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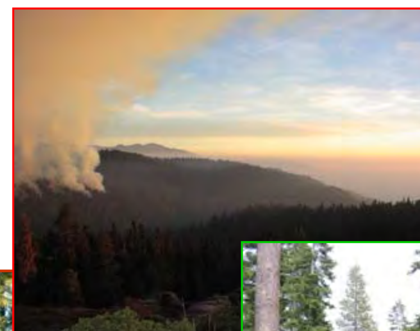
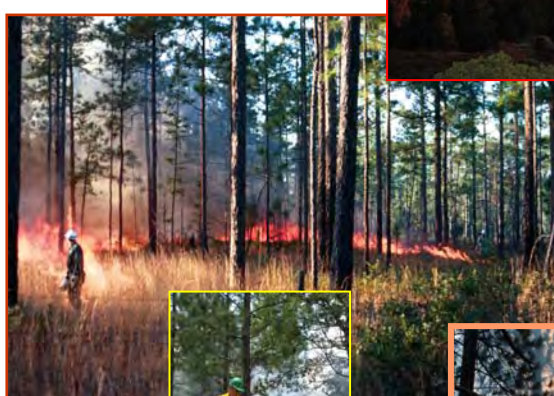
Restoring Fire-Adapted Ecosystems: Proceedings of the 2005 National Silviculture Workshop

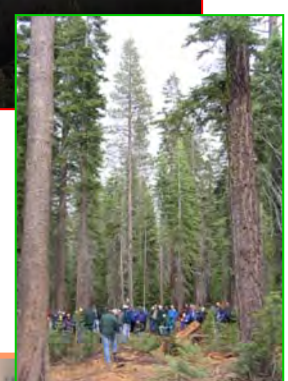
June 6-10, 2005 Tahoe City, California

Historic and Future Trends



Management Strategies





Silvicultural Options

Risks And Impacts

Abstract

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Many federal forests are at risk to catastrophic wild fire owing to past management practices and policies. Mangers of these forests face the immense challenge of making their forests resilient to wild fire, and the problem is complicated by the specter of climate change that may affect wild fire frequency and intensity. Some of the Nation's leading scientists and practitioner present approaches in tackling the problem.

Key words: wild fire, fuel management, thinning, climate change, fire history, resilience

Restoring Fire-Adapted Ecosystems: Proceedings of the 2005 National Silviculture Workshop

June 6-10, 2005 **Tahoe City, California**

Robert F. Powers, *Technical Editor*

Introduction

North America’s forests are magnificent. They include the world's tallest, oldest, and most massive trees and nearly every genus of conifers on Earth. Their history carries a legacy of massive ice that ground mountains to plains and sculpted river valleys, of great landforms that rose to channel air movement and moisture, of recurrent fire that shaped forest succession. These natural forces controlled where forests grew and where they are today. Human influence pales by comparison. Yet forest composition has changed in the span of a century. Except in the West, most of North America's aboriginal forest is gone, changed to young-growth stands, converted to other uses. Still, today’s forests are magnificent by any measure. They are among the most extensive and diverse in the world, and many forests have been protected administratively from further harvests. But the future of our forests depends on today’s management. And sound management centers on the art and science of silviculture.

From the perspective of human life spans, North American forests seem unchanging. But change is certain. Climate, seemingly immutable to our parents, is changing. And while the exact causes of climatic change remain arguable, evidence compels us to believe that the future will be different from the past and that we must be ready. Managers must develop strategies for coping with change. One expected change is the nature of wildfire. Our forests—particularly those of the West—are threatened. Each successive year seems marked by a rise in wildfire frequency, extent, and severity. Well-meant policies of decades of fire suppression plus shifts in forest management practices have led to changes in forest structure and diversity, physiological stress, and fuel accumulation. And a mantra is heard that our public forests should be managed toward conditions typifying pre-European settlement. But this is a vain hope akin to putting the genie back into the bottle, because our forests have a new complexion. Many of our forests are urbanized—some as traffic corridors, others as semimanaged interstices in a patchwork of community development. This has produced a mosaic of ownerships and a complexity of management challenges. Yet, as we fret with the bustle of everyday life, forests continue to grow. Change marches inexorably. The threat of catastrophic fire looms large.

Restoring fire-adapted or fire-resilient forested ecosystems in the specter of change and uncertainty demands intelligence and creativity. Among our most useful tools are models, both conceptual and mathematical, that project the likely outcomes of specific management strategies. But models are simplifications that are extrapolated to scales ranging from stand groups to landscapes, and those built on empirical data risk being nonfunctional if conditions in the future are different than in the past. How do we develop more flexible and reliable models? The hope for this rests on a better understanding of the principles that control processes. Not all processes are ecological;

some are administrative, and we need to sharpen our understanding of both if we are to adapt successfully to change. To this end, managers and researchers representing several disciplines assembled in 2005 at Granlibakken Resort, on the shore of Lake Tahoe. Speakers produced manuscripts, which, in turn, were subjected to a minimum of two technical reviews (one from a scientist, another from a practitioner), an independent statistical review, and a final review by the editor. Not all manuscripts met our standards for publication. Those that did are reported here.

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KEYNOTE ADDRESS: The Role of Silviculture in Restoring Fire-Adapted Ecosystems[1]

James K. Agee[2]

Abstract

Across the drier forests of the western United States, historical fire was a natural silvicultural process--thinning stands from below, cleaning surface fuels, and maintaining fire-resilient conditions. The 20th century fire exclusion policy, although initiated with the best of intentions, has been a disaster in dry forests, converting them to high-severity fire regimes. Restoring fire-safe forests will require the use of fire or silvicultural options that mimic fire to reduce surface fuels, reduce ladder fuels, and reduce crown density, while in the process retaining the largest, most wildfire-tolerant trees. Challenges include the lack of markets for small material, perceived environmental effects of large-scale operations, and the need to act within a global warming context.

Introduction

Forests across the West are in trouble. Wildfire area appears to be increasing year by year, and the severity of these fires appears to be outside of the historical range of variability. Yet the gamut of solutions ranges from "heavy harvest" to "do nothing". Only in limited instances do we seem to be able to develop consensus approaches for action. I'd like to provide a broader view of the problem and potential solutions, recognizing the importance of priorities and place when applying these principles.

How Did We Get Here?

The fire problems we now face have a complex history. They start with an attempt to forge a national forest management policy in the early 1900's, and the critical role that the large fires in Idaho and Montana in 1910 played in creating that policy (Pyne 2001). Foresters believed that European-style forest management could never be applied in America unless fire was controlled, and the fires of 1910 were the catalyst for a new fire exclusion policy. Up into the 1920's, there were voices of dissent, which advocated for the use of prescribed fire. The case for "light burning", as it was then called, was made primarily by industrial foresters, who were concerned that their old growth would be burned by intense fires due to fuel buildup before they could cut it. Their pleas for short-term conservation of fire in dry forests were rejected (Agee 1993), and a century of fire exclusion resulted.

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[1] A version of this paper was presented at the National Silviculture Workshop, June 6-10, 2005, Tahoe City, California.

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What happened? "Enormous areas are growing up in dense, even-aged stands… of reproduction." "Fire hazard has increased tremendously." "Fires, when they occur, are exceedingly hot and destructive…" These statements are as true in 2005 as they were in 1943, when they were penned by Harold Weaver (Weaver 1943) in the Journal of Forestry. In dry forests, this turned out to be the disaster predicted by Harold Weaver in the 40's. The high-canopied, low-fuel forests of the turn of the century morphed into fuel-choked forests that now burn with high severity--but not everywhere did this happen, and not everywhere was high-severity fire out of character--it was an ecology of place.

In places where severe fires have replaced those more benign, Smokey Bear has been blamed for being too effective. But fire prevention, as exemplified by Smokey, is still an important part of fire management, and is not the sole source of problems where they occur. Pick-and-pluck selective logging removed the most fire-tolerant trees from the forest, and where the forests were predominately large trees, the forest was functionally clearcut. Even low-intensity fires will have more severe effects when smaller trees, and trees of less fire-resistant species, replace large old ponderosa pines. Grazing removed many of the fine herbaceous fuels that carried pre-European fires. Over regional scale landscapes, the proportion of low-severity fire declined and the proportion of high-severity fire increased. In Forest Service regions 1 through 6, the area covered by historical fire regimes was about evenly divided between low, mixed, and high severity, but is now almost all mixed and high severity.

An Ecology of Place

High severity fires were always part of western landscapes, typically occurring in wetter coastal areas or in forests at high elevation. But they were uncommon in the drier forest types. We can provide a context for this variability using the concept of the historical fire regime (Agee 1993). High severity fire regimes historically had fire return intervals exceeding 100 years, and, when fires occurred, they tended to be mostly stand-replacement in character. Forest types included here would be subalpine fir, mountain hemlock, Pacific silver fir, and western hemlock (both the Douglas-fir and spruce types).

The fire ecology of western hemlock/Douglas-fir forests is described elsewhere, but fires were typically separated by centuries. After a fire, the growing space opened up for Douglas-fir allowed it to become a stand dominant in the next generation, and because it is long-lived, the stand dominant for hundreds of years. Without fire, the stand would eventually become dominated by western hemlock and western redcedar, but few stands ever reached this stage before another stand-replacement fire occurred. These fires were often weather-driven events, such that fuels were a secondary consideration in the definition of either fire size or severity (Agee 1997). Forests with historical high-severity fire regimes are a low priority for active management to reduce fire hazard. Fire risk is low--many of these stands have persisted for centuries with very high fuel loads, so that short-term mitigation of hazard is not justified except in limited circumstances: other catastrophic events that excessively increase dead fuels, or when adjacent to urban interface areas.

Historical mixed-severity fire regimes present a more intermediate situation. Drier Douglas-fir forests (southern Oregon/northern California), red fir, and grand fir/western larch forests fit into this category. Fire return intervals might range from

30-100 years, with intermediate-sized patches of varying severity. At a landscape scale, this provided diversity in both species composition and structure of these historical forests. This variability had significant effects on the ability of subsequent fires to spread, and helped to maintain this patchy character on the landscape. Fuels, topography, and weather interacted to affect both fire spread and severity. Fires in these forests might have started in July and burned into October, under a wide variety of weather patterns, in a wide variety of forest patches with different structures and fuels, and across topography where it burned upslope, downslope, at night, and during the day.

The case for active management in the mixed-severity fire regimes is easier to make than in the high-severity fire regimes. Fire risk is higher, and portions of these landscapes historically experienced low-severity fires. The case is weaker than in the low-severity fire regimes, where fire has been removed for many more “cycles”.

Low-severity fire regimes historically occurred in the warmer, drier forests where a substantial snow-free dry season existed. These forests, usually with some ponderosa pine or pine mixed with Douglas-fir, white fir, or grand fir, are found broadly across the western United States. Although some of the Colorado Front Range and South Dakota pine forests appear to fit into mixed-severity fire regimes, the Southwest, California, and Pacific Northwest pine forests appear to fit the classic low-severity fire regime pattern of frequent, low-intensity surface fires (Allen et al. 2002). It is these forests where the most dramatic shifts in fire severity have occurred.

Restoration of Firesafe Conditions

The principles of firesafe forests are clear (Agee et al. 2000, Agee 2002a, Brown et al. 2004): reduce surface fuels, reduce ladder fuels (those fuels that bridge the gap between surface fuels and overstory canopy fuels), keep the large trees, and reduce crown density. Also implied here is an order. At the end of treatment, the most important actions are also in the same order. Lowering surface fuels reduces the flame length of a potential wildfire. Removing ladder fuels reduces the probability that a surface fire will transition to a crown fire. Retaining large trees keeps the most fire-tolerant trees in the stand. Reducing crown density lowers the probability that an independent crown fire will occur.

We know that prescribed fire does a pretty good job of reducing surface fuels – those are the fuels that carry the fire, so by definition they have to decline after a burn. But like most resource management actions, prescribed fire can be applied in many forms, under different weather conditions, and as a heading, flanking, or backing fire. One thing that is often overlooked is that prescribed fire also creates fuels by killing live vegetation that is not consumed in the first fires. It can replace much of the original fuel load in five years, although usually resulting in a much higher height to live crown. Pile burning, chipping, mastication--in short, any treatment that removes or compacts surface fuels -- will reduce the surface fire flame length of a potential wildfire.

Increasing the height to live crown reduces ladder fuel contributions that might help a surface fire transition to a crown fire. The torching phenomenon is basically an interaction between the potential surface fire flame length, the moisture content of the understory foliage, and the height that the foliage occurs above the ground. Two of these three variables are under managerial control. By reducing surface fuels, potential surface fire flame length is reduced, and by increasing height to live crown by thinning understory trees, the required surface fire flame length to initiate crowning is increased. Prescribed fire can be effective in doing both if properly scheduled.

At the same time, it is important to keep the large trees, and conversely lower the density of the small ones. The efficacy of reducing the crown density depends largely on a tree removal process that does both: reducing crown density while keeping the large trees. It's also important to remember that as thinning intensity increases, there are tradeoffs with surface fire intensity caused by drier surface fuels and increased mid-flame wind speeds in the thinned stands. Often in the debates about active management, we hear, "Oh, we must thin the stand to save it!" But thinning comes in many forms, and only some forms will result in a firesafe forest condition. Consider three types of classic thinning. A low thinning removes trees from below: the smallest ones. A crown thinning takes a wider group of trees, and a selection thin is a thin from above: the largest ones first. These classic graphs suffer from the exclusion of a structural component found in most current mixed-conifer stands: an unmerchantable tree layer (*fig. 1*).

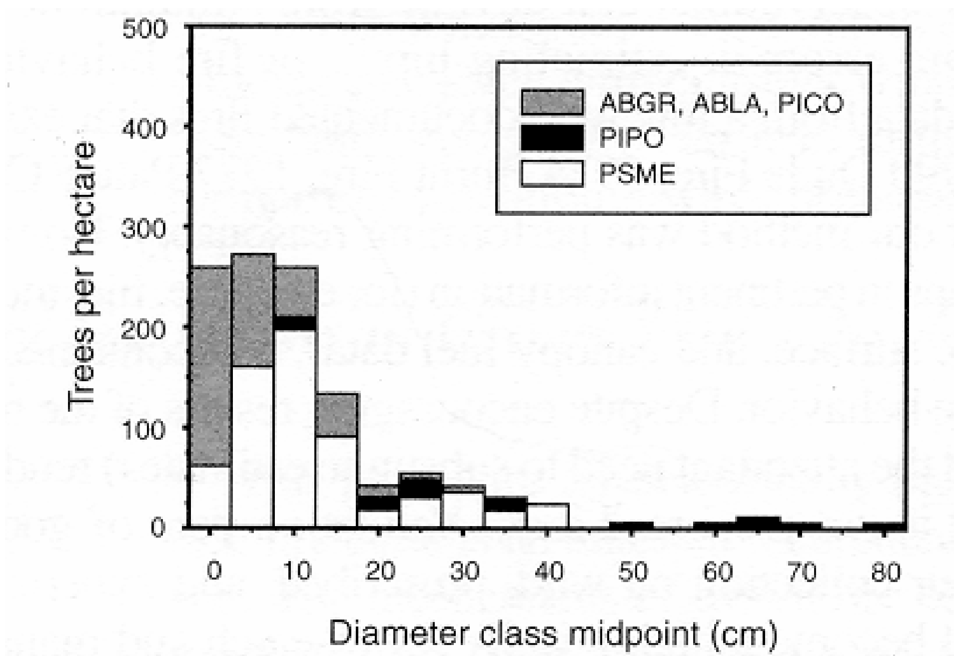

Figure 1—A typical dry forest size class distribution. Larger trees are mostly ponderosa pine (PIPO, *Pinus ponderosa*), intermediate sizes are dominated by more shade tolerant species like Douglas-fir (PSME, *Pseudotsuga menziesii*), and small size classes are other species (grand fir, ABGR, *Abies grandis*; subalpine fir, ABLA, *Abies lasiocarpa*; lodgepole pine, PICO, *Pinus contorta*). A majority of the trees are too small to be commercially viable. From Scott and Reinhardt (2001).

A simulation of the effect of various thinning and fuel treatment options was done using a stand much like this one (*fig. 2*). It has large trees (up to 100 cm [40 in] in diameter), but there are also a lot of small ones. It was assumed for this exercise that a commercial diameter limit was 15 cm (6 in). The simulation first applied a variety of alterations to the tree list in order to reduce basal area to a threshold of 15 m^2 ha^{-1} (60 ft^2 ac^{-1}). The simulated thinning treatments included no thin, low thin (start with smallest tree, increase tree size removed until basal area threshold is met), low thin-commercial limit (start with 15 cm tree, then increase as before until threshold is met), and selection thin (start with largest tree and move down in size until threshold is reached). The simulated fuel treatment options included no treatment, or a prescribed fire with a 0.6 m (2 ft) flame length to reduce post-treatment fuels. Then a worst-weather wildfire was simulated to burn across each stand, and survival was estimated using FOFEM (First Order Fire Effects Model [Reinhardt et al. 2002]). FOFEM essentially applies a flame length to the tree list and calculates mortality as a function of crown volume killed and bark thickness for each species/diameter class. Obviously, many other combinations could have been applied, and many other beginning stand structures could have been used. But some basic principles emerge from this analysis.

The treatments are arrayed from lowest to highest survival. The unmanaged stand (at left, *fig. 2*) suffers a stand replacement event. The surface fire flame length enables substantial torching in this stand. Equally severe was the selection thin with no fuel treatment, as the fire burned across increased surface fuels with increased fireline intensity, and only small trees were present. The first treatment with any residual survival were the low thin-commercial limit with no fuel treatment and the selection thin that had fuel treatment. In the former case, the unmerchantable understory, combined with additional surface fuels from the thinning, resulted in some torching and an intense surface fire, while in the latter case, survival was minimal because all the trees in the residual stand were small.

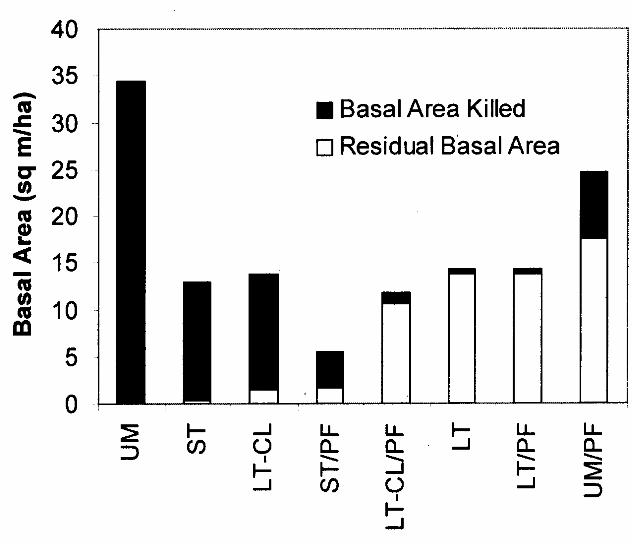

Figure 2—Survival from a simulated severe fire weather wildfire of the stand structures shown in fig. 1. Columns are organized by absolute amount of residual basal area (white part of column). UM= unmanaged, ST = selection thin, LT = low thin, CL = commercial limit (>15 cm), PF = prescribed fire. The unharvested stand was assumed to be NFFL (Northern Forest Fire Lab) fuel model 10, harvested stands with no prescribed fire were assumed to be NFFL Model 11, and any stand treated with prescribed fire was assumed to be NFFL model 9. Fuel moistures for 1-, 10-, and 100-hr fuels were 4, 5, and 6 pct for models 9 and 11 and 5, 6, and 7 pct for model 10. Open wind speed of 36 km hrP^{-1P} was adjusted to 0.4 for models 9 and 11 and 0.2 for model 10. From Agee and Skinner (2005).

The four options to the right had better survival. They all had either a low thinning (one with a commercial limit) or fuel treatment by prescribed fire. In this stand, the best result was obtained with the stand that was not thinned at all, where a prescribed fire had been applied that reduced surface fuels and raised the height to live crown. Of course, in the real world, such prescribed fires also create dead fuels, and those are not included in the simulation. Inclusion of those fuels would have shown the option of prescribed fire only to have been less effective.

This simulation involved a set of "worst-case" wildfire conditions that might vary across the West. The absolute outcomes of this set of simulations would be different if a wildfire with a different flame length had been applied using more or less severe weather (fuel moisture and wind speed). However, the relative order of treatment effectiveness would remain the same due to the deterministic nature of the fire behavior and fire effects programs used here.

The basic principles emerging from this analysis are that "no action" is a disaster, thinning from above is also a disaster as it removes the most fire-tolerant trees, and low thinning is the best thinning method (from the standpoint of creating firesafe forest structures). Prescribed fire shows up as being valuable, but in this simulation, the dead fuels it creates are not included. Treatments that reduced surface fuels, treated ladder fuels, and kept the large trees fared best.

Empirical Evidence for Firesafe Forests

The theory of firesafe forests derives primarily from research and empirical constants obtained from boreal (high latitude) forests. Experimental crown fires there have been studied for decades. Quantitative estimates of the relations between flame length and height to live crown, for example, or thresholds of mass flow rate (the quantity of crown fuel below which crown fire cannot operate) are largely derived from black spruce and jack pine forests. Over the last decade, evidence from the lower 48 states suggests that these principles also apply to western forests. Four examples illustrate how these firesafe principles have been successfully applied and mitigated wildfire damage.

