildland areas:

1. Structures, neighborhoods, businesses
2. Infrastructure (roads, bridges, rivers, dams, airports)
3. Lifestyle and economy (recreation, agriculture, extractive industries like logging, mining)
4. Environment (scenery, air quality, wildlife, natural hazards like fire, earthquakes, hurricanes)

Such diverse community values makes it difficult to justify any single overt fuel management tactic on the basis of "protecting" all aspects of a community from wildland fires. Protection afforded to one component by a given tactic (for example, localized fuel management for structure protection) may little benefit other values (like scenery, air and water quality, or recreation opportunities). The expected loss concept suggests that treatment tactics must be partitioned according to their specific fire behavior changes that are appropriate and relevant to the response of the values concerned. That means essentially treating wildlands separately from developed areas because the effects and scales of those effects are not uniformly applicable.

To benefit the urban portions of a community, fuel management research suggests that fuel management activities need only be concerned with the fuels in the immediate proximity of the structures – within their ignition zone. The material properties of the structures themselves are also important, and managing fuels within the home ignition zone is shown to be the most effective at reducing the nearby sources of firebrands and combustible fuels and vegetation that are commonly associated with structure ignition. When fires occur, structures are less likely to be "lost," thus reducing the expected net value change of the urban values.

Wildland fuel management in low-elevation forest types, extending perhaps many kilometers away from urban locations, however, is critical to reducing the likelihood that wildland fires will spread to urbanized areas and pose ignition threats. Wildland fuel treatments can reduce the probability portion of the expected net value calculation by changing fire behaviors at long distances as well as fire movement. These changes in fire behavior increase the effectiveness of fire suppression, especially during initial attack by slowing fire growth and limiting spotting. They also increase the survivability and resilience of low-elevation forest vegetation to the inevitable wildland fire, thereby benefiting the wildland values of the community. Because urban fire disasters often result from wildland fires igniting tens of kilometers away from urbanized areas under extreme weather conditions (e.g., the Hayman fire in Colorado and the Rodeo-Chedeski fires in Arizona in June 2002), wildland fuel management activities must be located broadly across those landscapes. Evidence that fuel breaks surrounding urban zones are sufficient to reduce threats to urban values is lacking. Because of their location on the periphery of wildlands, fuel breaks cannot reduce losses of wildland values associated with a community. Although it is commonly argued that fuel breaks will reduce wildfire intensities adjacent to residential development and thereby allows firefighters to protect homes, wildland-urban fire disasters tend to occur during severe fire conditions when fire behavior characteristics often overwhelm fire protection resources. These fuel treatments may facilitate firefighter effectiveness but only if firefighters are available to be effective. Given homes with high ignition potential and without fire protection, even a low intensity wildfire can result in severe wildland-urban fire destruction as exemplified by the 2000 Los Alamos fire destruction (Cohen 2000b).

In the context of fire risk management, the general goal of "community protection" can only be accomplished if treatments satisfy the principles of being *necessary* and *sufficient* for the specific elements of the risk equation. Wildland fuel management is *necessary* to change wildland fire behavior ($p(F_i)$), but to be effective at mitigating risk for landscape-level community values and adjacent developments it must be accomplished in *sufficient* amounts and patterns. Wildland fuel management is, however, not *sufficient* alone to abate threats to home ignition. Susceptibility of homes to damage involves different factors than wildland resources (e.g., construction standards vs. tree species) and relates to different fire behaviors than do wildland resources (e.g., firebrands vs. fireline intensity). To reduce expected loss from home ignition, it is *necessary* and often *sufficient* to manage fuels only within the home ignition zone (change $p(F_i)$) and abide by fire resistant home construction standards (change L_i).

Conclusions

We suggest that problems to society posed by wildland fires are analogous to those of traffic accidents. Traffic accidents cannot be stopped, either by increasing the police force, or by reducing speed limits. No government agency or politician believes it possible to stop them altogether. The consequences of traffic accidents that do occur, however, can be mitigated by engineering safety features into automobiles (airbags, seatbelts, frame design) as well as transportation infrastructure (modifying bridge abutments, steep curves, etc.). Likewise, wildland fires cannot be stopped, either with an increasing firefighting budget or fire prevention efforts. Wildland fires will always occur, and ecologically, we know that they must occur in many ecosystems; excluding them is not desirable even if it was possible. The challenge for fire management is to reorient the focus of efforts toward limiting the undesirable effects of fires on ecosystems and human development, not stopping fires. Similar to traffic safety engineering, this paper describes approaches to engineering wildland landscapes and home ignition zones that make our societies more compatible with wildland fires. Sustainability of wildland ecosystems can be accomplished by managing fuels and landscape pattern to change fire behavior. Structure survival can be greatly increased by separate efforts that adopt readily available construction standards and maintain fuel conditions in the home's immediate vicinity. Expectations of our society must also become aligned with the reality of coexistence with fire and its positive and negative effects.

Literature Cited

Agee, J.K. 1998. Fire strategies and priorities for forest health in the Western United States. In: Proceedings: 13th conference on fire and forest meteorology; Lorne, Australia; 1996: 297-303.

Agee, J.K; Bahro, B.; Finney, M.A.; Omi, P.N.; Sapsis, D.B.; Skinner, C.N.; van Wagtendonk, J.W.; WeatherspoonC.P. 2000. The use of fuel breaks in landscape fire management. Forest Ecolology and Management. 127: 55-66.

Alexander, M.E.; Stocks, B.J.; Wotton, B.M.; Flannigan, M.D.; Todd, J.B.; Butler, B.W.; Lanoville, R.A. 1998. The International Crown Fire Modelling Experiment: an overview and progress report. In: Proceedings of the 2nd symposium on fire and forest meteorology, 1998 January 11-16; Phoenix, AZ. Boston, MA: American Meteorological Society: 20-23.

Arno, S.F.; Brown, J.K. 1989. Managing fire in our forests: time for a new initiative. Journal of Forestry. 87(12): 44-46.

Arno, S.F.; Brown, J.K. 1991. Overcoming the paradox in managing wildland fire. Western Wildlands. 17(1): 40-46.

Baskent, E.Z. 1999. Controlling spatial structure of forested landscapes: a case study towards landscape management. Landscape Ecology. 14: 83-97.

Baskent, E.Z.; Jordan, G.A. 1996. Designing forest management to control spatial structure of landscapes. Landscape and Urban Planning. 34: 55-74.

Brackebusch, A.P. 1973. Fuel management – a prerequisite, not an alternative to fire control. Journal of Forestry. 71: 637-639.

Brown, A.A.; Davis, K.P. 1973. Forest fire: control and use. New York: McGraw-Hill Book Co. 686 p.

Cohen, J.D. 1995. Structure ignition assessment model (SIAM). In: Proceedings of the Biswell symposium: fire issues and solutions in urban interface and wildland ecosystems. Gen. Tech. Rep. PSW-158. Albany, CA: US Department of Agriculture, Forest Service, Pacific Southwest Research Station: 85-92.

Cohen, J.D.; Butler, B.W. 1998. Modeling potential ignitions from flame radiation exposure with implications for wildland / urban interface fire management. In: Proceedings of the 13th conference on fire and forest meteorology, vol 1. Fairfield, WA: International Association of Wildland Fire: 81-86.

Cohen, J.D. 2000a. Preventing disaster: home ignitability in the wildland-urban interface. Journal of Forestry. 98(3): 15-21.

Cohen, J.D. 2000b. A brief summary of my Los Alamos Fire destruction examination. Wildfire. 9(4): 16-18.

Cohen, J.D. 2001. Wildland-urban fire—a different approach. In: Proceedings of the firefighter safety summit. In process.

Davis, L.S.; Cooper, R.W. 1963. How prescribed burning affects wildfire occurrence. Journal of Forestry. 61: 915-917.

Dunn, A.T. 1989. The effects of prescribed burning on fire hazard in the chaparral: toward a new conceptual synthesis. In: Berg, N.H., tech. coord. Proceedings of the symposium on fire and watershed management. Gen. Tech. Rep. PSW-109. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station: 23-29.

Finney, M.A. 2001. Design of regular landscape fuel treatment patterns for modifying fire growth and behavior. Forest Science. 47(2): 219-228.

Finney, M.A. 2002. Fire growth using minimum travel time methods. Canadian Journal of Forest Research. 32: 1420-1424

Finney, M.A. 2003. Calculation of fire spread rates across random landscapes. Submitted to International Journal of Wildland Fire. 12(2).

Foote, Ethan I.D. 1994. Structural survival on the 1990 Santa Barbara “Painted Cave” fire: a retrospective study of urban-wildland interface fire hazard mitigation factors. Berkeley, CA: University of California at Berkeley. MS Thesis.

Green, L.R. 1977. Fuel breaks and other fuel modification for wildland fire control. USDA Ag. Handbook 499.

Helms, J.A. 1979. Positive effects of prescribed burning on wildfire intensities. Fire Management Notes. 40(3): 10-13.

Hirsch, K.; Kafka, V.; Tymstra, C.; McAlpine, R.; Hawkes, B.; Stegehuis, H.; Quintilio, S.; Gauthier, S.; Peck, K. 2001. Fire-smart forest management: a pragmatic approach to sustainable forest management in fire-dominated ecosystems. The Forestry Chronicle. 77(2): 357-363.

Howard, Ronald A.; North, D. Warner; Offensend, Fred L.; Smart, Charles N. 1973. Decision analysis of fire protection strategy for the Santa Monica mountains: an initial assessment. Menlo Park, CA: Stanford Research Institute.

Kallender, H. 1969. Controlled burning on the Fort Apache Indian Reservation, Arizona. Tall Timbers Fire Ecology Conference Proceedings. 9: 241-249.

Koehler, J.T. 1992. Prescribed burning: a wildfire prevention tool? Fire Management Notes. 53-54(4): 9-13.

Martin, R.E.; Kauffman, J.B.; Landsberg, J.D. 1989. Use of prescribed fire to reduce wildfire potential. In: Berg, N.H., tech. coord). Proceedings of the symposium on fire and watershed management. Gen. Tech. Rep. PSW-109. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station: 17-22.

Minnich, R.A.; Chou, Y.H. 1997. Wildland fire patch dynamics in the chaparral of southern California and northern Baja California. International Journal of Wildland Fire. 7: 221-248.

Parsons, D.J.; van Wagtendonk, J.W. 1996. Fire research and management in the Sierra Nevada. In: Halvorson, W.L. and G.E. Davis, eds. Science and ecosystem management in the National Parks. Chap. 3. Tucson, AZ: University of Arizona Press.

Pollet, J.; Omi, P.N. 2002. Effect of thinning and prescribed burning on crown fire severity in ponderosa pine forests. International Journal of Wildland Fire. 11(1): 1-10.

Salazar, L.A.; Gonzalez-Caban. A. 1987. Spatial relationship of a wildfire, fuel breaks, and recently burned areas. Western Journal of Applied Forestry. 2(2): 55-58.

Scott, J.H.; Reinhardt, E.D. 2001. Assessing crown fire potential by linking models of surface and crown fire behavior. Res. Pap. RMRS-RP-29. Fort Collins, CO: U.S. Department of Agriculture, Forest Service Rocky Mountain Research Station.

Van Wagner, C.E. 1977. Conditions for the start and spread of crown fire. Canadian Journal of Forest Research. 7: 23-34.

Van Wagner, C.E. 1993. Prediction of crown fire behavior in two stands of jack pine. Canadian Journal of Forest Research. 23: 442-449.

van Wagtendonk, J.W. 1995. Large fires in wilderness areas. In: Brown, J.K.; Mutch, R.W.; Spoon, C.W.; Wakimoto, R.H., tech. coords. Proceedings: symposium on fire in wilderness and park management. Gen. Tech. Rep. INT-GTR-320. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 113-116.

Weatherspoon, C.P.; Skinner, C.N. 1996. Landscape-level strategies for forest fuel management. In: Sierra Nevada Ecosystem Project: Final report to Congress, Volume II. Assessments and scientific basis for management options. University of California Davis, Centers for Water and Wildland Resources: 1471-1492.

Wilson, C.C.; Dell, J.D. 1971. The fuels buildup in American forests: a plan of action and research. Journal of Forestry. 69: 471-475.

Wood, D.B. 1982. Fuel management’s potential for reduction frequency of large fires in the Northern Rockies. Journal of Forestry. 80: 105-107.

Wright, Henry A.; Bailey, Arthur W. 1982. Fire ecology: United States and Canada. New York: John Wiley and Sons. 501 p.

Scheduling Removals for Fuels Management

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Abstract— *We explore management science options for scheduling the placement of fuels reductions. First, we look at approaches for creating and maintaining a pre-specified set of forest conditions that are deemed desirable from a fuels management perspective. This approach is difficult because the pre-specified forest is hard to determine and because it is an any-aged forest management problem that is intrinsically nonlinear. Second, we look at capturing the spatial relationships suggested by fire behavior in long-term fuels management. This approach is difficult because fire origins and behavior can be quite random and unpredictable. It is necessary to accept a particular fire event (analogous to the 500-year flood in flood-control planning) as the target for fuels management. Overall, we conclude that long-term fuels management presents a formidable problem for management scientists. Numerically intense stochastic programming methods, such as Monte Carlo repetitions with a fire simulator, may be the most promising approach.*

Background

The use of mathematical models for managing fires has a rich and varied history in North America. Martell (1982) traces the application of operations research methods in forest fire studies back to the early 1960s. The earliest potential applications of operations research techniques to wildland fire management were mentioned by Shephard and Jewell (1961). Follow-up work by Parks and Jewell (1962) generated considerable interest by examining the use of differential equations and calculus to identify the optimal suppression force for a forest fire. Swersey (1963) and McMasters (1966) extended Parks and Jewell's work by focusing on the optimal mix of different suppression units and the effects of labor constraints on resource allocation rules.

Growing familiarity with optimization techniques spawned additional fire management applications, notably analyses of detection options (Kourtz and O'Regan 1971) and airtanker retardant delivery systems (Simard 1979, Greulich and O'Regan 1982). Fire suppression has continued to receive considerable interest through the use of optimal control theory (Parlar and Vickson 1982), nonlinear programming (Aneja and Parlar 1984), and catastrophe theory (Hesseln et al. 1998). In addition to optimization, simulation modeling has provided useful insights for evaluating management alternatives, especially in an uncertain decision environment (Ramachandran 1988, Fried and Gilless 1988, Mees and Strauss 1992, Mees et al. 1993, Gilless and Fried 1999). Other simulation work oriented towards allocating management resources in fire containment efforts includes Mees (1985), Anderson (1989), and Fried and Fried (1996).

Boychuk and Martell (1988) used Markov chains to analyze seasonal hiring requirements. Martell et al. (1989) modeled seasonal variation in the occurrence of human-caused forest fires. Mills and Bratten (1982) described the use of an economic efficiency system for minimizing the "cost plus net value change" of various fire management alternatives. And, a number of expert

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systems have also been developed for wildfire containment applications, such as Kourtz (1987), Saveland et al. (1988), Fried and Gilless (1989), Stock et al. (1996), and Hirsch et al. (1998).

Introduction

Recent interest in fire and fuel management is particularly motivated by the high fuel levels in many forested areas, especially in ecosystems with short fire return intervals and where fire was excluded during the 20th century (Arno and Brown 1991, Covington and Moore 1994). These fuel conditions create a need for the long-term reduction of fuel loads. This management problem will require new and spatially explicit management science methods expanded to landscape scales. Most approaches in the current literature boil down to a greedy algorithm selecting areas according to some ranking of risk or effectiveness (e.g., Chew et al. in press, Omi et al. 1981). In this paper, we will explore a few options for attacking this problem that are a bit richer in either dynamic or spatial relationships, and point out the significant difficulties along the way. First, we will look at approaches for creating and maintaining a pre-specified set of forest conditions that are deemed desirable from a fuels management perspective. Second, we will look at capturing the spatial relationships suggested by a fire event targeted for long term fuels management.

Creating and Maintaining a Desired Forest

A number of authors have suggested that fire and fuels conditions in ponderosa pine and mixed conifer forests would be improved by creating a forest with densities and age structures that emulate historical conditions or natural processes (e.g., Brown et al. 1999, Keifer et al. 2000). A forest's fuel profile is quite complex, but let us assume that a pre-specified density and age structure can be defined that is "desirable" from a fuels management perspective. Scheduling the removals (through controlled burning, mechanical removals, commercial timber harvests, and other methods) to create and maintain such a forest presents the forest management scientist with a formidable problem in and of itself. Let us begin with the traditional timber harvest scheduling models that focused on removals to optimize some measure of present net worth.

Traditional Timber Harvest Scheduling Models

Traditional approaches to even-aged timber harvest (and other removals) scheduling are summarized in Johnson and Scheurman (1977). Using the "Model I" as the most typical formulation, the approach defines discrete time periods and limits to the age of harvest, so as to create a finite number of scheduling options on specified units of land. Assuming that the objective function is discounted net revenue maximization, the problem is typically formulated as:
Maximize:

$$\sum_{t=1}^{p} \left(R-L\right)/\left(1+r\right)^{T_t} H_t - \sum_{i=1}^{m} \sum_{i=1}^{n_i} \sum_{t=1}^{p} C_{ijt} / \left(1+r\right)^{T_t} X_{ij}$$

Subject to:

$$\sum_{j=1}^{n_i} X_{ij} \le A_i \quad \forall i$$

$$\sum_{i=1}^{m} \sum_{j=1}^{n_i} V_{ijt} X_{ij} = H_t \quad \forall i$$

Where:

Indexes
t indexes time periods
i indexes analysis areas
j indexes management regimes
p = number of time periods in the planning horizon
m = number of analysis areas
n_i= number of harvesting regimes for analysis area i

Variables
H_t= total harvest in the time period t
X_{ij} = number of acres in analysis area i allocated to management regime j

Parameters
A_i= number of acres in analysis area i
V_{ijt}= the yield volume per acre in analysis area i, in harvesting regime j, and in time period t
T_t= the time (in years) at the midpoint of time period t
r = the discount rate
R = the nominal revenue per unit of volume harvested
L = the nominal cost per unit of volume harvested
C_{ijt} = the nominal per-acre cost in time period t for analysis area i and harvest regime j

Notice that discrete removal regimes with a limited number of management actions are defined with discrete time periods, and yields are approximated accordingly. It should also be noted that this approach is quite conducive to constraints such as nondeclining yield, formulated as:

$$H_t \le H_{t+1} \quad t = 1,..., p - 1$$

If we were to try to use this approach to create and maintain the desired forest for fuels management purposes, we would probably start by defining our land units at a fairly fine scale (so that they are relatively spatially explicit). Then, state variables (S_{ikt}) would be defined that track the number of trees in different age classes (k) in each land unit (i), and each time period (t). If each management regime (X_{ij}) were defined such that specific-aged trees are removed, then the state variables could be defined as some function of the management regimes:

$$S_{ikt} = f\left(X_{ij}\right) \qquad \forall i, j, k, t$$

and the state variables could be used in objective functions or constraints to optimize the creation and maintenance of the desired forest from a fuel management perspective.

The primary problem with this approach is that there is an extremely large number of management regimes implied. For each land unit, a nearly infinite number of different options would be possible in terms of the number of different trees in different age classes harvested in different time periods. In order to even approach an optimal conversion and maintenance of the desired age structure over all the land units, the choice variables would have to include a reasonable representation of all options available. This problem arises because the Model I formulation presented above is an even-aged management model, and the problem of creating a forest with a particular location-specific age distribution is really an "any-aged management" problem.

An Any-Aged Management Approach

The early studies that formulated optimization models for uneven-aged management (e.g., Adams and Ek 1974) assumed a steady state solution and viewed the stand-level uneven-aged management problem as one of determining the optimal diameter class distribution, the optimal species mix, the optimal cutting cycle length, and also an optimal conversion strategy for stands not initially in the desired steady state (Hann and Bare 1979, Gove and Fairweather 1992). The steady state assumptions make the problem more tractable, with choice variables (based on diameter classes) and a cutting cycle that apply across the stand.

The less restrictive approach in Haight et al. (1985) is probably more conducive to fuels management, because it does not necessarily need to be used in the context of a steady state solution (see also, Buorngiorno and Michie 1980). Haight (1987) and Haight and Monserud (1990) subsequently coined the term "any-aged management" which we use here as the underlying basis for fuels management. The problem with the model in Haight et al. is that only relatively small problems can be solved (as it is nonlinear). For this apparent reason, Haight et al. only apply their model to a single stand. What is needed for fuels management is a formulation that can handle many diverse (and spatially explicit) stands and be solvable with large problems covering large areas of land.

The Haight et al. (1985) and Buongiorno and Michie (1980) models define choice variables that directly relate to the trees removed, rather than the area treated as in the traditional timber harvest scheduling models described above. The Haight et al. model is nonlinear because the periodic ingrowth and mortality of trees in each age class are nonlinear functions of the number of trees in (potentially) all the age classes. One approach to the large-scale fuels management problem might be to relax this assumption enough to formulate a linear program with many stands. At this point, let us abandon the economic objective function in favor of one that directly optimizes the creation and maintenance of the desired forest. A possible formulation along these lines might look like:

Minimize:

$$\sum_{i=1}^{m} \sum_{k=1}^{q} \sum_{t=1}^{p} \lambda_{ikt} \tag{1}$$

Subject to:

$$\lambda_{ikt} \geq S_{ikt} - \bar{U}_{ik} \quad \forall i,k,t \tag{2}$$

$$\lambda_{ikt} \geq \bar{U}_{ik} - S_{ikt} \quad \forall i,k,t \tag{3}$$

$$S_{ik1} = \bar{S}_{ik} - X_{ik1} \quad \forall i,k \tag{4}$$

$$S_{i1t} = \sum_{k=1}^{q} \left(r_{ik} S_{ik(t-1)}\right) + A_{it} - X_{i1t} \quad \forall i; t = 2,\ldots,p \tag{5}$$

$$S_{ikt} = \left(1 - M_{i(k-1)}\right) S_{i(k-1)(t-1)} - X_{ikt} \quad \forall i; k = 2,\ldots,q-1; t = 2,\ldots,p \tag{6}$$

$$S_{iqt} = \left(1 - M_{i(q-1)}\right) S_{i(q-1)(t-1)} + \left(1 - M_{iq}\right) S_{iq(t-1)} - X_{iqt} \quad \forall i; t = 2,\ldots,p \tag{7}$$

Where:

Indexes

t indexes time periods (there are p of them)
i indexes analysis areas (there are m of them)
k indexes age groups (there are q of them)
Note: t and k are defined with the same time step.

Variables

X_{ikt}= the number of trees harvested in time period t, in area i, and in age class k
S_{ikt}= the number of trees in time period t, in area i, and in age class k
A_{it}= the number of trees artificially planted in area i in time period t

Parameters

r_{ik}= the natural regeneration rate for age class k in area i
$\bar{U}_{ik}$ = the targeted number of trees in age class k in area i
$\bar{S}_{ik}$ = the initial number of trees in age class k in area i
M_{ik}= the mortality rate for age class k in area i

Equation (1) minimizes the sum of all deviations from the desired forest, over all land units, age classes, and time periods. Equations (2) and (3) define the lambda variables as the absolute value of the deviation of the state variables from the desired forest variables. Equation (4) calculates the state variables for the first time period as the initial conditions less any removals (as choice variables) that take place at the beginning of the first time period. It is assumed here that all removals happen at the beginning of each time period. Removals are also accounted for in equations (6) and (7). Equations (5)-(7) track the trees in each land unit as they move through the age classes from time period to time period. Note that the time periods and the age classes need to be defined with consistent time steps. Equation (5) applies to age class 1, equation (7) applies to the oldest age class, and equation 6 applies to all age classes in between. Natural regeneration is accounted for as a parameter and artificial planting is accounted for as a choice variable in equation (5). Mortality is accounted for as a parameter in equations (6) and (7).

Clearly, the weakness of this formulation (in terms of creating a given age structure) is the heroic assumption that mortality and regeneration in each age class and each area are linear functions of the number of trees in that age class and that area, such that M and r are fixed constants. Mortality and regeneration have always been problematic in timber harvest scheduling models because they are affected by so many factors and because they are subject to significant randomness. The impact of these simplifying assumptions would vary from case to case, but it is not hard to imagine situations where they would yield significantly incorrect results. As a simple example, imagine a case where very high densities of young seedlings already crowd the under-story, but equation (5) would continue to add regeneration at the same rate as if there were no seedlings at all.

An alternative approach that might be preferable in some cases would be to make mortality and / or regeneration a linear function of the number of trees in multiple (perhaps all) age classes (see Buorngiorno and Michie 1980). The *r* and *M* parameters would then apply to the total number of trees in multiple age classes, possibly the entire area. This would still, however, be a simplistic, linear characterization of very complex processes.

The analyst might be able to repeatedly solve the model, adjusting the *r* and *M* parameters over the planning horizon to closer match the state of the forest in solution (in each time period). Whether such an iterative approach would converge on a stable solution or not would, again, be expected to vary from case to case. It does help that the purpose of the model is to come as close as possible to a pre-specified forest. Still, if the initial state is much different from the desired state (which we would expect when the initial fuels condition is highly undesirable), then the forest mortality and regeneration rates for the desired state would not be very accurate during the conversion period. The fact of the matter is that the any-aged forest management problem is much more difficult from an optimization modeling standpoint than the even-aged problem, which means that no totally satisfactory large-scale approach is available at this time to optimally create and maintain a pre-specified forest structure that is not even-aged.

Before concluding our discussion on creating and maintaining a desired forest, we should reiterate that an "optimal" fuel condition would be much more complicated to determine than the age-structure problem discussed here. The fuel profile would be influenced by harvest practices, such as slash removal standards, as well as many forest dynamics not captured in the simple designation of an age structure. Further, judicious fuel management may involve much more than tree removals. For example, a manager may wish to leave trees behind for a shaded fuelbreak that requires needle-fall for periodic maintenance prescribed burns. The problem is dauntingly complex, and we have yet to discuss the topic of the next section—spatial relationships.

Spatial Relationships—Optimization With a Target Fire

The previous approach is spatially explicit in that choice variables are defined with sufficient spatial detail to emulate historical conditions across the landscape, but no spatial relationships (such as juxtaposition, proximity, fragmentation, or edge relationships) are really included. The obvious source of spatial relationships for fuels management is the spatial nature of fire itself. That is, the spatial layout of fuels and the spatial relationships between fuel loads in different areas are important because they can affect fire behavior. For example, a large conflagration involving vertically contiguous fuels at the head of a fire front can increase the probability of that fire moving to the crowns. Stand replacement usually results if wind, moisture, and fire intensity are conducive to sustaining such a crown-fire event. In order for an optimization model to account for these spatial relationships, however, a particular fire event needs to be accepted as the "target" of our fuels management efforts. Otherwise, it is uncertain where the fire of concern starts and what it does from there. If it is truly random where a fire of concern might start and how it might behave, then fuels management may be a spatial problem only in the sense that there might be general spatial guidelines for the desired fuel structure that we wish to obtain (which could be included with the approach discussed in the previous section). If, on the other hand, a particular fire can be accepted as a target for fuels management, then the placement of fuels management effort might be optimized to account for the implied spatial relationships.

Even if we assume that we can define a particular target fire to guide our fuels management efforts, it is still not completely clear what our specific objective should be. Traditionally, fire managers have focused on fire suppression strategies that emphasize direct control of the fire or containment of its perimeter within pre-determined or natural barriers. When confronted with fires that exceeded control or containment capabilities of available suppression resources, the fall-back position has called for the protection of valued resources (i.e., homes, communities, or other human developments). This approach is also relevant when it is decided to let a fire burn so as to restore natural fire processes or as a part of fuels reduction efforts. Recent policy changes (Zimmerman and Bunnell 1998) call for an expansion of strategies for managing fires, especially at the landscape scale. As Finney (2001:219-220) states, "Two basic strategies for landscape-level fuel management are to contain fires and to modify fire behavior...a spatial arrangement of treatments that primarily modifies fire behavior would involve area-based or dispersed patterns. For fire modification, it is clear that the greatest reduction in fire size and severity occurs when fuel treatment units limit fire spread in the heading direction." One option between letting a fire burn unhindered and attempting suppression is thus to slow its spread across the landscape, relative to any valued resources that it threatens. This is a fundamentally different objective than the traditional approaches, and its practical application would depend on acceptance by the fire management community.

Let us assume that there are distinct areas of concern (such as towns, summer homes, campgrounds, ski areas, and so forth), and that a fire management objective is to delay ignition of those "protection areas" in the target fire event as long as possible. The advantages of such a delay would include: (1) maximizing the chances that other suppression efforts or independent factors such as weather changes might cause the fire to subside before the protection areas are impacted; and (2) maximizing the time available for building fire line around the protection areas, for modifying fuels to reduce a fire's severity near the protection areas, or for evacuation of the protection areas.

If the objective of long-term fuels management is to mitigate the effects of a particular target fire, with known origin(s) and spread behavior, then one approach (from Hof et al. 2000) could be as follows. To begin, the landscape would be defined with a grid of cells to capture spatial location. The management variables would be defined as application of fuels reduction efforts in each cell (such as prescribed burning, mechanical removals, and so forth) and these efforts would be scheduled over a fairly long period of time because only so much fuels reduction can be accomplished in a given year (or season). Thus, discrete time periods of, say, one to ten years would be defined and indexed with t. The trajectory of each cell's fuel load over time, and its response to fuels management, would have to be tracked as well. Such a model might be formulated as:

Maximize: λ (8)

Subject to:

$$\lambda \le T^{\circ}_{mnt} \quad \forall t \tag{9}$$

$$T^{\circ}_{abt} = 0 \quad \forall t \tag{10}$$

$$T^{\circ}_{ijt} \le T'_{hkt} \quad \forall i, j, t \quad \forall (h, k) \in \Omega_{ij} \tag{11}$$

$$T'_{ijt} - T^{\circ}_{ijt} = f_{ij}\left(F_{ijt}\right) \quad \forall i, j, t \tag{12}$$

$$F_{ijt} = \sum_{k=1}^{K_{ij}} B_{ijkt} X_{ijk} \quad \forall i,j,t \tag{13}$$

$$\sum_{k=1}^{K_{ij}} X_{ijk} \le 1 \quad \forall i,j \tag{14}$$

$$\sum_{i} \sum_{j} \sum_{k}^{K_{ij}} D_{ijkt} X_{ijk} \le \overline{X}_t \quad \forall t \tag{15}$$

$$0 \le X_{ijk} \le 1 \quad \forall i,j,k \tag{16}$$

Where:

Indexes

i indexes cell rows, as does h
j indexes cell columns, as does k
k indexes management prescriptions (there are K_{ij} of them for cell ij)

Variables

T°_{ijt}= the time that the target fire front ignites cell ij, if it occurs in time period t
T'_{ijt}= the time that the target fire front leaves cell ij, if it occurs in time period t
T'_{hkt}= the time that the target fire front leaves cell hk (and potentially ignites cell ij) if it occurs in time period t
F_{ijt}= the fuel available for combustion in cell ij and in time period t
X_{ijk}= the proportion of cell ij allocated to management prescription k
f_{ij} =an empirical function that relates available fuels in cell ij to the duration of time between entry and exit of the fire front

Parameters

a,b = the row and column of the fire origin cell
m,n = the row and column of the protected area cell
Ω_{ij} = the set of row and column indexes for cells that can *potentially* ignite cell ij. Note: this would typically be some subset of the cells adjacent to cell ij and would be determined primarily by a combination of wind conditions during the target fire and topography.
B_{ijkt} = the available fuel in time period t, in cell ij, that results if management prescription k is applied
D_{ijkt} = a dummy parameter that is equal to one if management prescription k applies fuels reduction in time period t, in cell ij, and is zero otherwise
$\overline{X}_t$ = the total number of cells that can be treated with fuels reduction in time period t

The objective function (8) together with the inequalities in (9) maximize the minimum ignition time of the protection cell *mn* across time periods (the target fire might occur in any time period). Other objective functions that aggregate time periods would also be possible. Equation (10) sets the ignition time of the origin cell to zero. Equation (11) relates the ignition times of each cell to the times that the fire front leaves the cells which can potentially ignite it. The multiple inequalities in (11) cause each cell to be ignited by the first potentially igniting cell that the fire front departs from. Equation (12) relates the duration of time that it takes the fire front to move through each cell to the available fuel in that cell, given existing weather and topography. In practice, the spread rate functions (f) would be estimated with fire prediction models or empirical data. Linear approximation of this f function is discussed in Hof et al. (2000). Constraints (10) - (12) account for a potential target fire in each time period. Constraint (13) applies the management variables (that include scheduling) to determine the changing available fuel load in each cell *ij* over time. Constraint (14) restricts the sum of the management prescription variables to be less than or equal to one for each cell, and constraint (15) limits the amount of fuels management treatment that can be applied in each time period. It may be desirable to replace (16) with binary integer constraints on all X_{ijk}, to force either complete treatment or no treatment of each cell in any given time period. As the formulation stands, it is assumed that the B_{ijkt} parameter is applicable to fractional values of X_{ijk}. Extension of this approach to multiple protection areas is straightforward (see Hof et al. 2000).

This model, again, is based on using a target fire to guide long term fuels management. The approach is similar to the use of particular storm events (such as a 500-year flood) to guide watershed and flood control planning. Presupposing highly random fire behavior conditions such as wind speed and direction or location of lightning strikes, however, may be far less certain than the path of water flow on a landscape. If a different fire event eventually occurs, the fuels management strategy based on the target fire may or may not be desirable. At any rate, the model is readily solvable if the *f* functions are linear, because it is then a linear program with continuous variables.

Conclusion

Overall, long term fuels management presents a formidable problem for management scientists. Treating the problem as one of creating and maintaining a particular forest (which is believed to be desirable from a fuels perspective either because of historical conditions or some other criterion) assumes that a desirable fuel profile can be obtained by creating and maintaining a particular forest density and age structure. The resulting forest removals problem is difficult because it is an any-aged forest management problem that is intrinsically nonlinear. The assumptions necessary to make such a problem linear are rather heroic. Treating the problem so as to account for the spatial nature of fire itself is difficult because fire origins and behavior can be quite random and unpredictable. It is necessary to accept a particular fire event as the target for fuels management. An approach that focuses on spatial fuel pattern, per se, might show promise, but guidelines for desirable patterns are not apparent. Monte Carlo approaches that simulate many fires might show promise in accounting for the uncertainty of fire origin and behavior, but heuristics for finding near-optimal solutions have yet to be developed and the basic computing time necessary to adequately simulate an adequate number of fires may be prohibitive. Clearly, much additional work is needed on all aspects of the spatial and dynamic management of fuels at the landscape scale.

References

Adams, D.M.; Ek, A.R. 1974. Optimizing the management of uneven-age forest stands. Canadian Journal of Forest Research. 4: 274-287.

Anderson, D.H. 1989. A mathematical model for fire containment. Canadian Journal of Forest Research. 19: 997-1003.

Aneja, Y.P.; Parlar, M. 1984. Optimal staffing of a forest fire fighting organization. Canadian Journal of Forest Research. 14: 589-594.

Arno, S.F.; Brown, J.K. 1991. Overcoming the paradox in managing wildland fire. Western Wildlands. 17(1): 40-46.

Boychuk, D.; Martell, D.L. 1988. A Markov chain model for evaluating seasonal forest fire fighter requirements. Forest Science. 34(3): 647-661.

Brown, P.M.; Kaufmann, M.R.; Shepperd, W.D. 1999. Long-term, landscape patterns of past fire events in montane ponderosa pine forest of central Colorado. Landscape Ecology. 14: 513-532.

Buongiorno, J.; Michie, B.R. 1980. A matrix model of uneven-aged forest management. Forest Science. 26: 609-625.

Chew, J.; Jones, J.G.; Stalling, C.; Sullivan, J. [In press]. Combining simulation and optimization for evaluating the effectiveness of fuel treatments for four different fuel conditions at landscape scales. Proceedings of the 8th Symposium on Systems Analysis in Forest Resources; 2000 September 28-30; Snowmass, CO.

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

Finney, M.A. 2001. Design of regular landscape fuel treatment patterns for modifying fire growth and behavior. Forest Science. 47(2): 219-228.

Fried, J.S.; Fried, B.D. 1996. Simulating wildfire containment with realistic tactics. Forest Science. 42(3): 267-281.

Fried, J.S.; Gilless, J.K. 1988. Stochastic representation of fire occurrence in a wildland fire protection planning model for California. Forest Science. 34(4): 948-955.

Fried, J.S.; Gilless, J.K. 1989. Expert opinion estimation of fireline production rates. Forest Science. 35(3): 870-877.

Gilless, J.K.; Fried, J.S. 1999. Stochastic representation of fire behavior in a wildland fire protection planning model for California. Forest Science. 45(4): 492-499.

Gove, J.H.; Fairweather, S. 1992. Optimizing the management of uneven-aged forest stands: a stochastic approach. Forest Science. 38(3): 623-640.

Greulich, F.E.; O'Regan, W.G. 1982. Optimum use of air tankers in initial attack: Selection, basing, and transfer rules. Res. Pap. PSW-163. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station. 8p.

Haight, R.G.; Brodie, J.D.; Adams, D.M. 1985. Optimizing the sequence of diameter distributions and selection harvests for uneven-aged stand management. Forest Science. 31(2): 451-462.

Haight, R.G. 1987. Evaluating the efficiency of even-aged and uneven-aged stand management. Forest Science. 33(1): 116-134.

Haight, R.G.; Monserud, R.A. 1990. Optimizing any-aged management of mixed-species stands: II. Effects of decision criteria. Forest Science. 36(1): 125-144.

Hann, D.W.; Bare, B.B. 1979. Uneven-aged forest management: State of the art (or science?). Gen. Tech. Rep. INT-50. Missoula, MT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 18 p.

Hesseln, H.; Rideout, D.B.; Omi, P.N. 1998. Using catastrophe theory to model wildfire behavior and control. Canadian Journal of Forest Research. 28: 852-862.

Hirsch, K.G.; Corey, P.N.; Martell, D.L. 1998. Using expert judgement to model initial attack fire crew effectiveness. Forest Science. 44(4): 539-549.

Hof, J.; Omi, P.N.; Bevers, M; Laven, R.D. 2000. A timing-oriented approach to spatial allocation of fire management effort. Forest Science. 46(3): 442-451.

Johnson, K.N.; Scheurman, H.L. 1977. Techniques for prescribing optimal timber harvest and investment under different objectives: Discussion and synthesis. Forest Science Monograph 18.

Keifer, M.; Caprio, A.; Lineback, P.; Folger, K. 2000. Incorporating a GIS model of ecological need into fire management planning. In: Neuenschwander, L.F.; Ryan, K.C., Tech. Eds. Proceedings from the Joint Fire Science Conference and Workshop; 1999 June 15-17; Boise, ID: 122-137.

Kourtz, P.H. 1987. Expert dispatch of forest fire control resources. AI Applications in Natural Resources Management. 1(1): 1-7.

Kourtz, P.H.; O'Regan, W.G. 1971. A model for a small forest fire to simulate burned and burning areas for use in a detection model. Forest Science. 17(2): 163-169.

McMasters, A.W. 1966. Wildland fire control with limited suppression forces. Berkeley, CA: University of California. 141 p. Ph.D. Dissertation.

Martell, D.L. 1982. A review of operational research studies in forest fire management. Canadian Journal of Forest Research. 12: 119-140.

Martell, D.L.; Bevilacqua, E.; Stocks, B.J. 1989. Modelling seasonal variation in people-caused forest fire occurrence. Canadian Journal of Forest Research. 19: 1555-1563.

Mees, R.M. 1985. Simulating initial attack with two fire containment models. Res. Note PSW-378. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station. 7 p.

Mees, R.; Strauss, D. 1992. Allocating resources to large wildland fires: A model with stochastic production rates. Forest Science. 38(4): 842-853.

Mees, R.; Strauss, D.; Chase, R. 1993. Modeling wildland fire containment with uncertainty in flame length and fireline width. International Journal of Wildland Fire. 3(3): 179-185.

Mills, T.J.; Bratten, F. 1982. FEES: Design of a fire economics evaluation system. Gen Tech. Rep. PSW-65. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station. 26 p.

Omi, P.N.; Murphy, J.L.; Wensel, L.C. 1981. A linear programming model for wildland fuel management planning. Forest Science. 27(1): 81-94.

Parks, G.M.; Jewell, W.S. 1962. A preliminary model for initial attack. IER report no. 32. Berkeley, CA: Operations Research Center, Institute of Engineering Research, University of California. 29 p.

Parlar, M.; Vickson, R.G. 1982. Optimal forest fire control: An extension of Parks' model. Forest Science. 28(2): 345-355.

Ramachandran, G. 1988. Probabilistic approach to fire risk evaluation. Fire Technology. 8: 204-226.

Saveland, J.M.; Stock, M.; Cleaves, D.A. 1988. Decision-making bias in wilderness fire management: Implications for expert system development. AI Applications in Natural Resources Management. 2(1): 17-29.

Shephard, R.W.; Jewell, W.S. 1961. Operations research in forest fire problems, IER report no. 19. Berkeley, CA: Operations Research Center, Institute of Engineering Research, University of California. 29 p.

Simard, A.J. 1979. A computer simulation model of forest fire suppression with air tankers. Canadian Journal of Forest Research. 9: 390-398.

Stock, M.; Williams, J.; Cleaves, D.A. 1996. Estimating the risk of escape of prescribed fires: An expert system approach. AI Applications in Natural Resources Management. 10(2): 63-73.

Swersey, R.J. 1963. Parametric and dynamic programming in forest fire control models. IER report no. 93. Berkeley, CA: Operations Research Center, Institute of Engineering Research, University of California. 24 p.

Zimmerman, T.Z.; Bunnell, D.L. 1998. Wildland and prescribed fire management policy: implementation procedures and reference guide. Boise, ID: National Interagency Fire Center. 81 p.

Wildland Fire Use: A Wilderness Perspective on Fuel Management

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Abstract—*Current federal wildland fire policy recognizes wildland fire as an important natural process and emphasizes the need to reintroduce fire into ecosystems. The policy also recognizes that hazardous fuel accumulations may need to be reduced on vast acreages of land before fire can safely be returned to wildland ecosystems. Wildland fire and fuel managers have a variety of options for reducing fuels including wildland fire use, management-ignited prescribed fires, thinning, and other mechanical methods. All of these options will need to be exploited to accomplish the task of reducing hazardous fuels and restoring healthy fire-dependent ecosystems. Wildland fire use, while focusing primarily on restoring fire as a natural process and maintaining ecosystems, has the potential to be very effective for managing fuels. It may be the most appropriate strategy in wilderness and in other remote unroaded areas. To effectively implement wildland fire use, wildland fire managers will need to rely on comprehensive fire management plans. The development of these plans should include analyses needed to support the wildland fire use decision and should consider the potential benefits from wildland fire, long-term consequences of management decisions, and impacts of decisions across large landscapes.*

Introduction

Decades of effective fire suppression and land use change have led to fuel accumulations, escalating fire behavior and spread, increased risk to human life and property, and the deterioration of fire dependent ecosystems. The Federal Wildland Fire Policy Report of 1995 declares, "Wildland fire, as a critical natural process, must be reintroduced into the ecosystem." The policy also recognizes that hazardous fuel accumulations may need to be reduced before fire can be reintroduced. The magnitude of the hazardous fuel problem is substantial. It is estimated that fire regimes on over half the land under federal ownership (230 million acres) have been moderately or significantly altered from their historical range (Rocky Mountain Research Station 1999). These lands are therefore at moderate or high risk of losing key ecosystem components and may require moderate or high levels of restoration treatment. In addition to these at-risk lands, there are areas where healthy ecosystems already exist, and treatments may be required to maintain their condition.

A wide spectrum of strategies is available for reducing accumulated fuels and their associated risks including naturally or accidentally ignited wildland fires, management ignited prescribed fires, and a variety of mechanical and chemical methods (Omi 1996). The effectiveness and cost of different fuel treatments depends on a variety of factors including: location, fuel type, size of treatment unit, treatment method, and institutional factors (Rideout and Omi 1995, Schuster et al. 1997, Cleaves and Haines 1997, Cleaves et al. 1999, Gonzalez-Caban 1997). From local to national levels, managers and

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planners are seeking to maximize the effectiveness of fuel management programs while controlling costs. In FY2001, USDA and USDI treated 2.25 million acres for hazardous fuel reduction (USDA and USDI 2002). Although the goal for FY2002 is somewhat higher (2.4 million acres), this is only a small fraction of the total acreage in need of treatment. A variety of factors can limit the acres that are treated, including funding, inadequate staffing, lack of experienced and skilled personnel, unsuitable weather, and technological limitations (Barrett et al. 2000, Cleaves et al. 2000, Miller and Landres, in prep.).

The task of reducing hazardous fuels and restoring or maintaining healthy fire-dependent ecosystems is enormous. Despite the impressive commitment to hazardous fuel reduction being made at the national level (USDA and USDI 2002), available resources and opportunities to use prescribed fire and mechanical methods will always be limited. Wildland fire and fuel managers will need to employ all available options and opportunities for reducing fuels. One such option is the use of naturally ignited wildland fire, or wildland fire use (WFU). This paper provides a brief historical context for WFU and discusses WFU as a potentially effective method for fuel management.

Wildland Fire Use in Wilderness

When the Secretaries of Interior and Agriculture issued the Federal Wildland Fire Management Policy and Program Review in 1995, they provided policy direction for all federal wildland fire activities (USDA and USDI 1995). One of the guiding principles of the new policy is that “the role of wildland fire as an essential ecological process will be incorporated into the planning process.” The current direction provides for allowing fires from natural ignition sources to be managed for resource benefits if an approved fire management plan is in place (Zimmerman and Bunnell 1998).

The use of naturally ignited wildland fires to achieve resource objectives on federal lands began in the 1970s. At that time, these fires were called Prescribed Natural Fires (PNFs); with the policy change in 1995 came the new terminology of Wildland Fire Use (WFU). Since the early 1970s when policies were first implemented to use natural ignitions, well over 1 million acres have been allowed to burn by either PNF or WFU on National Park Service and Forest Service lands (S. Botti, USDI National Park Service, unpub. data; D. Bunnell, USDA Forest Service, unpub. data). As of 2001, 85 of the 403 FS wildernesses (excluding Alaska) have fire management plans that allow for the use of wildland fire.

The vast majority of PNFs and WFU have occurred within federally designated wilderness or national parks. The Wilderness Act of 1964 defines wilderness as “an area where the earth and its community of life are untrammeled by man,” and “which is protected and managed so as to preserve its natural conditions.” Wilderness is to be managed so that it “generally appears to have been affected primarily by the forces of nature, with the imprint of man’s work substantially unnoticeable.” Consistent with this language and with the current understanding of fire’s role in natural ecosystems, the wilderness policies of all four federal wilderness management agencies (NPS, USFS, FWS, BLM) recognize the importance of fire as a natural ecological process and the desirability of restoring the historical role of fire to wilderness ecosystems (Parsons and Landres 1998).

Suppression of lightning ignitions clearly does not allow the forces of nature to affect wilderness and therefore runs counter to the intent of the Wilderness Act. However, fire suppression has been and continues to be the dominant fire management strategy. Indeed, in many areas suppression has resulted in conditions where the “imprint of man’s work” is quite noticeable as large-scale successional changes and unprecedented fuel accumulations (e.g., Arno et al. 1997, Covington and Moore 1994, Parsons and DeBenedetti 1979). Most of the fires suppressed in wilderness are suppressed because there is no fire management plan that allows for WFU. Currently, only about one in five wilderness areas have fire management plans that allow the use of natural ignitions (Parsons 2000). Even in those wildernesses where the fire management plan allows for WFU, the majority of lightning ignitions are suppressed. For instance, the Bob Marshall Wilderness Complex is a large wilderness area in western Montana with a PNF / WFU program. Between 1988-1998, 80% of the lightning ignitions in the wilderness area were suppressed (Parsons 2000). Wilderness fires are suppressed for a variety of reasons: the potential for the fire to escape the wilderness boundary and threaten values outside of the wilderness; overextended staff and resources; the national or regional fire situation; air quality concerns; and a complex set of political risks (Poncin 1995, Miller and Landres, in prep.).

Wildland Fire Use for Fuel Management

In addition to its ability to help restore the natural process of fire and its ecological role in wildland ecosystems, WFU has the potential to be an effective strategy for accomplishing fuel management objectives. The federal wildland fire policy supports the use of wildland fire as a fuel treatment alternative (USDA and USDI 1995). Wildland fire reduces fuels through consumption, and interrupts fuel continuity by creating vertical and horizontal gaps within and between surface fuels and crown fuels (Brown and Smith 2000). Although the ability of prescribed fire and mechanical treatments to mitigate wildfire behavior and severity has been demonstrated (Pollet and Omi 2002, Omi and Martinson 2002), the effectiveness of WFU as a fuel treatment has not yet been formally assessed. However, many examples exist where fire behavior appears to be affected when the fire spreads into a previously burned area. For example, the area burned in 1996 by the Swet Fire in the Bitterroot NF appears to have inhibited fire spread in 2000, and in Glacier NP, the Moose Fire of 2001 burned around the area of the Anaconda Fire of 1999. These and other anecdotal examples suggest that the mosaic created from abutting burned areas of different ages can aid in tactical fire suppression and reduce the probability of fire escaping to lands with high values-at-risk (van Wagtendonk 1995, Mohr and Both 1996).

In the 105 million acres of federally designated wilderness as well as on other unroaded lands outside wilderness, WFU may be the most feasible option for reducing fuels. Reduced access to the interiors of these areas limits the ability to apply prescribed fire, thinning, and other mechanical methods for fuel management. Further, these more manipulative fuel treatment methods may be inappropriate for use in designated wilderness where their use is limited by current legal and policy constraints, as well as public acceptance (Ingalsbee 2001, Landres et al. 2001).

Planning for Wildland Fire Use

Wildland fire use is only an option if an approved fire management plan allows it (Zimmerman and Bunnell 1998). The fire management plan should provide the information needed to support the WFU decision and should contain comprehensive analyses of the resource and public values that may be affected by fire. Given the time-critical nature of the WFU decision, it is essential that these analyses be done prior to the fire incident. The fire management plan can serve as the instrument of this pre-analysis. To support the WFU decision, this pre-analysis should consider the following:

1. Wildland fire benefits and risks. When deciding whether to manage an ignition as WFU, the wildland fire manager needs to assess the benefits of fire use along with its risks. For example, fire's ecological benefits and its ability to reduce hazardous fuels must be weighed against the potential threats it poses to human life and property. The decision to suppress a fire is made when the potential negative consequences from fire outweigh its potential benefits. Conversely, the WFU decision is justified when the potential benefits outweigh the risks. The fire management plan can serve a valuable role in the WFU decision-making process by providing the wildland fire manager with the information needed to make a balanced assessment of the risks and benefits from wildland fire (Miller et al. 2000).

2. Long term consequences. The beneficial effects of wildland fire are often realized over much longer time scales than the negative impacts from fire. Landscape mosaics created by fire may be able to reduce the likelihood of property loss in the wildland urban interface but may also require many years of successful WFU implementation. In contrast, the social impacts from fire can occur immediately after, or even during, the fire. In evaluating an ignition for WFU, the wildland fire manager needs to understand the long- and short-term consequences of both WFU and continued fire suppression. A fire management plan could be prepared using the results of ecosystem simulation models that project future conditions. This information would allow the manager to compare the long-term consequences of his / her alternatives.

3. Landscape scales. Fire is a process that operates at large spatial scales and fire management activities affect entire landscapes. Implementing WFU in the interior of a large wilderness area may adversely impact air quality far outside the wilderness boundary. Decisions to suppress ignitions that start outside the wilderness boundary can affect the fire regime in the interior of the wilderness by preventing the natural immigration of fires spreading into the wilderness. To consider an ignition for WFU, a wildland fire manager needs to evaluate the potential impacts on a variety of values across a broad geographic area. If developed in conjunction with a Geographic Information System (GIS), the fire management plan can be used to organize and display information about the social, economic, cultural and ecological values that may be affected by fire management activities. In addition, the fire management plan could contain up-to-date information about fuels and the biophysical environment that affects fire spread—information that can be fed directly into fire behavior prediction tools (Finney 1994).

These three aspects of the pre-analysis (fire benefits, long-term perspectives, and landscape scales) will be essential for supporting a WFU decision. In addition, they could also help link the fire management plan to the land and resource management plan. A key element of the land management planning process is the identification of desired future conditions, and the potential benefits from WFU could help define these conditions. A long-term, landscape scale perspective is consistent with land and resource management planning, which is based on the principles of long-term sustainability and cross-boundary integration (Committee of Scientists 1999). Ideally, the land management plan would provide the goals and objectives for the fire management plan and these objectives could be framed in terms of long-term desired future conditions across the management area. To complete the linkage from the fire management plan back to the land management plan, the success of the fire management program should be evaluated in terms of these land management objectives. For example, the performance of a fire management program might be measured in terms of social impacts or desired future conditions that have been identified in the land management plan (Rideout and Botti 2002).

Summary

The task of reducing hazardous fuels and their associated risks on federal lands is enormous. To accomplish this task, wildland fire and fuel managers will need to utilize the full spectrum of fuel management strategies, including wildland fire use (WFU). WFU has the potential to be very effective for managing fuels and is likely the most appropriate strategy in wilderness and in other remote unroaded areas. The decision to manage an ignition for WFU will hinge on the analyses contained in the fire management plan. To adequately support the WFU decision, these analyses need to consider benefits from wildland fire, long-term consequences, and landscape scales.

References

Arno, S.F., Smith, H.Y. and Krebs, M.A. 1997. Old growth ponderosa pine and western larch stand structures: influences of pre-1900 fires and fire exclusion. Res. Pap. INT-RP-495. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 20 p.

Barrett, T.M., Jones, J.G., and Wakimoto, R.H. 2000. USDA Forest Service use of spatial information in planning prescribed fires. Western Journal of Applied Forestry. 15: 200-207.

Brown, J.K. and Smith, J.K. 2000. Wildland fire in ecosystems: effects of fire on flora. Gen. Tech. Rep. RMRS-GTR-42-Vol-2. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 257 p.

Cleaves, D.A. and Haines, T.K. 1997. Regulation and liability risk: their influence on the practice and the price tag of prescribed burning. In: Bryan, D. (ed.). Proceedings of the environmental regulation and prescribed fire conference; 1995 March 15-17; Tampa, FL: 165-183.

Cleaves, D.A., Haines, T.K., and Martinez, J. 2000. Influences on prescribed burning activity in the National Forest System. In: Fire and forest ecology: innovative silviculture and vegetation management; proceedings of the 21st Tall Timbers fire ecology conference; 1998 April 14-16; Tallahassee, FL. Tallahassee, FL: Tall Timbers Research Station: 170-177.

Cleaves, D.A., Haines, T.K., and Martinez, J. 1999. Prescribed burning costs: trends and influences in the National Forest System. In: Proceedings of the symposium on fire economics, planning and policy: bottom lines; 1999 April 5-9; San Diego, CA. Gen. Tech. Rep. PSW-GTR-173. Albany CA: U. S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 277-287.

Committee of Scientists. 1999. Sustaining the people's lands - recommendations for stewardship of the National Forests and Grasslands into the next century. [Online]. Available: http: / / www.fs.fed.us / news / science [August 16, 2002]

Covington, W.W. and Moore, M.M. 1994. Southwestern ponderosa forest structure and resource conditions: changes since Euro-American settlement. Journal of Forestry. 92: 39-47.

Finney, M.A. 1994. Modeling the spread and behavior of prescribed natural fires. In: Proceedings of the 12th international conference on fire and forest meteorology; 1993 October 26-28; Jekyll Island, GA. Bethesda, MD: Society of American Foresters: 138-143.

Gonzalez-Caban, A. 1997. Managerial and institutional factors affect prescribed burning costs. Forest Science. 43: 535-543.

Ingalsbee, T. 2001. Wildland fire use in roadless areas: restoring ecosystems and rewilding landscapes. Fire Management Today. 61: 29-32.

Landres, P.B., Brunson, M.W., and Merigliano, L. 2001. Naturalness and wildness: the dilemma and irony of ecological restoration in wilderness. Wild Earth. 10: 77-82.

Miller, C., Landres, P.B. [In preparation]. Information needs for wildland fire and fuels management.

Miller, C., Landres, P.B., and Alaback, P.B. 2000. Evaluating the risks and benefits of wildland fire at landscape scales. In: Neuenschwander, L.F.; Ryan, K.C., tech. coords. Crossing the millennium: integrating spatial technologies and ecological principles for a new age in fire management; proceedings of the joint fire science conference and workshop; 1999 June 15-17; Boise, ID. Boise, ID: University of Idaho: 78-87.

Mohr, F. and Both, B. 1996. Confinement: a suppression response for the future. Fire Management Notes. 56: 17-22.

Omi, P.N. 1996. Landscape level fuel manipulations in greater Yellowstone: opportunities and challenges. In: The ecological implications of fire in Greater Yellowstone; proceedings of the second biennial conference on the Greater Yellowstone ecosystem; 1993 September 19-21; Yellowstone National Park, WY. Fairfield, WA: International Association of Wildland Fire: 7-14.

Omi, P.N. and Martinson, E.J. 2002. Effect of fuels treatment on wildfire severity. Final Report to the Joint Fire Science Program Governing Board. Fort Collins, CO: Western Forest Fire Research Center.

Parsons, D.J. 2000. Restoration of natural fire to United States wilderness areas. In: Watson, Alan E.; Aplet, Greg H.; Hendee, John C., comps. Personal, societal, and ecological values of wilderness: Sixth World Wilderness Congress Proceedings on Research, Management, and Allocation, vol. II; 1998 October 24-29; Bangalore, India. Proc. RMRS-P-14. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 42-47.

Parsons, D.J. and DeBenedetti, S.H. 1979. Impact of fire suppression on a mixed-conifer forest. Forest Ecology and Management. 2: 21-33.

Parsons, D.J. and Landres, P.B. 1998. Restoring natural fire to wilderness: how are we doing? In: Pruden, Teresa L.; Brennan, Leonard A., eds. Fire in ecosystem management: shifting the paradigm from suppression to prescription, proceedings of the 20th Tall Timbers fire ecology conference; 1996 May 7-10; Boise, ID. Tallahassee, FL: Tall Timbers Research Station: 366-373.

Pollet, J. and Omi, P.N. 2002. Effect of thinning and prescribed burning on crown fire severity in ponderosa pine forests. International Journal of Wildland Fire. 11: 1-10.

Poncin, D.E. 1995. Prescribed natural fire strategies and tactics. In: Brown, James K.; Mutch, Robert W.; Spoon, Charles W.; Wakimoto, Ronald H., tech. coords. Proceedings: symposium on fire in wilderness and park management; 1993 March 30-April 1; Missoula, MT. Gen. Tech. Rep. INT-GTR-320. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 133-135.

Rideout, D.B. and Botti, S.J. 2002. Performance-based interagency fire program analysis system: blueprint for fire planning. Journal of Forestry. 100: 36-41.

Rideout, D.B. and Omi, P.N. 1995. Estimating the cost of fuels treatment. Forest Science. 41: 664-674.

Rocky Mountain Research Station. 1999. Coarse-scale spatial data for wildland fire and fuel management. [Online] Available: http://www.fs.fed.us/fire/fuelman [August 16, 2002].

Schuster, E.G., Cleaves, D.A., and Bell, E.F. 1997. Analysis of USDA Forest Service fire-related expenditures, 1970-1995. Res. Pap. PSW-RP-230. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 29 p.

USDA and USDI. 1995. Federal wildland fire policy. [Online] Available: http://www.fs.fed.us/land/wdfire.htm [August 16, 2002].

USDA and USDI. 2002. FY 2001 performance report: National Fire Plan. [Online] Available: http://www.fireplan.gov/fy_2001_performance.cfm [August 16, 2002].

van Wagtendonk, J.W. 1995. Large fires in wilderness areas. In: Brown, James K.; Mutch, Robert W.; Spoon, Charles W.; Wakimoto, Ronald H., tech. coords. Proceedings: symposium on fire in wilderness and park management; 1993 March 30-April 1; Missoula, MT. Gen. Tech. Rep. INT-GTR-320. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 113-116.

Zimmerman, G.T. and Bunnell, D.L. 1998. Wildland and prescribed fire management policy, implementation procedures reference guide. [Online] Available: http://www.fs.fed.us/fire/fire_new/fireuse/wildland_fire_use/ref_guide/refguide.pdf [August 16, 2002].

Modeling the Effects of Fuel Treatments for the Southern Utah Fuel Management Demonstration Project

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Abstract—*An integrated multi-scale analysis strategy using output from a variety of fire behavior and effects models has been developed for the Southern Utah Fuel Treatment Demonstration Project. Broader-scale analyses at the sub-basin or sub-regional scale will employ the models FIREHARM and LANDSUM across the entire study area. Sub-watershed and landscape level analyses will primarily use the FARSITE, FlammMap, and spatial versions of NEXUS and FOFEM on selected landscapes. Stand and polygon level analyses will be simulated using the Fire and Fuel Extension to the Southern Utah variant of the Forest Vegetation Simulator (FFE-FVS) and non-spatial versions of FOFEM and NEXUS for representative fuel and vegetation conditions in the study area.*

Background

Many areas throughout the United States are facing the triple threat of increasing fire intensity, increasing residential growth in areas prone to wildland fire, and increasing suppression costs and losses. In addition, substantial changes have and are occurring in the way we plan and implement management on the National Forests and Grasslands relative to use of wildland fire, prescribed fire, and mechanical fuel management. Past emphasis in fire management has been on wildfire suppression and prescribed fire in support of other resources such as hazard reduction and site preparation in harvested areas and wildlife habitat improvement. Allowing lightning fires to burn in wilderness areas to restore natural process has also been occurring for better than 30 years. It is only in the last five or six years that fire management has employed prescribed burning and mechanical fuel treatments to reduce unnatural fuel build-up in non-wilderness areas.

The 1995 Federal Wildland Fire Management Policy Review (USDA and USDI 1995) contributed to the development of a new federal fire policy that directs agencies to balance fire suppression capability and the use of fire to regulate fuels and sustain healthy ecosystems. The review directed fire managers to objectively evaluate and compare fuel treatment options including prescribed fire, thinning and other mechanical methods of fuel treatment, and increased utilization of biomass. Following the Cerro Grande Fire of 2000, Secretary of the Interior Bruce Babbitt and Secretary of Agriculture Dan Glickman directed the Interagency Federal Wildland Fire Policy Review Working Group to review the 1995 Federal Wildland Fire Management Policy and Program Review.

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The General Accounting Office (GAO) report of 1999 (General Accounting Office 1999) emphasized the need for the Forest Service to develop a cohesive strategy to address catastrophic wildfire threats. This report describes the extent and seriousness of problems related to the health of National Forests in the Interior West, the status of efforts by the Department of Agriculture's Forest Service to address the most serious of these problems, and barriers to successfully addressing these problems and options for overcoming them. The Forest Service responded with a document presenting a potential strategy for addressing fire management issues in fire adapted ecosystems (USDA FS 2000). This cohesive strategy establishes a framework that restores and maintains ecosystem health in fire-adapted ecosystems for priority areas across the Interior West.

In the 1998 appropriation, Congress, with the support of the Administration, provided a more flexible funding authority to support the aggressive use of fire and mechanical fuels treatments, with the goals of reducing the occurrence of uncharacteristically severe wildland fires and improving ecosystem health. In granting this new funding authority, Congress expressed a concern that "both the Forest Service and the Department of Interior lack consistent and credible information about the fuels management situation and workload, including information about fuel loads, conditions, risk, flammability potential, fire regimes, locations, effects on other resources, and priorities for treatment in the context of the values to be protected." This resulted in the creation of the Joint Fire Science Program, a concerted effort toward addressing the "fuels problem" and providing a scientific basis for implementing fuels management activities with a focus on activities that will lead to development and application of tools for managers. Development of methods for fuel characterization, mapping, and assessment must include examination of both available and needed fuel models. In turn, development of tools for managers requires an assessment of the role of information system technology.

Southern Utah Fuel Management Demonstration Project

The Southern Utah Fuels Management Demonstration Project is one such project funded by the Joint Fire Sciences Program. It will creatively link current technology in a consistent and comprehensive manner to allow comparisons of alternatives for fuel management for roughly 13 million acres of Southern Utah and 2 million acres of Northern Arizona. The databases and models will be used to support the planning and implementation of an integrated, interagency landscape level fuel management program for the region. Our goal in this respect is to improve the fuel management program in Southern Utah and a portion of Northern Arizona by establishing an interagency demonstration area. This area is undergoing rapid change in land use, which places some urgency on the need for this approach to fuel management.

The demonstration area (figure 1) includes contiguous state and federal lands within the administrative boundaries of the BLM, Forest Service, state of Utah, and the National Park Service, roughly encompassing the southern 15 percent of Utah (table 1). Several agencies have intermixed land ownership and a history of good interagency cooperation on management issues.

Southern Utah is at the ecological crossroads for much of the Western United States. It contains steep environmental gradients. This allows us to study a

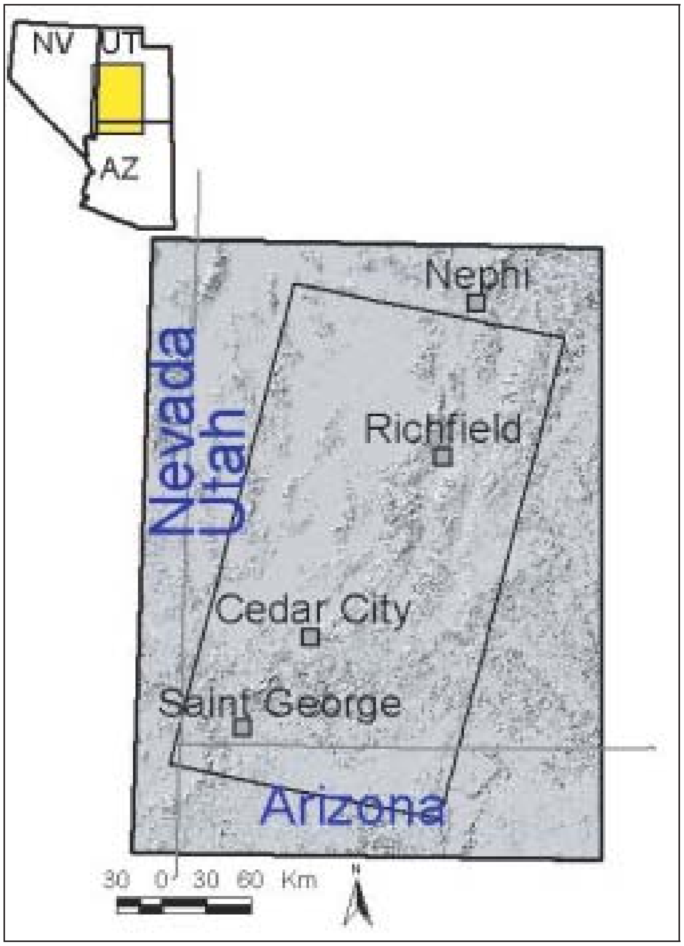

Figure 1—Southern Utah Fuel Management Demonstratioin Area location.

broad range of fuel and fire regimes associated with vegetation types representative of the High Plateaus of Utah, the Great Basin, the Colorado Plateau, and the Mohave Desert. Previous mapping efforts (Homer et al. 1997) identified 29 vegetation types in the demonstration area (table 2), including various associations of pinyon-juniper, ponderosa pine, sagebrush-grass, aspen, spruce-fir, mountain shrubs, and desert shrubs. These vegetation types are similar in species composition, stand structure, and ecologic function, so they have similar fire regimes to vegetation types found on hundreds of millions of acres in the 11 western states. Thus fuel treatment guidelines developed in the project have wide potential applicability.