1987 Hayfork Fires

These fires occurred during a massive outbreak of fires in northern California and southern Oregon. Weatherspoon and Skinner (1995) evaluated fire severity as evidenced by crown scorch visible on post-fire aerial photography. The forests burned in this study, mostly mixed-evergreen forests, were not specifically treated with firesafe principles in mind, but treated forests were classified as either cut-treated or cut-untreated. Cutting was largely selective overstory removal, so cut units were implied to have average tree size smaller than uncut units. Fuel treatment was either lop and scatter or patchy prescribed fire. Forests experiencing the least damage were uncut-untreated forests that had the largest trees. However, cut-treated forests did not significantly differ from uncut forests. Fire severity in cut-untreated forests was significantly higher.

Megram-Onion Fire, 2002

This fire in northwestern California burned largely in fuels created after a large wind-snap event in the winter of 1995-1996. The Forest Service created limited fuelbreaks in this Douglas-fir/white fir forest. In some fuelbreaks, surface fuels, ladder fuels, and crown density were reduced, while in others only the surface and ladder fuels were treated. From the air and the ground, the fuelbreak edge is obvious (*fig. 3*), and even though substantial crown density was left, the fuelbreak forest, although it burned, suffered only a low severity fire compared to the untreated area.

Figure 3—Area burned by the Megram Complex Fire in the vicinity of a fuelbreak. Untreated areas are upper left, and treated areas (surface and ladder fuels) are to the lower right. Untreated areas experienced high severity fire, while the fuelbreak survival was very high. Photo courtesy of USDA Forest Service.

Tyee Fire, 1994

A large Washington wildfire burned across ponderosa pine/Douglas-forest, and created huge patches of stand replacement fire. Areas where thinning and prescribed burning had been done fared much better than untreated areas, although scale of treatment was important. One area had trees less than 15 cm (6 inches) removed within three years of the fire, residual trees pruned, and surface fuels piled and prescribed burned. The crown fire approached this area, dropped to the ground within one tree length, and burned through as a surface fire, scorching about 50 percent of the crown volume and allowing a nearby residence to be saved. An older, nearby narrow fuelbreak also showed better survival than untreated areas outside. The fuelbreak was created in the 1970s, and the trees in this thinned area had grown such that their average diameter was about 50 percent greater than in adjacent unthinned areas. Again, a crown fire quickly transitioned to a surface fire upon encountering the fuelbreak, and then retransitioned to a crown fire on the far side of the fuelbreak.

Cone Fire, 2002

This fire entered Blacks Mountain Experimental Forest in northeastern California where thinning and burning experiments had been underway for several years. All treatments had been completed within five years of the wildfire. In areas thinned and burned, the wildfire would not even spread. A rapid transition in mortality occurred as one crossed into the boundary of treated units.

As a caution, the Hayman fire of Colorado (2002) must also be mentioned. Here, fuel treatment appeared to be effective under "normal" wildfire conditions, but treatments were not effective during exceptionally severe fire weather when the fire ran 29 km (18 miles) in one day. We appear to have good guidelines for stand-level treatment, and if the proper steps are taken, high-severity fire can be altered to low-severity fire under almost all conditions. Fuel does make a difference in low-severity fire regimes. Weather historically was responsible for the larger spread of these fires, but severity was fuel-related. This is still true. Several issues remain outstanding, though.

How much of a landscape need be treated? This depends on assumptions about what will be done when a wildfire occurs (Finney 2001). With aggressive fire suppression, probably 20-35 percent of a landscape will fragment fuels such that suppression can be effective. If an aggressive fire suppression response is unlikely, then untreated areas of the landscape will burn severely, and treated area will burn less severely. How much of the landscape do we want to place at risk? If suppression forces will concentrate on the wildland-urban interface, at the cost of more wildland area burned, then this argues for wider-scale treatment in the wildland.

How long are these treatments effective? The answer depends on what is meant by "effective". Historical research (e.g., Heyerdahl et al. 2001, Wright and Agee 2004) shows that historic fires in ponderosa pine forests often stopped at the boundaries of areas burned in the previous two years. After that, spread was likely to pass over into previously burned areas. So, the effectiveness from a spread perspective is probably five years or less. From a perspective of severity, which would appear to be a more relevant criterion, the effectiveness depends on how long ladder fuels and surface fuels remain low, and how they interact with the residual tree fire tolerance in the face of wildfire. In most cases, where the first fuel treatment was effective, the answer might be 10-20 years.

Many constraints on active management face today's land managers. Some segments of society so fear active management that they apply the precautionary principle to an extreme, ignoring the fact that in dry forests the "no action" option is itself a large risk. Species impacts, from large species, such as hawks or owls, to small organisms, like mollusks and lichens, often constrain even "light on the land" management. Soil impacts from any harvest, and effects of possible roadbuilding, limit the ability to treat large areas with thinning. Some harvest techniques, like helicopter, have minimal soil impacts, but require leaving much more surface fuel (tops, etc.) in the woods. The biggest constraint for wildlands is the focus on the urban interface, as if there were no values at all to protect away from the interface. That problem will only grow larger with time.

If we ever get serious about global climate change and carbon balances, more regulation of prescribed fire from strictly a carbon balance perspective is likely. But how will that affect the tradeoff between wildfire carbon emitted and that emitted by practices like prescribed fire intended to reduce wildfire carbon emissions? All of the issues involve risk management, and we do a very poor job of placing the choices for managers in a policy context. Current climate projections suggest that across the West, we are likely to experience warming temperatures and lower annual precipitation, concentrated in winter months (Lenihan et al. 2003). Fire seasons might be longer. Silviculture to restore firesafe forests can only become more important in the future.

Healthy Forests Restoration Act (HFRA) of 2003

I'd like to close by moving a bit closer to a policy issue at our doorstep: the Healthy Forests Restoration Act (HFRA) of 2003. In my view, the Act provides the appropriate technical policy guidance for "doing the right thing" in our drier forests. It is generally limited to drier forests (with some reasonable exceptions), it has an area limit, it directs a focus on small diameter trees, and allows both thinning and prescribed fire. But the effectiveness of HFRA remains to be demonstrated. If the agencies and their administrators choose to follow the intent of the HFRA, I think it will improve forest health across the West. It will engender trust on part of interest groups and help to move the restoration process forward. But if it used only to justify allowable cut quotas, it will fail both in the public eye and only exacerbate the fire problems we face today.

In our dry forest types, our 20th century choice, for better or worse, turned the friendly flame into a demon. As a society, we have difficult choices about how to correct this policy nightmare that covers millions upon millions of acres across the West. There are risks of action and no action, and people on both sides who want to do the right thing, and the wrong thing. If the larger, broader society has a better understanding of what the "right thing" is, we will take back the forests from the advocates of the extremes, and allow fire and its silvicultural surrogates to play ecologically appropriate roles in forest restoration.

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Forest Changes Since Euro-American Settlement and Ecosystem Restoration in the Lake Tahoe Basin, USA[1]

Alan H. Taylor[2]

Abstract

Pre Euro-American settlement forest structure and fire regimes for Jeffrey pine-white fir, red fir-western white pine, and lodgepole pine forests were quantified using stumps from trees cut in the 19th century to establish a baseline reference for ecosystem management in the Lake Tahoe Basin. Contemporary forests varied in different ways compared to the presettlement reference. Contemporary Jeffrey pine-white fir forests have more and smaller trees, more basal area, less structural variability, and trees with a more clumped spatial distribution than presettlement forests. The mean presettlement fire return interval for the period 1450-1850 for Jeffrey pine-white fir forests was 11.5 years, and most fires (>90 percent) burned in the dormant season, while no fire was recorded in the study area after 1871. Differences in the structural characteristics of contemporary and presettlement red fir-western white pine and lodgepole pine forests were similar to those for Jeffrey pine-white fir forests. However, 19th century logging changed the composition of red fir-western white pine forests, and these forests now have more lodgepole pine than red fir or western white pine. Comparison of contemporary Jeffrey pine-white fir forests with the presettlement reference suggest that restoration treatments should include: (1) density and basal area reduction, primarily of smaller diameter trees, (2) reintroduction of frequent fire as a key regulating disturbance process, and (3) increasing structural heterogeneity by shifting clumped tree distributions to a more random pattern. Restoration treatments in red fir-western white pine forests should include: (1) a shift in species composition by a density and basal area reduction of lodgepole pine, and (2) increasing structural heterogeneity by shifting tree distributions to a more random pattern. In lodgepole pine forests, the restoration emphasis should be: (1) a density and basal area reduction of small diameter trees, and (2) an increase in structural heterogeneity that shifts tree spatial patterns from clumped to a more random distribution. Re-introduction of fire as a regulating process into high elevation red fir-western white pine and lodgepole pine forests may be viewed as a long-term restoration goal.

Introduction

The concepts of reference conditions and the range of natural variability are central to forest management practices being developed under the rubric of ecosystem management (Agee and Johnson 1988, Grumbine 1994, Kaufmann et al. 1994, Overbay 1992). Reference conditions are usually considered the range of historical variability in forest structures and processes at the time of European settlement (Morgan et al. 1994, Swanson et al. 1994). Perspectives that emphasize the management of forests within their historical range of variability evolved from an

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understanding of what presettlement conditions were, and how and why contemporary conditions deviate from them (Covington et al. 1997, Kaufmann et al. 1994, Landres et al. 1999, Morgan et al. 1994, Swanson et al. 1994). Presumably, management for presettlement conditions will maintain important evolutionary and functional linkages between species and reduce the risk of unexpected ecological outcomes such as species extinction (Landres et al. 1999, Moore et al. 1999, Swanson et al. 1994). Identification of reference conditions is therefore an essential step in implementing ecosystem management. Moreover, reference conditions represent a framework for evaluating current ecosystem structures and processes and for designing restoration treatments to change current conditions if they fall outside their historic range of variability (Covington and Moore 1992, Fule et al. 1997, Grumbine 1994, Kaufmann et al. 1994, Morgan et al. 1994, Swanson et al. 1994). In cases where contemporary forest conditions are outside their range of historic variability, the presettlement reference can also be used to identify restoration goals and to develop restoration treatments (e.g., Fule et al. 1997, Moore et al. 1999, Morgan et al. 1994, Swetnam et al. 1999, White and Walker 1997).

In the Sierra Nevada, forests have been dramatically altered by Euro-American land use practices. Forest lands have been logged, grazed, and burned since the mid to late 19th century (McKelvey and Johnston 1992, SNEP 1996, Vankat and Major 1978). Consequently, resource managers need information on presettlement forest conditions where much of the evidence of the forest was removed by 19th or 20th century logging. In the Lake Tahoe Basin, forests were logged soon after initial Euro-American settlement (hereafter presettlement) (Leiberg 1902). Small scale logging to supply timber for local use began as early as 1861 and near clearcut logging occurred between 1873 and 1900 (*fig. 1*) (Lindström 2000, Strong 1984). Contemporary forests became established after logging.

In this paper, reference conditions from well preserved cut stumps on early cut-over lands were used to identify and compare historical and contemporary forest characteristics in the Lake Tahoe Basin as a basis for forest ecosystem management. The specific questions addressed in this study were: (1) What was the historical range of variability in forest structure, i.e., species composition, size structure, spatial pattern, on the east shore of Lake Tahoe, and how is contemporary forest structure different? (2) How did presettlement fire regimes vary spatially and temporally? (3) How can this information be used to guide restoration of forests in the Lake Tahoe Basin towards a presettlement condition?

Study Area

Forests were sampled in a 2900 ha area on the west slope of the Carson Range on the east shore of Lake Tahoe (*fig. 1*). Elevations range from 1910-2666m. The topography of the Carson Range is steep and complex, and variability in elevation and topography exert strong control on the distribution of forest types in the study area. Three forest cover types occur in the area: (1) Jeffrey pine (*Pinus jeffreyi*) forests occur at lower elevations, (2) red fir (*Abies magnifica*) -western-white pine (*P. monticola*) forests occur at mid-elevation, and (3) lodgepole pine (*P. contorta* var. *murrayana*) occupy at high elevations or wet sites within the red fir zone. Climate in the Lake Tahoe Basin is characterized by cold-wet winters and warm-dry summers. Mean monthly temperatures at South Lake Tahoe (1820m) range from -1°C in January to 18°C in July and mean annual precipitation is 78.4 cm. Most precipitation

(86 percent) falls as snow between November and April. The terrain is steep and complex and forests grow on shallow (< 1m), excessively drained, medium acidity soils derived from Mesozoic aged granite (Hill 1975, Rogers 1974). Stands sampled in this study were located where the original forests were nearly clearcut during the Comstock mining era, and where stumps were well preserved. Sampled sites included the observed variability in the density and size-class distributions of well preserved stumps and live trees in each forest type. A total of 20 stands (11 Jeffrey pine, six red fir-western white pine, three lodgepole pine) were sampled in this study.

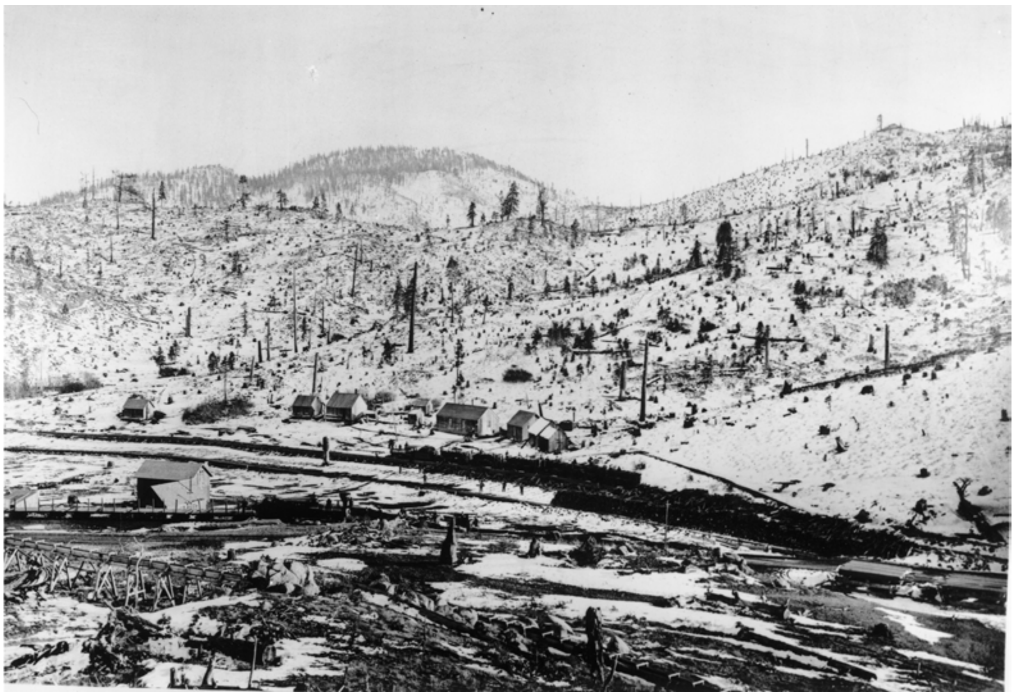

Figure 1—Forest conditions in the study area in 1876 at Spooner Summit in the Carson Range, on the east shore of Lake Tahoe, Nevada (C.E. Watkins). Note the near clearcut logging of the forest and the presence of cut stumps that served as the basis for the reconstruction of presettlement forest conditions in the Carson Range, Lake Tahoe Basin, Nevada.

Methods

Forest Structure and Composition

All forest stands were sampled using 50m X 100m (0.5 ha) plots, and the location (UTM coordinates with a GPS), slope aspect, slope pitch, and elevation of each plot were recorded. All stumps and live trees (stems ≥10 cm in diameter at breast or stump height) within a plot were mapped to the nearest 0.3m by establishing a 10m X 10m grid and then measuring the coordinates (x,y) of each tree from the origin (0,0) of each cell using a metric measuring tape. The species and diameter of each stump (stemwood) or live tree (outside bark at dbh) were then recorded. Bark thickness was added to stump stemwood diameter using bark thickness estimates from inside-outside bark diameter regression equations developed for ponderosa pine (*P. ponderosa*), sugar pine (*P. lambertiana*), and red and white fir (i.e., Dolph 1989, Larsen and Hann 1985, Walters and Hann 1986a). Lodgepole pine has thin bark (<1 cm) (Agee 1993) so bark thickness was not added to lodgepole pine stump diameters.

Stand Age Structure

The age of the post-logging cohort was estimated by coring 9 to 26 of the largest diameter contemporary trees 30 cm above the ground surface in each plot. Cores were sanded to a high polish, their growth rings were cross-dated (Stokes and Smiley 1968), and tree age was assigned based on the calendar year of the inner most ring. The ages of potential presettlement trees, i.e. established prior to 1850, were also determined in each plot. Presettlement trees were distinguished from the post-logging cohort by their height, crown form, large diameter, bark structure and thickness, and highly clumped spatial pattern. All potential presettlement trees were cored, cores were sanded and cross-dated, and their annual growth rings were measured to the nearest 0.01 mm. Total radial growth since the logging date was then subtracted from tree diameter to determine stem diameter on the date the stand was logged. All presettlement stems ≥10 cm on the date of logging were included in presettlement forest reference estimates. The date of logging in each plot was estimated by identifying the dates of sudden increases in radial growth in surviving presettlement trees. Growth releases were identified visually from graphs of a standardized growth index (actual width/mean width) (Veblen et al. 1991), derived from the radial growth measurements of presettlement trees in each plot.

Spatial Patterns

The type, scale, and intensity of spatial patterns of trees in the presettlement and contemporary forest were identified and compared using Ripley's K(t) function (Ripley 1977). Ripley's statistic examines the number of stems around each stem in concentric circles of a given radius (Duncan 1991, Kengel et al. 1997). The number of stems, occurring within a circle of radius t, is compared with the expected number of stems based on a Monte Carlo simulation of a randomly (Poisson) distributed point pattern. This pattern permits detection of significant ($P<0.05$) aggregation or hyper-dispersion in a population. The intensity of pattern is indicated by the magnitude of K(t), and the scale of pattern is determined by the radius of the distance class, which in this case was 3m.

Fire History

Fire occurrence in presettlement Jeffrey pine forests was identified for 13 sites, of about 10 ha each, from fire scars recorded in the annual growth rings of stumps (n=39) and recently dead standing presettlement trees (n=2). Samples were removed from the stumps and live trees with a chainsaw and the tree rings in each sample were cross-dated using standard dendrochronological procedures (Stokes and Smiley 1968). The calendar year of each tree ring with a fire scar lesion in it was then recorded as the fire date. Presettlement fire histories in red fir-western white pine and lodgepole pine forests could not be reconstructed from cut stumps. Stumps with fire scars in them shattered when samples were extracted with a chainsaw. The season of burn for a fire was determined from the position of the fire scar lesion within the annual growth ring, following Baisan and Swetnam (1990): (1) early (first 1/3 of earlywood), (2) middle (second 1/3 of earlywood), (3) late (last 1/3 of earlywood), (4) latewood (in latewood), (5) dormant (at ring boundary). In this area of strongly seasonal precipitation (winter wet, summer dry), dormant fires represent fires that burned in late summer or fall after radial growth ceased for the year, not fires that burned in winter or early spring (Caprio and Swetnam 1995).

Results

Dates of Logging and Forest Age-Structure

Logging dates were determined for 17 of the 20 plots. Four plots were cut in the 1870s, ten were cut in the 1880s, two were cut in the 1890s, and one was cut in 1905. Based on the dates of logging in adjacent plots, the other three plots were probably logged in the 1870s (n=1) and 1890s (n=2).

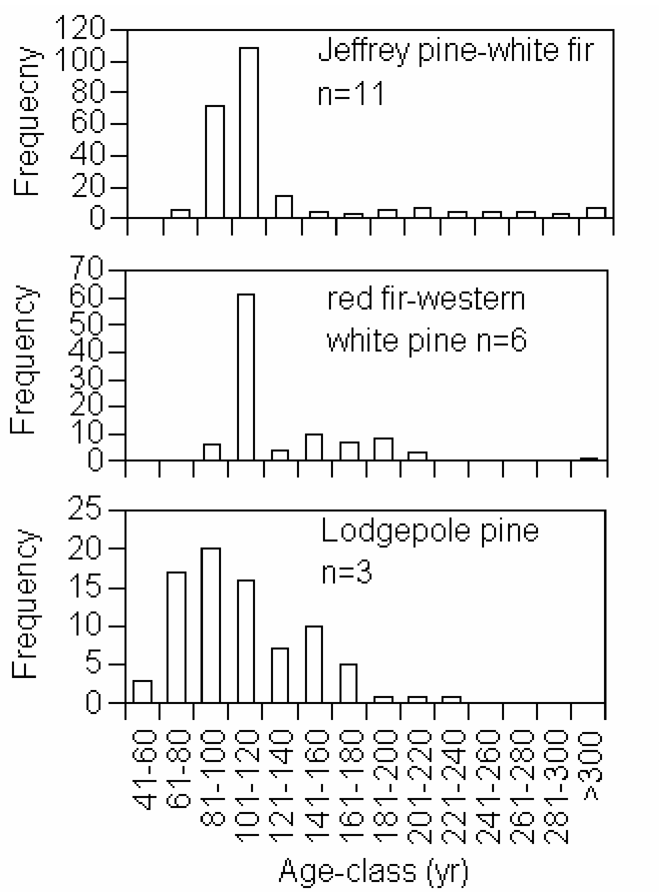

Figure 2—Frequency distribution of aged trees in contemporary forests in 20-yr age-classes, n is the number of plots.