Table 1—Percent composition of land ownership in Southern Utah study area.

Agency	Percent of study area
Bureau of Land Management	49
Forest Service	23
Private	19
State Lands	7
Bureau of Indian Affairs	1
National Park Service	1

Table 2—Percent composition of major vegetation types in Southern Utah study area from Utah and Arizona combined GAP Analysis cover type databases (Homer et al. 1997).

Cover type	Percent of study area
Pinyon-Juniper	28.4
Salt Desert Scrub	16.7
Sagebrush	11.4
Sagebrush-Grass	6.7
Grassland	5.6
Agriculture	4.0
Spruce-Fir	3.5
Blackbrush	3.2
Aspen	3.0
Oak	3
Ponderosa Pine	2.4
Creosote-Bursage	1.6
Mountain Shrub	1.6
Dry Meadow	1.5
Mountain Fir	1.4
Ponderosa Pine/Shrub	1.3
Barren	1.2
Desert Grassland	1.2
Water	.7
Alpine	.3
Greasewood	.3
Lowland Riparian	.3
Urban	.3
Mountain Riparian	.2
Aspen/Conifer	.1
Mountain Fir/Mountain Shrub	.1
Pickleweed Barrens	.1
Spruce Fir-Mountain Shrub	.1
Wet Meadow	.1
Wetland	.1

Project Objectives

This project has three main objectives. First is to develop Geographic Information System (GIS) data layers of fuel, vegetation, weather, and terrain inputs necessary to conduct fire behavior and fire effects analysis across the entire study area. We will use a number of computer models to accomplish this including FARSITE (Fire Area Simulator) (Finney 1998), FlamMap (Finney in progress), NEXUS (Scott 1999), FOFEM (First Order Fire Effects Model) (Reinhardt et al. 1997), LANDSUM (Keane et al. 1997a), a Southern Utah variant of the Fire and Fuel Extension to the Forest Vegetation Simulator (FFE-FVS) (Reinhardt and Ryan 1998), and FIREHARM (Keane in progress). In addition, we will develop criteria for landscape features and map those that deserve important consideration in fire management decisions including infrastructure features and other natural and cultural resource values such as municipal watersheds and wildland urban interface areas (WUI). The second objective is to apply and test these models on the various fuel types and weather conditions across the study area. This objective will facilitate review and validation of model inputs and outputs. The third objective is to demonstrate the use of these models and data at a variety of scales in order to aid in the implementation of fuel treatment activities within the study area. This includes the development of fuel treatment guidelines for the various vegetation types.

Current Efforts

Recently, a number of federal land management agencies have invested in efforts to produce management strategies using GIS data and fire models to reduce extreme fire effects to communities, and cultural and natural resources, while at the same time increasing the health of ecosystems and decreasing the probability of extreme fire events. An initial attempt at developing such a strategy was the "Development of Coarse-Scale Data for Wildland Fire and Fuel Management" (Schmidt et al. 2002). This effort integrated maps depicting potential natural vegetation, current cover type, historical natural fire regimes, and current fuel condition and identified areas in the continental United States where current fire regimes are significantly different from historic conditions and where fuel conditions are potentially suitable for treatment. In addition, this effort also combined maps depicting housing density, extreme fire behavior, and fire exposure from vegetation to assess potential threat of fire to the wildland-urban interface.

The Utah BLM has used Utah GAP Analysis GIS data, created from LANDSAT-TM data (Homer et al. 1997), to assess state level fire hazard and model fire potential for fire planning and dispatch purposes (Wimmer et al. 2000). When combined with GIS data layers depicting population densities and historical fire occurrence, it identified potential areas with a serious fire threat as well as areas where detailed interagency planning and tactical analyses and treatment may be needed.

These efforts indicate areas of Southern Utah are at risk and are suitable for fuel treatments. They provide a strategic basis for broad, regional scale programmatic direction, particularly with respect to fire suppression activities, but lack the spatial and thematic resolution required by current state-of-the-art models to prioritize and locate landscape or project-level fuel treatments. Generally, these models require 30 m resolution datasets that describe a wide range of fuel and vegetation attributes such as surface fuel loads, crown fuel, density and height of vegetation, and biophysical site potential. In addition, they do not utilize existing fire behavior and effects models to test the relative efficacy of fuel treatments. Concurrent with these efforts, a number of land management agencies in Utah have initiated numerous site and project level fuel treatment projects utilizing funding provided by the National Fire Plan. However, these efforts are not directly linked to existing strategic and forest level planning efforts and do not integrate these programs within and across multiple temporal and spatial scales.

Multi-Scale Fuel Treatment Analysis

Recently, a multi-scale, integrated planning approach has been identified as a way to link broader scale fuel management plans to more site-specific project level fuel treatment projects (Hann and Bunnell 2001). This approach maximizes efficiency at each scale and can be more successful at achieving objectives not only at the project level but upward to the regional and national scales (Hann and Bunnell 2001). Also, in order to rate the relative effectiveness of fuel treatments at a variety of scales, simulating potential fire behavior and fire effects in a modeling environment will require use of the different models at different scales (Reinhardt et al. 2001). The Southern Utah Fuel Management Demonstration Project will utilize a number of fire behavior and fire effects models at what we feel is the appropriate scale (table 3). This project has worked closely with fire managers within the study area to produce products that will allow them to prioritize, select, and implement fire restoration projects across a number of different spatial and temporal scales. We have sponsored

Table 3—Fire behavior and fire effects models used in the Southern Utah Fuel Management Demonstration Project with corresponding management application and spatial scale of application.

Model	Applications	Scale
LANDSUM	Management strategy comparisons to determine vegetation response and predict successional pathways to reduce future fire intensity, crowning potential and flame length.	Sub-basin/sub-regional
FIREHARM	Management strategy comparisons to assess the probability of extreme fire effects using various weather scenarios.	Sub-basin/sub-regional
FARSITE/ Flammap	Management strategy comparisons under various weather scenarios.	Sub-watershed/landscape
NEXUS	Management strategy comparisons for reducing crown fire risk including thinning, pruning, and other fuel removal.	Stand/polygon and Sub-watershed/landscape
FOFEM	Management strategy comparisons to determine optimum treatment schedules to reduce first-order -impacts on surrounding environments under wildfire and prescribed fire conditions.	Stand/polygon and Sub-watershed/landscape
FFE-FVS	Management strategy comparisons to determine optimum treatment schedules that reduce fire intensity, tree mortality, and surface fuel loading over time.	Stand/polygon

series of workshops attended by local fire managers to help us develop GIS layers, understand vegetation succession, assign fuel attributes to vegetation layers, and design weather scenarios to ensure these products are valid and usable.

The coarsest scale of analysis, across a sub-basin and sub-regional geographic extent, will require the use of a spatially explicit multiple pathway succession model called LANDSUM to assess general trends in vegetation distribution and fire regimes through multi-century time periods. This analysis will primarily determine departure of current fuel and vegetation conditions from historical conditions as well as allow us to simulate potential future conditions under changing climatic regimes. We will initially develop successional pathway models using the Vegetation Dynamics Development Tool (VDDT) (Beukema and Kurz 1998). We will also use LANDSUM to predict the consequences of the treatment versus no treatment of fuel over somewhat shorter time scales such as 10 years, 50 years, and 100 years within portions of the study area. The model will be used to programmatically simulate fuel treatment prescriptions throughout the study area to reduce extreme fire events and maintain healthy ecosystems and communities, mainly surrounding the higher priority areas.

The model FIREHARM will characterize potential fire hazard and fire risk for the entire study area across time periods more appropriate for operational fire management planning by using computed spatial and temporal probabilities of extreme fire events. This will integrate 18 years of daily weather data, from 1980-1996, with fire behavior and fire effects models to compute temporal probabilities of user-specified fire events occurring within the study area for every polygon on the landscape. Values at risk, including communities, natural, and cultural areas of importance in the southern Utah landscape, will be assessed with regards to the fire hazard and risk maps.

The middle scale of analysis, at a sub-watershed or landscape geographic extent, will consist mainly of varying the spatial pattern of fuel and vegetation on the landscape in order to test ways of reducing the intensity and severity of wildfires and protecting values at risk. The spatial pattern of fuel treatments has an important effect on disrupting fire growth across a landscape. Work by Finney (2000) indicates that one effective means to reduce the intensity and severity of wildfires is to treat fuel in such a way as to minimize the area burned by a head fire. We will work with local fire managers to identify areas on the landscape where they have planned or implemented fuel treatment projects and use these areas for intensive modeling. We will use FARSITE and FlamMap to game the portions of these sub-watersheds and landscapes in order to determine the size, shape, and configuration of fuel treatments necessary to form an effective barrier to the development of large fires or to the spread of large fires into sensitive areas identified as having high values at risk. The FARSITE model is a state of the art model for predicting the spread and intensity of fires across a landscape. It is designed to model continuous fire behavior through time using spatial data representing elevation, slope, aspect, surface fuel model, canopy cover, crown base height, crown bulk density, and stand height. FlamMap predicts fire behavior without the use of a fire spread algorithm and produces maps of surface and crown fire behavior characteristics using FARSITE input data layers for a given set of weather and / or fuel moisture data inputs for all points across the landscape simultaneously. Fuel treatment strategies will include a method for linking or modifying the landscape with respect to low spread rate landscape polygons identified through the use of FARSITE and FlamMap.

In addition, NEXUS will be used to determine the effect of combinations of thinning, pruning, and fuel removal on likelihood of surface fire vs. crown fire and FOFEM will be used to evaluate changes in fire effects (fuel consumption, smoke, tree mortality, soil heating, and mineral soil exposure) with wildfire vs. prescribed fire conditions for these selected landscape as well. The NEXUS model calculates instantaneous changes in fire behavior, as well as transitions from surface fire to torching to crown fire with varying harvest, pruning, and fuel treatment options. We are also developing a spatial variant of NEXUS that will allow retrieval of data from GIS databases, and will produce output in a format suitable for inclusion in GIS for spatial analysis and mapping.

Stand or polygon level analysis will consist of individual non-spatial model simulations of the full range of fuel and vegetation conditions identified within the study area. The Fire and Fuel Extension to the Southern Utah variant of the Forest Vegetation Simulator will be used to model forest and woodland fuel and vegetation types. Simulation runs will summarize fuel conditions and associated fire behavior and effects resulting from a variety of potential fuel treatment strategies. NEXUS will be used to determine the effect of combinations of thinning, pruning, and fuel removal on likelihood of surface fire vs. crown fire. Simulation runs using FOFEM will be used to summarize fuel conditions and associated fire behavior and effects resulting from a variety of potential fuel treatment in non-forested and non-woodland vegetation types. We will calculate first order fire effects such as fuel consumption, smoke production, and tree mortality using the FOFEM, including the use of a “batch” mode programmed to run for multiple stands at the landscape level. We will analyze stand or polygon scale treatment effects on long term fuel and fire dynamics with varying treatments to determine optimum treatment schedules the FFE-FVS model. We will be using a new regional variant that addresses species that occur in Southern Utah and the Intermountain West.

Summary

The Southern Utah Fuel Management Demonstration Project is a Joint Fire Sciences project that creatively links a number of fire behavior and fire effects models to allow comparison of fuel management strategies for roughly 13 million acres of Southern Utah and 2 million acres of Northern Arizona. The databases and models will be used to support the planning and implementation of an integrated, interagency landscape level fuel management program for the region. A multi-scale analysis strategy using output from a variety of fire behavior and effects models will be developed. Broader-scale analyses at the sub-basin or sub-regional scale will employ the models FIREHARM and LANDSUM across the entire study area. Sub-watershed and landscape level analyses will primarily use the FARSITE, FlammMap, and spatial versions of NEXUS and FOFEM on selected landscapes. Stand and polygon level analyses will be simulated using the Fire and Fuel Extension to the Southern Utah variant of the Forest Vegetation Simulator (FFE-FVS) and non-spatial versions of FOFEM and NEXUS for representative fuel and vegetation conditions in the study area.

References

Beukema, S.J.; Kurz, W.A. 1998. Vegetation dynamics development tool user's guide. Version 3.0. Prepared by ESSA Technologies Ltd., Vancouver, BC. 102 pp. Pp 309-324.

Finney, Mark A. [In progress]. FlamMap. Rocky Mountain Research Station, Fire Sciences Lab, Missoula, MT.

Finney, Mark A. 2000. Design of regular landscape fuel treatment patterns for modifying fire growth and behavior. Forest Science. 47(2): 219-228.

Finney, Mark A. 1998. FARSITE: Fire Area Simulator - model development and evaluation. Res. Pap. RMRS-RP-4. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 47 p.

Hann, Wendel J.; Bunnell, David L. 2001. Multi-scale land and fire planning. International Journal of Wildland Fire. 10: 389-403.

Homer, C.G., Ramsey, R.D., Edwards, T.C., Falconer, A. 1997. Landscape cover-type modelling using multi-scene thematic mapper mosaic. Phototgrammetric Engineering and Remote Sensing. 63: 59-67.

Keane, Robert E. [In progress]. Fire harm. Rocky Mountain Research Station, Fire Sciences Lab, Missoula, MT.

Keane, Robert E.; Long, D.G.; Basford, D.; Levesque, B.A. 1997. Simulating vegetation dynamics across multiple scales to assess alternative management strategies. In: Integrating spatial information technologies for tomorrow: GIS '97 conference proceedings; 1997 February 17-20; Fort Collins, CO : GIS World: 310-315.

Reinhardt, Elizabeth D., Keane, Robert E.; Brown, James K. 2001. Modeling fire effects. International Journal of Wildland Fire. 10: 373-380.

Reinhardt, Elizabeth D.; Ryan, Kevin C. 1998. Analyzing the effects of management actions including salvage, fuel treatment, and prescribed fire on fuel dynamics and fire potential. In: Pruden, Teresa L.; Brennan, Leonard A., eds. Fire in ecosystem management: shifting the paradigm from suppression to prescription, proceedings of the 20th Tall Timbers fire ecology conference; 1996 May 7-10; Boise, ID. Tallahassee, FL: Tall Timbers Research Station: 206-209.

Reinhardt, E.D.; Keane, R.E.; Brown, J.K. 1997. First Order Fire Effects Model: FOFEM 4.0, user's guide. Gen. Tech. Rep. INT-GTR-344. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 65 p.

Schmidt, Kirsten M.; Menakis, James P.; Hardy, Colin C.; Hann, Wendel J.; Bunnell, David. 2002. Development of coarse-scale spatial data for wildland fire and fuel management. Gen. Tech. Rep. RMRS-GTR-87. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Scott, Joe H. 1999. NEXUS: A system for assessing crown fire hazard. Fire Management Notes. 59(2): 20-24.

U.S. General Accounting Office. April 1999. Western National Forests: A cohesive strategy is needed to address catastrophic wildfire threats. Report to the subcommittee on forest and forest health, committee on resources, House of Representatives. GAO / RCED-99-65.

U.S. Department of Agriculture, Forest Service. 2000. Protecting people and sustaining resources in fire-adapted ecosystems: a cohesive strategy. The Forest Service Management response to General Accounting Office Report, GAO / RCED-99-65. 86 p.

U.S. Department of Agriculture, U.S. Department of the Interior. 2000. National fire plan: managing the impact of wildfire on communities and the environment. 35 p.

U.S. Department of Agriculture, U.S. Department of the Interior. 1995. Federal wildland fire management, policy and program review-final report. National Interagency Fire Center: Boise, ID. 45 p.

Wimmer, Sheldon; Shive, John C.; Sheffey, Jack. 2000. Utah statewide fire assessment. Proceedings Crossing the Millennium: Integrating Spatial Technologies and Ecological Principles for a New Age in Fire Management. Joint Fire Science Conference and Workshop; Volume 1. Boise, Idaho, 1999 June 15-17. 298 p.

Fire Regime Condition Class and Associated Data for Fire and Fuels Planning: Methods and Applications

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Abstract—*A pilot project was conducted in the Trout West watersheds of the Pike National Forest in Central Colorado. Maps and interpretations were developed to support prioritization, planning, and effects analysis for fuel and ecosystem restoration to achieve National Fire Plan Cohesive Strategy options. The area is about 65,000 hectares (135,000 acres) in size and representative of Southern Rocky Mt. Province ponderosa pine ecosystems. Fire regime potential vegetation-fuel types, departure from central tendency of the historical range of variability (HRV), fire regime condition class, wildfire ignition risk, wildland urban interface, fuel models, and associated information were mapped. An analysis was conducted indicating that treatment of about 10,000 hectares (25,000 acres) of high and moderate HRV departure areas, and maintenance of about 2000 hectares (5000 acres) of low departure areas, could achieve condition class 1 over a 5-year period. Treatment and maintenance focused on a landscape design substantially reduced wildfire risk to both wildland urban interface and ecosystems. A treatment option focused only on wildland urban interface and buffer areas did not substantially reduce risk to communities or ecosystems when compared to the no-treatment option.*

Introduction

The Pike National Forest is located in central Colorado and contains much of the Rocky Mountain Front between Pueblo and Denver. Wildfire is a substantial risk to National Forests as well as adjacent homeowners in the wildland urban interface. A Forest Service pilot project was conducted to evaluate methods for mapping and interpretation of hazardous fuel and associated data for prioritization and planning of restoration projects to reduce risks to ecosystems and people. The area selected for the project was the Trout West watersheds located west of Colorado Springs. The ecosystems of these watersheds are considered representative of the ponderosa pine ecosystems of the Southern Rocky Mt. Province (Bailey 1995). These watersheds also contain considerable wildland surban interface near the community of Woodside, Colorado.

Hann and Bunnell (2001) provide an overview of multi-scale methods for planning and implementation of the National Fire Plan using the Forest Service cohesive strategy (USDA Forest Service 2000) guidance across multiple scales of planning. In the overview they emphasize the importance of stepping down the coarse-scale fire regime condition class data developed by Hardy and others (2001) along with other key data for prioritization and planning at finer scales. Hann and Bunnell (2001) provide definitions of the natural fire

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regime groups and fire regime condition classes that we refined for this study (tables 1 and 2).

The primary objective of the Trout West pilot project was to develop the finer scale methods and applications for fire regime condition class. In addition, a number of other variables, including wildland urban interface, wildfire occurrence risk, and fuel models, were also developed. A suite of additional resource and geographic variables was also used in the integrated prioritization and planning process.

Table 1—Natural (historical) fire regime classes from Hardy et al. (2001) and Schmidt et al. (2002) as interpreted by the authors for modeling landscape dynamics in the Trout West watersheds.

Fire regime class	Frequency (mean fire return interval)	Severity	Modeling assumptions
I	0 – 35+ years, Frequent	Surface and mixed	Open forest, woodland, and savannah structures maintained by frequent fire; also includes frequent mixed severity fires that create a mosaic of different age post-fire open forest, woodland, shrub, or herb patches that make a mosaic of structural stages. Mean fire interval can be greater than 35 in systems with high temporal variation.
II	0 – 35+ years, Frequent	Replacement	Shrub or grasslands maintained or cycled by frequent fire; fires kill non-sprouting shrubs which typically regenerate and become dominant within 10-15 years; fires remove tops of sprouting shrubs which typically resprout and dominate within 5 years; fires typically remove most tree regeneration.
III	35 – 100+ years, Infrequent	Mixed and surface	Mosaic of different age post-fire open forest, early to mid-seral forest structural stages, and shrub or herb dominated patches generally <40 hectares, maintained or cycled by infrequent fire. Interval can range up to 200 years.
IV	35 – 100+ years, Infrequent	Replacement	Large patches generally >40 hectares, of similar age post-fire shrub or herb dominated structures, or early to mid-seral forest cycled by infrequent fire. Interval can range up to 200 years.
V	200+ years	Replacement, mixed, and surface	Variable size patches of shrub or herb dominated structures, or early to mid to late seral forest depending on the type of biophysical environment. Cycled by rare fire or other disturbance events. Often have complex structures influenced by small gap disturbances and understory regeneration.

Methods

Findings on assessment of methods for project and watershed scale fire regime condition class rating were reported by Hann (2003). Use of these methods for mapping fire regime condition class and associated variables were initially tested in a smaller, approximately 9,000 hectare (20,000 acre) watershed restoration-planning project on the San Isabel National Forest to the west, which resulted in a number of recommendations for improvement (McNicoll and Hann 2003). We decided to further test and develop these methods in the Trout West watersheds, a larger area of about 53,000 hectares (130,000 acres) that had both different ecosystems and different types of base vegetation, fuels, and fire data.

Table 2—Condition classes from Hardy et al. (2001) and Schmidt et al. (2002) as interpreted by the authors for modeling landscape dynamics and departure from historical or natural range of variability in the Trout West watersheds. Historical range of variability (HRV) is the variability of regional or landscape composition, structure, and disturbances, during a period of time of several cycles of the common disturbance intervals, and similar environmental gradients, referring, for the United States, to a period prior to extensive agricultural or industrial development (synthesized from Morgan et al.1994; Landres et al. 1999; Hann et al. 1997). Natural range of variability (NRV) - the ecological conditions and processes within a specified area, period of time, and climate, and the variation in these conditions that would occur without substantial influence from mechanized equipment.

Class	NRV or HRV departure	Description
Condition class 1	Low	Vegetation composition, structure, and fuels are similar to those of the natural regime and do not predispose the system to risk of loss of key ecosystem components. Wildland fires are characteristic of the natural fire regime behavior, severity, and patterns. Disturbance agents, native species habitats, and hydrologic functions are within the natural range of variability.
Condition class 2	Moderate	Vegetation composition, structure, and fuels have moderate departure from the natural regime and predispose the system to risk of loss of key ecosystem components. Wildland fires are moderately uncharacteristic compared to the natural fire regime behaviors, severity, and patterns. Disturbance agents, native species habitats, and hydrologic functions are outside the natural range of variability.
Condition class 3	High	Vegetation composition, structure, and fuels have high departure from the natural regime and predispose the system to high risk of loss of key ecosystem components. Wildland fires are highly uncharacteristic compared to the natural fire regime behaviors, severity, and patterns. Disturbance agents, native species habitats, and hydrologic functions are substantially outside the natural range of variability.

The project was organized through the identification and management of 10 steps. These involved:

1) Identify and map fire regime potential vegetation-fuel types (FRPVT)
2) Model historical range of variation (HRV)
3) Assess current conditions
4) Compare current vegetation-fuel conditions with central tendency of HRV
5) Compare current fire interval and severity with central tendency of HRV
6) Summarize fire regime condition class for each FRPVT
7) Summarize area to treat and maintain to achieve condition class change
8) Map wildfire occurrence risk
9) Map fuel models
10) Map the wildland urban interface (WUI)

Identify and Map Fire Regime Potential Vegetation-fuel Types (FRPVT)

Methods for classification of natural fire regime potential vegetation-fuel type (FRPVT) stratification were based on several criteria (Hann 2003). The overall objective was to stratify based on identification of biophysical conditions that create substantial differences in management implications for restoration of fire-adapted ecosystems and reduction of risk to people and property. These key management implications involved identifying those factors that have caused substantial change in the natural fire regime conditions, such as:

1) exclusion of fire through suppression and lack of wildland fire use that mimics the natural regime;

2) past management (for example, harvest and grazing practices) that have not mimicked the natural effects of fire and other disturbance regimes; and
3) exotic invasions (for example cheatgrass, knapweed, and blister rust).

Criteria recommended to aid in stratification of FRPVT include biophysical type differences in:

1) pre-suppression fire interval group (0-35+, 35-100+, >200 years);
2) pre-suppression fire severity (surface versus replacement versus mixed);
3) upper layer lifeform potential (herbland, shrubland, woodland, forestland, barrenlands, ag-urban);
4) dominant upper layer species complexity potential (if one, two, or greater than two dominant species);
5) lower layer (understory) lifeform indicator species;
6) standing and down fuels - duff/litter layer potential;
7) climate (temperature/ moisture zones not associated with slope-aspect); and
8) slope-aspect (such as flat, cool aspect slopes, and warm aspect slopes).

The Land Type Association (LTA) map was used to examine broad vegetation type differences, and then further divided topographically to reflect changes in fire regimes. FRPVT classes were not broken down as finely as may be available in detailed plant association or habitat type maps, as this would cause a large number of stratifications that would not be meaningful for development of management implications. As such, the FRPVT was identified first relative to the fire regime group and secondarily to vegetation-fuel type species indicators that are important for management implications. The key was to define the fire regime group first, then stratify the vegetation biophysical type associated with a fire regime group that provides the linkage for development of management treatments.

Reconnaissance transects were driven or walked to identify the elevation, aspect, and slope breaks associated with changes in fire regime group and associated vegetation-fuel types. Initial field classification of fire regime group and potential vegetation type followed the methods outlined by Hann (2003). Fire scarred trees were located and tree boring and scar counting methods were used to nondestructively classify the pre-suppression fire interval group. Fire scarred stumps were located and cross-sections were cut to allow more accurate counts of the intervals between scars in order to validate the classification of the fire interval group. Initial classification of fire severity group was assigned based on substantial presence (surface fire regime), presence (mixed fire regime), or absence (replacement fire regime) of fire scarred trees or stumps. In a later step, simulation modeling of historical range of variability (HRV) of fire interval and fire severity was used to crosscheck these classifications.

Intensive fire scar cross-dating, tree ring chronology, and ground mapping were not conducted. Methods for characterization of FRPVT and the HRV were designed for rapid ground reconnaissance combined with use of available data, review of the literature, and comparison of historic and current photographs for integration using simulation modeling. For this study we decided to limit ourselves to a level of inventory and analysis effort that could typically be expended on most fire and fuel management projects. Therefore we decided not to impose intensive and costly methods typically used for fire history research.

Most FRPVT(s) were mapped using GIS map query assignments to terrain model classes of elevation, aspect, and slope. Digital elevation models (DEM) were utilized to derive these aspect, slope, and elevation terrain models. However, the riparian valley FRPVT could not be mapped using this process so the land type association map was used to make this assignment. The high elevation grassland FRPVT also could not be mapped using the terrain model and was delineated using aerial photos and digital orthophotos. Urban polygons were also identified from photo interpretation. Lakes were available from existing map coverages.

Model Historical Range of Variation (HRV)

The HRV for vegetation-fuel conditions, fire frequency, and fire severity was simulated using the vegetation development dynamics tool software (VDDT) (Beukema and Kurz 2001). We opted to use the standardized succession and disturbance model called the "box" model in order to have an organizational framework with predefined successional stages and disturbances (Hann 2003).

Using the "box" model we conducted numerous simulations of HRV for vegetation-fuel class composition, fire interval group, and fire severity group by adjusting fire, other disturbance, and succession probabilities. Landscape conditions that would have existed before active fire suppression were simulated over a 500-year period with a climate similar to the current. Native American influences on fire frequency and intensity were considered part of the natural or native system (Barrett and Arno 1982). Utilizing fire scar interval counts and other historical clues from the reconnaissance transects, probabilities of fire occurrence and succession were calculated and used as a range of inputs to sensitivity test the models. This information was then combined with evaluation of historical and current photos, literature (Brown and others 1999, Kaufmann 2000, Kaufmann and others 2000 and 2001), local knowledge, and results of the sensitivity testing to determine the final combination of disturbances and succession probabilities. The final HRV was simulated 10 times to account for variability. The key was to develop an estimate of the variation in natural landscape dynamics that would occur without active fire suppression and other modern anthropogenic influences over a long time period under the current climate.

We did not have an objective to attempt to simulate HRV with high accuracy or conduct extensive validation. The objective was to simply identify the major trends of conditions and processes that occurred in HRV to use as a broad reference for determining departure of current conditions and processes. From this we calculated an average for the HRV class composition, fire interval, and fire severity. This average provided an estimate of the central tendency of the HRV to be used as a reference condition for comparison with current conditions. The methods for comparison of current conditions with the reference estimate of central tendency follow those of Clements (1934) and Mueller-Dombois and Ellenberg (1974). Because of the lack of intensive FRPVT specific HRV ground truth, we followed Hann (2003) in using plus or minus 33% from the HRV average as including the typical HRV. This is a compromise between the plus or minus 25% recommended by Keane et al. (1996, 1997, 2002) for simulation modeling and the 80% median range recommended by Hessburg (1999) for historical photo analysis.

For each FRPVT, the "box" model classes were cross-referenced with the current vegetation-fuel classification for cover type, size class, and canopy closure classes. The HRV composition was then calculated. This provided a characterization for the vegetation-fuel class composition specific to the FRPVT that could be cross-walked to the current vegetation-fuel map data. An example is provided in table 3 for the frequent surface fire regime lower elevation undulating ponderosa pine FRPVT. In addition, the average fire interval, amount of surface fire, replacement fire, and total fire were also determined from this final simulation data.

Table 3—Simulated average for each class of the "box" model during the historical range of variation (HRV) for the frequent surface fire regime lower elevation undulating ponderosa pine FRPVT. The average was used as the measure of central tendency for the HRV. A plus or minus 33% variation or range of 66% was used as a measure of the range of variation.

"Box" model class	Box model description	FRPVT current vegetation class description	HRV % average
A	Early development	Tree regeneration open/grassland	14
B	Mid development closed	Closed canopy pole	4
C	Mid development open	Open canopy pole	11
D	Late development open	Open canopy mid mature–mature	59
E	Late development closed	Closed mid mature - mature	12
F – L	Did not occur in HRV	Other vegetation classes	
Total			100

The valley riparian and high elevation meadows were not modeled for the HRV because there would be little to no active management in these areas (the grasslands are almost entirely on private land).

Assess Current Conditions

The Resource Inventory System (RIS) map coverage was used as the vegetation data source for cover type and structure. Many polygons lacked canopy closure data; some lacked cover type. Vegetation data of this scale was not available for private lands. Consequently, we digitized stand polygons across private lands and populated these with photo interpreted cover type and structure to fill the missing data. Missing data in National Forest polygons were also attributed utilizing aerial photo interpretation.

Canopy closures from the RIS maps were combined into three classes:

Shrub/herb/tree seedling (S): <5-10% sapling and larger tree cover, corresponds with HRV class A;

Open forest (O): generally 11-50% canopy closure of sapling and larger trees, corresponds with HRV classes C or D; and

Closed forest (C): generally > 50% canopy closure of sapling and larger trees, corresponds with HRV classes B or E.

Structural classes (as defined by size) from RIS were also combined:

Early seral (1): tree seedling, shrub, herb (attributed in RIS as Structural Habitat Stage 1; corresponds with HRV class A;

Mid seral (2 and 3): sapling (2) and pole (3) (attributed in RIS as structural habitat stage 2 or 3; corresponds with HRV class B or C; and

Late seral (4 and 5): mature saw timber (4) and old growth (5) attributed in RIS as Structural Habitat Stage 4 or 5; corresponds with HRV class D or E.

During the reconnaissance transects we recognized substantial Douglas-fir tree mortality in many stands. We decided to identify stands with substantial (>50%) recent mortality due to Tussock moth and Douglas-fir beetle. Local insect and disease inventories indicate that these landscape scale outbreaks were due to the much greater amount of Douglas-fir in the landscape than would naturally occur without fire suppression. This greater amount results in high current landscape level vulnerability to mortality, rather than the historical individual tree or group mortality in scattered stands. Although a few stands within the landscape may have been characteristic of mortality during the HRV, we concluded that many stands within the landscape would be uncharacteristic of conditions in the HRV. Unfortunately, the local forest insect and disease aerial survey maps were mapped at too coarse a scale for stand level attribution of mortality. Consequently, we utilized our most recent aerial photos (1997) and local knowledge to identify those stands with substantial mortality.

In summary, the following vegetation attributes were required for the analysis:

1. Cover type
2. Canopy closure
3. Structural (size) class
4. Substantial mortality

The current average fire interval was estimated to be 1% based on the fire occurrence data for the whole Trout West watershed area. This was determined as an average for the whole area rather than for each FRPVT because the fire occurrence data only exists for the past 30 years and locations and net size of fires have low accuracy. Current amount of replacement fire as a percent of total fire was estimated at 90% based on reconnaissance of recent wildfire areas and local knowledge of fire behavior in typical stand conditions.

Compare Current Vegetation-Fuel Conditions With HRV

Utilizing a variety of GIS and other computer tools (Spatial Tools, Arcview, Arcinfo, Excel), a frequency table was derived to depict each unique combination of FRPVT, cover type, size class, canopy cover, and mortality for the current map coverage. Each combination was concatenated and added to a new item labeled "Key."

A "Class" item was then added, and each unique combination (Key) was cross-referenced to the associated HRV Structural Class (A, B, C, D, E). In addition to the standard HRV classes A-E, "box" model Classes I-L applies to uncharacteristic conditions. These were not included in the HRV characteristic classes on the premise that historical conditions would have contained none or very minimal (<1%) amounts of such types at a landscape scale:

Class G: Uncharacteristic Timber Harvest (harvesting has produced a type that did not occur at a landscape scale in HRV);

Class I: Uncharacteristic Succession (succession has proceeded beyond the HRV range producing a type that did not occur at a landscape scale in HRV); and

Class L: Uncharacteristic Insect or Disease mortality (insect and disease mortality creating a type that did not occur at a landscape scale in HRV).

A corresponding item "Name" was created as a class descriptor, and attributed with a label, such as Open Mid-Development. An acre field was added to depict the corresponding acreages for each Class. Utilizing Excel, a sum is calculated within each FRPVT for each class. A similarity comparison was then made between HRV and current vegetation-fuel conditions for each FRPVT (table 4). This measure of similarity was developed by Clements (1934) and is considered to be one of oldest and most straightforward measures of similarity or its inverse, which is dissimilarity (100-similarity). In this study we use the term "departure" in the same manner as dissimilarity. The difference for any one class was calculated as (current – HRV average)/(current + HRV average) expressed as a percent. A departure contribution of low was considered to be within a range greater than -25% and less than +25% difference from the average for HRV. The contribution of moderate was considered to be less than or equal to -25%, but greater than -75%, and greater than or equal

Table 4—Example calculation of similarity of current conditions to HRV for the frequent surface fire regime lower elevation undulating ponderosa pine FRPVT. The similarity for any Class is the smaller of HRV or current amounts.