Tree ages in the plots were similar in all forest types. Most trees were 80-120 years old and became established soon after logging (*fig. 2*). There were surviving presettlement trees in the plots but they were not abundant. On average, Jeffrey pine-white fir plots had 9.2 (range 0-44), red fir-western white pine plots had 10.4 (range 0-16), and lodgepole pine plots had 16.6 (range 0-26) trees ha^{-1} >120 years-old.

Forest Structure and Composition

Jeffrey Pine-White Fir Forests

The characteristics of presettlement Jeffrey pine-white fir forests were different than those of contemporary forests (*table 1*, *fig. 3a*). Contemporary forests were more dense, they had greater basal area, and they had different shaped size-class distributions than presettlement forests. There were few presettlement trees <40 cm dsh and contemporary Jeffrey pine, white fir, and red fir trees were smaller in diameter than presettlement trees (*table 1, fig. 3a*).

The spatial patterns of trees were also different in the presettlement and contemporary forest (*table 2*, *fig. 4a*). Presettlement trees >10 cm dsh and >40 cm dsh were clumped at small scales (≤ 9m) in <50 percent of the plots, and they were randomly distributed at larger scales. Large presettlement trees (>40 cm dsh) in a few

Table 1—*Structural characteristics of presettlement and contemporary forest stands on the east shore of Lake Tahoe, Carson Range, Nevada. Density and basal area estimates are for stems >10 cm in diameter.*

	Presettlement				**Contemporary**			
Forest type	**Mean**	**Std. Dev.**	**Min.**	**Max.**	**Mean**	**Std. Dev.**	**Min.**	**Max.**
Jeffrey pine-white fir								
Density (trees ha^{-1})								
Jeffrey pine	55	9.7	26	90	297	171.5	132	758
White fir	13	9.7	0	32	38	21.6	8	78
Red fir	1	2.8	0	10	8	19.1	0	68
All	68	22.2	30	114	343	178.7	172	794
Basal area (m^2 ha^{-1})								
Jeffrey pine	19.4	5.3	11.6	29.3	38.9	6.6	23.4	48.1
White fir	5.7	4.1	0	12	5.1	2.8	0.4	11
Red fir	0.4	1	0	2.6	2.4	5.3	0	18.2
All*	25.5	8.1	12.6	38.1	46.4	6.3	28.4	58.7
Diameter (cm)								
Jeffrey pine	68	7.8	54	85.6	38.7	8.5	28.6	52.5
White fir	76.3	27.9	54.8	113	45.4	14.1	25.3	66.4
Red fir	75.5	78.9	56.2	97.2	43.7	12.8	39.2	58.2
All	67.5	8.1	54.7	85.3	39.4	8.8	27.8	52.7
Red fir-western white pine								
Density (trees ha^{-1})								
Red fir	94	32.1	68	142	184	142	14	328
Western white pine	53	17.9	22	74	71	52.4	14	146
Lodgepole pine	14	23	0	58	274	188.8	0	484
White fir					1	1.7	0	4
Jeffrey pine	1	0.7	0	4	3	2.8	0	6
All	162	33.1	118	208	538	259.1	214	842

Basal area (m^2 ha^{-1})								
Red fir	40	9.3	27	53	24	16.9	5.7	50
Western white pine	15.5	6.4	5.8	22.2	6.5	5.1	0.8	13.3
Lodgepole pine	0.3	3.2	0	8.2	17.9	10.5	0	31.6
White fir					<0.1	0.1	0	0.2
Jeffrey pine					0.1	0.1	0	0.3
All	55.8	9.3	40.9	67.8	48.5	15.4	31.7	71.4
Diameter (cm)								
Red fir	73.5	8.1	62.3	80.8	42.1	10.3	31.5	60.7
Western white pine	63.9	9.8	47.3	80.3	32.1	8.3	21.8	41.5
Lodgepole pine	33.8	10.3	27.3	41.5	28.3	5.1	23.2	31.4
White fir					27.6			
Jeffrey pine					20	7.6	16.8	23.1
All	64.9	7.1	56.6	75.1	33.1	5.6	26.7	39.8
Lodgepole pine								
Density (trees ha^{-1})								
Lodgepole pine	171	74	90	234	583	334	202	850
Red fir	12	17.3	0	32	27	3.5	0	76
Western white pine					4	6.9	0	12
White fir					1	1.2	0	2
Jeffrey pine	3	4.7	0	8	2	2.1	0	4
All	186	85.7	98	266	617	366	204	860
Basal area (m^2 ha^{-1})								
Lodgepole pine	55.6	32	29.7	91.4	40.3	14.5	26	55.1
Red fir	1.5	1.4	0	2.3	6.4	11.1	0	19
Western white pine	2.6	4.5	0	7.9	0.3	0.9	0	0.6
White fir					0.4	0.5	0	1
Jeffrey pine					0.1	0.2	0	0.2
All	59.7	87.6	37.6	93.5	47.8	18.9	26.1	59.6
Diameter (cm)								
Lodgepole pine	62.4	7.3	54.5	69.2	29.4	6.6	25.4	36.9
Red fir	59.5	44.2	33.9	85	42.9	17	33.2	52.5
Western white pine	107.3				41.4	21.7	28.8	54
White fir								
Jeffrey pine								
All	62.4	6.4	55	66.4	30.4	5.7	26.1	36.9

plots were clumped at intermediate scales (12-15m). Contemporary trees >10 cm dbh and <40 cm dbh in 90 percent of the plots, in contrast, had clumped distributions at all scales and large (>40 cm dbh) contemporary trees were clumped at all scales in about half the plots.

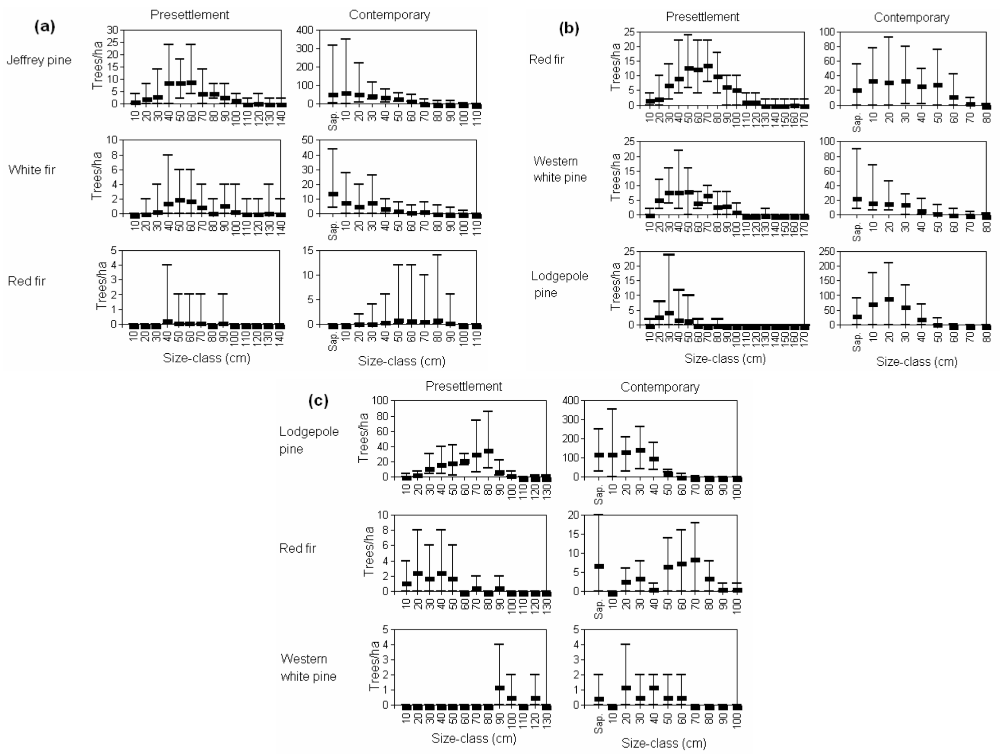

Figure 3—Average density and range of trees (stems ≥10 cm in diameter) and trees and saplings (stems >1.4m tall and <10 cm in diameter) in presettlement and contemporary (a) Jeffrey pine-white fir forest plots (n=11), (b) red fir-western white pine (n=6), and (c) lodgepole pine forest (n=3) forest plots. Note that the scale of the y-axis is not the same on the graphs. Not shown for contemporary forests of red fir-western white pine are Jeffrey pine in the Sapling (mean=0.7, range0-2), 10 cm (mean=1.3, range 0-4) and 30 cm (mean=0.7, range 0-2) size-classes, and white fir in the 20 cm (mean=0.3, range 0-2) and 30 cm (mean=0.3, range 0-2) size-classes. Not shown for contemporary lodgepole pine forests are Jeffrey pine in the 40 cm (mean=0.7, range 0-2) size-class, and white fir in the 30 cm (mean=1.0, range 0-2) size-class.

Red Fir-Western White Pine Forests

The characteristics of presettlement red fir-western white pine forests were different than those of contemporary forests *(table 1*). Presettlement forests were less dense than contemporary forests but basal areas were similar. The composition of contemporary and presettlement forests was quite different. Presettlement forests were mainly red fir and western white pine, while lodgepole pine was present in only two plots. Contemporary forests, on the other hand, have more lodgepole pine than red fir or western white pine, and >50% of the contemporary trees are lodgepole pine. The size-class distributions of the contemporary and presettlement forest were also

Table 2—*Frequency of plots with a clumped distribution (Ripley's K(t), P<0.05) at 3m distance steps for three diameter classes (>10 cm, <40 cm, >40 cm) of stems in presettlement and contemporary forests. Only populations with >13 individuals were analyzed and those that were too small are indicated by a dash (-); n=number of plots.*

Forest type	**Distance (m)**								
	n	3	6	9	12	15	18	21	24
Jeffrey pine-white fir									
Presettlement									
>10 cm	11	4	5	3	1	1	0	0	0
<40 cm	11	-	-	-	-	-	-	-	-
>40 cm	11	5	4	4	3	3	1	0	0
Contemporary									
>10 cm	11	11	11	11	11	11	11	11	11
<40 cm	11	11	11	11	11	11	11	10	10
>40 cm	11	3	6	5	5	5	7	5	5
Red fir-western white pine									
Presettlement									
>10 cm	6	5	4	2	0	1	1	1	1
<40 cm	6	2	1	1	1	1	1	0	0
>40 cm	6	4	2	1	1	0	0	0	0
Contemporary									
>10 cm	6	6	6	0	5	5	6	5	4
<40 cm	6	5	5	5	5	4	4	4	4
>40 cm	6	5	3	3	2	2	2	0	0
Lodgepole pine									
Presettlement									
>10 cm	3	1	3	2	2	2	1	1	1
<40 cm	3	0	0	0	0	0	0	0	0
>40 cm	3	1	1	2	1	1	1	1	1
Contemporary									
>10 cm	3	3	3	3	3	3	2	2	2
<40 cm	3	3	3	3	3	3	3	2	2
>40 cm	3	1	1	0	0	0	0	0	0

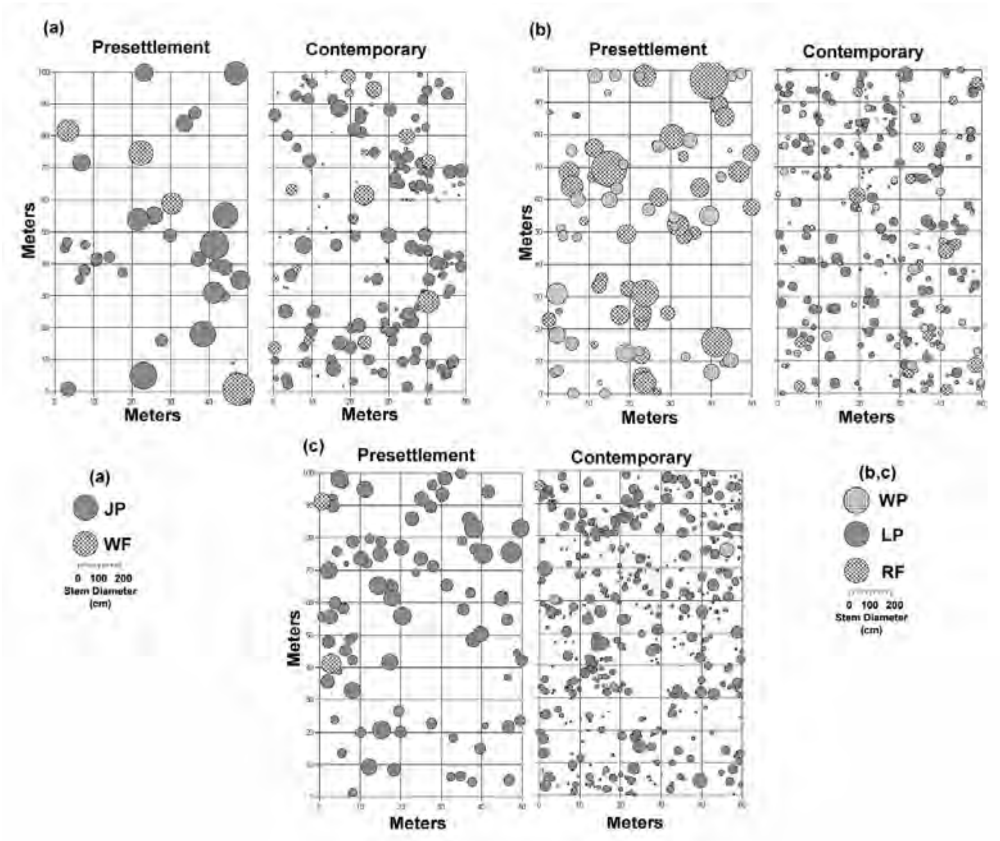

Figure 4—Stem maps of presettlement and contemporary (a) Jeffrey pine-white fir, (b) red fir-western white pine, and (c) lodgepole pine forest in the Carson Range, Lake Tahoe Basin, Nevada. Each pair illustrates stand structure in the same 0.5 ha plot. The plots shown for each forest type had the median presettlement stem density. Species acronyms are: JP-Jeffrey pine, LP- lodgepole pine, RF-red fir, WF-white fir, WP-western white pine.

different and presettlement red fir, western white pine, and lodgepole pine trees were larger than contemporary trees *(table 1*, *fig. 3b*). There were few presettlement trees <30 cm dsh.

The spatial patterns of trees were also different (*table 2*, *fig. 4a*). Presettlement trees >10 cm dsh and >40 cm dsh were most frequently clumped at small scales (<9m), and randomly distributed at larger scales (*table 2*, *fig. 4b*). Small and intermediate sized trees (<40 cm dsh) were usually randomly distributed at all scales. In most contemporary forest plots, all but the largest trees (>40 cm dbh) had a clumped distribution at all scales. Large contemporary trees (>40 cm dbh), in contrast, were most frequently clumped at the smallest scales (3-9m), and they were randomly distributed at larger scales (*table 2*).

Lodgepole Pine Forests

Presettlement lodgepole pine forests were less dense than the contemporary forests but their basal areas were similar (*table 1*). Presettlement trees were also larger than those in the contemporary forest, and they were present in a wider range of size-classes (*fig. 3c*). There were few presettlement stems <30 cm dsh in any of the plots. Spatial patterns of presettlement lodgepole pine trees varied by size class but they were different than for contemporary trees. Presettlement trees >10 cm and >40 cm dsh were clumped at all scales in some plots, and trees <40 cm dsh were randomly distributed at all scales. In contemporary forests, trees >10 cm and trees <40 cm were usually clumped at all scales, but large trees (>40 cm dbh) were mainly randomly distributed (*table 2*, *fig. 4c*).

Fire History in Jeffrey Pine-White Fir Forests

A long history of fire was preserved in Jeffrey pine stumps. One hundred and fifty-six fires were recorded as fire scars in the stumps. The fire record spanned the period AD 1160 to 1871, but only one site recorded fires before 1400. Four sites recorded fires by 1450, so the period 1450 to 1850 was selected as the period for the presettlement period fire disturbance analysis (*fig. 5*). Sample depths >10 percent are generally adequate to analyze temporal variation in fire occurrence in short fire return interval ecosystems (Caprio and Swetnam 1995).

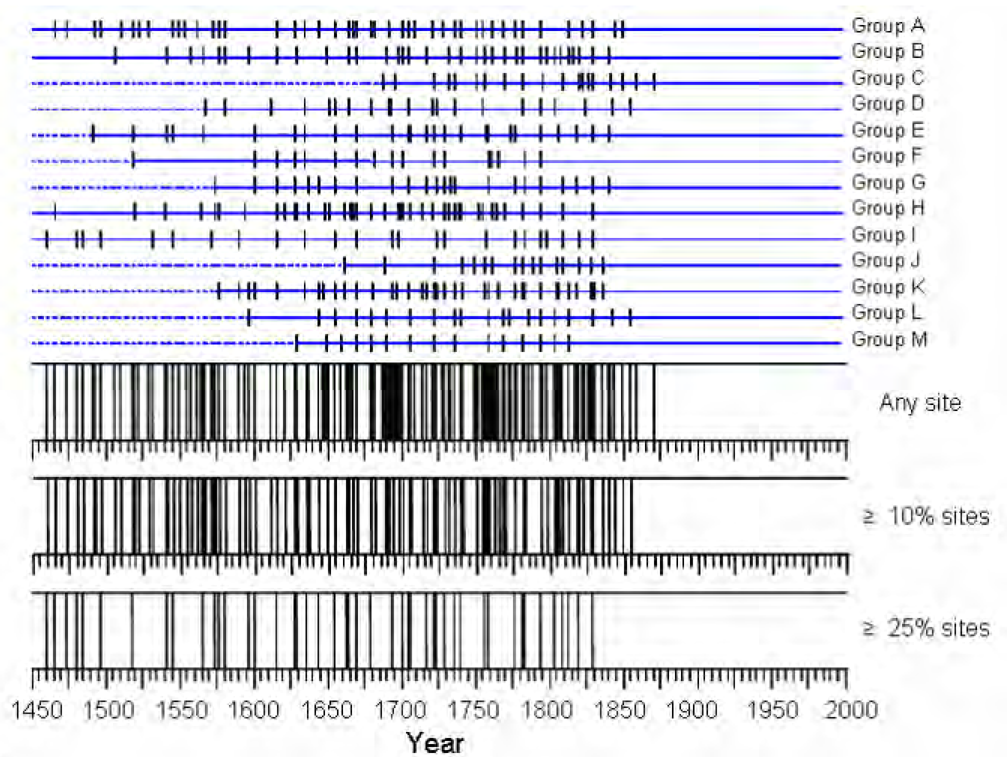

Figure 5—Fire chronology for Jeffrey pine-white fir forests for the period 1450-2000 for the 13 sample sites on the east shore of Lake Tahoe, Carson Range, Nevada. Fire dates are indicated by short vertical lines. The top portion of the graph shows fires record in each of the 13 sample sites. The three composite graphics at the bottom show fires recorded by all, >10% or more, and >25% or more of the sites.

Mean fire interval (MFI) estimates varied with sample area size and the MFI was shorter for the composite chronology for all sites (2.9 years) and longer for site chronologies (11.4 years) (*table 3*). The frequency of fires of different extent was inferred from the number of sites recording a fire. Small fires recorded by one or <10 percent of the sites were the most frequent type of fire and they had the shortest MFI (*table 3, fig. 5*). Larger fires recorded by 25 percent or more of the sites were less frequent and had longer MFI.

Table 3—*Fire return intervals statistics (years) for sites (n=13), and for a composite of all sites for the period 1450-1850. WMPI is the Weibull median probability fire interval.*

Type of sample	Number intervals	Mean	Median	WMPI	Std. Dev.	Min.	Max.	Skew-ness	Kurt-osis
Sites	321	11.4	9	9.8	8.6	1	82	2.8	15.6
Site composite									
All fires	135	2.9	2	2.5	2.3	1	12	1.5	2
>10% scarred	98	4	4	3.6	2.6	1	12	0.8	0.2
>25% scarred	45	8.2	7	7.1	5.7	1	23	0.9	0.1

The position of fire scars within annual growth rings indicate that fires burned mainly in the dormant season after trees had stopped radial growth for the year. This pattern suggests that dormant season fires burned in late summer or early fall, since radial growth of Jeffrey pine north of Lake Tahoe (120 km) is complete by late August (Taylor 2000).

Discussion

The structure and composition of forests that now cover the east shore of Lake Tahoe are very different from the pre Euro-American forests. Overall, the original forest was more open, less dense, and was composed of trees that varied widely in diameter. The original forest was removed over a period of 2-3 decades, beginning in the 1870s, to supply timber and cordwood to the Comstock mines in Virginia City, Nevada. A new forest began to establish immediately after stands were logged and forests in the Carson Range are now mostly dense 100-120 yr-old second growth, where trees are relatively small in diameter. This general description of forest changes in the Carson Range since the late 1800s is derived using dendroecological techniques, and it is consistent with written descriptions of the extent and severity of logging in the Lake Tahoe Basin in the 19th century (Leiberg 1902). However, written descriptions do not provide the quantitative data on presettlement forest characteristics needed by managers to design treatments to restore forests to the desired condition. In the case of the Lake Tahoe Basin, the desired condition has been identified by multiple stakeholders as the presettlement condition (Christopherson et al. 1996).

Sites for this study were chosen for conditions that promoted stump preservation to reduce uncertainty in reconstructed forest characteristics caused by disappearance of material. Complete decay, logging, or consumption of woody material by fire can eliminate evidence of the earlier forest (Fule et al. 1997). Small diameter stumps were present on most sites indicating that post-logging consumption of wood was unlikely. Wood decay varies with size and species and, in California forests, fir decays more quickly than pines, and small stems decay more rapidly than large ones (Kimmey 1955, Harmon et al. 1987). Therefore, reconstructed density and basal area estimates may be more reliable for pines than fir, and for large than small trees. Thus, managers should use the reference estimates conservatively in evaluating how contemporary forests deviate from the presettlement conditions, and for developing forest-ecosystem restoration goals.

Comparison of Presettlement and Contemporary Conditions

The presettlement reference conditions identified for the Carson Range can be applied by managers and stakeholders to: (1) evaluate contemporary forest conditions to prioritize management activities, (2) determine the causes of contemporary forest change, and (3) develop treatment strategies to restore highly altered forests to a desired condition (Christopherson et al. 1996, Covington et al. 1997, Kaufmann et al. 1994, Moore et al. 1999, Swanson et al. 1994, White and Walker 1997). Forests on the east shore of Lake Tahoe have a shared history and consequently they share certain common characteristics. They established immediately after near clearcut logging in the late 19th century, and then developed during a long post-logging, fire-free period. Yet, a comparison of presettlement and contemporary forest conditions and fire history suggest that contemporary forests vary in how and why they deviate from the presettlement reference.

Compared to presettlement forests, contemporary Jeffrey pine-white fir forests have smaller trees, less structural variability and, on average, five-fold more trees and nearly two-fold more basal area. The density and basal area change differences are greater in Jeffrey pine-white fir forests than for other forest types because of the key role of frequent fire in shaping presettlement Jeffrey pine-white fir forest structure, and the effect of fire suppression on post-logging stand development. The fire record demonstrates that low severity surface fires burned frequently. These fires maintained low stand density and basal area by thinning seedlings, saplings, and small diameter trees in presettlement Jeffrey pine-white fir forests. Moreover, the most widespread fires occurred during drought years that were preceded by wet years 2-3 years before the drought. Fuel buildup during the wet years appears to have predisposed the landscape to widespread burning during drought. No thinning fires have burned in Jeffrey pine-white fir forests since 19th century logging, and the post-logging fire-free period is unprecedented in length compared to the >400 year record of presettlement fire. Thus, fire regime changes caused by fire suppression have resulted in greater changes in Jeffrey pine-white fir forests from the presettlement condition than for the other forest types.

Fire regimes were not identified for presettlement red fir-western white pine and lodgepole pine forests in the Lake Tahoe Basin, but data on fire regimes for these types of forests elsewhere in the Sierra Nevada and southern Cascades suggest that return intervals are much longer (45-110 years) (Bekker and Taylor 2001, Caprio in press, Pitcher 1987, Taylor 2000). In fact, the 120 year post-logging fire free period may not exceed the longest fire free periods experienced in these forest types during the presettlement period. Consequently, although contemporary forests are denser and less structurally diverse than presettlement forests, the role of logging in these differences is probably more important than fire suppression. The post-logging compositional change in red fir-western white pine forests illustrates this point. Lodgepole pine was a minor component of presettlement stands, but its density in contemporary forests exceeds the combined density of red fir and western white pine. The mass establishment of lodgepole pine after logging the red fir-western white pine forests indicates that it is a successful pioneer species. The post-logging expansion of lodgepole pine into red fir forests may be temporary. Seedlings and saplings of red fir and western white pine are abundant in the understory of these stands suggesting that they may replace lodgepole pine as stands develop.

Lodgepole pine regeneration was also prolific after logging in presettlement lodgepole pine forests. In fact, contemporary lodgepole pine forests are the densest forests in the Carson Range. On these sites, however, lodgepole pine appears to be self-replacing and not successional. Lodgepole pine saplings and seedlings are abundant in the forest understory and continuous regeneration of lodgepole appears to be a characteristic feature of Sierra Nevada lodgepole pine forests (Parker 1986).

Despite uncertainty associated with possible disappearance of material, reference conditions identified from stumps are similar to reference conditions reconstructed from remnant old-growth stands in other parts of the Lake Tahoe Basin (Manley et al. 2000). Average presettlement tree density in old-growth Jeffrey pine (n=7) and mixed conifer (n=11) stands ranged from 63-67 trees ha^{-1}, similar to the value (68 trees ha^{-1}) reconstructed from stumps. Mean basal area from stumps and old-growth trees was also similar (25.3 m^2 ha^{-1} vs. 27 m^2 ha^{-1}). The tree density estimate from old-growth red fir stands (n=14) is lower (mean =107 trees ha^{-1}) than for stumps (mean=161 trees ha^{-1}), but the basal area estimates are similar (53 m^2 ha^{-1} vs. 55.8 m^2 ha^{-1}). There are no data for old-growth lodgepole pine stands for the Lake Tahoe Basin for comparison with stump reconstructions. The similarity in the estimates of density and basal area from stumps and old growth forest suggest that the reference estimates from stumps are sufficiently reliable for use as a guide for forest restoration planning in the Lake Tahoe Basin.

Application to Desired Conditions

In the Lake Tahoe Basin, the goal of institutional and citizen stakeholders is to return contemporary forests to a presettlement structure (Christopherson et al. 1996). The measurements of preserved cut stumps provide a range of quantitative estimates of presettlement forest conditions that can be used to meet forest restoration goals. Resource managers, however, need to use this reference information prudently in the process of developing plans. The reference conditions were established for limited parts of the landscape, where stump preservation was excellent. Reference conditions for other sites on the landscape, or other areas in the Lake Tahoe Basin, may deviate from those for sites with well preserved cut stumps. Resource managers can accommodate for some of these limitations by incorporating other ecological knowledge into restoration plans. For example, the Jeffrey pine-white fir forest reference is from the Carson Range, which is more xeric than sites on the west shore of Lake Tahoe (Barbour et al. 2002). In mixed Jeffrey pine-white fir forests, white fir is more abundant on mesic than xeric sites (Barbour 1988, Vankat 1982). Thus, resource managers could adjust the reference to include more white fir on more mesic parts of the landscape. For restoration planning, a reference should be viewed as a foundation for designing restoration treatments that will be complemented by multiple types of other ecological information, rather than as a rigid target that defines the acceptable outcome (Allen et al. 2002, Landres et al. 1999).

Contemporary forests on the east shore of Lake Tahoe vary in different ways from the presettlement reference forest, and plans to achieve desired conditions need to vary accordingly. Contemporary Jeffrey pine-white fir forests deviate more from the presettlement reference conditions than other forest types and restoration of these forests should receive highest priority. In Jeffrey pine-white fir forests, the reference conditions suggest that restoration objectives should emphasize: (1) density and basal area reduction primarily of smaller diameter trees (<40 cm dbh), (2) reintroduction of frequent fire as a key regulating disturbance process, and (3) increasing structural heterogeneity by shifting clumped tree distributions to a more random pattern.

Reference conditions for presettlement red fir-western white pine forests indicate that restoration objectives for contemporary forests should emphasize: (1) a shift in species composition by a density and basal area reduction of pioneer lodgepole pine, and (2) increasing structural heterogeneity by shifting tree distributions to a more random pattern. Reference conditions for presettlement lodgepole pine indicate that restoration objectives for contemporary forests should emphasize: (1) a density and basal area reduction of small diameter trees (<30 cm dbh), and (2) increasing structural heterogeneity by shifting tree distributions to a more random pattern. Given the relatively low fire frequency in higher elevation red fir-western white pine and lodgepole pine forests, re-introduction of fire as a regulating process may be viewed as a long-term forest restoration goal. Given that the physical legacy of the presettlement forests on lands affected by early-day logging (cut stumps) is still present in many landscapes (Fule et al. 1997), the method of estimating reference conditions from well preserved stumps may also be applicable for use on other early cut-over lands in other forest ecosystems in the western United States.

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Silviculture and Forest Management Under a Rapidly Changing Climate[1]

Carl N. Skinner[2]

Abstract

Climate determines where and how forests grow. Particularly in the West, precipitation patterns regulate forest growth rates. Wet years promote "boom" vegetative conditions, while drought years promote "bust." Are managers safe in assuming that tomorrow's climate will mimic that of the last several decades? For the last ~100 to ~150 years, climate has been warming at what appears to be an unusually rapid rate and is projected to continue into the foreseeable future. Increased temperatures are projected to lead to broad-scale alteration of storm tracks changing precipitation patterns in both seasonality and amounts. Multiple lines of paleoecological data show that such changes in the past, which were rarely as rapid, were accompanied by major reorganization of vegetation at continental scales. Exercises in modeling of possible ecological responses have shown the complexity in understanding potential responses of forests. Additionally, these exercises indicate that dramatic changes in natural disturbance processes are likely. Indeed, some believe that the responses of disturbance regimes to climate change may be emerging in the more frequent outbreaks of very large fires, widespread tree die-off across the southwest, expansive insect infestations in the Rocky Mountains, and more rapid and earlier melting of snow packs through the West. Developing both short- and long-term forest management responses will be challenging. Therefore, silviculturists must be aware of the nature of and implications of climate change in order to develop management strategies that may help to reduce adverse effects while sustaining healthy, productive forests.

Introduction

The successful practice of silviculture depends on a strong understanding of the relationships of species to climate in order to manage forests to meet many of society's needs from wood products to wildlife habitat. Climate is a great controller of our environment. Climate determines where and how forests grow. The type of climate for any particular place is a consequence of long-term, generalized weather conditions over gradients of time from days to seasons to decades or centuries. Climate includes not only the central tendencies or 'average' weather, but also the patterns of variation and nature of the extremes. Each species has a more or less unique geographical distribution that is related to its particular range of adaptations to climate and other environmental factors. Thus, climate strongly determines the potential for a species to grow and thrive in any particular place.

Some controversy surrounds the causes of the current climate change, but the evidence that climate is changing is compelling (Parmesan and Yohe 2003,

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Appenzeller and Dimick 2004). It is not the intent of this paper to spend time on the possible causes or 'blame' for the current rapid climate change. To do so often diverts attention from the more immediately important discussion of how climate change will likely affect the ability to be successful forest managers. Indeed, the relative contributions of natural climatic variation and human-induced climate change will likely not be known for decades (Hughes 2000). Rather, the objectives in writing this paper are to (1) describe the evidence indicating that climate is changing rapidly, and (2) discuss the implications of climate change for forest managers.

Climate Change

We know that climate has undergone large fluctuations in the past, oscillating between cold ice ages and warm temperate periods over many 1000s of years. The common view has been that climatic variations take place over long time scales and, though intellectually interesting, are not relevant to practical forest management. Thus, day-to-day forest management has largely been conducted from a perspective that climate is relatively stable over relevant time scales and what were successful practices over the last several decades will likely continue to be successful into the foreseeable future. However, in light of recent research on climate variation, continuing to manage from this perspective may leave one quite vulnerable to undesirable consequences.

Multiple lines of evidence developed in the late 20th Century are providing more complete descriptions of past climatic variation, the characteristics of past climate changes, and associated ecological responses. Since the instrumental record is so short (little more than a century in many parts of the country), the development of this evidence has relied on interpretations and inferences drawn from proxy or indirect records of climate. Proxy records used for reconstructing descriptions of past climate variation come from ice cores, sediment cores, packrat middens, corals, and tree-rings among others (Stokstand 2001).

Each proxy has different potentials for temporal depth and resolution. Sediment cores may include from several thousand to millions of years of pollen, charcoal, and other biological indicators (Davis 2001, Whitlock and Anderson 2003). Packrat middens may include up to ~40,000 years of information (Rhode 2001). Both methods are usually limited to 50+ years of resolution, although sediment cores occasionally may be found that are annually laminated. Ice cores provide seasonal to decadal resolution for up to 40,000 years, while annual resolution is provided by corals for up to 400 years and tree rings for up to several thousand years (Stokstad 2001).

Though each proxy has its pluses and minuses, together they provide complimentary information that allows us to gain a more comprehensive understanding of the temporal and spatial scales of climatic variability. The picture that emerges indicates climate is continually changing at varying rates due to multiple, nested scales of oscillations responding to a variety of climate forcing factors (Millar 2005). Abrupt shifts (major changes on a decadal scale or less) in climate have been found to be associated with periods of rapid, low-frequency climate change similar to what we are currently experiencing. This potential for abrupt change is thought to be due to response to the achievement of particular threshold conditions (NRC 2002). Notably, the current period is one of exceptionally rapid warming (Stanley 2000).

Temperature

The warming of the 20th Century that continues to today reversed a millennial-scale cooling trend (Mann et al. 1999). The rapidity of changing conditions is exemplified by the switch from the 1800s as the second coldest century to the 1900s being the warmest century (1900s) of the last millennium with much of the warming taking place from 1920-1945 and 1975 to the present (Jones et al. 2001).

Though the Earth's average atmospheric temperature has warmed considerably over the last century, the warming has not been temporally or spatially uniform. For the Northern Hemisphere, more pronounced increases have taken place through the raising of diurnal minimum, winter, high latitude, and high altitude temperatures than diurnal maximum, summer, low latitude, and low altitude temperatures (Jones et al. 2001, Walther et al. 2002). Average nighttime and winter temperatures have increased at twice the rate of daytime (Walther et al. 2002) and summer temperatures (Jones et al. 2001). Modeling exercises project that atmospheric temperatures across the continental United States are likely to continue to increase to between approximately 2^{O}C (3.6^{O}F) and 6.6^{O}C (11.9^{O}F) by the end of the 21st Century (Hansen et al. 2001).

Water

The influence of climate on vegetation is primarily manifested through the interactions of energy input (temperature) and the water available for plant growth and decomposition (Olsen 1969, Stephenson 1990). Most models predict increases in average global precipitation will accompany increasing temperature. The average annual precipitation has increased over the 20th Century across the United States (Hughes 2000). However, the regional distribution will vary from wetter in some areas to dryer in others with additional changes seen in seasonality and type of precipitation (Hughes 2000, Mote et al. 2003). For example, there has been a 10% reduction in annual snow and ice cover of the earth since the 1960s (Walther et al. 2002). Thus, the actual water available for tree growth may increase or decline depending upon how the interaction of temperature and moisture are manifested regionally and locally (Walther et al. 2002, Mote et al. 2003).

Climate change is expected to alter the energy/water balance influencing plant growth, disturbance processes, and patterns of habitat. In western North America, several important hydrologic patterns have emerged over the closing decades of the 20th Century – an earlier onset of spring peak stream flows, a shortened annual period of warm-season snow packs due to earlier and more rapid snowmelt (Cayan et al. 2001, Mote et al. 2003, Stewart et al. 2004), and greater interannual variation in stream flow (Jain et al. 2005). A consequence of these changes is a greater proportion of stream flows in winter-spring with lower flows in summer (Regonda et al. 2005). As the effects of evapotranspiration instead of snowmelt come to dominate many streams, they will change their diurnal peak flows to early morning from late afternoon (Lindquist and Cayan 2002). In terms of available soil moisture, the earlier onset to spring drying and earlier snowmelt reduces available soil moisture over the course of the warmer growing season potentially creating greater stress for vegetation (Stephenson 1998).

Potential Ecosystem Responses to Global Warming

Climate influences where and how forests grow through spatiotemporal variation in environmental factors that include atmospheric temperature, precipitation, wind, and humidity. Global temperature has been increasing rapidly along with atmospheric concentrations of CO_2 over the last several decades and is expected to continue through this century. These changes in temperature are expected to be associated with changes in spatiotemporal patterns of precipitation and water availability (Mote et al. 2003, Stewart et al. 2004) which in turn are expected to affect patterns of disturbances and other agents of change (Ayres and Lombardero 2000, Logan and Powell in review), all of which combine to affect where and how species of trees are able to grow (Davis and Shaw 2001, Iverson and Prasad 2002, Hughes 2000, Shafer et al. 2001).

Silvicultural strategies will benefit from an understanding of processes and likely biophysical responses to climate change that cascade ecosystems through effects on physiology, phenology, range and distribution, and abundance of species (Harrington et al. 2001, Walther et al. 2002). Though the response of ecosystems to projected changes is likely to be quite complex (Neilson and Drapek 1998, Walther et al. 2002), there are some key variables that will be particularly of interest to forest managers.

Disturbance

A warming climate is likely to increase the occurrence of extremes for many types of disturbance processes – wind, floods, drought, insects, pathogens, and fires. Each of these disturbance processes will affect our ability to manage forests for desired outcomes. This paper focuses on drought, insects, and fire, since they are more likely to affect broad, regional areas and their interactions have potential to significantly affect atmospheric carbon (Breshears and Allen 2002, Harrington et al. 2001).

Drought

North American droughts of the 20th Century, though known to have caused great economic hardship, were unusually benign and not representative of the full range of variability compared to multi-year droughts of the last millennium (Cook et al. 2004, Stine 1994, Woodhouse and Overpeck 1998). Regional drought may cause widespread changes to ecosystems both directly through mortality of susceptible species and indirectly by creating conditions that more readily support high intensity fires or insect outbreaks. Climate change modeling predicts that portions of the country, such as the Southwest, may experience more prolonged and severe droughts with continued warming.

The more severe effects of drought, especially under a rapidly changing climate, are apt to be realized along major ecotones where vegetation is normally under some stress. A severe drought in the 1950s in New Mexico provides an example of the potential influence of drought on forest species distributions along an ecotone boundary. The impact of this drought, enhanced by bark beetle activity in a landscape where vegetation had become increasingly dense over the early 20th Century, was sufficient to cause the ecotone between ponderosa pine forest and piñon-juniper woodland to shift up to 2 km as ponderosa pine retreated to higher elevation. Additionally, the shift of this ecotone boundary has persisted for over 40 yrs (Allen and Breshears 1998).

The direct effects of drought are usually seen in reduction of tree growth, but direct mortality may occur in more extreme situations (Guarín and Taylor 2005). More commonly, significant changes in stand conditions and mortality occur indirectly from other agents such as insects or fire that are facilitated by the dry conditions. Whether drought induced ecological changes become a more permanent condition will depend on the severity of the effects and the response of the ecosystem to the future climate trajectory.

Insects

Climate change is expected to alter host / insect relationships through effects on physiology, phenology, and species distributions (Hughes 2000, Harrington et al. 2001, Williams and Liebohold 2002). Though, resulting manifestations of these effects will likely be complex and not necessarily predictable, Harrington et al. (2001) summarize some of the more general potential effects:

- ***Physiology*** Warming temperatures, especially in regions where insects have been limited by severe cold, will allow some species to survive in regions that were previously unfavorable. Additionally, the longer annual warm periods will allow some species to produce more broods.

- ***Phenology*** For species whose life-cycle events are controlled by temperature, warming temperatures will likely lead to changes in timing of flowering and growth in trees as well as in activity in insect populations. Where species with life cycles controlled by photoperiod interact with species controlled by temperature, novel relationships are likely to develop.

- ***Distributions*** Altered distributions of both potential hosts and insect species are likely to be a result of the effects of climate change on physiology and phenology. These potential changes are expected to bring into contact populations of trees and insects whose distributions did not previously overlap.

The recent expansion of the mountain pine beetle (*Dendroctonus ponderosae* Hopkins) into environments where it was previously limited by effects of extreme winter cold on broods (Carroll et al. 2004, Logan and Powell in review) may be one dramatic example of response to such alterations. In recent decades the mountain pine beetle has moved northward into areas of British Columbia with previously unsuitable climate (Carroll et al. 2004). The effect of this migration has been dramatic mortality in lodgepole pine (*Pinus contorta* Dougl. var. *latifolia* Engelm.) populations not previously exposed to the beetle (Carroll et al. 2004, Logan and Powell in review). It has been suggested that a continued warming of the climate could allow this insect species to migrate into jack pine (*P. banksiana* Lamb.) populations on the eastside of the Rocky Mountains, presenting an avenue that may possibly allow the mountain pine beetle to move east and then south as far as the southern pine forests of Texas (Logan and Powell in review).

Fire

The combined effects of droughts and insects may lead to a pulse of tree mortality that increases the potential for intense fires. There is a short-term and a long-term facet to the increase in potential fire intensity. In the short-term, as long as the dead foliage remains on the trees, there may be a dramatic increase in the potential for intense crown fires. Once the dead foliage drops, this danger may be considerably reduced for a few years. However, as the trees decay over the next decade or so following the pulse of mortality, they fall and can help create an accumulation of large, heavy fuels. These large fuels contribute to a longer-term potential for intense fires since they may take many years to decompose, especially in the dry environments of the West.