"Box" model class	HRV %	Current %	Similarity %	Difference %	Departure contribution	Abundance	Management implication
A	14	9	9	- 22	Low	Similar	Maintain
B	4	5	4	+ 11	Low	Similar	Maintain
C	11	2	2	- 69	Moderate	Rare	Recruit
D	59	20	20	- 49	Moderate	Rare	Recruit
E	12	31	12	+ 44	Moderate	High	Reduce
F – L	0	33	0	+ 100	High	High	Reduce
Total	100	100	47				

to +25%, but less than +75%, while the high class accounted for -75% to -100% and +75% to +100%. Abundance was considered to be rare if less than -25%, similar if between plus or minus 25%, and high if equal or greater than 25%. The general management implication for landscape scale restoration to reduce departure would be to maintain similar classes, maintain and recruit rare classes, and reduce the high classes.

The sum of similarity for classes was calculated. An example for the frequent surface fire regime low elevation undulating ponderosa pine (FRPVT 1) is depicted in table 4.

We would emphasize that as discussed by Hann (2003), HRV similarity or departure, as well as fire regime condition class, are landscape and not stand variables. Any specific fine scale pixel or stand can occur in any one of the characteristic HRV classes (A through E). The similarity to HRV or departure depends on how much of each of these classes occurred during HRV vs. how much now occur.

However, in order to identify the risk that a given stand or pixel contributes to the condition class or departure from central tendency and the management implications, we calculate additional variables that represent the Departure Contribution and HRV Conditions for each Class (example in table 4):

Departure Contribution:
Low Contribution: Classes A-E; < ± 25% difference from HRV
Moderate Contribution: Classes A-E; ≥ ± 25% difference from HRV; <75%
High Contribution: Classes A-E ≥ 75% difference from the HRV; plus uncharacteristic types
HRV Conditions (management implications) or abundance classes:
Maintain: Classes A-E; < ± 25% difference from HRV
Similar Abundance
Recruit: Classes A-E; > -25% difference from HRV
Rare Abundance
Reduce: Classes A-E; > +25% difference from HRV
High Abundance
Restore: Classes F-I (Uncharacteristic types)
High Abundance

This difference was calculated as (current – historic)/(current + historic) * 100.

Utilizing GIS and other computer tools (Spatial Tools, Excel, Arcinfo, and Arcview) these departure contribution classes were displayed spatially and summarized.

Compare Current Fire Interval and Severity With HRV

To compare the current fire interval and severity with HRV, we followed the method outlined by Hann (2003). Since these values are measured in years (fire interval) and percent canopy replacement (fire severity) the similarity of historical to current can be determined by calculating a ratio of the smaller divided by the larger (Mueller-Dombois and Ellenberg 1974). The departure can then be calculated by subtracting the ratio from 1 and multiplying times 100. If the current interval is less than the HRV average (currently more frequent) the current is divided by the HRV average, while if the current interval is greater than the HRV average (less frequent fire) the order is reversed. If the current interval is determined to be still within the HRV range for the fire regime group then the HRV average is equal to the current interval resulting in 100 % similarity. Classification of departure from the HRV average assumes that the variation from 0 to 33 % is within HRV, while higher values of departure are outside the HRV.

The basis for using the larger of the current interval or the HRV average as the denominator was to provide an estimate of the proportional ratio of change irrespective of the direction (more or less frequent). As long as variation was allowed within the departure and condition classes to account for the HRV variation and a rule was imposed for those FRPVT and landscapes where fire interval was not outside of the HRV interval range, this methodology normalizes the differences.

The current fire interval probability was calculated as the current percent occurrence divided by 100. As discussed earlier, the current fire occurrence was estimated at 1 per cent, or a .01 probability for the Trout West landscape or for any FRPVT in the Trout West landscape. The HRV fire interval probability was calculated by dividing 1 by the HRV mean fire interval. An example calculation for the frequent surface fire regime low elevation gentle ponderosa pine follows:

Current fire probability = 1/100 = .01
HRV mean fire interval = 21 years
HRV fire interval probability = 1/21 = .047
Current to historical interval similarity = (.01/.047) * 100 = 21%.

The current severity probability is the percent occurrence of current replacement fire divided by 100. This was estimated to be 90 per cent, or a .9 probability for the Trout West landscape or for any FRPVT in the Trout West landscape. The HRV severity probability was calculated by dividing the average percent of HRV replacement fire by 100. An example calculation for the frequent surface fire regime low elevation gentle ponderosa pine follows:

Current replacement fire probability = 90/100 = .9
HRV mean replacement fire = 24 %
HRV replacement fire probability = 24/100 = .24
Current to historical severity similarity = (.24/.90) * 100 = 27%

The combined fire interval-severity similarity was calculated as the sum divided by 2, giving each component equal weight:
Current to historical fire interval-severity similarity = (21+27)/2 = 24%.

Summarize Fire Regime Condition Class for Each FRPVT

HRV departure from central tendency for any given attribute is calculated by subtracting the percent similarity from 100 (Hann 2003). In the frequent surface fire regime low elevation gentle slope ponderosa pine FRPVT example:

Vegetation-Fuels departure = 100 – 47 = 53%
Fire interval-severity departure = 100 –24 = 76%

The vegetation-fuel condition class was determined by calculating departure (100-sum of similarity) and classifying condition class 1 between 0-33% (considered to be within HRV), 2 from 34-66%, and 3 from 67-100% (table 5). The estimate of treatment in FRPVT 3 was designed to move this component two condition classes, from Condition Class 3 to the upper boundary (33% departure) of condition class 1, while the estimate in FRPVT 1, 2, and 4 were to move one condition class. This method of estimating treatment would respond to a management scenario focused on landscape scale restoration for reduction of risk from both wildland fires during severe fire weather conditions and risks to ecosystem sustainability of HRV departure. The choice of use of the upper boundary of a class, the midpoint of that class, or some other measure for the class depends on the management scenario.

Using the standard class breaks from Hann (2003) the two components (vegetation-fuels departure and fire interval-severity departure) of the fire regime condition class were categorized as follows for the frequent surface fire regime low elevation gentle slope ponderosa pine type:

Vegetation-fuel departure condition class = 2
HRV vegetation-fuel departure class = Moderate
Fire interval-severity condition class = 3
HRV fire interval-severity departure class = High

The intersection of the two departure points, rather than a sum and division, was used to assign the final natural fire regime condition class (Hann 2003).

Fire regime condition class = 3
HRV departure class = High

The break between frequent and infrequent fire regimes is typically an average fire interval of 35 years (figure 1 and table 1). However, this break should

Table 5—Summary of Trout West fire regime potential vegetation types (FRPVT) with associated area, vegetation-fuel condition class (Veg-Fuel CC), vegetation-fuel departure (Veg-Fuel Dep.), and estimates of area to treat and maintain.

FRPVT code	FRPVT description	Area hectares (acres)	Veg-fuel CC	Veg fuel dep.	Area to treat (acres)	Area to maintain (acres)	Total area (Acres)
1	Gentle ponderosa pine	16,662 (41,173)	2	53	3333 (8235)	583 (1,440)	3916 (9675)
2	Low elevation south aspect ponderosa pine	4788 (11,832)	2	39	287 (710)	287 (576)	520 (1286)
3	Low elevation north aspect ponderosa pine -Douglas-fir	2934 (7251)	3	73	1174 (2900*)	1174 (219)	1263 (3119)
4	High elevation ponderosa pine –aspen -Douglas-fir –spruce -lodgepole pine	25,617 (63,301)	2	55	5636 (13,926)	5636 (2947)	6829 (16,873)
Sum		50,002 (123,558)	2	54	10,429 (25,771)	2098 (5182)	12,528 (30,953)

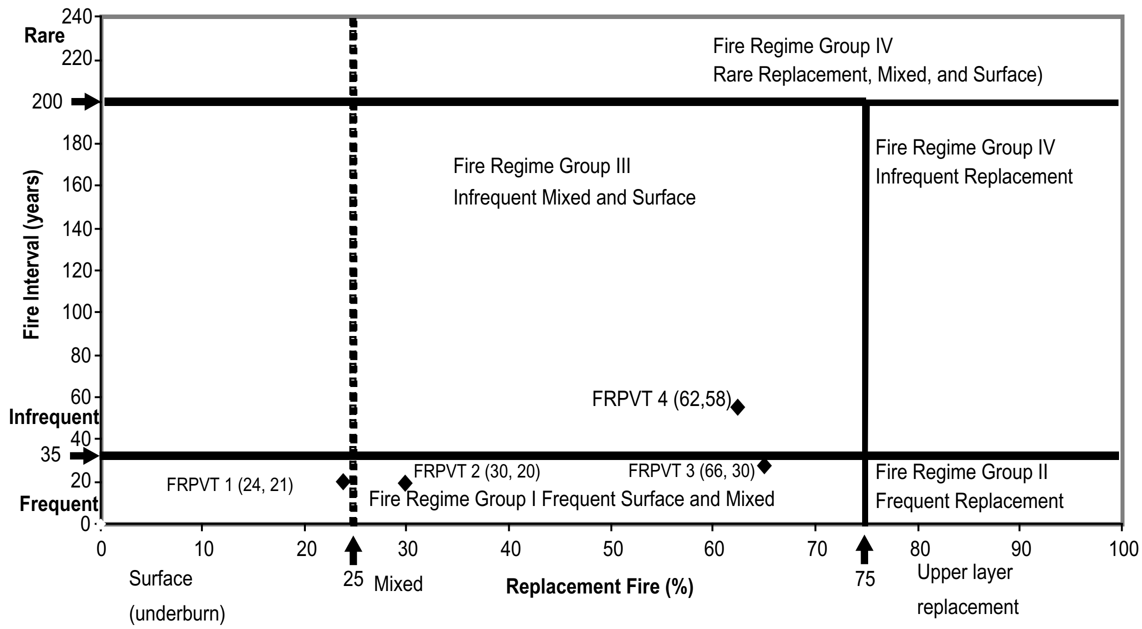


Figure 1. Graphical representation of natural fire regime group boundaries for inputs of fire interval (frequency) and fire severity (% replacement upper layer) showing the x and y axis intersects of FRPVT 1 (21, 24), 2 (20, 30), 3 (30, 66), 4 (58, 62) and the Trout West landscape (T-W-LS as a whole (40, 47).

be increased to 50 years for regimes with high temporal variation or the median should be used rather than the mean. The break between infrequent and rare is typically 100 years, but can range up to 200 years. Classification breaks for fire severity are 25% between surface and mixed and 75% between mixed and replacement. Fire regime group 1 is dominated by surface fire but contains substantial amounts of mixed fire regimes, while group 2 is limited to replacement regimes. Fire regime group 3 is dominated by mixed fire regimes but includes some surface fire regimes, while group 4 is limited to replacement regimes. In fire regime group 5 fires rarely occur but can range from replacement to mixed to surface in effects. In the Trout West analysis FRPVT 1 was the only regime to classify as a frequent, surface fire regime, and this is borderline. Given a plus or minus 33% variation, FRPVT 1, 2, and 3 could have a fire interval between 35 and 50 years, but given they are dominated by fires that are mixed in time they would still be classified in the frequent regime.

We would like to emphasize that the vegetation-fuel departure was based on cover type and structure, and the departure in these attributes only act as a proxy for departure in species composition, fuels, and mosaic patterns, which are important components of the natural vegetation-fuel system. In a similar vein the mean fire interval and simple classification of severity only acts as a proxy for departure in season of fire occurrence, variation in fire occurrence, fire behavior, and severity of fire effects, which are important components of the natural fire interval-severity system. However, in order to keep the analysis simple enough to complete in a short time with readily available data, the use of vegetation cover type-structure and fire interval-severity departure appears to be the best proxy to overall landscape departure from HRV and in fire regime condition class. An even greater reduction in complexity and time can be accomplished by using vegetation departure alone as a proxy for all vegetation-fuel and fire interval-severity components. However, this can be misleading, particularly for ecosystems that contain fire dependent or associated species that require fire effects for germination or to maintain a competitive advantage (Hann 2003).

Summarize Area to Treat and Maintain

It is very useful to have some estimate of the area to treat (recruit, reduce and restore) in order to achieve a fire regime condition class or HRV departure option. This was calculated for each FRPVT. An example is provided in table 5. In the example the option was to improve the FRPVT to the upper range of condition class 1 and lower HRV departure to conditions similar to the HRV regime based on the vegetation-fuel departure. The minimum area treated to achieve this option was calculated as follows:

Minimum area to treat = (Vegetation-fuel Departure- 0.33) * FRPVT area, where .33 is the upper boundary of Condition Class 1.

For the frequent surface fire regime low elevation gentle ponderosa pine FRPVT example, the calculation was: area to treat = (0.53 - 0.33) * 16,662 ha (41,173 acres) = 3333 ha (8235 acres). Given that condition class 1 includes the normal range of natural or historical variability, this level of treatment mimicking natural disturbance effects would move this FRPVT to a composition with borderline similarity to the HRV.

However, this calculation of treatment does not account for the need to maintain acres that are currently contributing to natural vegetation-fuel HRV conditions that could be lost due to natural succession over the project implementation period. For large landscape restoration projects the project implementation period can often range from 5 to 15 years. The amount similar to the HRV averages for classes A, B, C, and D were summed and multiplied times the area of the FRPVT to determine area similar to HRV. The amount from class E was not included in the calculation, because this class will not lose area to another class due to succession over the project implementation period. The amount of area similar to the HRV averages was divided by an average successional period for the implementation period. This successional period was estimated from the successional rates in the "box" model to be about 50 years. The project implementation period was estimated to be about 5 years. As an example (table 5), the amount that needs to be maintained in the frequent surface fire regime low elevation gentle ponderosa pine:

HRV similar area = (A + B + C + D similarity) * FRPVT Area
HRV similar area = (.09 + .04 + .02 + .20) * FRPVT Area
HRV similar area = (0.35 * 16,662 ha (41,173 ac.)) = 5832 ha (14410 acres)
Implementation successional period = 50/5 = 10
Area to maintain = HRV similar area / implementation successional period
Area to maintain = 5832 ha (14410 acres) / 10 = 583 ha. (1440 acres).

A simpler way to approximate amount to maintain would be to map the area that was classified with a "maintain" management implication (same as "similar" abundance class) in combination with moderate and high risk of departure, which can be used as a proxy for ecological sustainability risk. However, this does not account for differences in succession rates.

The calculation of area to treat and maintain uses the vegetation-fuel departure and not the fire interval-severity departure or both. This was because the management option was to improve sustainability of the current vegetation-fuel landscape, such that habitats would be more characteristic of natural conditions, and when wildfire occurs, the risk of uncharacteristic behavior and effects would be much less. The overall goal was not to mimic the frequency and behavior of the natural fire interval-severity system. However, the knowledge that this part of the system is in condition class 3 and high departure can be used to focus wildland fire use and prescribed fire programs on this FRPVT.

Map Wildfire Occurrence Risk

This analysis was designed to quantitatively predict future fire probabilities of wildfire occurrence based on past fire occurrences. The process was divided into three steps. The first step was to classify ignition frequency. Historic ignition point data was incorporated into a grid composed of .25 square kilometer cells. The ignition sources were grouped into two classes: L = Lightning and O = Non-Lightning. The number of ignition points present in each cell were counted and grouped into the following categories: lightning, non-lightning, and all sources. Ignition point classes were developed based upon: 1) the total number of ignition points in the map extent and 2) the range of total ignition points within the cells. The ignition point classes were defined as follows: low (L) = 1 ignition source; Moderate (M) = 2 to 3 ignition sources; and High (H) = 4 or more ignition sources.

The second step was to attribute ignition class to the vegetation polygon coverage. The vegetation polygon layer accounted for all ownership types (National Forest and private). Each polygon was assigned an ignition point class for each ignition point category (refer to the Ignition Class by Frequency process). This process set each ignition point class to a unique numeric value for the polygon by assigning class: H = 1000; M = 100; and L = 1. Then these numeric point values were summed for each polygon. The sum value was then used to classify the polygon ignition point class of: High = >999); Moderate = >100 – 999; and Low = 0 - 100.

The third step was to classify the wildfire occurrence risk by combining the polygon's ignition class with its FRCC departure contribution class. An ignition class and departure contribution matrix (table 6) was used as a guide to query for those combinations and assign the wildfire occurrence risk class to each vegetation polygon.

Table 6—Wildfire occurrence risk class matrix formed from the combination of FRCC departure contribution class and polygon ignition class. Polygon ignition class relative risk levels were calculated using recent wildland fire occurrence data for the Pike and San Isabel National Forests. Numbers of wildland fires were summarized across this larger area, the Trout West area, by watershed, and by departure contribution class to calculate and classify relative risk of wildland fire occurrence during the fire weather season. Final relative classes included very high (VH), high (H), moderate (M), low (L), and non-applicable (NA).

Departure contribution class	Polygon ignition class			
N	L	M	H	
High	L	M	H	VH
Moderate	L	M	M	H
Low	L	L	M	M
None, non-applicable, Water	NA	NA	NA	NA

Fuel Model Mapping

Anderson and National Fire Hazard Danger Rating System (NFDRS) fuel models were assigned using the descriptions from Anderson (1982) and local knowledge of fire behavior. Both fuel model classification systems focus on the fire behavior of the fuel model. Consequently, the assignment process focuses on the expected fire behavior of a vegetation-fuel type rather than the specific fuel loading and distribution characteristics of the vegetation-fuel type. Each unique combination of FRPVT, cover type, canopy closure, size class,

Table 7—The process for assigning Anderson and NFDRS fuels models from Anderson (1982) to vegetation cover type, canopy closure, size class, and mortality class combinations resulted in a large set of tabular values The data from FRPVT 1, the frequent surface fire regime lower elevation gentle ponderosa pine, is shown in the table as an example.

FRPVT	Cover type	Canopy closure	Size class	Mortality class	Anderson fuel model	NFDRS fuel model
1	Grass	Shrub-grass	0	N	1	L
1	Shrub	Shrub-grass	0	N	6	T
1	Ponderosa Pine	Closed	Pole	N	8	H
1	Douglas-fir	Closed	Large	Y	10	G

and mortality class was attributed with the corresponding fuel model class that best fit the expected fire behavior. There were about 25 to 50 unique combinations for each FRPVT, which were too many to display for this paper. An example set of combinations is provided in table 7 to display the assignment process. The data was displayed in both Anderson and NFDRS Fuel Model maps. Both fuel model classifications are very coarse but are useful for evaluating fire-planning scenarios or for use where fuels field data and custom fuel models are not available.

Map the Wildland Urban Interface

The wildland urban interface (WUI) and associated attributes were mapped and linked to the vegetation-fuel polygons. While mapping of WUI was not needed for FRCC analysis, this variable was important for defining and addressing fire management issues. A land use map was used to identify if a polygon was urban or not urban. Photo interpretation was used to identify all non-urban polygons as to being WUI or non-WUI. The wildland urban interface was defined as a polygon having at least one house in 16 hectares (40 acres). This value was selected because of the zoning regulations in Colorado that typically create subdivisions with one or more houses per 16 hectares (40 acres). Each WUI polygon was then attributed to a housing density class with the following definitions: low is 1 house per 16 hectares (40 acres) to 1 house per 2 hectares (5 acres); moderate is more than 1 house per 2 hectares (5 acres) to 1 house per .4 hectare (1 acre); and high is >1 house per .4 hectare (1 acre).

Once the Urban and WUI polygons were identified, a GIS buffer was created to depict areas in relatively close proximity to WUI polygons, and to quantify how many acres of National Forest land exist in the WUI buffer zone. The amount of buffer needed between a crowning wildland fire front and the urban interface varies depending both on fuels and fire weather conditions and the values of local residents. If structure protection alone is the key value then a much narrower buffer is viable if homeowners manage for defensible space. If local residents include values such as risk of smoke and loss of local scenic values then this buffer should be much broader. One of the biggest problems in effective fire management to suppress unwanted wildland fire and protect structures and utilities occurs when suppression forces are pinched into a narrow zone between a flaming wildland fire front and urban areas with one-way roads, non-defensible structures, and utilities. This substantially increases safety hazards to people, property, and firefighters and limits use of air support, equipment, and backfiring. By overlaying this buffer with the FRCC layers, areas close to homes can be displayed that are in high departure from natural conditions, within hazardous fuel models, and at high risk for future wildfire occurrence.

Although there is no standard buffer width for WUI, a two-mile width has commonly been used, as large wildfires can throw spot fires or make runs of this distance. However, the use of a two-mile buffer resulted in nearly the entire planning area being within the buffer, obviously defeating the intent of identifying a priority zone. We then reduced the buffer to a one-mile width, which was more effective in displaying a corridor around WUI areas sufficiently narrow to differentiate this high priority zone from other wildland vegetation polygons.

Results and Discussion

Identify and Map Fire Regime Potential Vegetation Types (FRPVT)

The Trout West planning area was stratified into six Fire Regime Potential Vegetation Types (FRPVT). Each type represents a broad aggregate of land with similar homogeneous fire regimes (both in historical fire interval and fire severity) and vegetation potentials. The six different types plus designation of urban areas and lakes included:

- FRPVT 1 – Low Elevation Gentle Slope Ponderosa Pine
  Natural fire regime group I: frequent surface fires
  Ponderosa pine/herb with aspen in draws
  Flat to undulating topography: less than 15% slope
  Montane / lower elevation: less than 8500 feet
  16,669 hectares (41,173 acres)
- FRPVT 2 South Slope Low Elevation Ponderosa Pine
  Natural fire regime group I: frequent mixed fires
  Ponderosa pine/shrub/herb – small amount of Douglas-fir
  South-facing slopes: >15% slope
  Montane/lower elevation: less than 8700 feet
  4790 hectares (11,832 acres)
- FRPVT 3 North Slope Low Elevation Ponderosa Pine – Douglas-fir
  Natural fire regime group I – frequent mixed fires
  Ponderosa pine – Douglas fir/shrub-herb
  North-facing slopes: >15% slope
  Montane/lower elevation: less than 8300 feet
  2936 hectares (7251 acres)
- FRPVT 4 High Elevation Mixed Conifer - Aspen
  Natural fire regime group III – infrequent mixed fires
  Ponderosa pine-Douglas fir-aspen-lodgepole pine-spruce
  Upper elevation: >8300 feet on north slopes; >8700 feet on south slopes
  Montane/all aspects
  25,628 hectares (63,302 acres)
- FRPVT 5 Riparian Valleys
  Natural fire regime group IV –infrequent replacement fires
  Valleys w/ meadow vegetation-willow-spruce – all elevations
  2124 hectares (5246 acres)
- FRPVT 6 High Elevation Grasslands
  Natural fire regime group II – frequent replacement fires
  High elevation grassy meadows with scattered ponderosa pine

Expansive meadow area specifically in the Woodland Park-Divide area
2252 hectares (5562 acres)

- URBAN
  Those areas of urban influence such as shopping areas, industrial lots, parking lots, irrigated golf courses, etc. The key is that these types do not have sufficient vegetation-fuel to carry a wildfire nor to threaten structures. Housing developments with trees and lawns that do have sufficient vegetation-fuel to carry a wildfire or to threaten structures were attributed as "Urban Interface" and included in the appropriate FRPVT.
  311 hectares (769 acres)
- LAKES 55 hectares (135 acres)

Model Historical Range of Variation (HRV)

Historical range of variation was modeled and summarized for FRPVT 1through 4. FRPVT 5 and 6 were not modeled because there was no expected restoration or maintenance in these types. Compositions of HRV vegetation-fuel classes, fire interval, and amount of replacement and surface fire were summarized (tables 3, 5, and 8). Using this data each of FRPVT 1 through 4 was classified into a fire regime group (figure 1) and cross checked with the field reconnaissance classification. FRPVT 1 on gentle slopes with a replacement of 24% was very close to the boundary between a surface and mixed regime, while FRPVT 2 on the steeper south slopes fell well within the mixed regime. Both had fire intervals that appear to average about 20 years, while FRPVT 3 on the north aspects was at the upper end of both the fire interval class and amount of replacement for the frequent mixed group. FRPVT 4 at the higher elevations was fairly different from the other FRPVT in that it fell well within the infrequent mixed group (58, 62) with a fire interval and replacement levels both at about 60. These average fire intervals appear to be somewhat more frequent than the average fire intervals identified by Kaufmann et al. (2000a, b). This may be because we underestimated the role large herb-shrub patch size with lack of seed source or competition from grasses and shrubs in comparison to the role of fire in slowing succession back to

Table 8—Summary of vegetation-fuel departure (Veg-Fuel Dep.) and fire interval-severity departure from the central tendency measure of the HRV average for FRPVTs in Trout West watersheds.

FRPVT Code	Description	Area (hectares) (acres)	Veg-fuel dep.	Fire interval	Fire interval departure	Replacement fire %	Fire severity departure	Fire interval -severity departure
1	Gentle ponderosa pine	16,662 41,173	53	21	79	24	73	76
2	Low elevation south aspect ponderosa pine	4788 11,832	39	20	80	30	67	74
3	Low elevation north aspect ponderosa pine -Douglas-fir	2934 7,251	73	30	70	66	27	49
4	High elevation ponderosa pine –aspen -Douglas-fir –spruce -lodgepole pine	25,617 63,301	55	58	42	62	31	37
Sum		123,558	54	40	60	47	48	54

dominance for forested vegetation. Or it may be because the Trout West watersheds are in somewhat more gentle terrain with soils that produce more grass and thus might have had a higher amount of fire. Given that condition class 1 includes plus or minus 33% variation around the estimate of central tendency for the HRV, and the difference for departure contribution, abundance, and management implication classes includes plus or minus 25%, the disagreement with Kaufmann et al. (2000a, b) did not have substantial influence on condition class or associated variable ratings.

Assess Current Conditions

Summary of the current conditions indicate only about 67% of FRPVT 1 area is in characteristic vegetation-fuel classes, which was similar to the amount for FRPVT 2 (tables 9 and 10). However, FRPVT 3, the north aspect ponderosa pine – Douglas-fir type had the lowest amount of characteristic types with only 28%, while FRPVT 4 had 45% characteristic types. Most of the uncharacteristic vegetation-fuel conditions in FRPVT 1 and 2 were a result of succession continuing past maximum fire return intervals and generating structures that did not occur in the historical landscape. Uncharacteristic insect and disease mortality was not a substantial factor in FRPVT 1, 2, or 4, but was substantial (17%) in FRPVT 3. Vulnerability of stands to epidemic levels of insect and disease mortality occurred because natural fire exclusion by suppression activities combined with historic timber harvest to reduce ponderosa pine and allowed Douglas-fir to dominate. FRPVT 1 and 2 appear to be too dry to have much Douglas-fir, while in FRPVT 4 much of the vulnerable or dead Douglas-fir has been removed in past harvest or salvage. Much of the area appears to have been affected by uncharacteristic harvest, burning, and livestock grazing activities that occurred during the late 1800s and early 1900s mining era. This may have contributed substantially to reduction in

Table 9—Each FRPVT was summarized for area, vegetation-fuel condition class (Veg-Fuel CC), HRV vegetation-fuel departure class, fire interval-severity condition class (CC), the HRV fire interval-severity departure class, the fire regime condition class and the HRV departure assignments. Condition classes were assigned as 1 for low HRV departure from central tendency, considered to be within the HRV, and 2 and 3 for moderate and high departure, considered to be increasingly outside the HRV.

FRPVT code	Description	Area (hectares) (acres)	Veg-fuel CC	HRV veg-fuel departure class	Fire interval-severity CC	HRV Fire interval-severity departure class	Fire regime condition class	HRV departure class
1	Gentle ponderosa pine	16,662 41,173	2	Moderate	3	High	3	High
2	Low elevation south aspect ponderosa pine	4788 11,832	2	Moderate	3	High	3	High
3	Low elevation north aspect ponderosa pine-Douglas -fir	2934 7,251	3	High	2	Moderate	3	High
4	High elevation ponderosa pine-aspen -Douglas-fir -spruce -lodgepole pine	25,617 63,301	2	Moderate	2	Moderate	2	Moderate
Sum		123,558	2	Moderate	2		2	Moderate

Table 10—Summary of current vegetation-fuel class conditions to compare amount of characteristic to uncharacteristic conditions. Characteristic vegetation-fuel classes were those considered to have composition and structure that occurred during the HRV, while uncharacteristic classes were considered to be those that did not occur during the HRV.

FRPVT code	FRPVT description	Area hectares (acres)	A-E composition %	F-L composition %
1	Gentle ponderosa pine	16,662 (41,173)	67	33
2	Low elevation south aspect ponderosa pine	4788 (11,832)	68	32
3	Low elevation north aspect ponderosa pine -Douglas-fir	2934 (7,251)	28	72
4	High elevation ponderosa pine –aspen -Douglas-fir –spruce -lodgepole pine	25,617 (63,301)	45	55
Sum		50,002 123,558		

ponderosa pine dominance and increase in small tree regeneration density (McNicoll and Hann 2003).

The closed late development class dominated FRPVT 1 (31%), with open late development also played a strong role (20%). This relationship was reversed in FRPVT 2 where the open late development was dominant (45%), with the closed late development only having 19 percent. Succession appears to be much slower on the south aspects that are much dryer and have more coarse and well-drained soils than on gentle terrain. Closed (12%) and open (13%) late development had similar amounts in FRPVT 3 apparently because insect and disease mortality had opened many of the previously closed stands. FRPVT 4 was dominated by the closed late development class (21%) with only half as much open late development (12%). This type receives more moisture and has cooler temperatures causing more rapid canopy closure. Past harvest and salvage was the primary causal agent in creating the open late development classes.

FRPVT 1 was the only type with substantial early development (8%) vegetation. This was primarily grass and some shrub, apparently maintained in this stage by heavy competition from the grass that limits tree seedling regeneration. FRPVT 1 and 4 were the only types having substantial mid development closed conditions (5 and 7% respectively). In the gentle low elevation ponderosa pine type, this appeared to be related to thick "dog hair" stands created from some past hot fire or excessive livestock grazing disturbance that maximized regeneration. In the higher elevation type, these stands were primarily the result of past harvest followed by tree planting. None of the types contained substantial open mid development conditions.

Compare Current Vegetation-fuel Conditions With HRV

FRPVT 3 had the highest departure in vegetation-fuel conditions (figure 2, 73%). In contrast FRPVT 2 had the lowest, with only 39% departure. This is

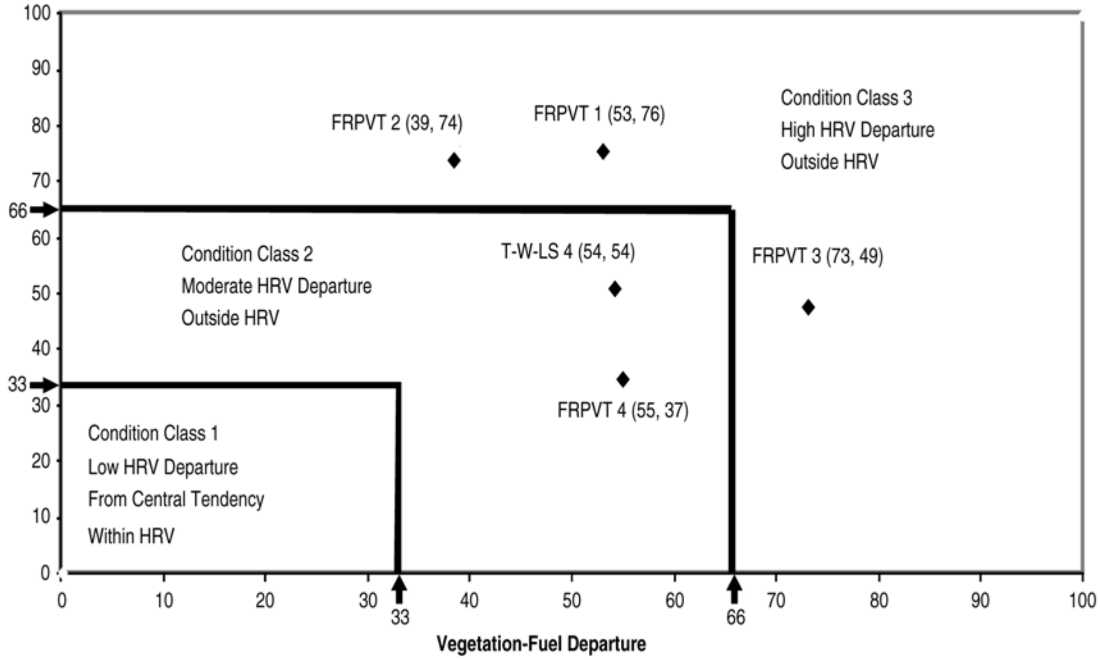


Figure 2. Graphical representation of fire regime condition class boundaries for inputs of vegetation-fuel departure and fire interval-severity departure showing the x and y axis intersects of FRPVT 1 (53, 76), 2 (39, 74), 3 (73, 49), 4 (55, 37) and the Trout West landscape (T-W-LS as a whole (54, 54). Condition class 1 can contain plus or minus 33 % variation around the estimate of central tendency for the natural or historical range of variability. This allows for a 66% range in variation. Condition class 2 and 3 are considered to be outside the natural or historical range of variability in successively higher levels.