Even in the absence of increased mortality from either drought or insects, a warming climate will likely alter fire regimes in ways that will make it more difficult to manage forests influenced by many decades of fire suppression and other activities. It is widely recognized that western forests have changed dramatically over the last century or so due to fire suppression, logging, grazing, and other activities. Fuel profiles have changed with stands becoming generally denser and often accompanied by increases in both ladder and surface fuels (Agee and Skinner 2005).

Climate change influences fire regimes in complex ways due to differentials in responses to variation in temperature and precipitation regimes. Both tree-ring records and modeling indicate that the probability of having fires is primarily driven by temperature whereas, the extent and intensity of fires is driven more strongly by precipitation patterns (Chang 1999, Flannigan and Van Wagner 1991, Swetnam 1993). Warmer temperatures lead to an earlier onset and later end for the drying period, thus increasing the probability of a fire during the longer fire season. Precipitation influences the growth of vegetation (fuel). The more/less precipitation in the wet season, the more/less fuel will be produced. In the occasional dryer years during moist periods, fires are likely to burn more extensively and intensely due to greater fuel accumulations. Conversely, fuels are likely to accumulate more slowly during longer periods of consecutively dry years. Once the initial fuel loads grown during the moist periods are consumed, though fires are expected to become more frequent in dryer periods, they are likely to be less extensive and less intense due to the limits to growth of new fuels in extended dry periods.

Under a warming climate, the general outlook is to expect a greater number of fires with more escaping initial-attack suppression activity due to the longer fire seasons (Fried et al. 2004). The past century of altering stand structures and accumulating live and dead fuels increases the probability that many fires in the dryer areas of the West will be of higher intensity than would have been likely under a historical fire regime (Agee and Skinner 2005).

Vegetation

Of great interest to silviculturists is how vegetation is likely to respond to changing climate. Different species will respond to climate change in different ways making it difficult to generalize about expected responses. However, altered productivity and changes in species distributions are likely to be two basic responses (Aber et al. 2001). Productivity will be affected by changes in temperature, precipitation, effective moisture, and competition among other factors. As climates change, species will migrate into new locations, while sometimes disappearing altogether from locations in which environmental factors become too stressful (Hansen et al. 2001, Joyce and Hansen 2001).

Climate change is likely to alter our potential to manage woody vegetation by affecting physiology, phenology, and distributions in ways similar to those described for insects. Such practices as spacing in tree plantations rely on assumptions about moisture availability and competition (Joyce and Hansen 2001), while scheduling of harvests depends on assumptions about productivity. Climate change is likely to alter competitive interactions and thus affect tree growth and the ability to achieve productivity goals.

Physiology

Moisture stress, especially in dryer regions, would tend to increase with warming temperatures. However, the increases in atmospheric CO_2 over the last century and projected into the next will increase efficiency of moisture use and offset, at least for sometime, the effects of increasing temperature. Plants with cold period requirements may not be able to receive sufficient cold exposure (Hughes 2000).

Phenology

Earlier onset of flowering and growth appears to be a response to warming temperatures in many species of trees. As a result, new interspecific relationships are likely to develop where species with life cycles responding to temperature interact with species controlled by photoperiod (Hughes 2000). For the silviculturist, these changes in competitive interactions will likely lead to new challenges in managing competing vegetation.

Distributions

Altered species distributions will likely be a result of the effect of climate change on physiology and phenology. One of the important things that paleoecology teaches us is that species distributions will change in response to climate (Davis and Shaw 2001, Whitlock 1992). Additionally, dramatic changes in species distributions can take place over scales of but a few decades to a century during periods of rapid climate variation (Davis and Shaw 2001, Peteet 2000). Notably, species respond to climate individualistically and not as plant communities or associations. Thus, current assemblages are likely to dissolve and coalesce into novel associations as species ranges adjust to a changing climate (Davis 1986, Whitlock 1992).

In order to better understand the potential re-distribution of species in North America, several projects projecting potential range shifts in tree species based on simulation of species responses to environmental changes have been undertaken (Iverson and Prasad 2002, Shafer et al. 2001). These studies use various models projecting climatic conditions (e.g., temperature, precipitation, growing days, etc.) coupled with spatial representation of a number of environmental variables (e.g., elevation and soils). These conditions are then compared to those in which species are currently found to create a ‘climate space’ for each species. This climate space is then presented geographically to represent where species would be expected to find favorable growing conditions under future climate projections. Prasad and Iverson (1999-ongoing) project that some species in the eastern United States may see little change in overall distribution (e.g., white oak [*Quercus alba* L.], red maple [*Acer rubrum* L.), whereas, other species (e.g., aspen [*Populus tremuloides* Michx.], paper birch [*Betula papyrifera* Marsh.], sugar maple [*A. saccharum* Marsh.]) may retreat northward into Canada as suitable climate space is lost.

The many changes in species distributions that are projected will bring into contact populations of trees and shrubs whose distributions did not previously overlap, introducing new inter-specific competitive relationships.

Genetics

Rapid climate change over the next century will likely render many species and local varieties less genetically suited to the environments in which they are currently found (Davis and Shaw 2001, Peteet 2000). Regeneration difficulties may be the earliest noticeable sign of the effects of climate change. Established, mature trees are often able to withstand a wide range of environmental conditions and will be able to survive for many years with effects primarily appearing as altered levels of productivity. However, establishing regeneration after logging or large fires may become more difficult since seedlings are often more sensitive to environmental conditions than are mature trees. Managers may want to consider mixing in seedlings from neighboring seed zones that may be better suited to the new local environment (Ledig and Kitzmiller 1992).

A lot of work has gone into selecting and breeding trees that perform exceptionally well under specific environmental conditions. Rapid climate change may reduce the ability of many plantations of such trees to perform to expectations. Using trees that do generally well under a variety of conditions may be better strategy than those that do exceptional under a narrow range of conditions (Ledig and Kitzmiller 1992).

Habitat

Highly altered habitat conditions for many animal species are likely to be a result of the complex changes in vegetation distributions that are expected with changing climate. Thus, the changing geography of favorable habitat for many animal species is likely to also lead to animal migrations and altered geographic distributions (Burns et al. 2003).

Managers are often required to manage for favorable habitat conditions in support of species of concern. Some examples are the requirements for 'old-growth' conditions for the northern spotted owl (*Strix occidentalis caurina*) in the Pacific Northwest, California spotted owl (S. o. *occidentalis*) in the Californias, and the Mexican spotted owl (*S. o. lucida*) in the Southwest. Many of the dominant species of trees important in the structure of these habitats regenerated and grew under very different past climates (Sprugel 1991). However, the influence of a warming climate on the likelihood of fires (as discussed above) may make it more and more difficult to sustain appropriate habitat without greater attention to landscape pattern, geographical context, and the realities of climate.

Specific types of locations in landscapes such as more mesic north facing slopes and the lower slopes of canyons are more likely to sustain 'old-growth' over longer periods of time than the landscape as a whole (Taylor and Skinner 1998, Beaty and Taylor 2001, Taylor and Skinner 2003). Therefore, it may be possible to learn from the past to help design landscape strategies that reduce the potential for unusually severe fires while improving the probability of maintaining sufficient habitat across landscapes (Taylor and Skinner 1998, Weatherspoon and Skinner 1996).

Conclusion

Climate is continuously changing, with some periods exhibiting rather stable, slowly changing climate, and other periods displaying great variation with rapidly changing climate. At present, we appear to be in one of the later situations, as climate appears to be changing rapidly and is expected to change considerably more over the coming decades.

Current expectations for forest productivity and maintenance of desirable habitat conditions are based mostly on how forests developed under past climate. In general, there is often little acknowledgment of the potential influence of our changing climate on those expectations. Management strategies that ignore the uncertainties associated with climate change are likely to fall short of expectations. Whereas, strategies that acknowledge ongoing climate change, incorporate relevant monitoring, and include capacity for adaptation will likely be more successful in the long run.

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Fuels Planning: Science Synthesis and Integration[1]

Rachel White[2] and Sarah McCaffrey[3]

Abstract
A century of fire suppression has created heavy fuel loads in many U.S. forests, leading to increasingly intense wildfires. Addressing this problem will require widespread fuels treatments, yet fuels treatment planners do not always have access to the current scientific information that can help guide their planning process. The Fuels Planning: Science Synthesis and Integration project was launched to compile relevant fuels treatment information for managers. Products include syntheses on various topics, a guidebook on silvicultural prescriptions, a set of models and information databases on possible environmental effects of fuels treatments, and a financial analysis tool for estimating costs and revenues of fuels treatments. The Fuels Planning project provides an example of how collaboration between managers and scientists can improve the utility of scientific findings. It is currently forming partnerships with several National Environmental Policy Act (NEPA) interdisciplinary teams who will use these decision support tools in planning fuels reduction projects starting in the summer of 2005.

Introduction
Even with increasing expenditures devoted to fire suppression in the US, the frequency, intensity, and annual acreage of wildfires continue to grow, due in part to heavy fuel accumulations after a century of aggressive fire suppression. Addressing this problem will require widespread, ongoing fuels treatments. But information overload makes it challenging for fuels treatment planners to integrate diverse scientific findings into their projects. The need for well-documented, accessible scientific information is crucial. To address this concern the Fuels Planning: Science Synthesis and Integration project was established to synthesize research findings relevant to fuels treatments, and to provide it to managers in a useful and accessible format. This information is of immediate need to managers, and the project staff worked to develop these products at an accelerated pace. Although the fire ecology and economics information was developed for application in the dry inland forests of the West, the social science findings are applicable to fuels planning activities throughout the US.

The tools produced by this project are designed to help field planning teams

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utilize current information without the burden of collecting and synthesizing disparate and rapidly emerging scientific findings. Target audiences include fuels management specialists, resource specialists, National Environmental Policy Act (NEPA) planning team leaders, line officers in the USDA Forest Service and the Department of the Interior, community leaders, and educators. Products developed will allow planners to quantify parameters and achieve greater consistency in their analyses by standardizing outputs. Information derived from the various tools can be used in environmental analysis processes such as Environmental Impact Statements, Categorical Exclusion documents, Environmental Assessments, and NEPA documents.

Products

The project divided into teams organized around four key topic areas: social science, forest structure and fire hazard, environmental consequences, and economics. Each team is publishing a complete synthesis of their topic in a peer-reviewed document and, where appropriate, developing decision support tools related to their topic area. In addition, each team has written a series of simple and approachable two-page fact sheets highlighting key information. All of these tools and publications are available on the project's website: http://forest.moscowfsl.wsu.edu/fuels/htm

Team Products

Social Science Team

Fire and fuels management projects must respond to complex social forces. Although research directly related to social implications of fuels management practices has only fully developed under the National Fire Plan, there are numerous areas where related research can provide managers with useful information. The social science team compiled and synthesized social science information relevant to fuels management in five key areas: (1) social acceptability of fuels treatments, (2) collaboration, (3) aesthetics, (4) communicating with the public about risk, fuels treatments, and defensible space, and (4) the social impacts of wildfire. Key findings from each of these five areas have been compiled in five individual GTRs (e.g., see Sturtevant et al. 2005), which will also be summarized in a manager oriented GTR. All the publications, including the team's fact sheets, are available at the project's website (see above).

Forest Structure and Fire Hazard Team

The forest structure team has published a GTR demonstrating how different silvicultural prescriptions would affect fire hazard and forest structure in dry forests of the western US, and highlighting ecological principles associated with managing forest fuels and vegetation for specific conditions (Peterson et al. 2005). Quantitative results are presented in tabular format for pre-and post-treatment effects for potential fire behavior attributes, such as crowning and torching potential, fire type, canopy base height, and canopy bulk density. Computer-based landscape simulations provide pre- and post-treatment images that help users visualize stand and landscape conditions, the effects of different management treatments, and fuel changes over time. This publication (another GTR on the science basis for changing forest structure) (Graham et al. 2004), and the team's fact sheets, are available at the project's website (see above).

Environmental Consequences Team

The environmental consequences team developed a set of models and information databases to assist managers and project planners in assessing the diverse possible environmental effects of fuels treatments.

- The Understory Response Model provides a comparative evaluation of treatment impacts to understory plants, including invasive species. It is a species-specific computer model that predicts qualitative changes in total species biomass for grasses, forbs, and shrubs after thinning, prescribed fire, or wildfire at one-, five-, and 10-year intervals.
- The Wildlife Habitat Response Model is a web-based computer tool that provides qualitative estimates of treatment impacts on the habitats of most vertebrate animals in the interior West. Using data gleaned from the scientific literature, the model identifies important habitat elements for a species, and displays expected suitability of the post-treatment environment.
- The *Armillaria* Response Tool helps to identify areas potentially susceptible to *Armillaria* root rot after stand treatment. This model can indicate how some fuels management activities may exacerbate root rot within high-risk stands, and helps determine an appropriate fuels management plan for reducing further damage from the disease.

The team also enhanced existing software to improve ease of use and applicability for fuels management. These products include:

- The Smoke Impact Spreadsheet--a smoke model that estimates particulate matter emissions, smoke production, and dispersion for comparison with appropriate federal or state air quality standards.
- The Water Erosion Prediction Project Fuel Management Tool--an erosion model that estimates the probability of sediment yield and flooding after a disturbance. It creates output tables that compare sediment generated for a variety of conditions (such as thinning, wildfire, undisturbed forest, etc.). These tables can be pasted directly into NEPA documents or similar analyses. (see Elliot 2004)

In contrast to almost all other tools currently used in fire management and planning, all these tools have been peer reviewed. This is one of the reasons they are valuable for NEPA and other science-based documents. These tools and related publications and fact sheets can be found at the project's website (see above).

Economics Team

The economics team developed a peer reviewed financial analysis tool called My Fuel Treatment Planner that assists managers in estimating costs, net revenues, economic impacts, and surface fuels associated with various fuels reduction treatments (see Biesecker and Fight in review). The planner provides insights on how to think through financial analyses, and interacts compatibly with existing planning tools. It was designed for fuels treatment planners, including those with little or no background in economics, forest management, or timber sales. It promotes common sense decision making by answering questions such as: what type of fuels treatment could pay for itself? What would it cost to treat this stand? Can I combine mechanical treatments and prescribed fire to make treatments less expensive? Easily navigable, this spreadsheet application is simple to use, yet the information behind it comes from years of data gathered from the western US. This tool and the team's fact sheets are available at the project's website (see above). To go directly to My Fuel Treatment Planner, go to: http://www.fs.fed.us/pnw/data/myftp/home.htm

Collaboration and Validation

To ensure that information was provided to managers in a useful form, the project worked to ensure involvement of a wide variety of people. The effort involved the collaboration of scientific experts from the North Central Research Station, Pacific Northwest Research Station, Pacific Southwest Research Station, and Rocky Mountain Research Station, their management counterparts, and university researchers, and received financial and technical support from Fire and Aviation Management. The cooperation has both improved the applicability of this project's results to fuels planning activities around the country, and boosted credibility, trust, and understanding on both sides. Managers were involved in initial development of primary research questions and initial product development. As products were developed, several data trials and beta tests were held with on-the-ground fuels planners and fire managers for ground-truthing and feedback. The testing also has helped to fine-tune product packaging and delivery.

Management Applications

By synthesizing current information and presenting potential treatment effects through computer models, these products support fuels managers and project planners as they select and execute fuels treatments. With more complete access to relevant scientific information, and with tools for improved environmental analysis, managers have an improved capability to make informed, defensible decisions. Products will help managers consider a range of options, including no-action alternatives. By comparing predicted effects delineated by the computer models against a threshold of acceptability, a manager has a clear and thoroughly reviewed logic for evaluating a final decision for treatment. For example, the financial planning tool helps managers determine cost-effective approaches to fuels treatments, while the suite of environmental consequences models can help identify areas in need of habitat protection. The silviculture GTR can help managers visualize potential impacts of their fuels treatment activities over space and time, and can allow them to make better decisions based on a range of options.

So far, users who have tested the tools have responded favorably. One user commented that these tools will be "useful on the smaller categorical exclusion-type projects, where fuels Assistant Fire Management Officers could do much of the analysis themselves." "Great for a small project," said another user. Other types of projects sample users anticipated using the tools for include Environmental Assessments, Environmental Impact Statements, and Hazardous Fuels Reduction projects. The models also provide a way to present outcomes, to create a framework for understanding other fuels planners and managers, and to forge a common language. All these facets also make these tools particularly valuable when managers have to explain a fuels treatment plan to the public.

Ongoing Work

The project currently is forming partnerships with NEPA interdisciplinary teams who will use the decision support tools in planning fuels reduction projects starting in the summer of 2005. A recently established technology transfer team for the project will work with these teams to provide initial on-site training sessions and on-going support throughout the planning process. This process will both help raise awareness of the various products and enable further refinement of the tools and science delivery materials. My Fuel Treatment Planner also has been incorporated in the Fireshed Assessment process that has been developed in California. All national forests in the state are using this process so that managers can assess their progress toward meeting the hazardous fuel reduction goals of the National Fire Plan, the Healthy Forests Restoration Act of 2003, and national forest land and resource management plans. These assessments rely on My Fuel Treatment Planner to provide financial analyses of different scenarios.

The Fuels Planning project provides an example of how active collaboration between scientists and managers can help facilitate the delivery of relevant scientific findings to managers. As a trial effort in such work on a national scale, and one also at an accelerated pace, it required significant dedication on the part of both the scientists and managers. Hopefully, as such actions become more routine and normalized, the resource requirements, while still significant, will become less demanding.

Lead Scientists

- Project leaders: Russell T. Graham, Sarah M. McCaffrey, and Leslie Sekavec
- Environmental Consequences team: Elaine Sutherland and Anne Black.
- Wildland Fire Behavior and Forest Structure team: David Peterson and Morris Johnson
- Economics team: Jamie Barbour and Roger Fight
- Social Science team: Pamela Jakes and Susan Barro

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Stewardship and Fireshed Assessment: A Process for Designing a Landscape Fuel Treatment Strategy[1]

Bernhard Bahro,[2] Klaus H. Barber,[3] Joseph W. Sherlock,[4] and Donald A. Yasuda[5]

Abstract

Natural resource land managers today face a difficult challenge of developing a cohesive fuels and vegetation management strategy that addresses the widely acknowledged wildfire threat. Treatments must also be compatible with a wide variety of other land management goals, such as managing for wildlife habitat, watersheds, and forest health. In addition, funding will always be a limiting factor for management of public lands; managers will always have to prioritize and strategize where funding provides the most benefits. Stewardship and Fireshed Assessment (SFA) is an interdisciplinary, collaborative process for designing and scheduling fuels and vegetation management treatments across broad landscapes to help natural resource managers balance goals for reducing potential for large, severe wildland fires with other ecological and social goals. The approach for modifying landscape-scale fire behavior (how large it gets, where it burns, and how severely it affects communities, habitats, and watersheds) is anchored in the concept that, by using a carefully designed pattern of treatment areas, managers can treat a fraction of the landscape to achieve intended modifications in wildland fire behavior. The SFA process uses existing data, robust assumptions, and data models in a geographic information system to provide a rapid assessment that informs land managers and the public on the trade-offs of different management strategies. The SFA process implements the "Plan, Do, Check, Act" model of the Forest Service's Environmental Management System. Using the concepts of active learning, this type of assessment is designed to increase public participation and understanding of forest management and develop support for forest restoration. Ultimately, it is hoped that active public dialog will help garner advocacy for a balance of active and passive management, and hopefully, reduce controversy and conflict regarding individual hazardous fuel projects.

Introduction

Since 1999, national focus has been placed on addressing the problem of wildland fire effects to communities and forest resources. This has resulted in the National Fire Plan, 10 Year Comprehensive Strategy, Healthy Forest Initiative and the Healthy Forest Restoration Act which provide direction, funding, and

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performance measures to address the hazardous fuels problem across the country. Both Congress and the public are concerned with ensuring efficient and effective use of funds directed for hazardous fuels reduction. In particular, managers are being asked to demonstrate how treatments are addressing threats to communities along the wildland urban interface. Thus, land managers are challenged with evaluating not only how individual treatments change wildfire behavior but also how patterns of treatments collectively perform at the landscape-scale to reduce the size and severity of wildland fires.

Despite this emphasis, implementation of fire and fuels management direction by Federal land management agencies has come under criticism in 27 separate Government Accountability Office (GAO) reports since 1999 (summarized in GAO-05-147). Collectively, these GAO reports reference the inability of federal land management agencies to adequately assess landscape strategies for hazardous fuels treatment, set priorities, develop out year plans, and collaborate with partners. However, a recent GAO report (GAO-04-705) noted: “One [approach] that appears promising for national implementation is the Fireshed Assessment process, an integrated interdisciplinary approach to evaluating fuel treatment effectiveness at reducing fire spread across landscapes.”

The Stewardship and Fireshed Assessment (SFA) process is a rapid assessment process that has been developed for the national forests in California. The SFA process frames and evaluates the performance of hazardous fuels treatments at a landscape-scale, where treatments are designed to change the outcome of a “problem” fire in a particular landscape. A “problem” fire is a hypothetical wildfire that could be expected to burn in an area that would have severe or uncharacteristic effects or result in unacceptable consequences. While the primary objective of strategic treatments is to reduce the wildfire risk to communities in the wildland urban interface, treatments must also be designed to integrate broader stewardship objectives, such as improving forest health, meeting habitat needs, and maintaining and improving watershed conditions. Given these multiple objectives, it is important that a landscape treatment strategy be reasonable and feasible and, critically, that it have public support. This is accomplished by evaluating treatment scenarios, which are combinations of treatment locations, treatment prescriptions, and implementation timelines, in an open and transparent manner. Through repeatedly testing and improving assumptions, public understanding of ecological processes, the effects of management, and management constraints and opportunities can be enhanced.

The individual Fireshed Assessment is a core component of the SFA process. The landscape is divided into firesheds, which are conceptually analogous to watersheds. These firesheds surround areas of similar wildfire threat where a similar response strategy could influence the wildfire outcome. Given that it is impossible to treat all of the hazardous fuels across a landscape, the identification and prioritization of the most critical and beneficial hazardous fuels to treat is critical. A Fireshed Assessment is based on the premise that fuels treatments strategically located to modify fire behavior can positively affect the outcome of a wildland fire by limiting the area severely burned and reducing negative effects on communities, habitat, and watersheds. The underlying assumption is that as landscape-scale wildfire behavior is modified over time, fire suppression and fire management opportunities will be enhanced, leading to fires that are less damaging and less costly (Finney et al. 1997).

Ultimately, managing landscapes to influence potential large wildfires requires careful prioritization and scheduling of fuels treatments across large areas over time.

Since federal, state, and private lands are often intermingled, developing a coordinated program of work requires close collaboration with other landowners and interested parties. Hence, two critical pieces must come together to change large wildfire outcomes: (1) collaboration and coordination with other agencies, landowners, and the interested public, and (2) on-the-ground implementation of a program of work, which establishes spatial locations, priorities, and schedules for multiple hazardous fuel treatment projects, ideally across all land ownerships in an area.

Core Components

Stewardship and Fireshed Assessment describes an overarching assessment process that is composed of several analytical and process components. The focus of SFA is collaborative resource problem-solving. In a dynamically linked system, each component informs and learns from other parts of the SFA process. *Table 1* provides a brief description of the core components of the SFA process.

Table 1—*Core Components of the SFA Process*

SFA Component	Description
Fireshed Assessments	Characterizes the potential "problem" fire. Map and description of treatments that could be implemented to address the threats from a problem fire. Considers existing fuel conditions, treatment opportunities, and resources of value.
Spatially Explicit Program of Work	Schedule and map showing how needed work can be accomplished in annual increments. Tests costs and feasibility of doing entire program over time. Provides temporal and spatial display of future activities to inform project-level cumulative effects analyses.
Individual Project Evaluations	Detailed site-specific analyses of individual projects that implement the program of work.
Project Implementation	On-the-ground implementation of individual projects.
Project Feedback and Monitoring	Compares actual treatments with planned treatments to determine if assumptions were reasonable and identify minor and major adjustments that may be needed.
Fireshed Assessment Review and Update	Review project feedback and trends of actual implementation to assess if overall strategy is still feasible and desirable. Revise individual assessments as needed.
Program of Work Update, Review, and Adjustment	Reviews and modifies out-year Program of Work, treatment strategy, and/or treatment scenario.
Bioregional or Regional Evaluation	Assesses trends of conditions to inform bioregional strategies.

The SFA process is not just “another planning exercise.” It is also not something that can be easily packaged into a standard “cookbook” because it dynamically responds to local ecological and social data and issues. It is designed to assist local land managers and their staffs in the development and implementation of a strategy designed to accomplish hazardous fuel treatments in a logical and feasible manner. The process can be used to streamline the planning process so that more dollars and resources can be used for project implementation and monitoring.

Successful implementation of the SFA process depends upon understanding and adopting key principles related to: (1) learning in action, (2) data, models and addressing uncertainty, (3) monitoring and feedback for adaptive management, and (4) collaboration and advocacy. This paper will first describe the importance of those principles and then provide a description of the steps involved with the first component of the SFA process, completing Fireshed Assessments.

Learning in Action

Learning in action is the fundamental principle at the heart of the SFA process. Learning in action occurs when highly functioning teams or groups work together effectively to identify and solve problems (Garvin 2000). Such groups are characterized by adaptability and flexibility as well as respect and trust among all members and their peers. Successfully using the SFA process occurs when all participants adopt and apply the tenets of learning in action.

The process of conducting Fireshed Assessments and developing a program of work is an ideal platform for learning in action. During the process, participants work together to identify and analyze problems and explore possible solutions. Natural resource problems, such as addressing wildfire threats, are ideally suited to a learning environment because they possess several key characteristics (Garvin 2000, p 123) as shown below. Participants learn by developing and testing the performance of spatial patterns of treatments in meeting landscape-scale goals and objectives. The knowledge of local conditions, by both managers and the public, greatly facilitate learning in action because discussions can focus on real-world scenarios rather than hypothetical situations.

Problem characteristics that stimulate learning:

- They are significant (the issues matter to people in the organization).
- They are complex (the solution is not obvious).
- They are multifunctional (participants must work across boundaries).
- They involve difficult people issues (the problems are organizational as well as technical).
- They are action-oriented (the goal is to do something, not simply analyze a situation).
- They are ill-structured (participants must frame and define problems as well as solve them).
- They involve surprises (neither the data nor the results are completely predictable).

A key outcome of collaboration and learning in action is bi-directional learning. Agency partners and the public learn about the ecological and social dilemma of managing for multiple resources (Allen and Gould 1986, USDA Forest Service 2004, pp 38-42) and land managers learn about the limits of scientific certainty and public concerns for balancing management of resources.

Data, Models, and Addressing Uncertainty

The SFA process takes advantage of an array of modeling tools to assess the potential of different treatment scenarios to meet landscape-scale goals. The modeling tools facilitate the evaluation of scenarios. However, models are not required to complete the assessment process. Rather, successful completion requires a group to work through a series of data gathering and synthesis steps. The process is focused on asking the right questions at the right scales, rather than a specific modeling tool or suite of tools.

Participants use existing data, recognizing that incomplete or imprecise data are the norm in natural resource management. Definitive cause and effect relationships are rarely known for most ecological systems, particularly related to the effects of management. Without these relationships, it is difficult to know what data to collect that would inform managers on the effects of management actions. The SFA process requires participants to make robust assumptions to fill these knowledge and data gaps using the best available information. Credibility is derived by openly declaring, discussing, and documenting these assumptions, and then moving forward with the assessments. The initial assessments can be based on coarse-scale assumptions, which are evaluated and replaced with finer-scale assumptions as more information becomes available. Sensitivity testing helps to identify which assumptions are likely to have the most influence on outcomes and are good candidates for further refinement. In general, assumptions that affect the short-term and local conditions are more critical to refine than those that affect long-term and landscape outcomes.

Computer models and computer data processing with databases, spreadsheets, and geographic information systems facilitate rapid assessment. A core suite of vegetation attributes are used to generate fuel models, wildlife habitat types, and forest health characteristics. This efficient use of data eliminates discrepancies that would occur if each resource area used different vegetation data to assess outcomes.

By modeling scenarios, experimentation and learning occur before significant resources (time and money) are committed to planning. In addition, competing assumptions can be explored and evaluated before decisions are made on where and how to implement on-the-ground projects. Results from learning in action inform the design of future projects at both the local level as well as at higher levels. Testing and improvement of assumptions also occurs during the modeling phase, planning phase, and implementation phase of a hazardous fuels treatment strategy.

Monitoring and Feedback for Adaptive Management

The challenge of natural resource management is not just the inherent uncertainty related to our current state of knowledge of forest dynamics and the relationship of management to ecosystem functions, but also the range of public knowledge and understanding of these ecological and social systems. This inherent uncertainty contributes to costly delays in implementing projects due to the increased efforts required to document the rationale for risk-taking and explain all of the potential outcomes from both taking an action as well as not taking any action. An adaptive management approach can be a powerful way to address this uncertainty and support collaboration and advocacy.

Both formal Adaptive Management (Kendall 2001, USDA Forest Service 2004, pp 64-88) and informal adaptive feedback are important to refining data and assumptions. Since formal adaptive management studies conducted in a research framework may require many years before findings can be documented, monitoring and evaluation of trends and observational inferences are used as feedback to test and refine assumptions. It is expected that learning occurs during the sensitivity testing mentioned above and that key assumptions are identified for more rigorous evaluation. Since time and funding prohibits studying all potential uncertainties, focused and purposeful evaluation of priority questions must occur and is facilitated by the collaborative environment of the SFA process. After Action Reviews or learning after doing is another important method to gather information and inform future actions and occurs throughout the entire process (Garvin 2000).

The Forest Service has adopted an Environmental Management System[6] (EMS) to systematically review and lessen the environmental impacts of its programs (Executive Order 13148, April 21, 2000). This EMS process uses a "Plan-Do-Check-Act" loop to make incremental and continual improvement. The SFA process follows this same continual improvement loop, using Adaptive Management and adaptive feedback to fulfill the "Check" part of the loop.

Collaboration and Advocacy

Collaboration is the cornerstone for successfully developing and implementing a strategy aimed at changing large wildfire outcomes and meeting other resource goals and objectives. Land managers are expected to work hand-in-hand with other agencies, groups, and individuals in designing and scheduling treatments. Key collaborators include Federal, State, and local government agencies, American Indian tribes, stakeholders--including fire safe councils, communities with Community Wildfire Protection Plans, and adjacent landowners--and interested organizations and individuals. The Healthy Forests Restoration Act of 2003 emphasizes collaboration during the preparation of hazardous fuels reduction projects, and regional efforts such as the Forest Service's Sierra Nevada Forest Plan Amendment directs managers to develop treatment patterns "using a collaborative, multi-stakeholder approach" (USDA Forest Service 2004, p. 49).

Fireshed Assessments, conducted in a collaborative environment, are expected to yield the following key outcomes: (1) development of a broadly supported strategic, spatial, multi-year program of work consistent with landscape-scale goals and objectives, (2) shared involvement, understanding, trust, and coordination among agency partners, stakeholders, collaborators, and the public, and (3) information (including activities and data from other ownerships) that can be used to inform regional and project-scale cumulative effects.

An important aspect in gaining collaborative support is to develop a common set of performance measures that can be used to evaluate the extent that potential strategies meet landscape objectives. Performance of a strategy is evaluated at two scales: (1) at the treatment or stand scale, and (2) at the landscape scale. At the treatment or stand scale, fire effects are simulated by evaluating changes in vegetation attributes based on the type of treatment that might occur at the treatment location. Often prescriptions are defined as a series of treatments. For example, an untreated area may require three entries of prescribed fire with three to four foot

[6] Unpublished data available on Forest Service Washington D.C. headquarters web site: http://www.fs.fed.us/emc/nepa/ems/index.htm

flame lengths over a 15-year period to accomplish desired fuel conditions. For a rapid, coarse scale assessment, it is only this final condition that is modeled to assess performance, while still recognizing that the fuel environment will be different after these interim treatments than in the final outcome. At the landscape scale, fire effects are measured by differences in projected changes in fire spread, in flame length (fire intensity), fire size (acres burned), and the overall efficiency of the treatment pattern. Using the predicted changes in vegetation structure, assessing potential outcomes for other resources, such as wildlife habitat, forest health, and watershed condition, allows a collaborative discussion around balancing treatments with effects to these other resources.

Steps to Conduct a Fireshed Assessment

Assemble Baseline Data

Fireshed Assessment is conducted rapidly using available information and computer models to simulate tree growth, treatments, and wildfires. The models depend upon Forest Service vegetation mapping linked to Forest Inventory and Analysis (FIA) plot data. This linkage allows the Forest Vegetation Simulator (Stage 1973) and Stand Visualization System (McGaughey 2004) to be used to characterize vegetation across the landscape. The vegetation information is updated to account for recent treatments and disturbances (forest mortality from insects, disease, and wildfires), since the vegetation map was created so that fuel model types and habitat types could be assigned. All of this information is managed through a geographic information system using vector and grid data along with databases and spreadsheets. Maps, tables, charts and graphs are all created to display the status of data and facilitate collaborative discussion about the current condition.

Determine Wildfire Threats by Describing the "Problem" Fire(s) Across the Landscape

A key step to building collaborative support for the location and intensity of treatments is to establish agreement on the threat to be addressed. A variety of exploration techniques are used to help identify the fire threat and conditions for problem fires that are of greatest concern for impacts to lives, property, forests, and watersheds.

The nature of the "problem" fire varies widely in different geographic areas, based upon vegetation types, fuels, weather, and topography. In California, "problem" fires are typically the few wildfires that escape initial attack and are therefore the most costly and damaging fires. The "problem" fire in the forested lands in the Sierra Nevada burns where there is an alignment of hot aspects (south and southwest aspects), deep river drainages, and winds. Fires in these drainages often spot across the river and develop multiple fire fronts and access in the canyons is often limited and dangerous for firefighters once the fire escapes initial attack. These fires typically become large over several days of active burning. "Problem" fires in forests in the northern portion of the state are often the result of multiple lightning fires, erratic winds, and an inversion layer resulting in large fires in steep topography with heavy vegetation. In southern California, the "problem" fire situation often occurs when multiple ignitions during Santa Ana wind conditions result in large, wind-driven fires that threaten multiple communities.

Agreeing on the threat in a fireshed allows diverse groups to work together to explore potential solutions and objectively compare different solution strategies. When the group is committed to addressing a problem, opportunities for compromise and rational tradeoff discussions become possible. The Fireshed Assessment process is designed to foster an environment where agreement on the problem and exploration of potential solutions can be done in a manner that advocacy for a solution strategy for a particular location becomes possible.

Examining the assessment area's fire history is the primary method for determining the characteristics of the fire threat. Exploring the size, duration, and spatial pattern of fires that have escaped initial attack in the past provides tremendous insight into the types of fires that are likely to occur in the future. An interagency agreement is in place between the Forest Service's Pacific Southwest Region and the State of California to annually map large fires across the state. Federal fires over 10 acres and state fires larger than 100 acres since the early 1900's have been mapped and are updated annually with new fires. This arrangement provides a rich source of information for evaluating trends in wildfires across the state and is a tremendous resource to land managers.

Calibrate the Fire Models and Validate the Fuels and Vegetation Data

One of the best methods for building confidence in the tools and databases is to use them to reconstruct past fires. Models like FARSITE (Finney 2004) and FLAMMAP (Finney 2005) are used to "re-create" a nearby, recent wildfire through simulations. During this process, local calibration of the fuel model data and weather conditions occurs so that the fire models more accurately simulate real fire behavior. Fuel model validation includes examining the assignment of fuel model, height to live crown, and crown bulk density attributes (Stratton 2004, van Wagtendonk 1996, Weatherspoon and Skinner 1996). Weather condition validation includes appropriate values for wind direction and strength, temperature, relative humidity, nighttime humidity recovery, fuel moisture levels, the presence of inversions, and other parameters that have influenced past "problem" wildfires. In addition to these fuel conditions and fire weather parameters, assumptions about the duration of the fire (number of active burning periods), potential ignition locations, and spot fire rates are documented. The calibration and gaming step allows the group to have an open and transparent discussion concerning the assumptions and limitations associated with fire behavior modeling. This sets the stage for simulating the potential "problem" fires across the landscape.

Delineate Firesheds to Frame the Assessment Area

Based on similarities in historical large fires and potential "problem" fires, the broader landscape (e.g., a national forest) is divided into firesheds. Unlike watersheds, firesheds may vary widely in size depending on how fuel types (e.g. grass, brush, or forest) and local topography (e.g. steep canyons, foothills, or high elevation/alpine) and weather (e.g. hot south-facing slopes, cool drainages and north slopes, upslope winds, or wind chutes) influence potential fire behavior. Fireshed boundaries are also influenced by the values they contain (e.g. communities in the wildland urban interface, domestic water supplies, high value infrastructure, habitats for wildlife species of concern, or unique natural areas) and by fire management opportunities (e.g. full suppression or wildland fire use). Firesheds cover large areas, usually encompassing several times the size of the largest potential problem fire. The purpose of delineating firesheds is to identify areas that are sufficiently large to assess the effectiveness of fuel treatments at changing the outcome of a large wildfire. Fireshed boundaries are not fixed and are defined at a coarse scale. Fireshed boundaries will change over time as fuel conditions and the characteristics of the fire threat change in response to management and natural changes in the landscape.

Develop a Treatment Pattern and Prescription Scenario Aimed at Reducing the Negative Effects of the "Problem" Fire

The approach for modifying landscape-scale fire behavior used in the national forests of California is anchored in the concept of treating a fraction of the landscape in the right places to achieve intended modifications in wildland fire behavior. The landscape-scale fire modification strategy is based on the premise that disconnected fuel treatment areas arranged in an appropriate overlapping pattern interrupting the general direction of fire spread are theoretically effective in reducing overall fire spread. Finney (2001) suggests that fire spread rates can be reduced, even outside of treated areas, as a fire is forced to flank areas where fuels have been reduced or otherwise modified. From a mathematical standpoint, Finney calculates that strategically treating a small proportion of the landscape (20 to 30 percent) can have the same change in landscape fire spread rates as randomly treating higher proportions of the landscape (60 percent). Theoretically then, for a given burning duration, a wildfire in the treated landscape should be smaller and have more areas burning at lower intensity when compared to the same wildfire burning in the untreated landscape. While fire suppression is not actively included in the simulations, logically, fire suppression opportunities should be greater where fires are burning less intensely and with a lower rate of spread.

The most effective pattern would be to align overlapping treatments oriented to the direction of expected fire spread. For each fireshed, a default treatment pattern is identified considering the expected fire behavior under "problem" fire conditions and the size a fire can get before it typically escapes suppression on initial attack. Using this pattern as a template, the assessment team identifies potential treatment areas, considering operational feasibility (e.g. equipment access, steep slopes and machinery limitations), environmental sensitivity (e.g. habitats, soils, archaeological sites), and logistical constraints (e.g. proximity to private lands, costs, limitations on operating season). The local knowledge of participants is critical in ensuring that all identified treatment areas are physically feasible to implement and reasonable in terms of costs and likelihood of accomplishment since the efficiency and effectiveness of the treatment pattern is evaluated under the assumption that all treatments are actually implemented.

Each treatment location is assigned a treatment prescription designed to create more desirable fire behavior (Agee and Skinner 2005). Specifically, surface fuels are reduced, crown base height is increased where ladder fuels are a problem, and canopy fuels are reduced as needed to reduce the potential for crown fire spread (Stephens 1998, Agee et al. 2000, Scott and Reinhardt 2001, Agee and Skinner 2005). Both the treatment location and treatment prescription are guided by the local management direction that may limit the extent of changes allowed in the diameter of trees removed or canopy cover that must be retained. These changes are simulated by changing the fuel models within the treatment areas.

The combination of the treatment pattern and individual treatment areas with assigned prescriptions constitute a simulation scenario. Each scenario generally follows a theme that applies a distinct spatial strategy to attempt to solve the problem situation. Usually, several simulation scenarios, each with different spatial strategies, are tested. These scenarios are not alternatives in the sense of the National Environmental Policy Act (NEPA), they are meant to allow exploration of short-term and long-term effectiveness, efficiency, and feasibility of different courses of action. They will help to frame alternatives to be more formally evaluated at a later time as individual projects are ready for site-specific evaluation.

This process is accomplished by projecting geographic information system displays onto a whiteboard using a laptop computer and LCD projector. The collaborative group then uses dry erase pens to delineate potential treatment areas which are then captured by heads-up digitizing. During this process, all members of the group are encouraged to participate in drawing potential treatment areas and the entire group is encouraged to openly discuss the perceived pros and cons of a potential treatment. By rotating the drawing amongst all group members, different perspectives on treatment considerations and design are brought to the discussion. This can be extremely powerful for groups that are not used to working in a truly integrated interdisciplinary manner and when diverse stakeholders participate in the process. To ensure that this step moves quickly, the group must consciously remember that this is a coarse scale assessment and is not site-specific project planning.

Test and Adjust Treatments and Consider Additional Scenarios

Understanding how fires are projected to spread and affect vegetation, soils, air, and water is very important in evaluating the performance of a scenario. Fire effects are modeled so that the projected differences between several possible outcomes can be characterized. At a minimum, four outcomes are assessed for each scenario, as displayed in *table 2.*

Table 2— *Comparison Outcomes for Scenario Assessment.*

	No Wildfire	**Wildfire**
No Treatment	No treatment and no wildfire occurs. Vegetation growth simulated for 20 years.	No treatment, but wildfire occurs. No treatment after wildfire and post-fire vegetation growth simulated for 20 years.
Treatment	Treatment occurs and no wildfire follows. Post-treatment vegetation growth simulated for 20 years.	Treatment occurs and then wildfire occurs. No treatment after wildfire and post-fire vegetation growth simulated for 20 years.

The FARSITE and FLAMMAP models generate the key parameters of flame length, fire type, rate of spread, and fire size. This information is overlaid with vegetation information and used to calculate projected vegetation changes. The Fire and Fuels Extension (Reinhardt and Crookston 2003) of the Forest Vegetation

Simulator (FVS) (Stage 1973) and the First Order Fire Effects Model (FOFEM) (Reinhardt et al. 1997) use flame length and fire type to predict mortality of the dominant tree species found in the vegetation database. FVS is used to predict the additional mortality that may be indirectly caused by fire--for example, from fire damage or post-fire insect infestations.