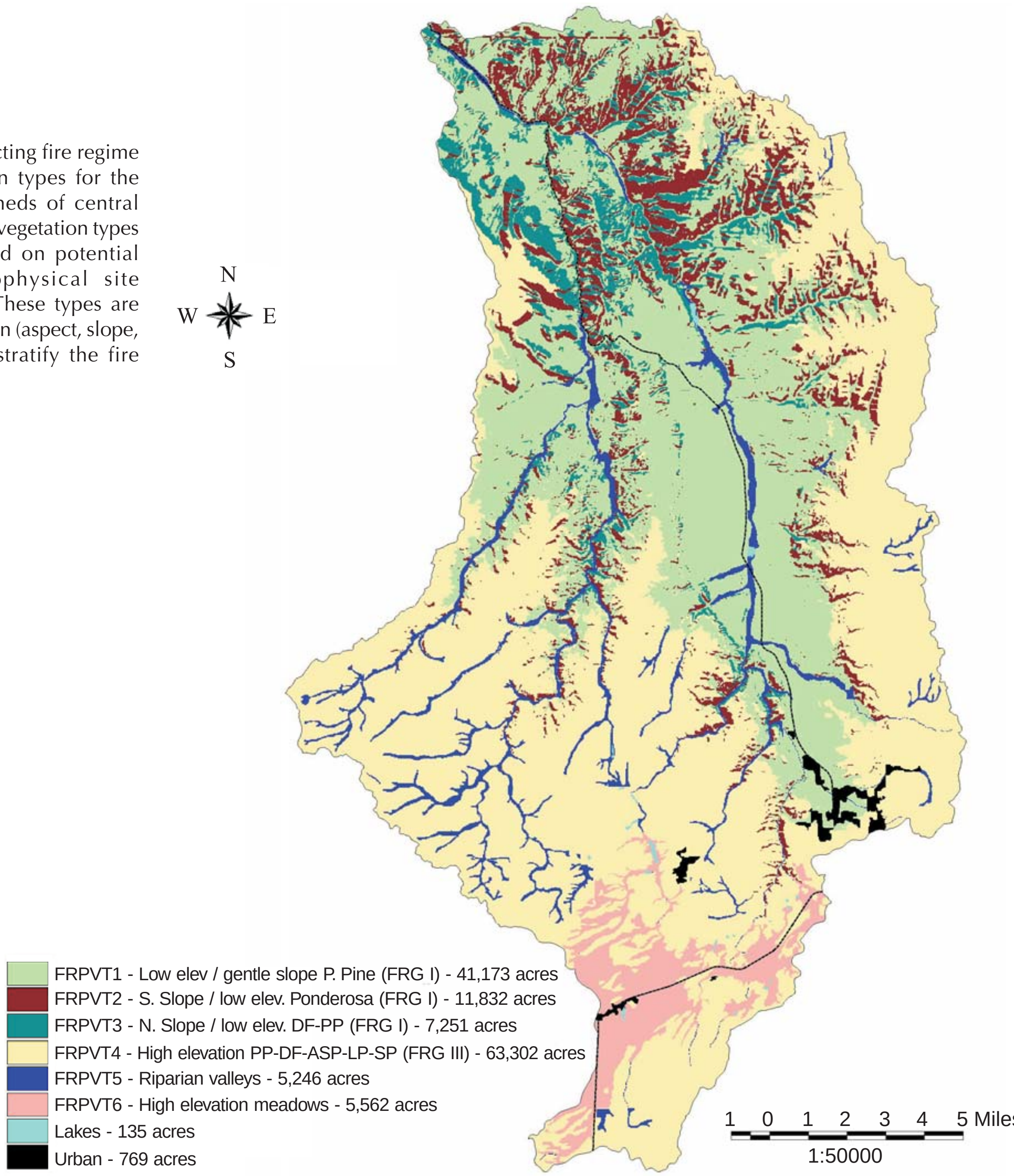
interesting given these two types are on contrasting slopes. In general the south slope has the least change because of dryer conditions that have slowed uncharacteristic succession, with the north aspect slopes showing the greatest departure because of moist conditions that allow fairly rapid uncharacteristic succession combined with a lack of past harvest due to the steep terrain.

FRPVT 1 and 4 had intermediate departures (53% and 55% respectively) that were similar to the landscape average for Trout West (figure 2, T-W-LS 54%). Departure in FRPVT 1 was somewhat less because of fairly slow uncharacteristic succession due to grass competition with tree regeneration and some past prescribed fire and harvest treatments. Although succession may be faster in FRPVT 4, the departure was slower because natural fire intervals are infrequent as compared to the more frequent interval in FRPVT 1 (figure 3). In addition, past harvests, salvage, and prescribed fire reduced departure to some degree.

Compare Current Fire Interval and Severity with HRV

In contrast to vegetation-fuel conditions, FRPVT 1 had the highest departure in fire interval-severity (figure 2, 76%). FRPVT 2 had almost as high a departure level with 74%. This is a logical relationship since both types are in a frequent fire regime and the primary causal agent of departure is fire exclusion. Past excessive livestock grazing in the late 1800s and early 1900s could have also been a related factor by reducing fine fuels, thus increasing the natural fire interval even prior to the active fire suppression efforts initiated in the 1920s and ‘30s. These causal factors may be interacting with increased fire ignitions from mining-related burning that occurred during that time and with decreased fire ignitions from Native American burning. However, given

Figure 3. Map depicting fire regime potential vegetation types for the Trout West watersheds of central Colorado. Potential vegetation types are classified based on potential lifeform and biophysical site indicator species. These types are then split using terrain (aspect, slope, and elevation) to stratify the fire regime.



the amount of lightning in the Trout West area, the lack of fine fuel from excessive livestock grazing would likely be the driving force in increasing the fire interval, as compared to changes in amounts of cultural burning.

FRPVT 3 and 4 had lower levels of fire interval-severity departure (figure 2, 49% and 37% respectively). This follows with these types having less frequent fire intervals. Although FRPVT 3 classified into the frequent fire regime, the average (table 8, 30 years) is very close to the upper boundary for the frequent regime. The range in variability of this type would take it into the infrequent regime for some cycles.

One of the key effects of the departure in fire interval-severity appears to be related to native plant diversity. The understory of mid and late development stands and of the early development stands appears to have very low diversity of native herb and shrub species. Many of these species are fire adapted or fire associated in terms of regeneration mechanisms. Even though the vegetation-fuel class conditions may allow for these species, the lack of regenerative fire effects precludes their development.

Summarize Fire Regime Condition Class for each FRPVT

FRPVT1 had the highest departure (76%) in fire interval-severity, but only moderate departure (53%) in vegetation-fuel class composition (figure 2) thus classifying as a condition class 2 for vegetation-fuel conditions, 3 for fire interval-severity conditions, and overall a fire regime condition class 3 for combined conditions (tables 8 and 9). FRPVT 4 had the lowest departure in both components (55%, 37%) with a moderate departure and overall condition class 2 assignment. FRPVT 3 had the highest departure in vegetation-fuel class composition (73%), only a moderate departure in fire interval-severity (49%), and thus an overall class 3 assignment.

FRPVT 1 and 2 have sufficient departure in fire interval and severity to classify as condition class 3 although the vegetation and fuel departure would classify as condition class 2. This would likely have implications that this regime may lack natural fire effects and have lost composition of fire associated species. FRPVT 3 has sufficient departure in vegetation and fuel to classify as condition class 3 although fire interval and severity departure would classify as condition class 2. This would have implications for high fuel loading and loss of natural cover type and structure diversity. FRPVT 4 classified as condition class 2 for both types of departure and because of its large area extent caused the average departure for Trout West as a whole to classify in condition class 2.

Summarize Area to Treat and Maintain

One scenario for the Trout West watersheds was to treat and maintain enough area to change the condition to a class 1 in a landscape pattern that would reduce risk to the urban interface. Additional secondary options included reducing potential large fire suppression costs and reducing ecosystem risks to air, water, native species habitats, and sustainability. Given that fire interval-severity outcomes are very difficult to measure and evaluate, it appeared that the vegetation-fuel condition class would be the most useful indicator to estimate area to treat and maintain, and to monitor relative to achievement of an option. Focusing on an option resulted in the need to calculate the area to treat and maintain based on the option of changing the Vegetation-fuel condition class from 2 to 1 for FRPVT 1, 2, and 4, and from 3 to 1 for FRPVT 3. This focus on vegetation-fuel condition class does not de-emphasize the need to focus on the fire interval-severity condition class and departure. The fire interval-severity condition class was identified as a focus for identification of type of treatments, particularly prescribed fire as a tool for treatment and maintenance of polygons in FRPVT 1 and 2.

Summary of the area to treat indicated approximately 10,000 hectares (76,000 acres) in order to achieve the condition class option (table 5). In addition about 2100 hectares (5200 acres) would need to be maintained during a typical project implementation period. A little over half would be focused at FRPVT 4, about one third to FRPVT 1, with the rest in FRPVT 2 and 3. Given that 2 and 3 are located on the steeper slopes with less road access, a strategy may be developed to treat these with prescribed fire or wildland fire use following treatment of surrounding areas in FRPVT 1 and 4 with mechanical and prescribed fire, and only emphasizing mechanical or hand treatment where FRPVT 2 and 3 polygons abut urban interface areas. Treatment polygons would be focused at reducing high and moderate departure (figure 4) and maintaining low departure. Given there is much more high and moderate departure (45,593 hectares, 112,663 acres) than needed to achieve the outcome, polygons would be prioritized based on most effective design to reduce risks of wildfire to urban interface and ecosystems combined with operational considerations (such as access, soils, terrain, and visuals). Treatment and maintenance prescriptions can be focused at those needed to reduce certain types, recruit other types, and maintain low departure conditions.

Fuel Model Mapping

Fuel models in themselves do not indicate potential for uncharacteristic wildfire behavior and effects, fire regime condition class, or departure from natural (historical) conditions (Hann and Bunnell 2001). However, the combination of an indicator of departure such as in figure 4 with fuel models (figure 5) has considerable value. Fuel model 2 (open grassy forest) would have been the most common fuel model in the natural (historical) regime. This fuel model can have rapid rates of spread in grassy fuels, but typically does not crown, have potential for blowup fire behavior, have severe fire effects, throw mass firebrands, and spread with long distance spotting fires. This fuel model still exists in scattered polygons (figure 5) but has been replaced in most polygons with fuel models 8, 9, and 10. Fuel model 8 (closed short needle single and multi-layer young forest without heavy ground fuels) in a moist or cold forest setting does not have high potential for ignition, spread, and crown fire. However, this fuel model would be uncharacteristic in a forest setting that is subject to drought conditions. In this kind of setting this fuel model can exhibit extreme crown fire behavior and long distance spotting (1.5-3 km, 1-2 miles), such as occurred during the fire seasons of 1988, 1994, and 2000. Fuel model 9 (closed long needle forest with litter-duff) can display even more extreme fire behavior than fuel model 8 in the dry forest setting. Fuel model 10 (closed forest with heavy ground and ladder fuels) typically displays the most extreme fire behavior and long distance spotting. The current vegetation-fuel conditions in the Trout West watersheds produce fuel model 8-9-10 complexes that are associated with high departure and uncharacteristic vegetation-fuel conditions.

Fuel models have shifted from the historical dominance of fuel models 2, 9, 1, and 8 to the current dominance of fuel models 8, 9, 2, and 10. This has resulted in a fire behavior shift during severe fire weather conditions from what were historically fast moving, but low intensity mixed and surface fires to current fast moving, but high intensity crown replacement fires and mixed fires. One of the biggest additional differences that affect landscape scale fire behavior is the current lack of non-forest fire maintained herbaceous-shrub (grass, forb, shrub) patches that were interspersed between the forested patches where fire would drop to the ground (Kaufmann et al. 2000a, b).

Map Wildfire Occurrence Risk

The wildfire occurrence and uncharacteristic fire risk indicates that the likelihood of current and near future ignitions, rapid rates of spread, and resistance to initial attack and wildfire containment would occur in the northern portion of the Trout West watersheds in the more rugged terrain (figure 6, moderate and high classes). The low class is strong to the southerly area of the watersheds indicating a lower likelihood that wildfires would initially ignite, be difficult to control, and spread from these areas. However, based on the departure map (figure 4), once a wildfire ignited and spread from inside or from adjacent watersheds uncharacteristic behavior (rapid rates of spread, crown fire, potential blowup fire behavior, mass firebrands, and long distance spotting) would be just as severe in the southerly end of the watersheds as in the

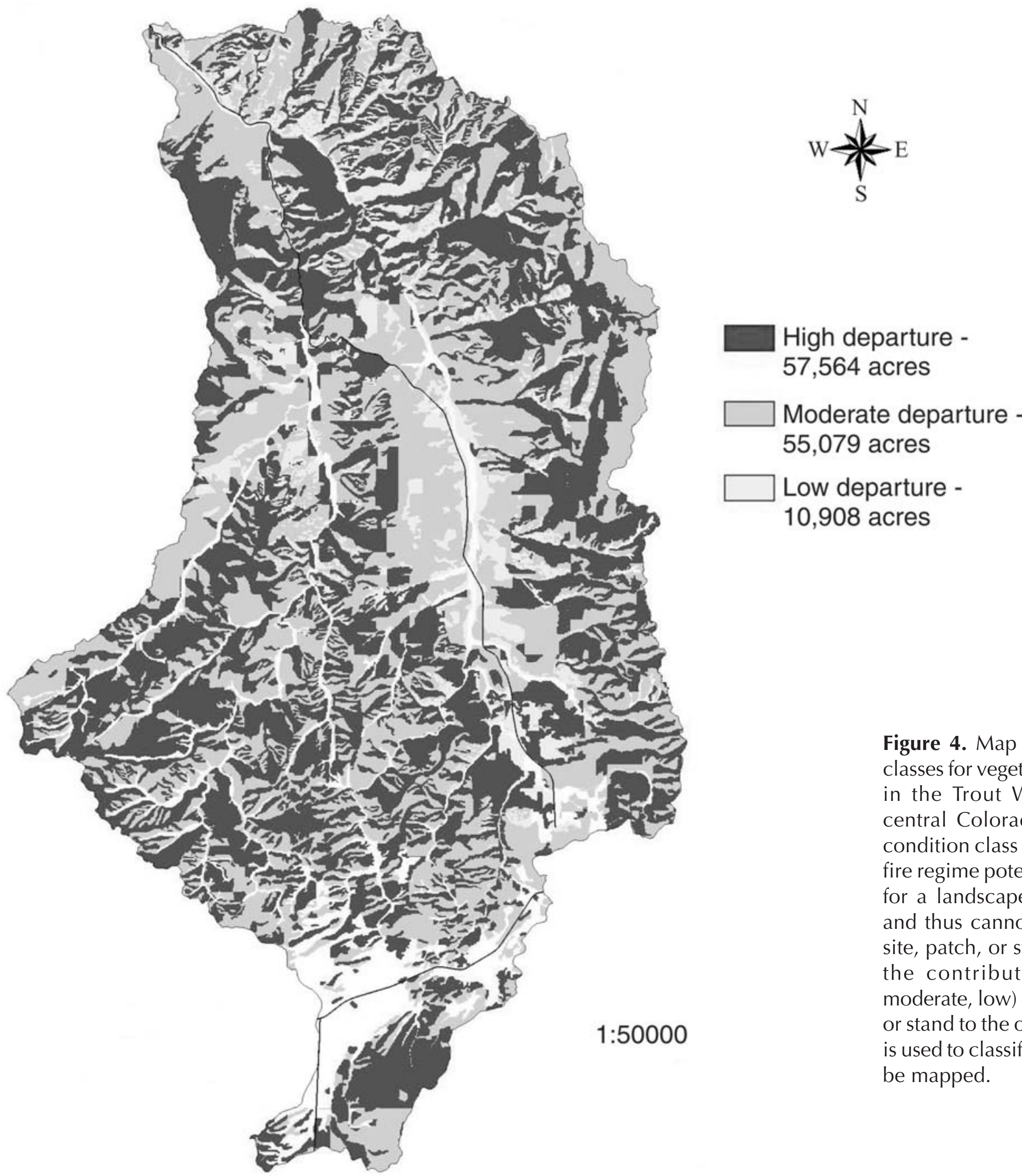
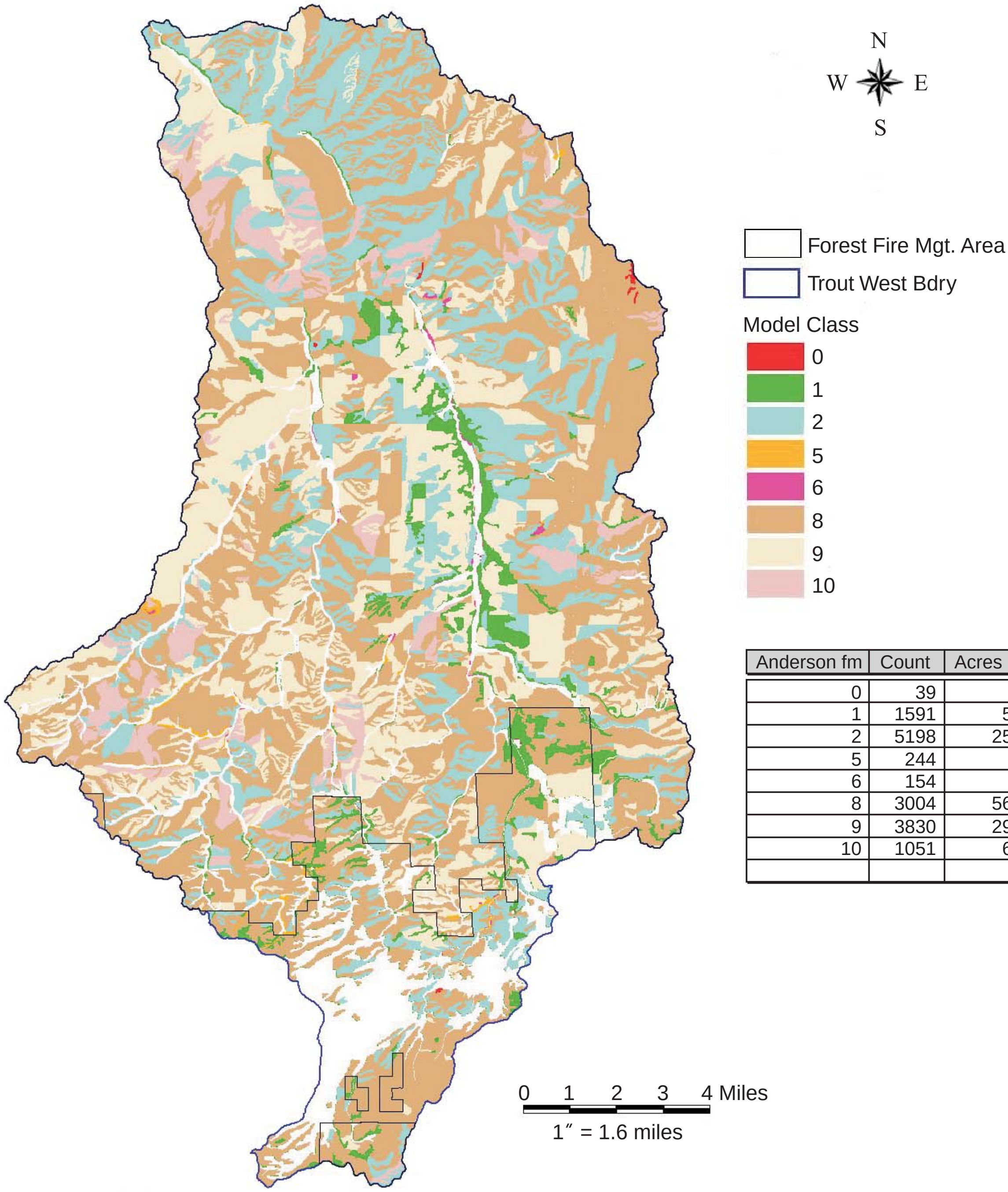


Figure 4. Map depicting departure classes for vegetation-fuel conditions in the Trout West watersheds of central Colorado. The fire regime condition class is determined for the fire regime potential vegetation type for a landscape or watershed area and thus cannot be mapped to the site, patch, or stand level. However the contribution or risk (high, moderate, low) of a given site, patch, or stand to the overall departure that is used to classify condition class can be mapped.

northern area. Amount of wildland fire ignitions or fuel flammability were found to not limit the current wildland fire occurrence. Initial attack to suppress wildland fires was found to be the primary cause of reduced fire occurrence compared to historical fire occurrence.

The high wildfire risk associated with uncharacteristic vegetation-fuel conditions occurs in a dry forest environment that is subject to cumulative multi-year drought and windy conditions with a high probability of ignition and spread from the northerly end of the watersheds or adjacent watersheds. The ignition and initial fire spread could come from the northerly portion of the landscape, from the landscape to the west or from the landscape to the east, driven by westerly or northwest winds, or Rocky Mt. Front easterly winds. The



Anderson fm	Count	Acres Final
0	39	64
1	1591	5338
2	5198	25055
5	244	399
6	154	159
8	3004	56237
9	3830	29591
10	1051	6680

Figure 5. Map depicting fuel models for the Trout West watersheds of central Colorado. Fuel models descriptions come from Anderson (1982) and are used along with weather and patch or stand canopy and structure attributes for modeling fire behavior (Andrews and Chase 1989, Finney 1998).

landscape as a whole, and in the context of adjacent landscapes, presents a high risk of eventually having a large wildfire event that could consume 60 to 80% of the watersheds, similar to the Buffalo Creek fire that occurred to the north in 1994.

Map the Wildland Urban Interface

The map of the wildland urban interface (WUI) indicates most of this area is in the southerly end of the watersheds on the higher elevation benches of

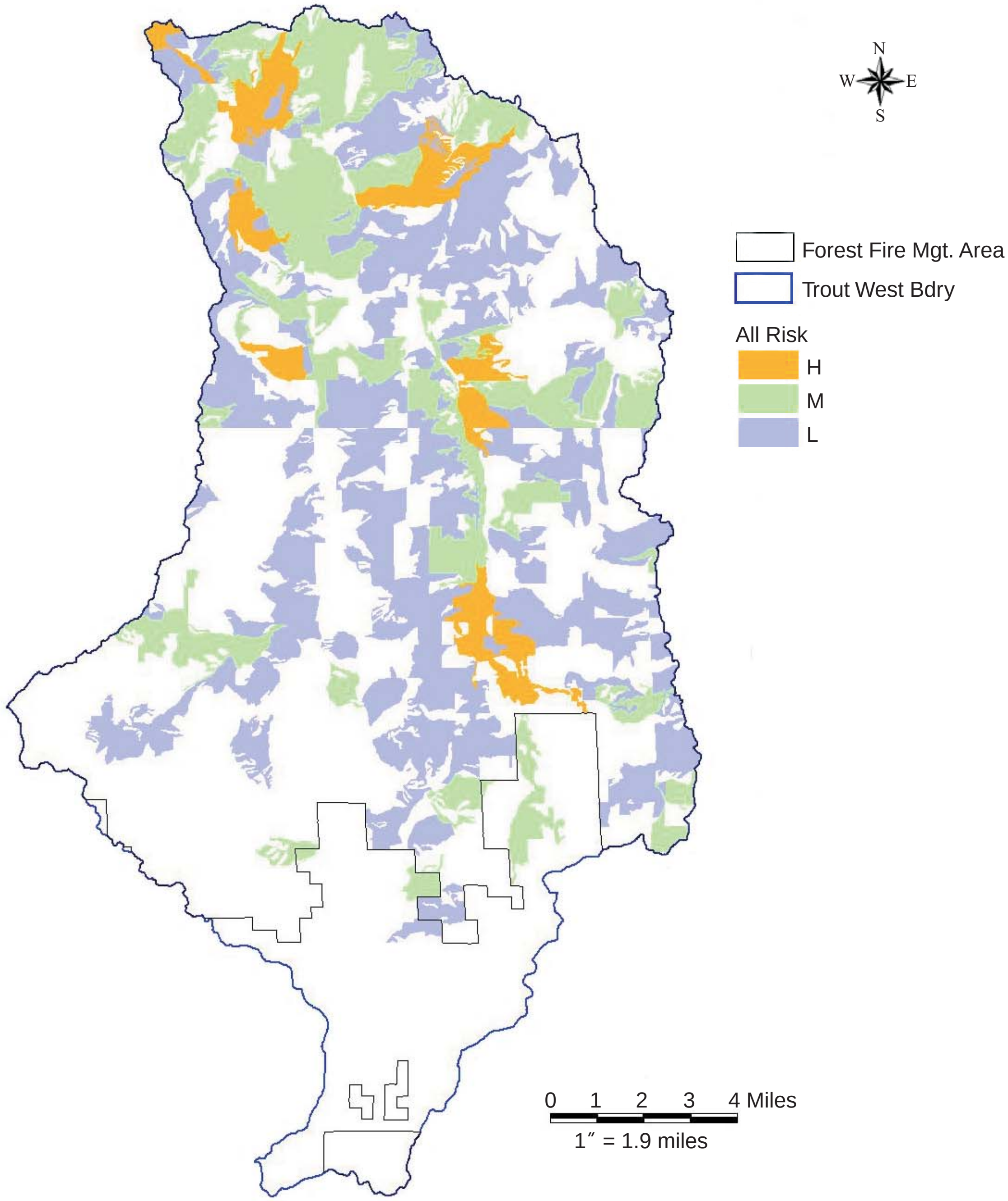


Figure 6. Map depicting wildfire occurrence risk for the Trout West watersheds of central Colorado. Relative risk levels were calculated using recent wildland fire occurrence data for the Pike and San Isabel National Forests. Numbers of wildland fires were summarized across this larger area, the Trout West area, by watershed, and by stand scale departure contribution class to calculate and classify relative risk of wildland fire occurrence during the fire weather season.

the watersheds with some area extending to the north down the primary stream valleys (figure 7). The one-mile buffer indicates about 17,000 hectares (42,000 acres) of public land are adjacent to the WUI. Most of the WUI is associated with the high or moderate departure uncharacteristic vegetation-fuel conditions. Very little is associated with low departure conditions. This is consistent with findings of Hann et al. (1997) in an area as far away as the northwestern U.S. (Interior Columbia Basin).

No Treatment, WUI Focus, and Landscape Focus Scenarios

An initial view of risk to WUI (figure 7) may falsely conclude that if most of the interface and buffer area were treated to reduce fuels and uncharacteristic fire behavior, objectives for change in overall condition class and WUI risk reduction could be achieved. However, large wildfires in contiguous uncharacteristic fuels would not be substantially slowed with this type of treatment (Finney and Cohen this volume). Nor would risk to urban interface be substantially reduced. In the most probable outcomes there is not much difference in risk between no action and a WUI focus option. A large wildfire event would spread as depicted in figure 8; most probably starting in the north end or coming from adjacent west or east landscapes and spreading to the south, initially pushed by winds from the west, north, or east, and then pushed by its own fire pre-heating and drying green and dead fuels, then burning at even higher intensities, and developing its own wind. Until the weather changes with rain or cooler temperatures and a drop in winds, fire behavior would be severe, of potential blowup nature, and spreading through long distance spotting. Fire suppression crews would be unable to attack this fire at the head even if the urban interface buffer areas had been treated for crown fire and fuel risk reduction, because of the mass fire brands raining into the area and fire jumping lines constructed by dozer or hand crews. Mass firebrands would potentially ignite many vulnerable structures causing most of the suppression resources to focus on protecting structures rather than on fire suppression. Although the WUI focused fuel treatments may not substantially change landscape level fire behavior, these treatments would somewhat reduce the severity of post-fire effects. Where fuels had been treated in the WUI buffer zone the severity of fire effects would be reduced to a more characteristic level within the interior of the treated polygons. However, the exterior of the treated polygons would be subjected to extreme heating from the fire in adjacent uncharacteristic fuels.

There is a different design option that can reduce wildfire risk to WUI and have the added benefit of reducing risk to ecosystems at landscape scales. This would be a landscape design. This type of design would involve treatment and maintenance to achieve the condition class 1 landscape option across the Trout West watersheds to change large wildfire behavior and effects. This option would focus on treatment of high departure and maintenance of low departure polygons throughout the watersheds in a pattern most effective at slowing large wildfire spread and reducing risk of negative ecosystem effects (Finney and Cohen this volume; Hann and Bunnell 2001). The first set of treated polygons could focus on mechanical and prescribed fire treatment of operationally accessible high departure polygons and maintenance of low departure polygons inside the line depicted in figure 7. The line generally surrounds both the urban interface and the higher risk areas from a landscape perspective. The second set of treatments would tie in the intermingled less operationally accessible high departure polygons through use of hand cutting and prescribed fire by being able to anchor into the first set of treatments. In addition, prescribed fire with minimal mechanical or hand treatment could be used outside the line and in a relatively small adjacent portion of the landscape to the west, which is primarily roadless and wilderness, to reduce the potential for uncharacteristic fire spreading from or to that area. This would allow wildland fire use or prescribed fire to be effectively used within the core of the adjacent roadless and wilderness area. Similar treatments and maintenance could be used outside the line and in the landscape to the east, which is a mosaic of WUI and non-WUI, similar to Trout West. This would reduce the potential

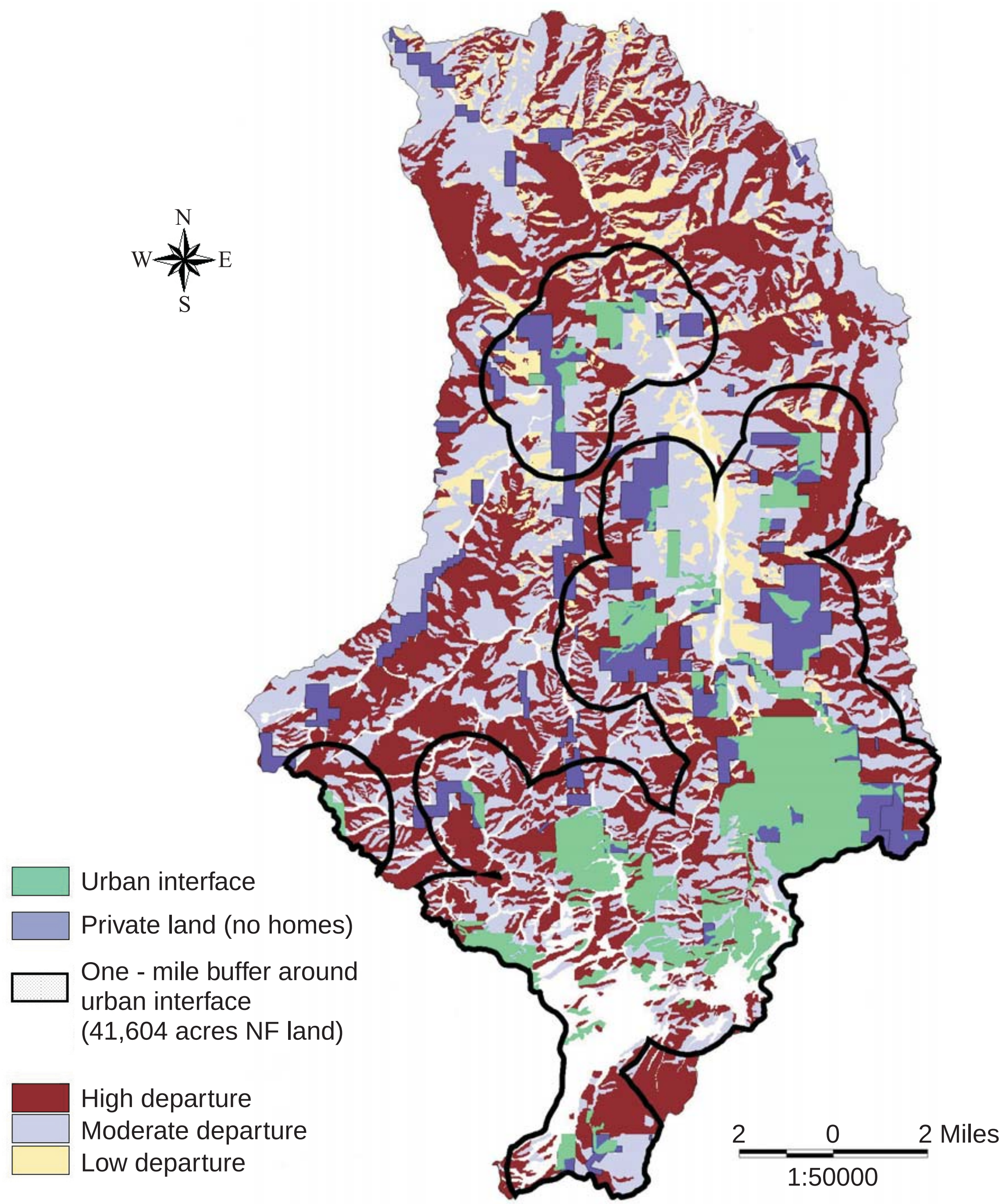


Figure 7. Map depicting urban interface, a one mile buffer around urban interface, and associated risks of uncharacteristic wildfire (high, moderate, low departure) for the Trout West watersheds of central Colorado. The urban interface was considered to be the area with one or more structures per 16 hectares (40 acres).

for uncharacteristic fire spread from that landscape to the Trout West watersheds or vice versa. In addition, the design could take into account ecosystem objectives for reducing risks to air, water, native species habitats, and sustainability; in essence achieving risk reduction for multiple benefits at the same cost.