The FVS system (Dixon 2003) and the Stand Visualization System (SVS) (McGaughey 2004) are used to describe and display forest characteristics in both tabular and graphic formats. This base information can then be used to evaluate many different resource effects. For example, forest health is examined by evaluating stand structure and stand density parameters (Reineke 1933), and wildlife habitat is evaluated by cross-walking the vegetation data into the California Wildlife Habitat Relationship habitat types (CA Dept. of Fish and Game 2002) to assess changes in the amount of breeding, foraging, and dispersal habitats for wildlife species of interest. This same base vegetation data can also be used to evaluate cumulative watershed effects, scenic visual quality, and other vegetation-based changes of interest to the collaborative group.

Once an initial assessment is done, the assessment team considers making adjustments to treatment location and treatment prescriptions based on what they learn from the fire simulation exercises. Often teams find that there are “holes” in their pattern of treatments. The FARSITE modeling can identify areas where the distance between treatment areas is too great, or is oriented in the wrong direction relative to slope or predominant winds, allowing a potential “problem” fire to become too large before it bumps into a treatment area. The FLAMMAP modeling can identify areas where the fire is likely to be a surface fire and where it is likely to be active and passive crown fire types. If the modeled “problem” fire could get large but is mostly of a surface fire type, then additional treatment areas might not be needed. In other areas, the team may find that the shape of a treatment area could be modified so that fires might not burn through them as fast or spot over them as easily. In other areas, there may be limited or no opportunities to feasibly develop treatments. The assessment team uses fire modeling to learn how fire spreads across their landscapes under many different wind conditions, ignitions patterns, and fire durations. Each round of simulation provides more insight into the potential pattern of treatments.

In addition to the treatment location, the assessment team can adjust the treatment prescription. Selecting a different prescription changes what is modeled to be removed and what is left. Fires are then modeled against these changes and the projected results are evaluated. The fire gaming is a process that requires multiple iterations, each time adjusting treatment locations, changing prescriptions, and evaluating scenarios based upon the collective learning of the collaborative group.

Discussion

Fireshed assessment involves a rapid, iterative process to guide interdisciplinary teams along a logical, step-by-step process, to design, test, and schedule fuels and vegetation management projects in order to reduce landscape-level fire hazard while achieving multiple resource objectives. Collaboratively defining the problems to address in a landscape allows agencies, working directly with the public, to develop scenarios and use a process of gaming to evaluate and compare the tradeoffs between strategies. Using the concept of learning in action, assumptions and data limitations are noted, and computer models such as FARSITE, FLAMMAP, and the Forest Vegetation Simulator are used to simulate and evaluate changes across the landscape.

Once individual fireshed assessments are completed across the entire landscape (e.g., a national forest or entire management area), the scope of the workload can be assessed. Individual treatments are grouped into proxies for projects that could be implemented in a given year. This allows costs, outputs, and cumulative effects to be aggregated for each proxy project.

Then, based upon factors such as expected funding, organizational infrastructure, treatment costs, industry and contractor capacity, community support, and administrative and regulatory limits, proxy projects can be grouped into a program of work. The program of work is not simply a list of upcoming projects with generalized project descriptions and locations. Instead, it is a spatially-explicit road map of where and when the vegetation and fuels treatments that implement an overall strategy are likely to occur. In addition, the program of work provides the rationale for (1) why specific areas are slated for treatment, and (2) the timing of each project in the overall schedule. Typically, the program of work describes details for the first five years, but it is grounded in a schedule to complete all of the anticipated treatments. Typically, this spans about 10-20 years based upon expected budgets and limits on the amount of treatments that can be physically accomplished each year. The program of work not only shows where activities are planned, it also shows areas that may be either deferred from treatment or approached under a different fire and fuels strategy, such as wildland fire use.

Once a spatially explicit 5-year program of work is completed across the forest, the performance of the schedule, in meeting forest, regional, and national goals, objectives, and impacts, can be assessed over time. The performance and impact results can inform the need for changes or refinements to the schedule. The program of work should be robust enough to: (1) allow land managers to make adjustments as budgetary, environmental, legal, and social conditions change; (2) game different outcomes as a result of these adjustments; and (3) determine when adjustments or a major change in the overall strategy should be evaluated. Decision makers should be able to communicate how the program of work is expected to change outcomes for potential wildfires, forest health, habitats, and watersheds both internally within the agency as well as with external groups.

The individual Fireshed Assessments and program of work can then be used to assess the projected effectiveness of treatments to provide protection to communities as well as estimate changes to other resources, such as wildlife habitats and watershed condition. Because these models can simulate changes over time, they are an ideal platform to assess projected cumulative effects at scales from the landscape to a forest to a bioregion. The ability to rapidly integrate adaptive feedback from participants helps build confidence in the process, which is an important first step at re-gaining the public’s trust in management of their lands.

The ideal situation would be where fuels are compatible with fire as a disturbance agent over space and time, such that fire plays its ecological role in shaping and maintaining vegetation and the social effects of fire in the environment are acceptable. This initial strategy to use strategically placed treatments is intended to be a short-term “triage” to moderate the rate of forests affected by large, uncharacteristically severe wildfires. This is designed to provide the opportunity for land managers to devise long-term management strategies that address the larger, holistic social and ecological issue of forest health and forest sustainability.

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Integrating Stand Density Management With Fuel Reduction[1]

Joseph W. Sherlock[2]

Abstract

The widespread effort to reduce fuel hazards in western forested ecosystems places significant emphasis on surface and small ladder fuels. Changes in canopy density, for purposes of either reducing potential crown fire impacts or insect/pathogen-related mortality, are less frequently considered. Providing a sound basis for treating more than surface and small ladder fuels is possible and can be smoothly integrated with most fuel reduction proposed actions. This paper illustrates a strategy that has potential to accomplish this goal.

Introduction

In January of 2004, the Record of Decision (ROD) for the Sierra Nevada Forest Plan Amendment Supplemental Environmental Impact Statement (SEIS) (USDA Forest Service 2004) was signed. This was the second ROD associated with approximately 25 years of analysis and evolving management objectives. The first (USDA Forest Service 2001), signed in 2001, was determined to be difficult to implement, pointing to possible failure in meeting the core goals. In 1993, the focus was largely limited to issues surrounding the protection of California spotted owl habitat. Then, in 1998, concerns were expanded. In the Sierra Nevada Forest Plan Amendment Final Environmental Impact Statement, one of the identified problem areas was fire and fuels management.

The 2004 ROD, while continuing to emphasize wildlife habitat conservation, added needed flexibility to address the increasingly hazardous fuel environment. Treatments modeled in the SEIS rely heavily on surface and ladder fuel reductions, accomplished by both mechanical and prescribed fire. Canopy fuel reduction is limited by SEIS Standards and Guidelines that set standards for residual canopy cover.

The major Standards and Guidelines that guide project design are essentially characterized by a canopy cover lower limit. Depending on land allocation, the limit is generally set at 50%, although 40% is acceptable in limited situations. For perspective, in Sierra Nevada forests, typically-applied thinning from below would likely yield canopy closures closer to 40% than to 50%.

Management direction focuses treatments toward Wildland Urban Interface (WUI) areas as well as the adjacent landscapes. Despite existing and threatened lawsuits, general support for reducing hazardous fuel remains high. There remains,

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however, controversy regarding treatment extent and intensity. In an effort to avoid potential controversy and get fuel reduction projects accomplished, canopy fuel reduction may be overlooked or minimized to ineffective levels. When this occurs, canopy fuel and stand density are often left at undesirably high levels. Project planners can benefit from a strategy that provides for compliance with the ROD as well as a clear linkage to the developing science basis for canopy fuel and stand density reduction.

Strategy

This paper describes an effective, ROD compliant strategy for increasing tree vigor while ensuring fuel hazards are reduced. The following sections describe an approach that relies on a combination of treatment unit design, density management, and tree selection principles.

The basis for fuel reduction treatments is well established (Graham et al. 2004). Reductions in surface fuel are almost always an essential step in a successful project. Significant reductions in fire intensity are provided by effective surface fuel treatments. Reducing ladder and canopy fuel, without surface fuel reductions, may actually increase wildfire hazard. The following discussion will focus on the related and complementary management of the larger tree portion of the fuel ladder and the canopy fuels.

Treatment Unit Design

The core principle related to treatment unit design is that a strategic pattern of effective fuel reduction will provide for landscape-scale benefits. To be regarded as strategic, treatment units need to be spatially located in a partially overlapping pattern, with a general alignment across the predicted fire spread direction. To be effective, significant reductions in rate of spread must occur within the treated areas, resulting in an increase in flanking fire behavior around the treatment units, while fire moves slowly within it. The basis for this strategy has been developed by Finney (2001).

Assuming an effective strategic placement, significant fuel reduction inside the treatment unit is essential. While multiple Standards and Guidelines limit forest structure change, the SEIS assumes effective treatment of surface and a portion of the ladder fuels. In practice, excepting the steepest slopes and complicated property boundaries, reducing surface and small ladder fuels is generally feasible, given adequate budgets.

The geographic area for assessing ROD compliance is the treatment unit. Its size can vary and was expected to vary from less than 150 to several hundred acres. In the modeling associated with the SEIS, canopy cover standards typically limited treatment intensity. However, specific values are not expected to be met on every acre, but, rather, over the treatment unit. When attempting to meet both fuel and stand density reduction objectives, project implementation can be adjusted to meet forest structure objectives at the individual tree or group level. I refer to this as the tree neighborhood level. The tree neighborhood is the geographic area where inter-tree competition is assessed. It can be thought of as a single tree and all the neighboring trees that influence its growth (vigor) environment.

An increasing body of evidence is accumulating to validate that effective fuel treatments play a critical role in reducing the adverse effects of wildfire (Landram and Hermit, 1996). In the specific case of the 2003 Cone fire, the incidence of fire-caused tree mortality rapidly declined as fire moved into treated areas (Skinner et al. 2002, Ritchie in press, Nakamura 2002).

Figure 1 illustrates a stylized approach to treatment unit design that can meet unit average canopy cover standards and guidelines. In this example, fires approaching from the southwest or north are received by effective fuel treatments, incorporating surface, ladder, and canopy fuel reductions. Fire behavior can be expected to change in response to these reduced fuel levels. Active or passive crown fire would be expected to shift to surface fire as it enters the treatment unit. Surface and small ladder fuel reductions in the core would decrease the likelihood of torching throughout most of the remainder of the treatment unit, despite higher canopy cover levels, especially if the spatial connectivity of high density forest is low. Higher residual canopy cover, on other portions of the unit, can be designed to obtain the desired treatment unit average.

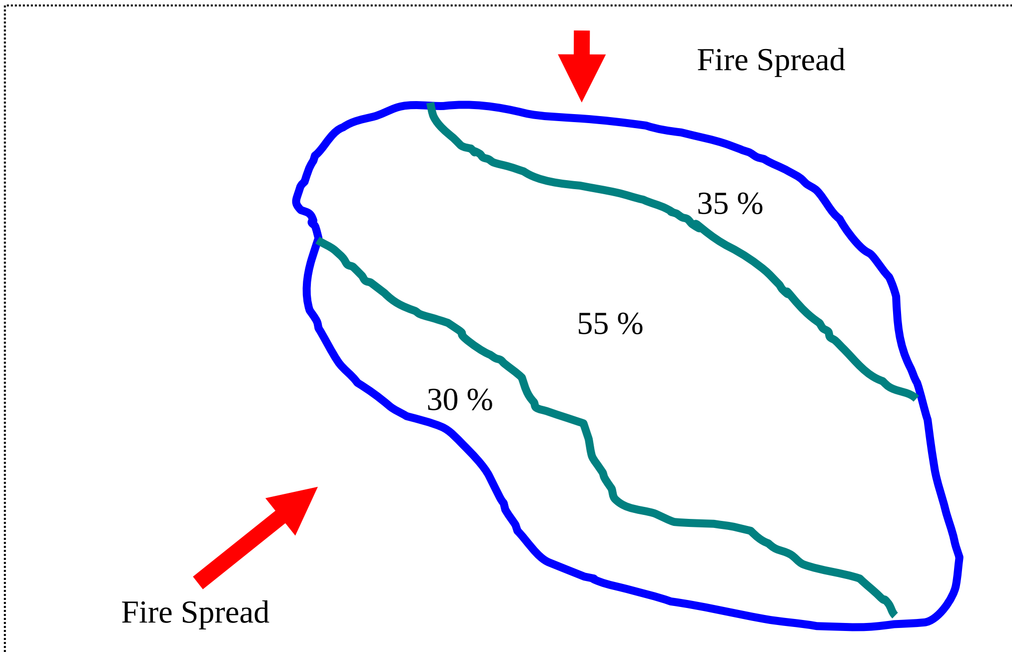

Figure 1—Strategy for canopy cover design that provides for increased probability for changing crown fire to surface fire upon entry into the treatment unit.

Figure 2 illustrates a variation of the above strategy, attempting to exploit the typical vertical and horizontal pattern variation of mixed conifer forests. Actual zones may be more highly varied than illustrated in this simplified graphic. This approach continues to provide for a reduced fuel profile along at-risk unit boundaries, as well as providing for tree size and arrangement patterns within. Important wildlife habitat features can be cultured at the tree neighborhood scale without placing extensive areas of high-density forest at risk of loss from wildfire.

Density Management

After treatment unit design, identifying a residual density that reduces crown fire spread hazard and unacceptable insect/pathogen-related mortality will ensure that the

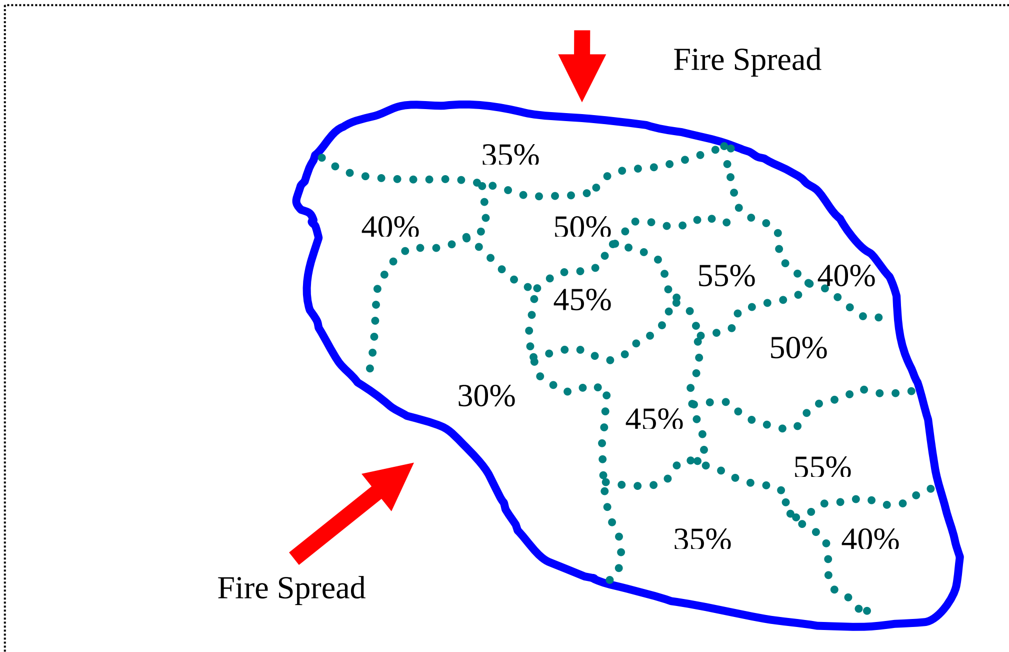

Figure 2—Strategy revision that provides for increased probability for changing crown fire to surface fire upon entry into treatment unit. In this case, variations are more complex, favoring habitat features as they exist and/or providing for even greater reductions in canopy fuel levels in places that will increase treatment efficacy.

post-treatment conditions meet the composite goals of the ROD. Given the desire to restore higher levels of large diameter trees, steps to reduce the potential for loss seem essential. Stand density index (SDI) estimates can be used to indirectly assess individual tree vigor at the per acre scale. The use of crowning and/or torching indices can add fuel hazard additional perspective to stocking levels.

While maximum SDI levels have been estimated for many Sierra Nevada species, specific data identifying critical, or threshold, values are not common. An SDI management range has been suggested by multiple researchers. This range is meant to characterize densities between the onset of competition and the lower limit of the zone of imminent mortality. Dean and Baldwin (1996) describe a management range between 20-30 and 50-55 percent of maximum. Similarly, Long and Shaw (2005) describe the range as 35 to 55-60 percent. Mortality projections, made by the Western Sierra Nevada variant of the Forest Vegetation Simulator (FVS WESSIN), initiate density-related mortality at 55% of maximum and peak mortality levels at 85%.

Ponderosa pine stand density appears to be regulated by *Dendroctonus* bark beetles more than by competition-induced mortality (Oliver 1995). Bark beetles define ponderosa pine's maximum SDI at 365, with a threshold for the zone of imminent mortality at 230 (Oliver and Uzoh 1997). For ponderosa pine stands, reducing existing densities to levels that remain below or near 230 appear to provide for the highest assurance that desired trees persist over time.

Limited data for natural white fir and red fir stands indicate a maximum density of 800 and 1,000, respectively (Oliver and Uzoh 1997). In contrast to ponderosa pine, mortality appears to be driven by intertree competition. Oliver and Uzoh (1997) suggest that, currently, the quantity of this data may not be sufficient to justify the replacement of other recommendations.

In conjunction with an effort to improve the characterization of mortality risk, Forest Inventory and Analysis (FIA) data from California plots has been summarized for the period from 1980 to 2000[3]. In contrast to earlier efforts, individual plots were assessed, instead of cluster plot averages. Plots were stratified, based on species dominance (≥80% by basal area), and maximum SDI levels were identified for several species. *Table 1* lists maximum SDI and the associated lower limit of self-thinning values. Many of the maximums derived from the FIA data are higher than those previously identified (and used in FVS WESSIN). This was expected, as the recent calculations are based on individual points and not from cluster plot averages. Also, while the sheer quantity of plots used in the FIA analysis easily outnumbers those available to Oliver and Uzoh, the nature of the stands is fundamentally different. High levels of variation exist within the FIA plot data; in particular, significant differences in age and development history make this collection distinct. Oliver and Uzoh used data from even-aged planted and natural stands. For even-aged ponderosa pine stands, an SDI value of 230 should be regarded as a threshold, beyond which mortality levels can be expected to increase. For other species, the general principles underlying the lower and upper limits of maximum stocking levels should be used as a guide. Maintaining stocking near the lower limit of self-thinning should provide for minimum mortality losses. The calculated FIA maximums should be regarded as preliminary and, most likely, more suitable at the tree neighborhood scale.

The use of density measures, that are characterized at the acre level, need to be further supported by tree-level assessments, as described in the following section. The vigor status of individual trees is a function of its position within its local neighborhood, and that is where the appropriate focus for final decisions should be.

Tree Selection

The final component of the strategy is the identification of individual trees that are intended to satisfy the goals of treatment at the tree neighborhood scale. The flexibility provided for canopy cover variability allows silviculturists to design treatments that are responsive at this scale. Recognizing that the applicable canopy cover standard and guideline is assessed at the treatment unit scale, portions of higher canopy cover need to be recognized and included in the overall strategy. Specific sites, where, historically, higher canopy cover was more likely to have persisted, may be a suitable approach. This may lead to the designation of portions of north- and/or east-facing slopes, as well as selected lower slopes on all aspects. Combined with site-specific identification of key habitat features, this approach may provide for environmental conditions that provide for even higher levels of suitable habitat than would be achieved with a more generalized approach.

This last aspect provides for the tree-specific decisions that can maintain or improve the vigor of individual trees. The use of risk-rating systems, crown classes,

[3] Unpublished data on file, F. Michael Landram, Regional Silviculturist, Pacific Southwest Region, Vallejo, California.

Table 1—*Maximum SDI values as indicated by Oliver and Uzoh, FVS WESSIN, and FIA data, with associated lower limit of self-thinning values.*

Species	Maximum SDI			Lower Limit of Self-Thnning		
	Oliver & Uzoh	FVS (WESSIN)	FIA	Oliver & Uzoh	FVS (55%)	FIA (55%)
Abies concolor	800	759	900		417	495
Pseudotsuba menziesii		547	800		301	440
Pinus ponderosa	365	571	650	230	314	358
Pinus jeffreyi		571	600		314	330
Abies magnifica	1,000	800	1,050		440	578
Calocedrus decurrens		706	700		388	385
Quercus chrysolepsis			750			413
Quercus kelloggii		382	550		210	303
Pinus contorta			850			468
Pinus lambertiana		647	400		356	220

and individual tree characteristics can be combined to provide an increased level of confidence when identifying which trees to favor.

Risk-rating systems, developed by Dunning, Keen, Salman and Bongberg, and Ferrell, can be used to guide how choices are made between individual trees (Dunning 1928, Miller and Keen 1960, Keen 1943, Salman and Bongberg 1942, and Ferrell 1989).

In 1928, Duncan Dunning published the first classification system for ponderosa pine in the Sierra Nevada. Subsequently, Keen expanded the classification from seven to sixteen classes (*fig. 3*), refining Dunning's effort to distinguish important distinctions by age, crown size, and dominance (Keen 1943). *Figure 4* illustrates the work of Salman and Bongberg, who, in 1942, published a four class risk-rating system. These efforts attempted to interpret tree vigor in an effort to predict susceptibility to pine beetle-caused mortality. Each of the illustrations included in *figs*. *3*, *4*, *5* were designed to enable field interpretation of crown characteristics regarded as key indicators of tree persistence.