This landscape type of treatment would substantially change the behavior and effects of a large wildfire run originating from within the Trout West watersheds or from adjacent landscapes. Wildfire from any of these sources

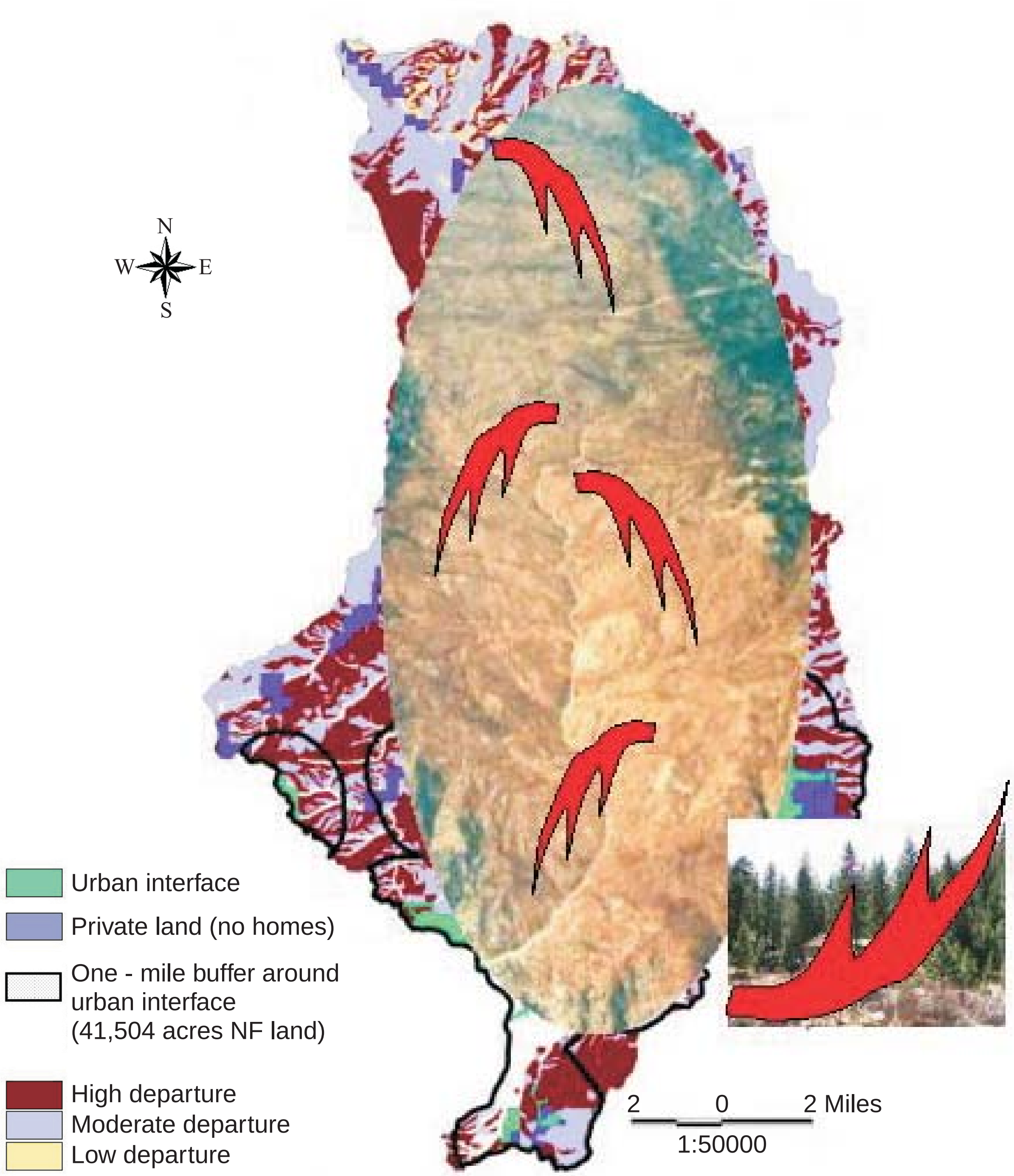


Figure 8. Map depicting the outcome of a Buffalo Creek type wildfire event upon the Trout West watersheds of central Colorado. This type of fire behavior spreads through mass long distance spotting across the urban interface, one mile buffer around urban interface, and into adjacent areas. Treatment of just the urban interface or a narrow buffer does little to improve management options for effective fire suppression and protection of wildland urban interface values.

would still spread fairly rapidly in grass and shrub surface fuels, but would have low risk of torching and spotting and little risk of sustaining a running crown fire. Initial attack would have a much higher chance of containing the fire and if the fire escaped initial attack suppression efforts could contain the fire using hand or dozer lines anchored across strategic areas. There would be little spotting into urban interface structures, thus reducing risk to both vulnerable and non-vulnerable structures. We again emphasize that the vulnerability of structures primarily exists within the narrow zone of the structure and surrounding area that typically is in the ownership of the structure owner (Cohen 2000, Finney and Cohen this volume). However, by substantially reducing firebrands and changing fire behavior from crown to surface, the risk even to vulnerable structures becomes less. This type of wildfire behavior could be managed within the availability of typical suppression resources without having to redirect most of the resources to protection of structures. We have generally found that even in communities with high awareness of wildfire risks and ability of structure owners to reduce these risks with mitigation of structure vulnerability and fuel management, there is at best only about half of the structure owners that will take action. Some redirection of suppression resources would probably be necessary to protect vulnerable structures in areas with torching, but this would be for a small number of areas compared to the WUI wide vulnerability that exists under the no treatment or WUI focus options. Costs of suppression would be much less under this scenario than the no treatment or WUI focus scenario and damage to resources would be minimal.

We think it is important to emphasize that restoration of the WUI area should not be a small area (donut hole) treated to reduce crown fire and fuel risks, within a surrounding landscape (donut) of untreated area. This can result in wildfire behavior in the surrounding donut that presents just as high a risk to the WUI. In a similar sense if the WUI publics also consider visual, air quality, water, and habitat values to be important, this type of donut hole treatment will do little to reduce the risk of loss of these values from wildfire. The key to design of successful restoration and maintenance is to reverse the relationship, such that the WUI area and surrounding landscapes present little risk of sustaining a running crown fire with high severity effects. High risk fuel conditions are generally relegated to operationally inaccessible polygons that are embedded within low risk conditions with a substantial distance to WUI areas.

Cost Comparison of Mapping and Analysis and Treatments

Costs of mapping and analysis were estimated to total about $10,000 (table 11). For the total area of 54,775 hectares (135,352 acres) this cost an average of $.18 per hectare ($.07 per acre). This information for the total landscape can be used to identify the treated acres, assess effects over the whole area for future planning and other resource planning efforts, and edge-matched with adjacent landscapes for broad-scale assessment to support Forest or Resource plans. Even if the total cost is paid for by the project area to be treated to meet the condition class 1 option (12,528 hectares, 30,953 acres), this only averages $ .80 per hectare ($ .32 per acre). This is a low cost compared to the typical costs of treatment in WUI of $200 per hectare ($500 per acre) and $120 per hectare ($300 per acre) in non-WUI, to have information to prioritize and plan what, where, when, and associated scenario outcomes.

We compare the costs of treatment for the three different scenarios of no treatment, WUI focus, and landscape WUI and ecosystem focus including the potential cost of large wildfire suppression (table 12). We developed restoration treatments costs of $988 per hectare ($400 per acre) for the WUI focus and $494 per hectare ($200 per acre) for the landscape focus using similar methods for estimation as from Hann et al. (2001). When assessing the amount of WUI buffer in different risk classes, we estimate that about 6070 hectares (15,000 acres) would be treated in this scenario as compared to 10,117 hectares (25,000 acres) for the landscape scenario. A similar approach was used to

Table 11—Estimated costs for Trout West watersheds fire regime condition class mapping and analysis. Vegetation data (cover type, size class, canopy closure) for Forest Service lands were already available in the resource information system (RIS). Some data correction was necessary and other public and private lands data was obtained through photo interpretation.

Task	Person days	Cost ($)
Coordination and design	10	2,000
Field reconnaissance	10	2,000
HRV modeling	12	2,400
Current maps and GIS analysis	18	3,600
Total	50	10,000

* $200 per person day = $10,000

assess the costs and amount treated for maintenance (table 12). We assumed that wood product values could be produced from about half of the restoration treatments with no product values produced from the maintenance treatments. Based on Hann et al. (2001) we estimated the average product value return to be about $247 per hectare ($100 per acre). Suppression costs were estimated at $500 per acre (Hann et al. 2001) for this area without treatment, with a size of 4048 hectares (10,000 acres) for a typical large fire size in central Colorado. Some benefit (10% reduction) in reduction of average suppression cost was applied for the WUI focus treatment scenario, but the gross size was assumed to be similar to the no treatment scenario. Both reductions in average suppression cost and in wildfire size were applied to the

Table 12—Estimate of costs for planning and implementation of Trout West watersheds restoration and maintenance comparing three different scenarios: 1) no treatment, 2) WUI focus, and 3) landscape WUI and ecosystem focus. The cost estimates include cost estimates for a large wildfire during severe fire weather conditions for each scenario.

Cost item	No treatment $	WUI focus $	WUI area treated hectares (acres)	Landscape WUI and ecosystem focus $	Landscape area treated hectares (acres)
Restoration		6,000,000	6,070 (15,000)	5,000,000	10,117 (25,000)
Maintenance		500,000	1012 (2,500)	500,000	2,023 (5,000)
Product Value		- 750,000	3,035 (7,500)	-1,250,000	5059 (12,500)
Suppression	5,000,000	4,500,000	4,047 (10,000)	500,000	4,047 (10,000)
Property	2,000,000	1,800,000		200,000	
Burn Rehabilitation	175,000	140,000	1,619 (4000)	35,000	405 (1000)
Total	7,175,000	12,190,000		4,985,000	

Restoration – WUI focus $400 per acre average; Landscape focus $200 per acre average.
Maintenance – WUI focus $200 per acre average; Landscape focus at $100 per acre average.
Product values - $100 per acre for 50% of restored acres.
Suppression - $500 per acre; 10% reduction WUI focus; 90% reduction for Landscape focus.
Property - 10 structures $200,000 each; 10% reduction WUI focus; 90% reduction Landscape focus.
Burned area evaluation and rehabilitation – No treatment results in 50% severe damage with rehabilitation costs of $35/acre; WUI focus results in 40% severe damage with rehabilitation costs of $35/acre; Landscape focus results in 10% severe damage with rehabilitation costs of $35/acre.

landscape scenario for a combined 90% decrease. A typical large fire loss of 10 structures was assumed with a value of $200,000 each for the no treatment scenario, with a 10% and 90% reduction in risk for the WUI focus versus landscape focus, respectively. The net sum cost for the three scenarios was approximately 7, 10, and 5 million dollars, respectively. Sensitivity testing of the estimated costs and assumptions on area indicate that even with major changes the no treatment will still be similar to the WUI focus and the landscape focus will consistently be substantially lower than the other two scenarios.

Design of Treatments to Achieve an Option

A problem that has emerged with many types of urban interface and ecosystem risk reduction restoration and maintenance treatments has been the application of measures that do not address the issue (Hann et al. 2001). Because of the history of timber management and silviculture in our forest ecosystems, measures such as crown closure, stand density, size, and basal area are commonly used to design treatments to reduce risk of uncharacteristic wildfire behavior and effects. In a similar vein the history of range management in rangeland ecosystems has resulted in common measures such as canopy cover, basal cover, density, and utilization. Because of this history, many treatments in forest ecosystems, with objectives for reducing risks to communities and ecosystems, continue to be focused on a tree growth, crown closure, basal area, or stand density measure, which may not achieve the objectives. Similarly many treatments in rangeland ecosystems with objectives for reducing risks to communities and ecosystems become focused on shrub or herb canopy cover or density. Measures of canopy biomass distribution, canopy depth, canopy base height, number of tree clumps, and surface fuel and ecosystem characteristics may be much more applicable for assessing and designing treatments to reduce crown fire potential, uncharacteristic fire behavior and effects, and coarse-filter approach to sustaining ecosystems (Finney 1988; Hann et al. 1997 and 1998; Keane et al. 1998; Reinhardt et al. 1997; Scott and Reinhardt 2001).

In addition to selection of applicable measures, projects designed to sustain ecosystems should avoid systematic “rules of thumb” or “one size fits all” prescriptions across all treated polygons (Hann et al 2001). Treatments with objectives that are very prescriptive in specifying numbers of trees by size, snags, down logs, and distance from riparian areas without allowing for natural variation can create a systematic landscape that does not allow for the fine scale variation needed by the diversity of native organisms and processes. Treatments designed to represent the range of historical or natural variability or even a median range of variability must be implemented in a way that allows for that variability. This can be achieved by prescribing variation, which may in itself constrain natural variation. A more useful technique may be to remove the desired amount of woody biomass and then use variation in prescribed fire effects to create the variation in polygon features such as shape, size, numbers of dead standing and down, litter and duff reduction, and species response. The response should be monitored and assessed against the understanding of natural variation. As implementation proceeds, the prescribed fire prescription should be adjusted to shift variation in effects.

Textbook or coarser scale mapping applied to project area site-specific fire regime and condition class can result in the greatest error in outcomes. We consistently find that these coarser scale results are not appropriate for fine-scale project design. More often than not, the lodgepole pine type is a mixed or surface fire regime rather than a replacement fire regime. In a given area, the sagebrush type may be a mixed regime, rather than a replacement fire regime. The coarse scale infrequent fire regime may be a frequent interval regime at the finer scale.

These potential errors also apply to restoration and maintenance of the Trout West watersheds. To avoid error in selection of measures we developed methods focused at fire regime condition class and fuel model, combined with cover type, canopy closure, and size class. To avoid prescriptive numbers without variation we used broad classes of canopy closure and size and focused on the fire regime, condition class, potential fire behavior and effects, and urban interface relationships to wildlands. To avoid the textbook or coarser scale fire regime condition class mapping implication that "all ponderosa pine types are frequent surface fire regimes," we developed and applied methods to develop site-specific fire regime condition class.

Management Implications and Recommendations

Methods

In retrospect, it would have been advantageous to have used the standard RIS density classes (habitat structural stage) of 10-40%; 40-70%; and 70%+ canopy closure. This would have fit in better with existing vegetation data. It is critical that the canopy closure density classes used as a basis for modeling HRV be the same as those used to describe current vegetation conditions. In this case, existing RIS density data was lacking in over half of the NF polygons, and an additional 25,000 acres of private land had no vegetation data. Since we had to do such a major renovation of the RIS tabular data to utilize it as a depiction of current conditions, we opted to develop our own set of density classes. This resulted in a tedious and complicated process. It is important to note that the breakpoint for canopy closure for open versus closed in the "box" model HRV structural stages is relatively flexible for two reasons: 1) estimates of canopy closure from historical photographs and stand reconstruction have high variability; and 2) ecosystems vary in what is considered naturally open versus closed (Hann 2003). Consequently, the canopy closure classification should be one standardized for the current vegetation, and cross-referenced to the open and closed categories for the "box" model structural stages.

We again emphasize that the FRCC map depicts the departure contribution across the entire FRPVT and does not apply to any one individual stand. The natural HRV landscape includes amounts in each of Classes A-E. In the Trout West area, FRPVT 1 has a central tendency for about 12% in Class E (closed mature/mid-mature forest). Currently, Class E comprises 31% of the area, nearly three times as much. It is only possible to show this entire existing Class E component as contributing moderately towards the FRPVT departure class (this is categorized as "moderate" departure because the difference between current and HRV is >25% and <75%). It is not possible to ascertain that any particular Class E stand is in moderate departure, because it would have been expected to occur with a range around the central tendency of 12% of the landscape naturally. This gives the manager the option of deciding how much of the existing 33% in Class E should and should not be treated based on operational accessibility.

A "priority treatment" map may be a useful venue to display those areas contributing significantly to overall departure that are likely most in need of fuels reduction treatments. This would only depict those areas with Moderate or High departure contributions *and* associated "Reduce" management implications or "High" abundance. It may be helpful to distinguish between those areas contributing to substantial departures because of underrepresentation across the landscape, and those that are overrepresented. Our departure or "risk" map (figure 4) depicts the entire area by low, moderate and high departure or "risk" contributions. While an important analysis product, this may be difficult to translate into on-the-ground implications. This is because the moderate classes are categorized as such because they are either overrepresented or underrepresented across the landscape, and thus may need to be either reduced or recruited. Only the overrepresented moderate class that may need active management would be depicted on the priority treatment map. The low departure contributions would not show up in the "Priority Treatment" map, as these are classified as "maintain" or "similar." However, we would also caution that this could result in managers not expending enough effort in developing restoration options for "recruitment" that would grow large trees, produce large snags and logs, regenerate to a different species composition, or maintain what currently has low ecological sustainability risk and is similar to the HRV.

While stratifying WUI polygons into low, moderate, and high housing or population densities may be helpful as a means of further prioritization of the Urban Interface zone, we found it to be less critical than the initial attribution as urban interface. The most accurate method of determining housing densities would be a housing map with precise point locations. A density function could then be applied to quantify home densities to meet varying definitions. This data, however, was not available for our analysis area. One county had no GIS housing data at all. The other county could display private parcels and identify how many homes were on each parcel, but could not depict the homes spatially. GIS maps of planned housing developments would also have been helpful, but did not exist. As an alternative, housing density for all RIS polygons meeting the minimum of one house in 16 hectares (40 acres) was attributed through aerial photo interpretation. The drawback to tying this attribute to a polygon is that the size of the polygon determines the minimum threshold. For example, an 80 hectare (200 acre) lodgepole pine polygon may have 4 houses, but it does not meet the WUI classification, as it represents only one house in 50 acres. Where the vegetation was more dissected, the polygons would be smaller and the houses would likely meet this minimum threshold. Acknowledging this limitation, these WUI data were infinitely more detailed and useable than the previously available data source that depicted very broad housing density zones. For our purposes, it worked very well. Because of the unique patterns of land use and housing development that occur for different areas, the housing density classification and wildland urban interface buffer distance may need to be locally defined. We recommend further research and assessment in other areas with different patterns of land use and housing development before standardization of methods.

There was little doubt that use of the "box" model with standardized definitions of HRV stages, succession, and disturbances greatly reduced the time and costs of analysis and resulted in much greater consistency between models for different FRPVT(s). Although we have no way of determining accuracy without an independent comparison, there was general consensus among the interdisciplinary team that use of this type of standardized model limits the variation to that of the ecosystem rather than to model framework, and thus reduces potential for errors. Allowing development of models with unconstrained successional paths and disturbances would have resulted in substantial variation between and within FRPVT(s). This would be a result of "splitters versus lumpers" as well as lack of understanding to attribute detailed succession and disturbance probabilities. The five conditions (A-E) and limited succession and disturbance pathways were scaled at about the same level of the understanding we could achieve from reading the local literature and conducting ground reconnaissance.

Findings

We summarize five implications from the results of this work:

1) Standardized methods for fire regime condition class that has a context to the national definitions can be cost effectively and consistently applied at project and landscape scales across all land ownerships.

2) These methods differ substantially from those applied at the coarse scale by Hardy et al. (2001) and Schmidt et al. (2002) because the scale of landscape composition and structure, and associated management implications, are much finer.

3) Fire regime condition class can and should be developed from the same basic vegetation data that are used for other resource management analyses and implications. This results in more consistent and logical outcomes in analyses and project design.

4) The analysis of no treatment, WUI focus, and landscape focus scenarios indicates that the typical approach to focusing on WUI and buffer areas may not be a viable option to reducing risk to communities. In contrast, a landscape focus reduces risk to communities and ecosystems with a more effective expenditure of funds.

5) Potential errors in design and implementation of treatments to achieve objectives for reduction of wildfire risk to communities and ecosystems can occur. These are typically associated with: a) choosing traditional forest or range management measures versus those focused on fuels, fire behavior and effects, and ecosystem characteristics; b) using fixed or "one-size-fits all" treatment prescriptions at a polygon level, rather than designing for variation in polygon outcomes across the landscape; and c) application of textbook or coarser scale fire regime condition class findings for fine-scale project design.

References

Anderson, Hal E. 1982. Aids to determining fuel models for estimating fire behavior. Gen. Tech. Rep. INT-122. U.S. Department of Agriculture, Forest Service. Intermountain Forest and Range Experiment Station, Ogden, UT. 22 p.

Andrews, Patricia L.; Chase, Carolyn H. 1989. BEHAVE: fire behavior prediction and fuel modeling system—BURN sub-system, Part 2. USDA For. Serv. Gen. Tech. Rep. INT-260. 93 p.

Arno, Stephen F. and Kathy M. Sneck. 1977. A method for determining fire history in coniferous forest of the mountain West. . Gen. Tech. Rep. INT-42. U.S. Department of Agriculture, Forest Service. Intermountain Forest and Range Experiment Station, Ogden, UT. 28 p.

Bailey, Robert G. 1995. Description of the ecoregions of the United States. 2d ed. revised and expanded. Misc. Pub. 1391. Washington, DC: U.S. Department of Agriculture, Forest Service. 108 p. + United States [Ecoregions] map; 1:7,500,000.

Barrett, Stephen W. and Stephen F. Arno. 1988. Increment-borer methods for determining fire history in coniferous forests. Gen. Tech. Rep. INT-244. U.S.

Department of Agriculture, Forest Service. Intermountain Forest and Range Experiment Station, Ogden, UT. 15 p.

Barrett, Stephen W. and Stephen F. Arno. 1982. Indian fires as an ecological influence in the Northern Rockies. J. Forestry. 647-651.

Beukema, S.J., Kurz, W.A., 2000. Vegetation Dynamics Development Tool and User's Guide, version 4.0. ESSA Technologies Ltd., #300-1765 West 8th Avenue, Vancouver, BC, Canada. www.essa.com. 117 p.

Brown, Peter M., Merrill R. Kaufmann, and Wayne D. Shepperd. 1999. Long-term, landscape patterns of past fire events in a montane ponderosa pine forest of central Colorado. Landscape Ecology 14: 513-532.

Clements, Frederick E. 1934. "The relict method in dynamic ecology." The Journal of Ecology. 22:39-68.Bray, R.T. and J.T. Curtis. 1957. An ordination of the upland forest communities of southern Wisconsin. Ecological Monographs. 27: 325-349.

Cohen, Jack. D. 2000. Preventing disaster: home ignitability in the wildland-urban interface. Journal of Forestry 98 (3): 15-21.

Finney, Mark A. 1998. FARSITE: Fire Area Simulator —Model development and evaluation. USDA Forest Service General Technical Report RMRS-GTR-4. 47 p.

Finney, Mark A. and Jack Cohen. 2003. Expectation and evaluation of fuel management objectives. This volume.

Gruell, George E. 1983. Fire and vegetative trends in the Northern Rockies: interpretations from 1871-1982 photographs. Gen. Tech. Rep. INT-158. U.S. Department of Agriculture, Forest Service. Intermountain Forest and Range Experiment Station, Ogden, UT. 117 p.

Hann, Wendel J. 2003. Mapping fire regime condition class: use of different methods to support different scales of prioritization, planning, and implementation. Presented at the October 14-18, 2001 Tall Timbers 22nd Fire Ecology Conference on "Fire in Temperate, Boreal and Montane Ecosystems", Alberta, Canada. Proceedings published by Tall Timbers Research Station, Tallahassee, Florida. Approx. 20 pages, In Press.

Hann, Wendel J.and David L. Bunnell. 2001. Fire and land management planning and implementation across multiple scales. Int. J. Wildland Fire. 10(3&4): 389–403.

Hann, W., R.E. Keane, C. McNicoll, and J. Menakis. 1994. Assessment techniques for evaluating ecosystem, landscape, and community conditions. In: Volume II Ecosystem management: principles and applications. Gen. Tech. Rep. PNW-GTR-318. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. P 237-253.

Hann W.J.; Jones, J.L.; Keane, R.E.; Hessburg, P.F.; and Gravenmier, R.A. 1998. Landscape dynamics. J Forestry 96(10)10-15.

Hann, W.J.; Hemstrom, M.A.; Haynes, R.W.; Clifford, J.L.; and Gravenmier, R.A. 2001. Costs and effectiveness of multi-scale integrated management. Forest Ecology and Management 153: in press. 19 p.

Hann WJ, Jones JL, Karl MG "Sherm", Hessburg PF, Keane RE, Long DG, Menakis JP, McNicoll, CH, Leonard SG, Gravenmier RA, Smith BG, 1997a. Landscape dynamics of the basin. In: Quigley TM, Arbelbide SJ, technical editors. An assessment of ecosystem components in the interior Columbia basin and portions of the Klamath and Great Basins: volume 2. Gen. Tech. Rep. PNW-GTR-405. Portland (OR): U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. p 337-1055.

Hardy, Colin C., Kirsten M. Schmidt, James P. Menakis, R. Neil Samson. 2001. Spatial data for national fire planning and fuel management. Int. J. Wildland Fire. 10(3&4): 353-372.

Hessburg, Paul F., Bradley G. Smith, and R. Brian Salter. 1999. Detecting change in forest spatial patterns from reference conditions. Ecological Applications 9(4): 199-219.

Kaufmann, Merrill R., Laurie Huckaby, and Paul Gleason. 2000a. Ponderosa pine in the Colorado Front Range: long historical fire and tree recruitment intervals and a case for landscape heterogeneity. In: Proceedings, Joint Fire Science Conference and Workshop, Boise, ID June 1999, Vol. 1, pp. 153-160.

Kaufmann, Merrill R., Claudia M. Regan, and Peter M. Brown. 2000b. Heterogeneity in ponderosa pine/Douglas-fir forests: age and size structure in unlogged and logged landscapes of central Colorado. Can. J. For. Res. 30: 698-711.

Kaufmann, Merrill R., Paula J. Fornwalt, Laurie S. Huckaby, and Jason M. Stoker. 2001. Chessman Lake—a historical ponderosa pine landscape guiding restoration in the South Platte Watershed of the Colorado Front Range. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 20 page draft.

Keane, Robert E., Donald G. Long, James P. Menakis, WendelJ. Hann, and Colin D. Bevins. 1996. Simulating course-scale vegetation dynamics using the Columbia River Basin Succession Model—CRBSUM. Gen. Tech. Rep. INT-GTR-340. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 50 p.

Keane, Robert E., Donald G. Long, Douglas Basford, and Barbara A. Levesque. 1997. Simulating vegetation dynamics across multiple scales to assess alternative management strategies. In eleventh Annual Symposium on Geographic Information Systems. “Integrating Spatial Information Technologies for Tomorro”, Vancouver, BC. Feb 17020, 1997 Conference Proceedinmgs. P 310-315

Keane, Robert E., Russell A. Parsons, and Paul F. Hessburg. 2002. Estimating historical range and variation of landscape patch dynamics: limitations of the simulation approach. Ecological Modeling 151: 29-49.

Keane, Robert E., Garner, Janice L., Schmidt, Kirsten M., Long, Donald G., Menakis, James P., and Finney, Mark A. 1998. Development of input data layers for the FARSITE fire growth model for the Selway-Bitterroot Wilderness Complex, USA. Gen. Tech. Rep. RMRS-GTR-3. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 66 p.

Keane, Robert E.; Mincemoyer, Scott A.; Schmidt, Kirsten M.; Long, Donald G.; Garner, Janice L. 2000. Mapping vegetation and fuels for fire management on the Gila National Forest Complex, New Mexico, [CD-ROM]. Gen. Tech. Rep. RMRS-GTR-46-CD. Ogden, UT: U.S. Department of Agriculture, Forest Ser-vice, Rocky Mountain Research Station, 126 p.

Landres P.B.; Morgan P.; and Swanson F.J. 1999. Overview of the use of natural variability concepts in managing ecological systems. Ecological Applications 9(4):1179-1188.

McNicoll, Cecilia H. and Wendel J. Hann. 2003. Multi-scale Planning and Implementation to Restore Fire Adapted Ecosystems, and Reduce Risk to the Urban/Wildland Interface in the Box Creek Watershed. Presented at the October 14-18, 2001 Tall Timbers 22nd Fire Ecology Conference on “Fire in Temperate, Boreal and Montane Ecosystems”, Alberta, Canada. Proceedings published by Tall Timbers Research Station, Tallahassee, Florida. Approx. 15 pages, In Press.

Menakis, J.P., J. Cohen and L. Bradshaw. 2000. Mapping wildland fire risk to flammable structures for the conterminous United States. In: L.A. Brennan, et al. (eds.) National Congress on Fire Ecology, Prevention, and Management Proceedings, No. 1. Tall Timbers Research Station, Tallahassee, FL. P xxx-xxx.

Morgan, Penelope; Aplet, Gregory H.; Haufler, Jonathan B. [and others]. 1994. Historical range of variability: a useful tool for evaluating ecosystem change. In: Sampson, Neil; Adams, David L., eds. Assessing forest ecosystem health in the inland west. New York: Haworth Press, Inc.: 87-111.

Morrison, Michael. 1998. Red-Zone Analysis for the Pike and San Isabel National Forest.

Mueller-Dombois, Dieter and Hienz Ellenberg. 1974. Aims and Methods of Vegetation Ecology. John Wiley & Sons. NY. 547 p.

Reinhardt, Elizabeth, Robert E. Keane, James K. Brown. 1997. FOFEM user's guide. Gen Tech. Rep. INT-344. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 65 p.

Schmidt, Kirsten M., James P. Menakis, Colin C. Hardy, Wendel J. Hann, and David L. Bunnell. 2002. Development of coarse-scale spatial data for wildland fire and fuel management. General Technical Report, RMRS-GTR-87, U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO. 39 p. plus CD.

Scott, Joe H. and Elizabeth D. Reinhardt. 2001. Assessing crown fire potential by linking models of surface and crown fire behavior. Res. Pap. RMRS-RP-29. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 59 p.

Swetnam, T.W. and J.H. Dieterich. 1985. Fire history of ponderosa pine forests in the Gila Wilderness, New Mexico. In: Proceedings—Symposium and Workshop on Wilderness Fire, 15-18 Nov. 1984, Missoula. Technical coordinators: J.E. Lotan, B.M. Kilgore, W.M. Fisher, and R.W. Mutch. Gen. Tech. Rep. INT-182. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Research Station, pp. 390-397.

USDA Forest Service. 2000.Protecting People and Sustaining Resources in Fire-Adapted Ecosystems, A Cohesive Strategy. USDA Forest Service Washington Office Fire and Aviation Management, Washington DC. 85 p.

Closing Comments: Fire, Fuel Treatments, and Ecological Restoration—Proper Place, Appropriate Time

G. Thomas Zimmerman[1]

Introduction

The title of this conference, "Fire, Fuel Treatments, and Ecological Restoration: Proper Place, Appropriate Time," is indicative of a wide range of elements critically important to ecosystem management. While in general, it encompasses many attributes of a comprehensive fire management program, it is also specific to the recently emerging focus on fuel treatment and ecological restoration. This conference brings together an assembly of individuals to discuss the range of information currently available pertaining to the issues of treating hazardous fuel buildups and restoring healthy, functioning ecosystems. The enormity of these issues, the situation surrounding current and future activities, and the developing program to answer questions about these subjects cannot be overstated. These are indeed huge issues that have been developing for nearly a century and will take massive efforts, committed workforces, and large budgets to reverse the current state of both altered fuel complexes and ecosystem health and condition. So, this conference is not only timely, but significant in terms of the importance and quantity of information that has been shared and discussed here.

I feel very fortunate to be able to present the closing comments at this conference. First, I am privileged to be involved with such an important effort. Secondly, when presenting comments, one has the opportunity to discuss personal viewpoints and is not tied to interpreting actual data. In most cases, this tends to make preparation much easier. What I will discuss here and close this conference with are my personal perspectives and a summary of the three days of this conference.

The Nature of the Fuel Treatment and Ecological Restoration Problem

Are we facing a problem? I would submit that we are not facing just a problem, but an enormous one. All practitioners involved in wildland fire management, and even casual observers, can attest to the magnitude and harshness of the 2000 fire season. What has caused this problem? Nearly a century of fire exclusion combined with other land uses has served to greatly increase fuel accumulations. This, in turn, has promoted changes in fire behavior, fire severity, and frequency. These changes in fuel and fire dynamics have led to a general decline in ecosystem health, which in combination with an expanding

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wildland-urban interface, is seriously affecting our ability to protect both ecosystems and communities. After years of stating the compounding effects of uncontrolled fuel accretion, attention is now focused on this problem. Our credibility has come under scrutiny and we must demonstrate the ability to overturn the current direction that fuel and fire dynamics are following.

Have our fuel treatment and ecosystem restoration activities developed over the years? I would submit that yes they have developed and will continue to do so. Indicative of these developments are the accomplishments of this conference. While here, you have heard 35 presentations in the topics of:

- Fuel treatment performance – fire hazard reduction,
- Restoration case studies and ecosystem effects,
- Treatment economics and social issues,
- Fire regime considerations,
- Landscape planning, and
- Field trips and discussion.

Over 300 people registered for this conference, and at a related conference, the National Fire Plan Conference in Madison, WI, another 300 people were in attendance to discuss the current state and future of fuel treatment and ecological restoration. The response to these two conferences clearly demonstrates the importance of this subject, the changing nature of current knowledge, and the increasing demands for additional information.

Programmatic Development–Putting the Pieces Together

How have fuel treatment and ecosystem restoration activities developed? If we look at the past, present, and future attributes associated with our fuel treatment and ecological restoration programs, such as historical experience, new initiatives, application focus, changing needs, accountability and scrutiny, changing application focus, scheduling, scale, information acquisition, collaboration, and performance measures, we can begin to see what has shaped this program, what current influences are, and what the guiding principles of the future will be. If we accept these attributes as the pieces that will make up the fuel treatment and ecological restoration programs of the future, we must fully understand what they are and how they fit into the puzzle.

Historical Experience

Earlier in this conference, Phil Omi stated that fuel treatment began in the 1930s. The actual extent of the program at this time was limited to specific geographic areas and accomplished only localized treatments. This activity, however limited, provided a foundation for the future. We have built on this experience over the years, but our efforts have almost exclusively focused on the application of prescribed fire. Limited collective experience has been gained with mechanical treatments, but this has resulted from support of silvicultural practices rather than from treatment of fuels or restoration of ecosystems. Wildland fire use, formerly categorized as prescribed natural fire, has been a fire management strategy used by some federal agencies since 1970. This practice has served as a principal technique for ecosystem maintenance and restoration in undeveloped areas such as large national parks and wildernesses and represents an extremely valuable strategy for ecosystem maintenance and restoration.

New Initiatives

Both Lyle Laverty and Governor James Geringer have talked about the various initiatives and directives that are now driving fuel treatment activities. These initiatives are propelling this program to greater levels than ever before. These initiatives consist of numerous reports, reviews, directives, and other agency documents that describe the enormity of the problem and make recommendations for resolution. The most significant initiatives include the 1995 Federal Fire Policy and 2001 Review and Update; multiple General Accounting Office audits; the National Fire Plan; the combined federal Cohesive Strategy (Restoring Fire-Adapted Ecosystems on Federal Lands – A Cohesive Strategy for Protecting People and Sustaining Natural Resources); and the 10-Year Comprehensive Implementation Plan (A Collaborative Approach for Reducing Wildland Fire Risks to Communities and the Environment, 10-Year Comprehensive Strategy, Implementation Plan).

Application Focus

Fuel treatments and ecological restoration activities have historically been focused on remotely based treatments conducted away from population centers. As a result, treatments could be conducted in lower risk situations, with lower threats to values to be protected. These earlier treatments were conducted on a small-scale basis and achieved only highly localized, stand maintenance results. These treatments were conducted for resource management objectives, silvicultural support, and hazard fuel reduction around federal infrastructures and may be representative of the "passive restoration" described by Greg Aplet in his presentation.

Changing Needs

The ever-increasing complexities of these programs are now trending into areas where the needs warrant expanding our range of treatment techniques to fully accomplish objectives. Historical experience may not have given us the requisite tools and knowledge we need for the demands of today and tomorrow to keep pace with changing societal values.

In the rush to quickly implement these programs, assign funds, and obtain results, we have sought quick and easy solutions. A full range of treatment techniques is available, and no single treatment is best suited for all situations. Some techniques are not even possible in certain situations. Our fuel treatment experience has been primarily a result of past prescribed burning. Some limited experience in mechanical treatments has also been gained but is the result of activities completed in support of silvicultural applications and not specifically designed for fuel treatment or ecological restoration. We need to utilize the various treatment types at our disposal and maximize the application to the land use situation. Treatment types including chemical, biological, mechanical, pile burning, mechanical plus pile burning, other multiple treatment combinations, small-scale prescribed fire, landscape-scale prescribed fire, and wildland fire use are viable options. However, these treatments must be applied in the correct land use situation. For example, landscape-scale prescribed fire and wildland fire use are not realistic or feasible in the wildland-urban interface areas while such site-specific treatments as biological, chemical, mechanical, and pile burning are not economically efficient in wildernesses or other large undeveloped and inaccessible areas. There are optimal areas of operation for each treatment type. Windows of opportunity exist and must be capitalized on. In terms of treatment types across the diverse range of land use situations, there is no panacea to the current issues of fuel reduction and management and ecosystem maintenance and restoration.

Accountability and Scrutiny

Now, probably more than ever before, accountability and scrutiny are forefront in the fuel treatment and ecological restoration programs. This is due to the potential outcomes of fuel treatments, outcomes of lack of treatments, values to be protected, recent occurrences (including Lowden Ranch, Cerro Grande, and North Shore of Kenai Lake prescribed fires), and the infusion of funding. Over the last several decades, land managers have stressed the changes that were occurring in fuel complexes and ecosystem health. After the 2000 fire season, the President and Congress listened and responded. The National Fire Plan was formulated and additional funds were provided to federal land managers to take action on reducing fuel accumulation and protecting ecosystems and communities. This development, though, is a double edged sword. The acceptance and endorsement of the problem by politicians and Washington level bureaucrats places it in a priority status and provides additional funding. This is one of the few times that partisan politics has not dominated the funding of strategic land management programs. The fuel treatment and ecosystem restoration program we are now embarking on is enjoying support from eastern and western Governors and Washington politicians, all with a common goal. We are seeing greater inter-departmental cooperation in the federal land management agencies, and markedly increased collaboration among federal and state agencies, tribal representatives, and local groups. However, with this increased support and funding comes additional scrutiny and attention. To maintain continued support, accountability and productivity are paramount. There will always be competition for funds and to maintain and continue to expand this program, we must make good decisions yet show sustained accomplishments with programmatic growth. We must reduce the barriers to implementation and avoid such issues like "analysis paralysis" that Tim Ingalsbee described in his presentation.

Changing Application Focus

The current need for fuel treatment and ecological restoration is shifting to include an area that we have not previously given full attention to. This is due, in part, to the fact that the wildland-urban interface did not always occupy such a prominent position in wildland areas. In addition, the wildland-urban interface is generally not on public lands, but on private lands and correspondingly has not been the subject of federal land management programs. Now, the wildland-urban interface requires significant attention and action. Our fuel treatment program is moving from one associated with remotely based treatments only to one having significant applications in both remotely based and population-based areas. Our fuel treatment program is also taking on greater levels of risk associated with this changing focus.

The enhanced focus that is being placed on wildland-urban interface area treatments does not mean a pendulum swing to the point where the goals of natural resource management and ecosystem maintenance and restoration will be minimized or ignored. Some very good points have been made during this conference about wildland-urban interface treatments versus ecological restoration. We have to continue to respond to both sets of objectives and we must fully understand that fuel treatment is not synonymous with ecosystem restoration.

External scrutiny over the wildland-urban interface is high and stems from recent General Accounting Office Reports and Testimony, Congressional

Hearings, budget hearings and language, accountability, and increased reporting requirements. Collaborative delineation and mitigation treatment development and implementation for wildland-urban interface areas is underway and providing a higher degree of local involvement. Reducing fuels in wildland-urban interface areas is about creating vegetative complexes that will burn with less intensity and severity and be less resistant to control. It is about protecting communities at risk. Wildland-urban interface treatments are being viewed by some as the priority. But, these types of treatments are very costly and labor intensive. Since the implementation of the National Fire Plan, wildland-urban interface treatments have required about two-thirds of the available funding and accounted for about one-third of the total acres treated.

You have heard earlier in this conference from Jim Menakis about using condition classes (fire regime condition classes) as a method to describe current conditions of vegetation and fuels. These classes depict the degree of departure from historical fire regimes resulting in alterations of key ecosystem components such as species composition, structural stage, stand age, and canopy closure. Wendel Hann provided a description of the joint federal cohesive strategy for protecting people and sustaining natural resources ("Cohesive Strategy") and explained the use of the condition classes to gauge the current status of ecosystem health across the country and to set long-term strategic goals for maintenance and restoration. When we look at the proportions of areas in the various condition classes, we can readily see the ongoing trend. Ecosystem health and condition is worsening. Without restoration treatments, the long-term prognosis is for continued worsening of ecosystem health and continued, if not increased, frequency, intensity, and outcome of wildland fire. Areas not classified as wildland-urban interfaces are in need of treatments. Remember during the last one and one-half years, fuel treatments and ecosystem restoration activities in non-wildland-urban interface areas have accounted for about one-third of expenditures and two-thirds of accomplishments.

How important are the condition class descriptions? A cursory response to this problem might indicate that we need to treat only the condition class 3 areas, those in the most degraded state. However, we must ask ourselves, Can we afford to not treat areas in all three condition classes? Greg Aplet of the Wilderness Society presented examples where lower intensity fires burning in the understory still were responsible for structural losses. Does condition class matter in the wildland-urban interface?

I submit that all these things are important to the future of this program. We need to develop a comprehensive fuel treatment and ecological restoration program that is responsive to both protection and restoration objectives. This program must closely depend on condition class designations, not focus on only the worst case, but provide for maintenance of the better situations while restoring the lesser situations to better conditions. It must effect change in the wildland-urban interface areas regardless of condition classes there. No matter what the vegetative condition is, the ability to control wildland fires and minimize burning intensity must be enhanced.

Scheduling

Scheduling affects both the proper place and the appropriate time for fuel treatment and ecological restoration. Scheduling determines the sequencing of projects (i.e., away from highest risk areas on continuing maintenance work), sets priorities, coordinates across jurisdictional boundaries, and influences long-term activities. Scheduling must be an integral part of fuel treatment and ecological restoration activities to ensure the maximum efficiency and productivity.

Scale

Scale is critically important. Bill Romme's presentation provided us with a solid example of the scale issues that face us and stand-versus-landscape considerations. We can no longer only deal with stand maintenance or site-specific treatments. We must now expand to landscape-scale fuel treatment applications. However, landscape-scale does not simply infer that larger treatments must be applied. A program of this scale will include a combination of site-specific and large-scale treatments using much of the full spectrum of treatment types to effect significant change over an entire landscape. This type of program, a comprehensive fuel treatment and ecosystem restoration program, must be developed and implemented. This type of program will be representative of "active restoration" carried out at the ecoregional level as Greg Aplet described in his presentation.

Information Acquisition

There are many unanswered questions involving fuel treatment and ecological restoration. Some you have heard during the last three days, some have not been discussed. Examples of the types of questions that have surfaced here include:

- What are we restoring the system to?
- What are the effects of treatments on fire severity?
- What are reference conditions?
- What are future conditions?
- What are the best restoration parameters?
- How do we treat the wildland-urban interface (you heard Mark Finney provide new information on this subject)?
- How can decision-making be improved?

Information acquisition is critical to maintaining and improving this program. Management must be dynamic and accept new developments and incorporate them into actions. The Federal Fire Policy stresses the importance of incorporating the best available science into fire management. Both Wally Covington and Greg Aplet reinforced this in their presentations. Wally Covington very succinctly summarized the importance of information acquisition by stating that without a sound understanding of the ecosystem that we are dealing with, our decisions will degenerate into ill-informed speculation, subjective judgment, bias, ideology, and personal policy viewpoints. Jim Menakis' and Wendel Hann's presentations provided examples of science-based management analyses and support to decision-making.

Collaboration

The future of this program is in collaboration. We must support collaboration at all levels but we must advocate and ensure local collaborative decision-making. Some examples exist of increased collaboration around the country but this must become a universal process. Tim Ingalsbee presented an example of successful local collaboration and how diversity and societal recognition helped implement a difficult project.

Performance Measures

What are measures of performance? These are standards that have developed in response to the new levels of accountability and scrutiny associated with the program. Performance measures have important value in that they identify standards associated with the program that managers need to meet. Thus, they help to provide structure and framework for program implementation. However, in the haste to implement, we have begun to confuse performance outputs with performance outcomes. A performance output is something that occurs from short-term operational activity, such as numbers of projects proposed, numbers of projects completed, or numbers of acres treated. A performance outcome, the desired result of the activities, represents what has resulted from long-term programmatic execution, such as numbers of communities protected, values to be protected with changed fire protection capability, proportions of condition classes maintained and restored, numbers of areas influenced or affected from treatments, and sustained incremental increases in programmatic accomplishments.

Trying to describe performance in terms of outputs merely focuses on the short-term actions. Using performance outcomes provides a better description of long-term influences of programmatic activities and can offer a basis for program evaluation.

Measuring Success

How do we measure success? To do this effectively, we must review the goals of fuel treatment and ecological restoration and the program outcomes. Success cannot be measured in small-scale increments but in establishment and implementation of a long-term, pro-active restoration-based landscape-scale fuel treatment program. This program must be accountable and meet the scrutiny accompanying increased funding and attention. It must effect change in the form of increased protection capabilities and reversed ecosystem degradation. It must enact and utilize collaborative decision-making at all levels. It must be responsive to dynamic ecological situations and not be static in time. Research has been a fundamental component of such a dynamic program. I have talked about information acquisition, but its importance warrants repeating. We must learn more, acquire information, and apply this information. Additional topical areas that have not been discussed include:

- What are the interactions with climate change and fire season duration, fire severity, and changing vegetation?
- What is the relevance of historical reference conditions?

Information such as that which Mark Finney presented must be acquired and applied to produce the most efficient efforts.

Summary

It is certainly an exciting time to be involved in natural resource management. The advent of the National Fire Plan means many things but principally an effort to resolve a worsening problem in fuel accumulation and ecosystem health. The National Fire Plan is not the final answer, but a beginning. We have a long way to go; you know that this situation is the result of a century of actions based on 100-year-old state-of-the-knowledge. We have learned more during this time about fire, fuel, and ecosystem dynamics and believe that charting a new course is necessary. This program of action will not be easily or quickly achieved. Along the way, there will be much to learn and much to incorporate into actions.

Currently, we have three basic questions that have arisen at this conference to respond to. Information related to these questions has been pervasive to the various sessions of this conference. These questions are:

- What are we restoring the system to? What about permanently altered ecosystems? What about not restoring but defining and stabilizing permanently changed ecosystems into new resilient systems?
- Have we created a situation that will allow the program to succeed?
- Will our actions be sufficient to protect communities and ecosystems?

We must answer these questions and evaluate our actions in light of them. Defendable and supportable responses to these questions will facilitate our ability to proceed with a successful long-term strategic program of action to meet the objectives of fuel treatment and ecosystem maintenance and restoration.

I would like to thank all the organizers, speakers, and participants at this conference; it has been an exciting, worthwhile, and motivational undertaking. I hope everyone here will take the information and enthusiasm back to your job and channel them into our common goals of fuel treatment and ecological restoration.

This has been a great conference and I ensure all of you that this will not be the last one concerning this subject.

Thank you for this opportunity.

Poster Abstracts

Poster Abstracts

Fire Severity and Salvage Logging Effects on Exotics in Ponderosa Pine Dominated Forests

Suzanne Acton, Payette National Forest, New Meadows, ID, and Forest Sciences Department, Colorado State University, Fort Collins, CO

The effects of fire severity and salvage logging on the understory vegetation response in ponderosa pine dominated areas of the South Fork Salmon River Drainage of west-central Idaho were examined five years post-fire. The study objectives were to correlate the effects of fire severity and salvage logging on exotic species richness and cover; and to examine the effects of scale in sampling in post-fire understory vegetation. Five years following fire, areas that burned with greater fire severity had increases in total species cover and exotic species richness. Low severity fires had lower exotic species richness compared to high severity and unburned areas. Salvage logging increased exotic species cover, but not exotic species richness. The only result that was consistent at multiple scales was the relationship between low severity fires and low exotic species richness. Measurement scale should be an important consideration when designing future research experiments in this area, with preference to multiple-scale measurements.

Using the FARSITE Model to Predict the Differential Effects of Fuel Treatments on Potential Wildland Fire Behavior in Dry Forests of the Wenatchee Mountains

James K. Agee and M. Reese Lolley, College of Forest Resources, University of Washington, Seattle, WA

A pressing issue in fire management is understanding the differences in the effects of fuel treatments on wildfire behavior. To gain insight into this issue Fire Area Simulator (FARSITE) was used to model fuel treatment effect on short-term wildland fire behavior to test differences between mechanical and prescribed burn fuel treatments. Intensive fuels data were collected as part of the pre-assessment for the Fire and Fire Surrogates Treatments Study at the Mission Creek site in the Wenatchee Mountains of Washington. These data were used to estimate mechanical and prescribed fire treatment effects on forest fuel profile and loadings using quantitative and qualitative methods. Finally, FARSITE was used to predict potential short-term wildland surface and crown fire differences between fuel treatments.

Effects of Seasonal Fire on Mortality and Oleoresin Pressure of Old-Growth Ponderosa Pines

James K. Agee and Daniel D. B. Perrakis, College of Forest Resources, University of Washington, Seattle, WA

This project examines techniques for evaluating the survivorship of old-growth ponderosa pines (*Pinus ponderosa* Laws.) following restorative prescribed burning at Crater Lake, OR. The study area was divided into spring burn units, fall burn units, and control units (no burning). Parameters measured (before and after burning) include tree species and density, understory vegetation, fuel loading, and ponderosa pine crown vigor. Selected pines will be intensively monitored following burning by measuring their oleoresin exudation pressure (OEP) using modified pressure gauges. OEP is considered a reliable indicator of susceptibility to bark beetle attacks, a frequent cause of mortality following burning. Pre-burn data shows considerable woody fuel accumulation, limited understory diversity, and fair to poor vigor in overstory ponderosa pines. It is expected that burning in the spring will result in lower OEP and higher pine mortality compared to burning in the fall. A threshold OEP reading that predicts future bark beetle mortality will hopefully be identified.

Consequences of Wildfire on Understory Vegetation in Untreated Ponderosa Pine Forests

Noah Barstatis, W. H. Moir, Julie Crawford, and Carolyn Hull Sieg, USDA Forest Service, Rocky Mountain Research Station, Flagstaff, AZ

Heavy fuel loading and high densities of small trees increase the risk for large stand-replacing wildfires in ponderosa pine forests of the Southwest. In addition, there is a concern that these high intensity fires will lead to the spread of a variety of exotic plant species that have been introduced to the region in recent years. A series of wildfires in 1996 in northern Arizona provided an opportunity for us to examine the understory response through time following a large, high intensity fire. In 1997 we established permanent plots on the Hochderffer wildfire and in adjacent unburned sites in northern Arizona. We sampled plant canopy cover and frequency in a total of 38 plots each year through 2001. Our preliminary analyses document dramatic differences in the vegetative composition and diversity between burned and unburned sites, including the presence of multiple exotic species as well as fire-following native shrubs.

Variation in Fuel Loadings Within Existing Ponderosa Pine and Mixed Conifer Forests in the Southern Colorado Front Range

M. A. Battaglia, W. D. Shepperd, and M. J. Platten, USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO

Assessment of the quantity and type of fuels present in a forest is important for the proper application of fire behavior model predictions. The objective of this study was to obtain quantitative data on fuel loadings that exist under ponderosa pine and mixed conifer forest types typical of the southern Colorado Front Range. Fuels inventory and stocking data were collected during 2001 from 479 plots on the Manitou Experimental Forest (MEF) in south central Colorado. The plots studied contained a variety of stocking and physiographic conditions and these relationships were explored to determine whether any correlations existed between fuel load and community condition. This poster describes the distribution of fuel loadings and explores relationships between aspect, elevation, stocking, species composition and fuel characteristics.

Forest Health and Tree Removal Equipment in the Wildland-Urban Interface—An Educational Video

David R. Betters and Robert Avera, Department of Forest Sciences, Colorado State University, Fort Collins, CO

Forest health is a key issue in the wildland-urban interface. Operational programs to improve forest health conditions in the interface typically involve thinning, or tree removals, of smaller diameter trees on privately owned lands. Implementing these programs requires 1) persuading private landowners, through education, to adapt and implement tree thinning operations on their properties, and 2) using tree removal equipment that is more cost effective for small diameter tree harvesting and suited for smaller acreages characteristic of private interface land holdings. The objective of this project is to develop a video and an accompanying information packet that addresses the two needs mentioned above. The video / packet will be created, critiqued via a survey, and then finalized. The video / packet will be applied in two wildland-urban interface community case studies concerning the adoption of fire mitigation programs. The finalized video / packet will be made available through Colorado State University Cooperative Extension.

A Community-Level Process for Adoption of Forest Restoration and Fire Mitigation Programs

David R. Betters and Christy L. Higgason, Department of Forest Sciences, Colorado State University, Fort Collins, CO

The National Fire Plan and its emphasis on restoring health to fire-adapted ecosystems and reducing community risk of wildland fires has created an overwhelming need for collaboration at all levels in Colorado. The nature of this problem requires the involvement not only of a variety of local organizations and citizens but also the coordination and participation of a number of resource agencies having separate but somewhat overlapping roles regarding communities and natural resource sustainability and forest health. To facilitate community involvement and reduce confusion in a situation where several agencies offer public service through land stewardship, a process for local community-level decision-making is key. The objective of this study is to develop a process coordinated amongst agencies in Colorado that facilitates community / interest group agreement for adopting and implementing a forest restoration / fire mitigation strategy. The analysis of information collected in the study includes two separate reviews of process. The first review involves community decision-making processes in general. This entails generalized process and communication theory applicable regardless of problem area. The second review involves existing processes used in fire mitigation in the wildland-urban interface. This information includes an overview of current processes used in fire mitigation programs in western states that include educational packages. These processes and education messages are examined to determine if consistency exists and whether improvement can be made overall by providing the best delivery content and means. The reviews are used to propose an overall process to facilitate adoption of community fire mitigation programs. It is envisioned that the information gathered from this project will also serve as a supplement to related land management planning efforts.

Response of Understory to Thinning and Prescribed Burning in Northeastern California

Melissa Borsting, Division of Ecosystem Sciences, College of Forest Resources, University of Washington, Seattle, WA

My research explores the short-term response of understory vegetation to silvicultural and fuel reduction treatments in ponderosa pine forests of northern California (Blacks Mountain Experimental Forest). It is important to understand these responses, as understory vegetation contributes significantly to biological diversity, nutrient cycling, wildlife habitat, and fire regime in these forests. The experimental design consists of two thinning treatments (high and low structural diversity), with and without prescribed burning, replicated at each of three locations. In this poster, I discuss the short-term responses of the dominant understory species and growth forms (e.g., annuals vs. perennials, herbs, graminoids, and shrubs). I compared changes in plant cover, species richness, and community composition, and address treatment-level responses, within-treatment heterogeneity, and relationships with burn intensity.

The Effects of Herbivory on Quaking Aspen (*Populus tremuloides*) Regeneration Following Various Treatments in South-Central Utah

Shauna-Rae Brown, Arizona State University, Chandler, AZ

Aspen (*Populus tremuloides*) ecosystems are known for their high biodiversity and are believed to have been a dominant type in the Intermountain West since the early Pleistocene. However, aspen ecosystems have dramatically declined since the 1870s. Over the past eight years, Fishlake National Forest, Utah has been working to stem the tide of aspen decline by treating for aspen restoration. During the summer of 2001 (June - October), 34 sites were surveyed to monitor the success of aspen regeneration following a variety of treatments including wildland fire, clearcutting for aspen regeneration, and clearcutting followed by prescribed fire. All clearcut sites that were monitored had some type of fencing erected to serve as study controls. Sites fenced to preclude all cattle and wildlife herbivory produced the highest number of aspen suckers compared to adjacent unfenced or cattle-excluded sites. Of the thirty-four sites surveyed, only one had no measurable aspen regeneration, which was due to high ungulate utilization.

Cerro Grande Fire Restoration at Los Alamos National Laboratory

Kevin Buckley, Sam Loftin, Jeff Walterscheid, and Mike Alexander, Los Alamos National Laboratory, Los Alamos, NM

The Cerro Grande Fire of 2000 had a huge impact on forests on and around Los Alamos National Laboratory (LANL). Immediately after the fire, there were concerns about increased erosion and flooding and its impacts on contaminated soil. An Emergency Rehabilitation Project was created to evaluate and estimate the impacts of the Cerro Grande Fire on LANL property, design appropriate methods to mitigate the effects of increased erosion and storm water runoff, and implement these measures to prevent further damage to people, property, and the environment. Seeding, raking, contour felling, log erosion barriers, straw wattles, straw mulch, and hydromulching were among the treatments used to stabilize soils and retain water on approximately 648 ha (1,600 acres) of moderately and severely burned hillslopes at LANL. Assessments show a highly successful response from combinations of seeding, raking, and mulching.

Mechanical Fuels Reduction Contract Underway at Richfield Bureau of Land Management

Linda Chappell, Doug Thurman, and Russell Ivie, Richfield Field Office, Utah Bureau of Land Management, Richfield, UT

Continuous pinyon-juniper fuels on the south, east, and north sides of Mayfield in south-central Utah are a potential wildfire threat. The nearby previously chained lands administered by the Bureau of Land Management were divided into three separate units to treat. This was in an effort to allow bidders of any size an opportunity to obtain the contract. Twenty-one hundred acres of pinyon / juniper fuel are now contracted for cutting in 2001-2002. A copy of the contract will be available. Following the treatments some broadcast burning will occur on areas with heavier fuel loads. Pre- and post-cut photos are included.

Definition of a Fire Behavior Model Evaluation Protocol: A Case Study Application to Crown Fire Behavior Models

Miguel G. Cruz, School of Forestry, University of Montana, Missoula, MT; Martin E. Alexander, Canadian Forest Service, Northern Forestry Centre, Edmonton, Alberta, Canada; and Ronald H. Wakimoto, School of Forestry, University of Montana, Missoula, MT

Testing and evaluation of models is an important and fundamental component of the scientific method, leading to model understanding and the increase of their credibility. The process of model evaluation has been seen differently by several authors due to the philosophical interpretations of what a model is. An important aspect when considering model evaluation is the definition of the criteria that should be followed, which depends on the type of model being evaluated and its potential application. When considering fire behavior science, theoretical models developed to understand certain physical and chemical phenomena should be evaluated in a different form than operational models built to support decision-making. In the present paper a model evaluation protocol is proposed to encompass different aspects, such as: (1) model conceptual validity, (2) data requirements for model validation, (3) sensitivity analysis, (4) inter-model comparison, (5) predictive validation, and (6) statistical tests. The proposed protocol was applied to evaluate fire behavior models aimed at predicting crown fire initiation and spread with potential application in fire management decision support systems. The evaluation protocol highlighted the limitations of the discussed models and the implications of such limitations when applying those models to support fire management decision-making. The particular case of definition of fuel treatments aimed at reducing crown fire hazard was analyzed. The model limitations identified through the results reveal deficiencies in the state of knowledge of determinant processes in crown fire behavior and point to pertinent research needs.

Monitoring Fuel Conditions and Predicted Fire Behavior Before and After Varied Fuel Treatments in the Quincy Library Group Area: Site and Landscape Scale Findings and Implications

Jo Ann Fites-Kaufman, USDA Forest Service, Adaptive Management Services Enterprise Team; Bernie Bahro, USDA Forest Service, Pacific Southwest Region; and Larry Hood, USDA Forest Service, Lassen National Forest, CA

A forest health pilot including mechanical treatments (thinning and biomass) and prescribed burning was conducted on portions of three National Forests. We monitored site and landscape scales of fuel conditions. The 85 pilot projects encompassed over 45,000 acres and 993 individual treatment units. All projects were mapped digitally and detailed data were collected on surface, understory, and crown fuels. Vegetation was collected for random units. We evaluated several statistical approaches given constraints of non-random treatment assignments and lack of before and after data. Randomization tests and Impact vs Reference Sites analyses were conducted. Key findings were that sites selected for burning had lower canopy covers, tree densities, and crown bulk than those selected for mechanical fuel treatments. Regardless of treatment, many plots were predicted to have crown fire at high fire weather conditions. At the landscape scale, the pilot resulted in treatment of less than 10 percent of the landscape.

Applying and Evaluating Ecosystem Reinvestment A Case Study of the Bitterroot Watershed, Montana

Stephanie Gripne, Crockett Wildlife Conservation Program, School of Forestry, University of Montana, Missoula, MT

Ecosystem reinvestment is one of the emerging frameworks associated with ecosystem management. Ecosystem reinvestment describes any activity that increases the value of ecosystems and communities by directing revenues from natural resource activities toward efforts to restore and maintain these systems. Ecosystem reinvestment builds off the premise that ecosystem goods and services can be thought of as sources of capital. Natural and social capital, just like physical capital, can degrade over time and needs investments of resources in order to maintain their value / function. My research objectives are to apply the ecosystem reinvestment framework to the Bitterroot Watershed ecosystem and evaluate the potential of this framework to actually increase the level of resources that are reinvested back into the ecosystem. Specifically, I propose to determine 1) the market and nonmarket economic value of several ecosystem goods and services generated in the Bitterroot Watershed, 2) the cost of restoring different scenarios of ecosystem goods and services, 3) the individuals who benefit from these ecosystem goods and services, and 4) the available mechanisms for providing the support needed to maintain these forest ecosystems using a economic analyses, a contingent valuation willingness-to-pay survey, stakeholder analysis, and literature review. My proposed study will provide one of the first formal evaluations of the application of ecosystem reinvestment as a model for maintaining and restoring ecosystems. Additionally, this study will improve the state of knowledge about market and nonmarket economic values of forest ecosystem goods and services in the Bitterroot Watershed, which to date, has not been comprehensively been explored.

Demonstration Plots for Comparing Fuel Complexes and Profile Development in Untreated Stands Vs. Stands Treated for the Management of Spruce Beetle Outbreaks

Elizabeth Hebertson, USDA Forest Service, Ogden Field Office, Ogden, UT; Michael Jenkins, Department of Forest Resources, Utah State University, Logan, UT; and Linda Wadleigh, USDA Forest Service, Williams, AZ

The activity of insects, diseases, and abiotic agents is known to contribute to changes in the characteristics of fuels complexes and associated fire behavior. Landscape-scale density management strategies have been proposed as viable alternatives to sanitation or salvage for managing insect and disease outbreaks. The effect of various density treatments on fuels complexes, or fuels development, however, is not known. An important agent of disturbance in Intermountain Region National Forests of the United States is the spruce beetle. Outbreaks have caused extensive mortality resulting in significant loss of timber, recreational opportunities, and aesthetics. Mortality resulting from outbreaks has also resulted in increased dead fuel loads and will likely alter the fuels complex of infested stands over time. The Fishlake National Forest in cooperation with the Utah State Division of Forestry, Fire and State Lands will implement two density management treatments in several spruce-fir forests with the purpose of reducing susceptibility of these stands to spruce beetle infestation while enhancing tree vigor. This situation provides an excellent opportunity to establish permanent demonstration plots in treated versus untreated stands. Fuels inventories and appraisals will be conducted to determine changes in the fuel complex and profile over time. Based on the fuel inventory and appraisal data fuels treatment strategies including a combination of mechanical and biomass, utilization will be implemented on the demonstration plots. Prescribed fire will also be considered for treating fuel provided all prescription variables are met. Other benefits derived from this information include the demonstration of strategies for managing insect outbreaks and fuel development and manipulation.

The Effects of Long-Term Repeated Prescribed Burning on Nutrient Regimes in Loblolly Pine Stands in the Gulf Coastal Plain in Arkansas

Jennifer J. Henry and Hal O. Liechty, School of Forest Resources, University of Arkansas, Monticello, AR; and Michael S. Shelton, USDA Forest Service, Southern Research Station, Monticello, AR

Nutrient levels of various forest components were measured in loblolly pine stands within the Gulf Coastal Plain of Arkansas. The stands evaluated had either been burned every 2-3 years for the past 20 years or had not received any prescribed burning. Comparisons were made between stand types (burned and unburned) to assess the long-term effects of repeated prescribed fire on nutrient regimes. Due to low fuel loadings and wet / cool weather conditions, fires were of low to moderate intensity. Pine litterfall collected in the burned stands was higher in K, Ca, and Mg but lower in C than in the unburned stands. Although nutrient levels of the forest floor, live foliage, and the soil did not generally differ between stand types, N concentrations were lower in the understory vegetation of the burned compared to unburned stands. Additional research is being conducted to determine the nutrient dynamics from the latest prescribed fire.

The Citizen's Call for Ecological Forest Restoration: Forest Restoration Principles and Criteria

Timothy Ingalsbee, Western Fire Ecology Center, Eugene, OR

In February 2001, a diverse group of conservationist and community forestry groups from around the country held a Restoration Summit in Boulder, Colorado to develop scientifically sound and socially progressive restoration principles. Their goal was to create a document to help grassroots conservationists and forest practitioners advocate for Congressionally funded restoration programs, and evaluate federal restoration policies and projects. The Boulder

Summit initiated a year-long process resulting in a consensus-based document called “The Citizen’s Call for Ecological Forest Restoration: Forest Restoration Principles and Criteria.” This document and the growing coalition of organizations and individuals who endorse it and plan to utilize it have the potential to affect forest restoration programs, policies, and practices on federal lands in the near future. This poster presents some of the core principles concerning ecological restoration, ecological economics, and community and workforce development, with a special focus on fire restoration and fuels treatments.

A Decision Support System for Spatial Analysis of Fuel Treatment Options and Effects at Landscape Scales

J. Greg Jones and Jimmie D. Chew, USDA Forest Service, Rocky Mountain Research Station, Missoula, MT

We present an approach for analyzing fuel treatment activities both spatially and temporally at the landscape scale that employs strengths of both simulation and optimization modeling. SIMulating Vegetative Patterns and Processes at Landscape ScaLEs (SIMPPLLE), a stochastic simulation modeling system, is used for projecting vegetative change in the presence of interacting natural processes, with or without management treatments. The Multi-resource Analysis and Geographic Information System (MAGIS), an optimization modeling system, integrates biophysical and socioeconomic information and schedules management practices spatially and temporally. The combination of SIMPPLLE and MAGIS provides a powerful analytical methodology for: 1) analyzing the extent and likely location of insects, disease, and fire both in the presence and absence of treatments, 2) developing spatial and temporal treatment alternatives for addressing fuels treatment along with other resource objectives, and 3) evaluating those alternatives in a manner that captures the combined effects of treatments and disturbance processes.

Invasive Plants and Wildfire on the Cerro Grande Fire, Los Alamos: Integration of Spatial Information and Spatial Statistics

Mohammed A. Kalkhan, Natural Resource Ecology Laboratory; Philip N. Omi and Erik J. Martinson, Department of Forest Sciences; Thomas J. Stohlgren and Geneva W. Chong, Natural Resource Ecology Laboratory; and Molly A. Hunter, Department of Forest Sciences, Colorado State University, Fort Collins, CO

The integration of spatial information and spatial statistical modeling can be used to investigate the spatial relationships among fuels, wildfire severity, and post-fire invasion by exotic plant species through linkage of multi-phase sampling design and multi-scale nested sampling field plots, and pre- and post-fire. This technique provides useful information and tools for natural resource managers, especially in describing landscape-scale fire regimes, invasive plants, and ecological and environmental characteristics for the Cerro Grande fire site, Los Alamos, NM, USA. We integrated filed data and spatial information

(Landsat TM Data, transform TM Data, and topographic data) with spatial statistics for modeling large-scale and small-scale variability to predict the distribution, presence, and patterns of native and exotic species. All models were selected based on lowest values of standard errors, AICC statistics, and high R^2.

Guidelines for Restoring an Unlogged Ponderosa Pine/Douglas-Fir Landscape in the Colorado Front Range After a Century of Fire Suppression

Merrill R. Kaufmann , Paula J. Fornwalt, Laurie S. Huckaby, and Jason M. Stoker, USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO; and William R. Romme, Department of Forest Sciences, Colorado State University, Fort Collins, CO

Current forests in the unlogged ponderosa pine / Douglas-fir landscape at Cheesman Lake in the Colorado Front Range are denser with fewer and smaller openings and more young Douglas-fir trees than occurred historically, largely the result of fire suppression. We estimate from spatial maps and temporal patterns of historical fires that four significant fires would have occurred since the last major fire in 1880. The ages of old trees were used to estimate the severity of past fires. Fire and tree age data were used to estimate plausible locations and severities of excluded 20th century fires. Landscape structure for 1900 was estimated by "de-growing" forest polygons derived from recent aerial photos and field plot data. FVS was used to estimate landscape structure during the 20th century with the simulated effects of 20th century fires. A frame model depicts changes in the structural components of the landscape from around 1500 to the present, and compares the effects of fire suppression with a plausible 20th century natural fire scenario.

A Device and Method for Conducting Small-Plot Experimental Burn Treatments

John L. Korfmacher, Jeanne C. Chambers, and Robin J. Tausch, USDA Forest Service, Rocky Mountain Research Station, Forestry Sciences Laboratory, Reno, NV

Large prescribed burns are valuable tools in assessing fire effects on soils and vegetation, but administrative and safety concerns may require smaller-scale efforts. We constructed a 3.4 m diameter fire enclosure for conducting experimental burn treatments on small (~10 m^2) circular plots, using sheetmetal, electrical conduit, and other commonly available materials. We field tested the enclosure in a sagebrush-grass ecosystem in central Nevada, and evaluated peak fire temperatures using small metal tags striped with temperature sensitive paint. Under-shrub microsites averaged 381°C, significantly hotter than under-grass (307°C) and bare-ground (310°C) microsites. Subsurface (2 cm depth) temperatures rarely exceeded 79°C, the lowest temperature detectable by our method. The fire enclosure contained the fire and did not permit escape of embers or firebrands. The fire enclosure, burn technique, and temperature monitoring method we use here are inexpensive, easily deployed, and desirable for experiments where larger-scale burns are impractical.

Direct Fire Effects Following the Jasper Fire, Black Hills National Forest, SD

Leigh Lentile and Skip Smith, Department of Forest Sciences, Colorado State University, Fort Collins, CO; Wayne Shepperd, USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO; and Tricia Balluff, Department of Forest Sciences, Colorado State University, Fort Collins, CO

In late August 2000, the Jasper Fire burned 33,000 ha of the Black Hills in South Dakota. The Jasper Fire burned under a variety of fuel and weather conditions creating a mosaic pattern of tree mortality within patches of varying size and extent. Remotely sensed canopy reflectance indicates that 39, 32, and 24 percent of the fire burned under high, moderate, and low severity conditions. Within the context of a larger monitoring design, we will provide an initial characterization of severity by quantifying direct fire effects on the overstory and the forest floor. An understanding of why particular trees die and what we can expect in terms of recovery based on different fire effects may be useful in planning.

We observed an increase in the proportion of crown, bole, and forest floor affected by fire along a severity gradient from low to high. In low and moderate severity treatments, 21-70 percent of the crown was scorched. In high severity treatments, 92 percent of the crown was consumed. We indexed fire severity as the product of the proportion of the ground area charred and the severity of char scaled from low (1) to high (3). Fire severity was 115 for low and 236 for high treatments on a scale from 100-300. Forest floor litter depths decreased from 2.54 cm in unburned stands to 0.15 cm in high severity treatments. Litter mass decreased from 1265 g m^{-2} in unburned areas to 82 g m^{-2} in high intensity treatments.

Tree mortality due to direct fire effects increased from low to high fire severity. Fire selectively killed smaller trees in low and moderate severity treatments. However, no trees, independent of diameter, survived in high severity treatments. This observation suggests that a threshold may exist between diameter and fire survivability in low and moderate severity fires in ponderosa pine systems in the Black Hills.

Experimental Approach for Monitoring the Jasper Fire, Black Hills National Forest, SD

Leigh Lentile and Skip Smith, Department of Forest Sciences, Colorado State University, Fort Collins, CO; Wayne Shepperd, Anna Schoettle, José Negrón, and Kevin Williams, USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO; and Tricia Balluff, Department of Forest Sciences, Colorado State University, Fort Collins, CO

In late August 2000, the Jasper Fire burned ~33,000 ha or 7% of the Black Hills National Forest, South Dakota. This was the largest recorded fire in the Black Hills. The Jasper Fire burned under a variety of vegetative, topographic, and meteorological conditions creating a mosaic of vegetative mortality in patches of varying size and extent. This fire appears to be similar to other historical fires in interior forests of the Black Hills in that it was an extensive, late season burn. The heterogeneity of fire effects suggests that fire behavior was spatially and temporally variable, exhibiting characteristics of surface, crown, and a combination of these fire types.

As part of a five-year effort, we established a network of permanent sites within the burn to monitor the influence of pre-fire vegetative conditions, fire size, and consumptive patterns on post-fire ecological succession. Direct fire effects on overstory, understory, forest floor, and soil components were assessed in 2001. Measurements of direct fire effects on soil and vegetation will be translated to quantitatively-based severity indices. Given the spatial extent and heterogeneity of the burn we can address questions relating to fire behavior and the timing and response of vegetation recovery. An assessment of mixed fire behavior within the signature of the Jasper Fire may provide a more unified and plausible idea of common fire behavior that has shaped landscapes within the Black Hills ecosystem.

Non-Structural Value Added Application of SDU Materials in Landscape Architecture Applications

Kurt H. Mackes and David G. Buech, Department of Forest Sciences, Colorado State University, Fort Collins, CO

Most materials culled from densely packed, even-aged stands are small-diameter trees (4- to 8-in. stems). Material of this size is difficult to process and extract economic value from. Most of this material is limited to nontraditional markets because of lack of grading standards / rules and reliable mechanical properties. There is demand for markets that require minimal material processing, are local, do not require grading and can be used without building code approval. This paper outlines applications in Landscape Architecture and the opportunities that they present. Applications include: fencing, retaining walls, walkways, nonstructural decks, and other “garden structures.”

Wood Utilization and Marketing Strategy for Small Diameter Wood in Boulder County

Kurt H. Mackes and Julie E. Ward, Department of Forest Sciences, Colorado State University, Fort Collins, CO; and Craig Jones, Winiger Ridge Ecosystem Management Pilot Project, Colorado State Forest Service, Boulder District

The Winiger Ridge Ecosystem Management Pilot Project in Boulder County, Colorado is one of 28 demonstration sites that are part of an innovative land management initiative between public agencies and landowners to collaboratively manage resources to improve overall conditions of the ecosystem. The purpose of this research is to develop a marketing strategy for utilizing the characteristically small diameter wood (<12" d.b.h.) that would be removed from this project and other fuel hazard reduction projects in Boulder County. This will be accomplished by characterizing the resource being harvested (quality, quantity, availability, etc.), identifying existing markets for materials, and exploring the potential for new markets that could utilize this resource.

Research on Stand Management Options for Reducing Fuels and Restoring Two-Aged Lodgepole Pine Communities on the Tenderfoot Creek Experimental Forest

Ward McCaughey, Forestry Sciences Lab, USDA Forest Service, Rocky Mountain Research Station, Bozeman, MT

Fire-dependent lodgepole pine stands comprise significant acreages of mid- and upper-elevation forests in the Northern Rockies, providing wood products, wildlife habitat, livestock forage, water, recreational opportunities, and expansive viewsheds. Many lodgepole pine stands are in late-successional stages and at risk to pests and catastrophic-scale fires. Tenderfoot Creek Experimental Forest is located on the Lewis and Clark National Forest in the Little Belt Mountains of Central Montana. Twenty percent of the lodgepole pine stands on the experimental forest were found to be two-aged and another 30 percent were in an indistinct mosaic of a duel-fire complex. This paper describes preliminary results of the Tenderfoot Research Project designed to evaluate two-aged harvest methods in lodgepole pine stands by integrating silviculture and prescribed fire. Research studies evaluate the effects of harvesting and prescribed fire on several resources such as water quality and quantity, wildlife, forest fuels, and vegetation response.

The Role of Wildland Fire and Subsequent Insect Attack on Ponderosa Pine Mortality

Joel McMillin, Linda Wadleigh, and Carolyn Hull Sieg, USDA Forest Service, Rocky Mountain Research Station, Flagstaff, AZ; José Negrón, USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO; Ken Gibson, USDA Forest Service, Region 1; Kurt Allen, USDA Forest Service, Region 2; and John Anhold, USDA Forest Service, Rocky Mountain Research Station, Flagstaff, AZ

The unprecedented fire year of 2000 provided an opportunity to quantify cumulative impacts of wildland fires and subsequent insect attack on ponderosa pine mortality over a large region. In 2001 we established plots in four National Forests: Black Hills in South Dakota, Custer in Montana, Arapaho-Roosevelt in Colorado, and Coconino in Arizona. In each area, we sampled 1500+ trees in burned areas and 500 trees in unburned areas. For each tree, we measured height, dbh, pre-fire live crown ratio, percent crown scorch, percent crown consumption, percent scorched basal circumference, scorch height on the bole, and insect presence. In addition, we collected four phloem samples from each of 200+ additional trees in each area to quantify the relationship between exterior signs of fire-caused damage and cambium damage. Tree mortality will be monitored for three years post burn. Our goal is to provide land managers with quantitatively based guidelines for assessing potential tree mortality following wildland burns.

Roundheaded Pine Beetle, *Dendroctonus adjunctus*, and Fuel Loads in the Sacramento Mountains, New Mexico

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The roundheaded pine beetle (RPB), *Dendroctonus adjunctus*, is one of the most important bark beetles associated with ponderosa pine, *Pinus ponderosa*, in the Southwest. Outbreaks of this insect have caused extensive mortality in the Lincoln National Forest, NM, in the 1970s and in the 1990s. During the mid-1990s a network of plots was established in infested and uninfested stands to develop models to estimate the probability of infestation by the RPB. During remeasurement of these plots during the summer of 2001, we quantified fuel loads associated with tree mortality caused by the RPB. We used Brown's method to inventory coarse woody debris (cwd) in the plots. Transects were placed in a randomly selected direction but with the center of the transect coinciding with the center of our plots. We observed no differences in cwd in the 0 – ¼ inch or the ¼ to 1.0 inch size classes between the infested and the uninfested stands. We observed significantly higher cwd debris levels in the 1.0 to 3.0 inch class; in the >3.0 inch sound and rotten size classes; and in total cwd in the infested stands. We used BehavePlus (USDA Forest Service) to simulate spread and flame length using default values for the timber with litter and understory fuel model but adjusting the 1-, 10-, and 100-hour fuels to the levels we observed. We obtained increases in flame length and rate of spread with corresponding increases in 100-hr fuels. The largest increases in fuel loads are associated with cwd >3.0 inches. These increases will likely result in increased fire severity. These findings suggest that bark beetle outbreaks can cause significant increases in fuel loads, influence fire behavior, and perhaps increase the severity of the fires. These findings can have profound implications for managing dead trees resulting from extensive bark beetle outbreaks.

Historic Fire-Vegetation Relationships and the "Reference Conditions" Concept

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In the Interior West, historic stand densities and disturbance histories have been reconstructed to describe how vegetation has changed due to a century or more of livestock grazing, logging, and fire suppression. We provide reconstructions for the ponderosa pine / Jeffrey pine forests of Lassen National Forest, California, using tree ring-based analyses from multiple half-hectare plots. We show that pre-Euroamerican fire regimes and vegetation dynamics varied both spatially and temporally. Changes in fire intervals, season, and extent varied among sites in part from the destabilizing effects of climate. Tree establishment was often episodic and only somewhat consistent with changes in disturbance. These results suggest that reference conditions, as defined by either reconstructed vegetation structure or fire intervals, may be contingent on both the time period and location considered. In complex landscapes, such landscape heterogeneity may limit the application of the reference condition concept for fire and fuel reduction strategies.

Sustainability of Thinning and Prescribed Fire Programs to Improve Forest Condition Along the Front Range of Colorado

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Fire suppression initiatives of the 20th Century have contributed to unprecedented fuel loadings and subsequent wildfire severities. The fires of 2000 will long be remembered for their social, economic, and ecological impacts, as well as the $1.8 billion presidential initiative that resulted. While this initiative is well-intended, many information gaps exist relative to the acceptability of the expanded thinning and prescribed burning programs included under this initiative to improve forest condition in Colorado. The two primary objectives of the proposed project are to (1) conduct an interdisciplinary examination (hypothesis tests) of the sustainability of expanded thinning and burning programs to improve forest condition along the Front Range of Colorado and (2) conduct a series of collaborative workshops involving professional land managers and affected publics, based on knowledge gained from hypothesis tests. Four primary information sources (i.e., agency records, field sampling, survey, focus groups) will be used to test a series of hypotheses related to the social, economic, and ecological concerns associated with the proposed thinning and burning programs. The collaborative workshops will also be used to evaluate additional hypotheses. This project will attempt to fill the information voids that exist by developing social, economic, and ecological criteria that can be used to evaluate the sustainability of expanded thinning and burning programs. The collaborative workshops, to be held during the final stages of the study, will use knowledge gained during the study to involve professional land managers and affected publics. The project will provide information and criteria that could be used by county faculty to develop forest restoration programs that are socially, economically, and ecologically sustainable. The results will be relevant to all current and proposed programs that involve mechanical thinning and / or prescribed fire treatments.

Using RX Fire to Maintain Shaded Fuelbreak Areas in Southern Spain

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The use of broadcast burning as fuel treatments is not used in Southern Spain nowadays. This is especially due to the characteristics of the vegetation in these Mediterranean areas, such as an absence of fine fuels or existence of multiple strata. However, the use of prescribed fire could be an interesting alternative for the maintenance of shaded fuelbreak areas. With the aim of studying the viability of such fuel treatment, eight shaded fuelbreak areas were burned in Malaga province. All of them were *Pinus pinaster* stands, but with different understory structures and species, and using different firing techniques. The objectives were specially focused on comparing fuel consumption among the plots and defining the minimum base crown height to secure the survival of the trees as well as maintain the burn within prescription. The results will ease the use of RX-fire as a fuel treatment in forest fire prevention in Southern Spain.

Cerro Grande Rehabilitation Project

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Following the Cerro Grande Fire of May 2000, the Facility and Waste Operations at Los Alamos National Laboratory (LANL) received emergency funding to develop and support the Cerro Grande Rehabilitation Project. The project is responsible for landscape level mitigation of catastrophic wildfires on 10,000 acres of LANL property. A LANL moratorium on prescribed burning exists, so thinning utilizing site-specific defensible space, fuel break, and general thinning prescriptions are being used to reduce tree density and mitigate fuels. In general, prescriptions are based on prior piñon-juniper restoration and fuel mitigation research and on historical ponderosa pine density and expected fire behavior. Site-specific fuel mitigation implementation plans were developed that protect threatened and endangered species, the wildland urban interface, forest health and sustainability, streamside management zones, soil, air quality, cultural sites, utility corridors, and public safety. The goals of the site-specific implementation plans include the creation of defensible space and fuel breaks around structures, ecologically sound slash disposal and fuel reduction methods, utilization of equipment and performing operations at the appropriate time and place, and maintaining or improving aesthetic quality. Desired future conditions include a reduction of the risk of catastrophic wildfire and its associated effects on ecosystem health and the urban interface to a politically and socially acceptable level, improved and maintained forest health and wildlife habitat, and support of fuel mitigation efforts across the southwest. Management tools presently in development include building a post-treatment continuous forest inventory and a comprehensive modeling package to guide adaptive management and maintenance of desired future conditions. The project is in its second year and scheduled for completion in 2003.

Mapping Fuels and Fire Regimes Using Remote Sensing, Ecosystem Simulation, and Gradient Modeling

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Maps of fuels and fire regimes are essential for understanding relationships between fire and landscapes and for enlightened restoration of fire regimes to within historical ranges. We present an approach to fuels and fire regime mapping using gradient-based sampling, remote sensing, ecosystem simulation, and statistical modeling to create maps of fuels and fire regimes. We developed a database including 40 variables describing indirect, direct, resource, and functional gradients of physiography, spectral characteristics, weather, and biogeochemical cycles for a 5,830-km^2 area in Northwestern Montana. Using a variety of statistical techniques we created maps of fuel loads, fuel model, fire interval, and fire severity with independently assessed accuracies from 50 - 80 percent. Incorporating direct, resource, and functional gradients significantly improved map accuracy over maps based purely on indirect gradients. These maps provide information to assess the hazards and risks of fire when deciding how best to restore forests to within historical ranges of variation.

Role of Fire in the Future of White Pine Populations in Colorado

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Limber pine (*Pinus flexilis* James) and Rocky Mountain bristlecone pine (*Pinus aristata* Engelm.) are two white pine species that grow in Colorado. Limber pine is distributed throughout western North America while bristlecone pine's distribution is almost entirely within Colorado. Limber pine grows from lower treeline to upper treeline (1600 m to 3400 m) in Colorado while bristlecone pine is restricted to the subalpine and upper treeline areas (2750 m to 3670 m). Both species are seral on mesic sites and form persistent stands on xeric sites. White pine blister rust (*Cronartium ribicola*) was introduced to the Vancouver area of North America in the early 1900s and is spreading southward through the white pine ecosystems. White pine blister rust was identified in Colorado in 1998 on limber pine and has not yet been reported on bristlecone pine.

Fire may provide a management tool to aid in the coexistence of white pine populations with the rust. Both pines are pioneer species that depend on fire to open sites for regeneration. In the presence of blister rust, these colonization sites offer opportunity for selection of rust-resistant genotypes. The balanced application of fire to increase colonization opportunities while preserving seed sources may accelerate the development of rust-resistant white pine populations. The incomplete understanding of the ecology of limber and bristlecone pine constrain our ability to rapidly develop and implement management and conservation programs. An interdisciplinary research program applying ecological, physiological, and meta-population approaches has been initiated to improve our understanding of the regeneration dynamics, population genetics, rust resistance, and adaptive variation of high elevation white pines to facilitate our ability to develop potential management and conservation options.

Effects of Fall Vs. Summer Prescribed Burns in SW Ponderosa Pine Forests: A Case Study from Saguaro National Park, Arizona

Kathy Schon, National Park Service, Saguaro National Park, Tucson, AZ

Two years of fire effects monitoring data are analyzed to evaluate changes in ponderosa pine forest structure, composition, fuel loading, and vegetation characteristics of Saguaro National Park in southeast Arizona. In June of 1993 and 1996 two prescribed burns were conducted in the Rincon Mountains of Saguaro National Park. In October 1996, 1997, and 1998, three more prescribed burns were implemented in the same mountain range. All five burns were in ponderosa pine forests between 7000 and 8600 ft elevation. The burn units were ignited using strip-firing techniques. Minimal mechanical treatment was applied around the burn perimeters to secure the fire line. Twenty-five 50x20m fire effects monitoring plots were installed throughout the burn units to monitor trends and evaluate objectives. These plots were burned in the same manner as the rest of the burn unit and no mechanical treatment was applied in or within 50 m of each plot. Preliminary data have shown similar results for both summer and fall burns in reducing fuels and measurable differences in reducing pole sized trees and overstory tree mortality. Preliminary herbaceous data have shown an increase in herbaceous cover with fall burning and a reduction with summer burning.

Building the Scientific Basis for Fuel Treatments and Restoration in the Southwest

Martha Schumann and Tori Derr, Southwest Community Forestry Research Center, Forest Trust, Santa Fe, NM

We are undertaking a project to describe and compare fuel reduction and restoration treatments implemented in ponderosa pine forests in New Mexico, Arizona, and Colorado. The analysis will evaluate whether the prescription and outcome meet the treatment objectives. Preliminary data from sites show a wide range of site conditions, prescriptions, and treatments. This project will contribute to the development of appropriate prescription guidelines for the Southwest. We have also developed a protocol for community-based assessments of fuel reduction projects, composed of a framework of monitoring questions, and methods for data collection and analysis. Protocols will be piloted in two areas on the Cibola National Forest. The goal of assessing and monitoring treatments is to build a regional knowledge base shared between communities, scientists, and managers to inform forest management.

First Year Postfire Erosion Rates on the Bitterroot National Forest

Kevin M. Spigel, Department of Geography, University of Wisconsin-Madison, Madison, WI; and Peter R. Robichaud, USDA Forest Service, Rocky Mountain Research Station, Moscow, ID

Accelerated runoff and erosion are common occurrences following forest fires due to loss of protective forest floor material. The combustion of organic forest floor material may cause high percentages of bare soil exposed to overland flow, raindrop impact, and water repellent soil conditions. Twenty-four silt fences were installed to study postfire erosion on a storm-by-storm basis on the Bitterroot National Forest in Western Montana after the 2000 fire season. This study focused on erosion rates on steep slopes between two burn severities and two stand densities. Erosion rates were determined from sediment trapped behind the silt fences from a contributing area of 100 m^2. Sediment collected was weighed onsite and a sub-sample taken for further analysis. Short duration high intensity thunderstorms (10-min max. intensity 79 mm hr^{-1}) caused the highest erosion rates (80 t ha^{-1}). Whereas the long duration low intensity rain events produced little erosion (0.01 t ha^{-1}).

Fire and Fire Surrogate Treatments for Ecosystem Restoration

Scott Stephens, Division of Forest Science, Department of Environmental Science, Policy, and Management, University of California, Berkeley, CA; and James McIver, USDA Forest Service, Pacific Northwest Research Station, La Grande, OR

Compared to presettlement times, many dry 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 treatments, however, the appropriate balance among cuttings, mechanical fuel treatments, and prescribed fire is often unclear. Resource managers need better information on the comparative effects of alternative practices such as prescribed fire and mechanical "fire surrogates." An integrated national network of 13 long-term research sites has been established to address this need, with support from the U.S. Joint Fire Science Program and the National Fire Plan. Four alternative treatments will be applied in replication at each site: (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. Response to treatment will be determined through the repeated measurement of a comprehensive set of core variables at each site, including aspects of fire behavior and fuels, vegetation, wildlife, entomology, pathology, soils, and economics. The experiment is designed to facilitate inter-disciplinary analysis at the site level and meta-analysis for each discipline at the national level. The inter-disciplinary nature of the study will provide managers with information on how their practices affect whole ecosystems, while meta-analysis will provide insight on which responses are general, and which are dependent on specific environmental conditions. At present, two sites have collected post-treatment data, five are in the midst of treatment application, and the remaining six have collected all pre-treatment data and are poised to apply treatments.

Identifying the Most Appropriate Decision Framework for Assessing Fire and Fuel Management Options

David A. Tallmon, Danny C. Lee, Steve Norman, and Christine May, USDA Forest Service, Redwood Sciences Lab, Arcata, CA

Fire and fuels management decisions can be very complex. The complexity inherent in these decisions often comes from the ecological, social, and economic issues that surround them. We identify and describe a number of different decision processes or frameworks that are available to decision-makers to help them articulate and explore the possible effects and risks of management actions for competing issues. The decision processes vary in terms of their complexity, sophistication, and underlying philosophies. We evaluate several of these different decision processes using simple criteria and suggest which ones are most appropriate for each of the institutional levels at which fire and fuels management decisions must be made.

Fuel Load Changes Associated With Pinyon-Juniper Woodland Expansion in Central Nevada

Robin J. Tausch, USDA Forest Service, Rocky Mountain Research Station, Reno, NV; and Alicia Reiner, Department of Environmental and Resource Sciences, University of Nevada-Reno, Reno, NV

Since European settlement there has been a pronounced increase in distribution and dominance of Great Basin pinyon-juniper woodlands. Two-thirds to three-quarters are post-settlement in age. Largely attributed to the reduced occurrence of fire resulting from the heavy livestock grazing that followed settlement, this expansion is impacting a wide range of sagebrush ecosystems. Dramatic changes in the levels and types of fuels present, and increases in fire size and severity, are resulting. Fuel load changes responsible for these changes have not been quantified. Studies of the expansion and growth of woodlands at a central Nevada location provide a first look at the patterns of increasing fuel loads over time. At mid successional stages, fuel loads can be increased by 500 lbs. per acre per year. At full tree dominance, total fuel loads can be 10 or more times those of the former sagebrush ecosystem.

Using Prescribed Fire for Fuel Reduction and Ecological Restoration

Laura Trader and Sarah DeMay, Bandelier National Monument, Los Alamos, NM

Research shows that fire suppression has resulted in dramatic changes in the fire regimes at Bandelier National Monument, NM, and has produced significant ecological effects on the fire-prone landscapes. High accumulations of woody fuels and increased tree densities are a few of these effects. These unnatural conditions have resulted in catastrophic fire events in areas, such as Bandelier’s ponderosa pine forests, where they were once anomalous. In 1977, the stand-destroying La Mesa Fire consumed 16,000 acres in Bandelier’s ponderosa pine forest. This fire event has changed the landscape from a pine forest to a grass / shrubland. In 1998, 21 years post fire, Bandelier fire and resource managers decided to use fire in this area to reduce the accumulations of woody fuels, reduce seedling tree densities, and increase native plant cover. Our poster presents the 2-year post burn results from data collected on permanent vegetation plots in this area. It appears that prescribed fire has been effective in reducing woody fuels and promoting forest health.

Importance of Fire Occurrence in Simulation of Fuel Landscapes on the Angeles National Forest, Southern California

David R. Weise, M. Arbaugh, USDA Forest Service, Pacific Southwest Research Station, Forest Fire Laboratory, Riverside, CA; J. Chew, G. J. Jones, USDA Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory, Missoula, MT; M. Wiitala, USDA Forest Service, Pacific Southwest Research Station, Forest Fire Laboratory, Riverside, CA; J. Merzenich, USDA Forest Service, Portland, OR; J. van Wagtendonk, USGS, Western Ecological Research Center, Yosemite Field Station, El Portal, CA; M. Schaaf, Air Sciences, Inc, Portland, OR; S. Schilling, USDA Forest Service, Pacific Southwest Research Station, Forest Fire Laboratory, Riverside, CA; J. Sullivan, USDA Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory, Missoula, MT; and R. Kimberlin, USDA Forest Service, Pacific Southwest Research Station, Forest Fire Laboratory, Riverside, CA

Understanding the trade-offs between short-term and long-term consequences of fuel treatments on ecosystems is needed before a comprehensive fuels management program can be implemented nationally. We are evaluating three potential trade-off models at eight locations in major U.S. fuel types. The selected models / modeling approaches are (1) the Fire Effects Trade-off Model (FETM), (2) sequential use of the SIMPPLLE and MAGIS models, and (3) the Vegetation Dynamics Development Tool (VDDT). Each model simulates natural succession and vegetation changes following various disturbances including wildfire. We are evaluating the sensitivity of these models to various inputs including description of fire occurrence. Fire occurrence rates estimated from data for the Angeles National Forest were multiplied by 0.25, 0.5, 2, and 4. One hundred years of vegetation change were simulated by each model using the modified fire occurrence rates. The effects of change in fire occurrence rate on simulated wildfire acreage, vegetation composition, and smoke emissions will be presented.

Simulation of Historical Fires and Their Impact on Fuels in Yosemite National Park

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Understory Vegetation Response to Fire and Overstory Reduction in Black Hills Ponderosa Pine Stands

Cody L. Wienk, USDI National Park Service, Wind Cave National Park, Hot Springs, SD and School of Renewable Natural Resources, University of Arizona, Tucson, AZ; Guy R. McPherson, School of Renewable Natural Resources, University of Arizona, Tucson, AZ; and Carolyn Hull Sieg, USDA Forest Service, Rocky Mountain Research Station, Flagstaff, AZ

The role and extent of fire in ponderosa pine forests in the Black Hills, South Dakota, has been reduced in the last century. In some dense stands, a thick layer of pine needles has replaced understory vegetation. We experimentally addressed the effects of prescribed burning and overstory reduction on

understory vegetation, and examined the extent to which lack of a soil seed bank constrained understory recruitment. Response of understory vegetation during the first growing season post-treatments was sparse. There were, however, significant treatment effects during the second growing season. Total understory biomass ranged from 5.8 kg / ha on untreated plots to 1724 kg / ha on clearcut, unburned plots. Herbaceous dicots comprised over 90 percent of the total understory biomass. Only 57 individual plants, or 186 seeds / m^2, emerged from 1080 soil samples. Nonetheless, paucity of viable seeds in the soil seed bank does not appear to constrain recruitment of understory vegetation in dense ponderosa pine forests of South Dakota.

Equipment to Reduce Submerchantable Biomass in Anticipation of Future Prescribed Fire Treatments

Keith Windell, Missoula Technology & Development Center, Missoula, MT; and Sunni Bradshaw, USDA Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory, Missoula, MT

The USDA Forest Service's Technology and Development Center in Missoula, Montana, was asked to identify or develop equipment and techniques that would help managers prepare difficult sites for future prescribed fire treatments. These difficult sites have fuel loadings and distributions that increase the chance of fire escape and unacceptable resource damage. Much of the equipment identified during this project has direct application for fuels reduction in the wildland urban interface. The results of this project have been documented in two publications: Understory Biomass Reduction Methods and Equipment (0051-2828-MTDC), and Understory Biomass Reduction Methods and Equipment Catalog (0051-2826-MTDC). The reports document operational characteristics, capabilities, and limitations of equipment with potential to reduce biomass.

Restoring the Mixed-Oak Forest With Prescribed Fire and Thinning

Daniel Yaussy and Mary Boda, USDA Forest Service, Delaware, OH

The oak forest type is one of the largest endangered ecosystems. Central hardwoods ecosystems dominated by an oak overstory are seldom replaced with oak regeneration, regardless of harvesting method. This failure in oak development is the result of changed disturbance regimes, in particular, the suppression of human-caused fire since the 1930s. Studies of prescribed fires have established that fire alone will not remove the larger midstory trees that inhibit oak regeneration. As part of the national Fire and Fire Surrogate (FFS) study, four treatments have been implemented: control, fire only, thinning only, and fire plus thinning. In addition to the integrated ecosystem studies required by the FFS study protocols, collateral investigations include the effects of the treatments on: acorn production and predation by weevils and deer, bats, and flying squirrels; mycorrhizae; fire intensity; and soil moisture.

Modernizing Fire Weather Data Collection: The RAWS Review

John Zachariassen and Karl F. Zeller, USDA Forest Service Rocky Mountain Research Station, Fort Collins, CO; Thomas McClelland, USDA Forest Service, National Weather Program, Washington, DC; and Richard W. Fisher, USDA Forest Service Air Program, Washington, DC

The Remote Automated Weather Station (RAWS) network has been providing weather data to the United States Forest Service (USFS) and other land management agencies for the past 23 years. Prior to that, most fire weather observations were taken manually and a few made automatically with a RAWS prototype. Current USFS requirements for weather information have grown from strictly fire support – for which RAWS was originally designed and deployed – to significantly broader applications and uses. For example: physical and regional climatology, weather forecasting, support of air quality and pollutant monitoring in the atmospheric boundary layer, aerosol and trace gas flux measurements, environmental aerodynamics, and ecosystem (process) modeling. Even though current and future fire applications will continue to be the primary use for RAWS data, the growing list of uses for weather data presents an urgent need to assess the RAWS network. Such an assessment will include reviewing the individual stations themselves for density, distribution, redundancy, and overall function in addition to the RAWS system's ability to provide the Forest Service with the meteorological data and weather intelligence products it needs now and into the future. Preliminary results indicate that the RAWS network is a national asset and functioning fairly well given that it is a multi-agency network and with many user / owner choices for funding and operating individual stations. Individual stations meeting National Fire Danger Rating System standards are providing data in support of fire weather forecasting and for calculating fire danger rating indices and components. The entire network is in perpetual transition, undergoing hardware and software upgrades as well as streamlining data transmission.

Measuring, Modeling, and Tracking Western Spruce-Fir Forest Water Vapor Fluxes as a Fire Danger Indicator

Karl Zeller, USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO; Ned Nikolov, USGS, Fort Collins, CO; and John Zachariassen, USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO

Measurements of vertical water vapor flux made by eddy covariance were taken at 23-meter height above a subalpine spruce-fir forest in the USDA Forest Service's Glacier Lakes Ecosystem Experiment Site, Snowy Range, Wyoming.

Seasonal and diurnal water-vapor fluxes were studied. Seasonally during the growing season, water fluxes had much larger amplitudes. The FORFLUX biophysical model was used to analyze eddy covariance flux measurements and track water availability in different components of the ecosystem (e.g. soil, ground litter, and vegetation) on an hourly basis. Seasonal and inter-annual variations of ecosystem water fluxes and water content in vegetation and litter can provide indications for critical ecosystem drying and help improve assessments of potentially fire danger.

The FORFLUX model mechanistically couples all major ecosystem processes and predicts seasonal dynamics of water vapor, carbon dioxide (CO2), and ozone (O3) exchange between a terrestrial ecosystem and the atmosphere. FORFLUX consists of four interconnected modules: leaf photosynthesis; canopy flux; soil heat-, water- and CO2- transport; and snowpack. All biophysical interactions are computed hourly while model projections can be output for either hourly or daily time steps.

FORFLUX requires input for: climate, soils, vegetation structure, species physiology (20 parameters); hourly ambient temperature, relative humidity, incident short-wave radiation, precipitation, above-canopy wind speed; soil texture, depth, and volumetric rock content; and canopy leaf area index and foliage clumping factor. Output: net ecosystem water, carbon, and ozone fluxes and their components, such as vegetation net primary production (NPP), plant ozone uptake, canopy photosynthesis and stomatal conductance, woody respiration, soil CO2 efflux, canopy transpiration and rainfall interception, soil evaporation, snow melt and sublimation, surface runoff, and subsoil drainage.

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RMRS
ROCKY MOUNTAIN RESEARCH STATION

The Rocky Mountain Research Station develops scientific information and technology to improve management, protection, and use of the forests and rangelands. Research is designed to meet the needs of National Forest managers, Federal and State agencies, public and private organizations, academic institutions, industry, and individuals.

Studies accelerate solutions to problems involving ecosystems, range, forests, water, recreation, fire, resource inventory, land reclamation, community sustainability, forest engineering technology, multiple use economics, wildlife and fish habitat, and forest insects and diseases. Studies are conducted cooperatively, and applications may be found worldwide.

Research Locations

Flagstaff, Arizona	Reno, Nevada
Fort Collins, Colorado*	Albuquerque, New Mexico
Boise, Idaho	Rapid City, South Dakota
Moscow, Idaho	Logan, Utah
Bozeman, Montana	Ogden, Utah
Missoula, Montana	Provo, Utah
Lincoln, Nebraska	Laramie, Wyoming

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