While the use of these systems cannot provide absolute certainty, the underlying principles contribute to the set of factors that can be used when making informed selection decisions. Significant discussion regarding these and other systems is well documented (Miller and Keen 1960) and the reader is referred to this publication for additional information. Despite the absence of a perfect classification system, a focus on the crown characteristics will reveal important information about the vigor status of the individual tree. Combining several of the crown characteristics will likely increase the probability of selecting trees most likely to thrive.

A risk-rating system for red fir and white fir growing in northern California adds additional information related to tree selection (Ferrell 1989). Three factors, all crown-related, can be used to predict mortality *(fig. 5).* The most useful characteristics were live crown percent, crown density, and ragged percent. This modern-day use of crown conditions seems to reinforce earlier efforts to recognize tree vigor by visual characteristics.

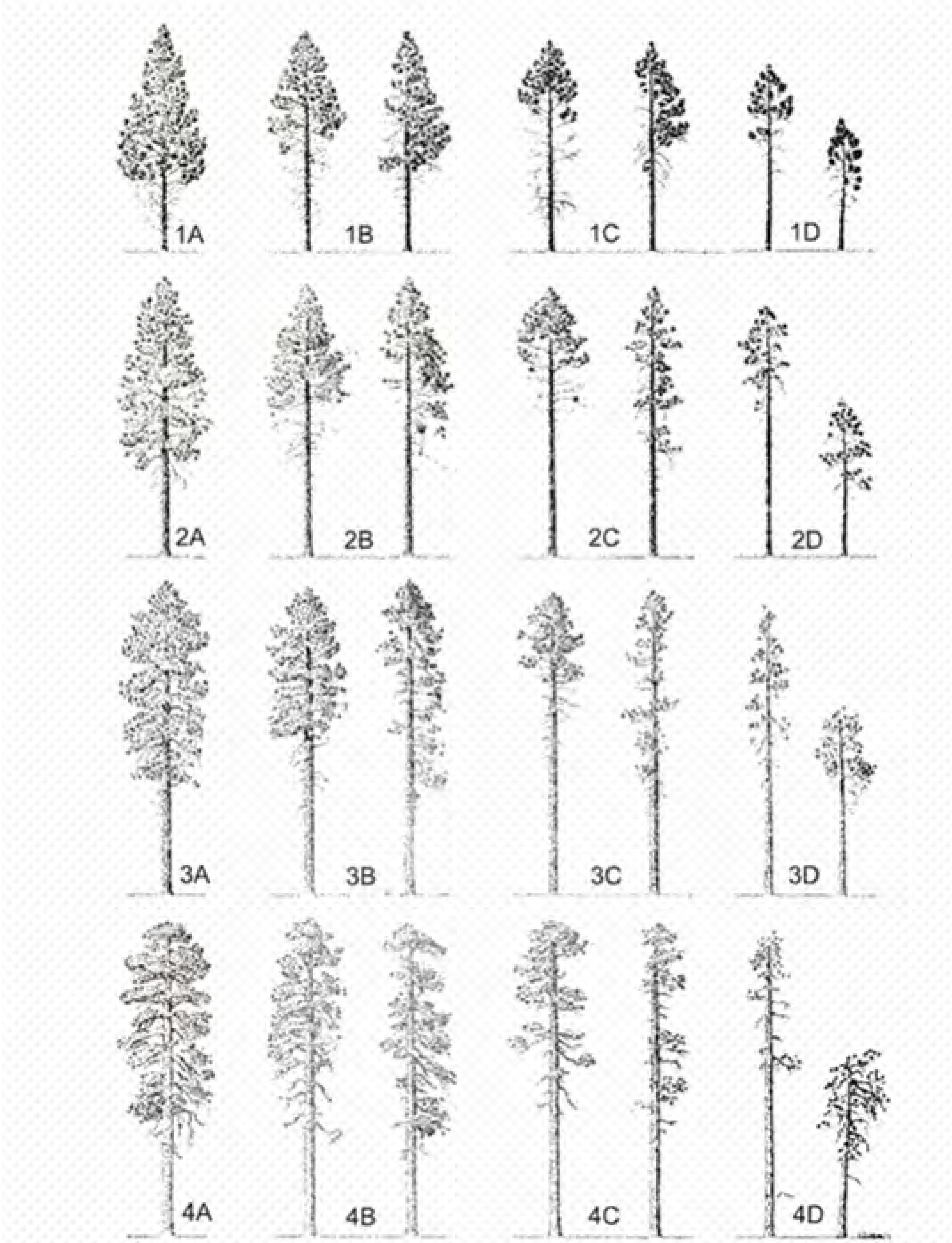

Figure 3—A ponderosa pine tree classification based on age (increasing from 1 to 4) and vigor (decreasing from A to D), from Miller and Keen 1960, page 178.

Figure 4— Degree of risk in ponderosa pine tree, from Miller and Keen 1960, page 183.

Crown classification adds additional tree-specific information to assist with the identification of trees capable of sustained and vigorous growth. The linkage between health and crown class is regarded as very high, when assessing trees within a cohort (Smith 1962). The Forest Inventory and Analysis (FIA), National Core Field Guide defines crown classes as displayed in *fig. 6.* This, and similar illustrations, attempt to differentiate tree crowns based on the relative share of sunlight and growing space afforded to individual trees. Favoring crown classes that have already exhibited an ability to acquire a larger share of the light resource, especially when combining

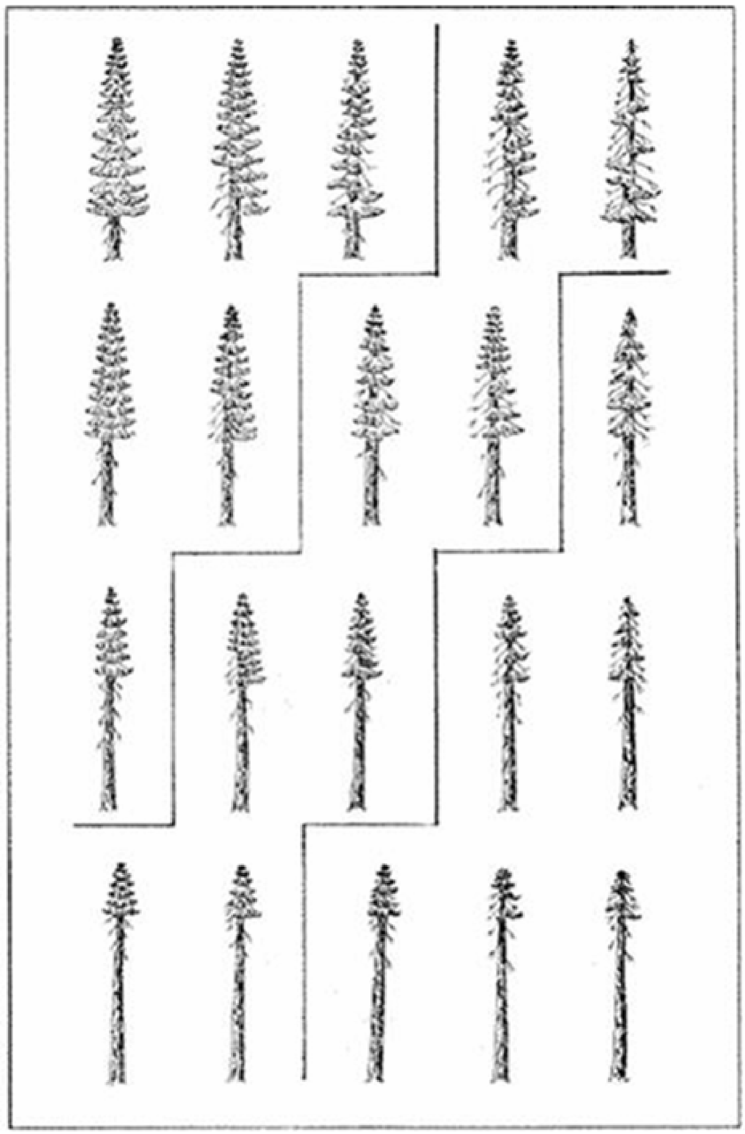

Figure 5—Risk classes for rapid visual prediction of 10-year mortality in California red fir and white fir: (left to right) low, medium, and high risk (Ferrell 1989, page 9).

indicators described above, will likely provide for higher levels of resilience and sustained vigor. Altering inter-crown spacing will allow for increased persistence of the lower crown, maintaining or increasing crown ratios and photosynthetic area.

The use of traditional external indicators of tree vigor can also be relied on to assess the status of individual trees. Crown indicators include shape, patchiness, and ratio. Stem indicators include bark fissure depth and color. Needle characteristics, such as length, color, and retention can be used as well.

Utilizing the combined strength of risk-rating systems, crown classification, and tree characteristics can increase the likelihood of selecting trees most capable of sustaining vigorous growth rates and to benefit from any associated fuel reduction treatments.

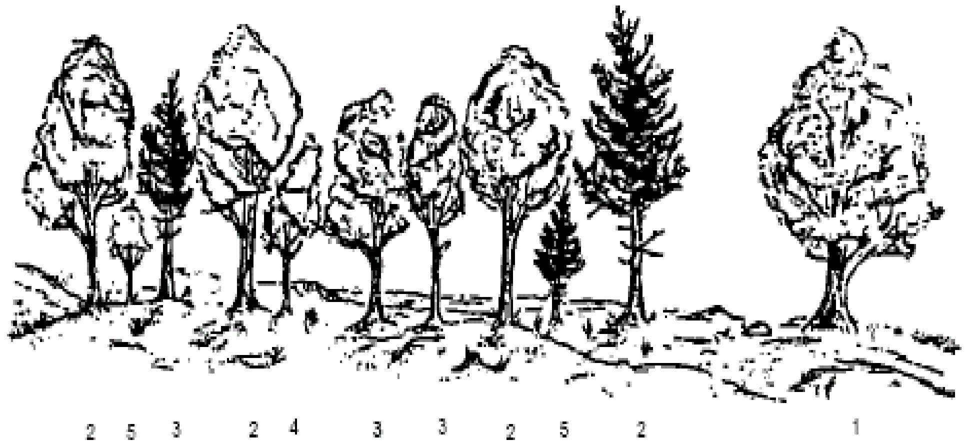

Figure 6—Examples of FIA crown class code definitions. (1 = open grown, 2 = dominant, 3 = co-dominant, 4 = intermediate, and 5 = overtopped. Taken from the FIA Field Guide for Phase 2 Measurements (2004), page 76.

Summary

The 2004 ROD provided additional flexibility needed to be more successful in meeting management goals. Fuel reduction projects that focus on surface and small ladder fuel attract relatively few appeals and are making progress, especially adjacent to forested residential areas. In some cases, larger fuel ladder and/or canopy fuel reduction projects, implemented via timber sale contracts, may be at risk of appeal or lawsuit. It may be possible to increase the level of advocacy among those who threaten appeals or lawsuits by more clearly describing the basis for management action. The threat of harm by wildfire appears to have motivated many to advocate actions that reduce hazards. Perhaps the adverse effects of stand density hazards need to be more clearly described. The loss of forest structure and composition, especially if the larger diameter trees are reduced, appears to be in clear opposition to the desired conditions of current Forest Plans. Managers may benefit from the described strategy as they strive for successful implementation of projects.

As canopy cover is assessed at the treatment unit scale, which can be several hundred acres in size, variation would be both inevitable and desirable. Using this allowance can provide for low-density environments to favor key trees, or groups of trees, as well as provide a fuel environment that will not support crown fire. Likewise, higher-density tree environments can be spatially intermixed to provide for higher-density habitat needs in places that minimize the risk of wildfire loss. Although Standards and Guidelines are commonly expressed as per acre criteria, treatment unit objectives are met by the summation of multiple tree neighborhood decisions.

Using the flexibility that exists, projects that reduce density and, simultaneously, reduce canopy fuel are possible. Unit design strategies can be implemented to assure that adverse landscape-level fire effects are reduced. At the treatment unit scale, strategic variations in density are likely to provide for beneficial changes in fire behavior, with reductions in acres affected by crown fire. An informed choice of residual stand density will increase the likelihood that excessive insect, pathogen, and/or intertree competition-related mortality will not cause adverse effects to wildlife habitat and that individual trees will be more resilient in the face of wildfire, drought, and other environmental stress agents. At the tree neighborhood scale, the use of tree selection criteria, as described above, will likely ensure that the remaining trees persist, providing for higher levels of large-diameter trees as well as intact forest habitat.

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Reintroducing Fire to the Oak Forests of Pennsylvania: Response of Striped Maple[1]

Patrick H. Brose,[2] Gary W. Miller,[3] and Kurt W. Gottschalk[3]

Abstract

We studied the role of fire exclusion in the formation of striped maple (*Acer pensylvanicum*) understories in mixed oak (*Quercus* spp.) forests of Pennsylvania and the response of this species to the reintroduction of fire. Prescribed fires were applied to parts of three mixed oak stands and data from the burned and unburned portions were compared. Increment cores and basal cross sections were collected from the unburned portions to document the dates when the different species had regenerated. In all three stands, the striped maple understories originated in the 1950s and 1960s when fire was no longer a disturbance. The prescribed fires initially reduced density of striped maple by 25 to 50 percent with delayed mortality increasing this rate to more than 80 percent. These data suggest that prescribed fire could be a viable means of controlling striped maple in mixed oak forests.

Introduction

There is growing appreciation and understanding of the important role periodic, low-intensity, surface fires played in the historic dominance of mixed oak forests throughout eastern North America, including the mid-Atlantic region (Abrams 1992, Brose et al. 2001, Yaussy 2000). This fire regime was largely the result of American Indian burning practices and, in conjunction with other environmental factors, helped perpetuate mixed oak forests on a wide variety of soils, especially mesic upland sites. The advent of effective fire control policies and practices ended the periodic surface fire regime of the mid-Atlantic region, like they did in the Southeast and the Interior West. However, unlike those other regions, the exclusion of fire did not translate into an increased loading of hazardous fuels that contributed to catastrophic, stand-replacing wildfires. Rather, the cessation of periodic surface fires in the mid-Atlantic region led to a new forest succession pathway, one in which fire-sensitive, tolerant shrubs and trees invade and eventually impede successful oak regeneration in mixed oak forests.

One beneficiary of the cessation of periodic surface fires is striped maple (*Acer pensylvanicum*). Striped maple is a small- to medium-sized, shade tolerant tree found from Nova Scotia west to the Great Lakes region and south along the Appalachian Mountains to North Carolina (Gabriel and Walters 1990). Within that range, it generally occurs in northern hardwood forests and is most common on cool, moist

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slopes. However, it is being found more frequently and abundantly in mixed oak forests, an environment from which it was historically absent or sparse.

Striped maple lives only about 40 years, but can subsist as a small seedling for 40 years before that (Hibbs 1979). It is a prolific seeder and, in conjunction with its seedling banking strategy, can develop high density populations in forests. When such populations develop, striped maple becomes a serious silvicultural problem as it casts a dense shade on the forest floor that impedes oak seedling survival and growth. In Pennsylvania, striped maple is considered the most troublesome woody understory species that competes against oak regeneration (pers. comm. Gary Rutherford, Silviculture Section Chief, Pennsylvania Department of Natural Resources, Bureau of Forestry).

Glyphosate-based herbicides often are used to control striped maple when its density becomes an obstacle to forest regeneration (Horsley and Bjorkbom 1983, Marquis et al. 1992). However, there are times and places when herbicide use is not possible, so there is growing interest in using prescribed fire as an alternative control method. Striped maple exhibits several attributes that suggest it might be quite sensitive to fire. Striped maple bark is quite thin regardless of stem diameter; its root collar (the location of dormant buds) is relatively high in the litter layer; and its root system is small and shallow. Striped maple leaves also emerge earlier in the spring than many other species. As a result, root carbohydrate reserves are depleted earlier thus rendering striped maple susceptible to surface fires for a longer period.

Surprisingly, literature on the effects of fire on striped maple is sparse. Swan (1970) compared burned and unburned northern hardwood stands in southern New York. He found unburned stands to have five times more striped maple than those that had been burned. Unfortunately, fire behavior was unknown and pre-fire striped maple density between stands was not documented. Conversely, Collins and Carson (2003) reported that nearly all striped maple sprouted vigorously following prescribed fires in West Virginia. Again, fire behavior was poorly described.

The objectives of this study were twofold. First, we wanted to document the establishment timelines of the striped maple populations in mixed oak forests, especially in regard to the establishment of the overstory oaks. Second, we wanted to determine whether striped maple densities increased or decreased following prescribed burning. Understanding both of these aspects of striped maple ecology will help foresters deal more effectively with the species when it poses a regeneration obstacle.

Methods

Study Sites

This study was conducted between 2002 and 2005 on three Pennsylvania State Forests: Bald Eagle, Clear Creek, and Moshannon. The Bald Eagle State Forest is located in the Ridge and Valley region of central Pennsylvania. The study site was a 10-acre stand situated at the bottom of an 18 percent, north-facing slope. Elevation was approximately 1400 feet. Soil was a stony loam (Typic Fragidult) formed from sandstone alluvium (Braker 1981). Consequently, it was moderately acidic, fertile, and well drained. Severe gypsy moth (*Lymantria dispar*) defoliation occurred there in the late 1980s and early 1990s, resulting in substantial overstory mortality. Salvage logging occurred in late 1993. The remaining overstory trees resembled a shelterwood stand and had a relative density (a measure of stocking) of 54 percent as per SILVAH stocking criteria (Marquis et al. 1992). Common overstory species included black oak (*Quercus velutina*), chestnut oak (*Q. montana*), northern red oak (*Q. rubra*), white oak (*Q. alba*), and red maple (*A. rubrum*). A dense sapling layer, more than 20 ft^2 of basal area, formed in response to this reduction in canopy cover and included striped maple, sweet birch (*Betula lenta*), red maple, and witch-hazel (*Hamamelis virginiana*). The forest floor contained abundant blueberry (*Vaccinum angustifolium*), huckleberry (*Gaylussacia baccata*), mountain laurel (*Kalmia latifolia*), and seedlings of several hardwood species, especially chestnut and northern red oak.

The Clear Creek State Forest is located on the Allegheny Plateau region of northwestern Pennsylvania. The study site was a 12-acre stand found at midslope of a five percent, east-facing hill. Elevation was approximately 1800 feet. Soil was a loam (Typic Dystrochept) formed in place by the weathering of sandstone and shale parent material (Zarichansky 1964). Consequently, it was moderately acidic, fertile, and well drained. The stand experienced light gypsy moth defoliation in the 1980s, with little attendant overstory mortality (relative density was 100-percent). Dominant canopy species included northern red oak, sugar maple (*A. saccharum*), black cherry (*Prunus serotina*), and yellow-poplar (*Liriodendron tulipifera*). The sapling layer was quite dense, more than 20 ft^2 of basal area, and consisted almost entirely of striped maple with a few American beech (*Fagus grandifolia*). The hardwood regeneration layer was virtually nonexistent, but the forest floor was covered with hundreds of thousands of northern red oak acorns because of a bumper mast crop in fall 2001. A few scattered pockets of hay-scented fern (*Dennstaedtia punctilobula*) comprised the herbaceous plant community.

The Moshannon State Forest also is located in northwestern Pennsylvania in the Allegheny Mountians region. The study site was a 12-acre stand situated on an upperslope bench with a northwest aspect and slope of two percent. Elevation was approximately 2100 feet. The stand experienced light to moderate gypsy moth defoliation and mortality in the 1980s but relative denisty was nearly 100 percent. Soil was a loam (Typic Fragiudult) formed in place by the weathering of sandstone and shale parent material (Hallowich 1988). Consequently, it was moderately acidic, fertile, and moderately drained. Dominant canopy species included northern red oak, sugar maple, black cherry, and yellow-poplar. The sapling layer was quite dense, more than 25 ft^2 of basal area, and consisted almost entirely of striped maple with a few American beech. The hardwood regeneration layer was virtually nonexistent but the forest floor was covered with hundreds of thousands of northern red oak acorns because of a bumper mast crop in fall 2001. A few scattered pockets of hay-scented fern comprised the herbaceous plant community.

The Prescribed Fires

The objective of all three fires was to remove the sapling layer that was competing with the oak regeneration. Personnel of the Pennsylvania Bureau of Forestry conducted the prescribed burns on April 19, 2002 at Clear Creek State Forest, May 23, 2002 at Bald Eagle State Forest, and May 3, 2004 at Moshannon State Forest. Fuel, weather, and fire behavior data are presented in *table 1*. Fires were lit by hand with drip torches in a strip-headfire pattern commencing at the downwind or uphill side of each burn unit. The Clear Creek fire minimally burned as it had only compacted leaf litter as a fuel. Observed flame lengths were only a few inches. Conversely, the Bald Eagle fire produced flame lengths of four to eight feet because that site had an abundance of ericaceous shrubs for fuel. The Moshannon burn displayed widely varying fire behavior. Some areas minimally burned due to a paucity of fuel while other areas produced enough heat to damage and/or kill overstory trees. Leaf expansion of the striped maples was as follows: Clear Creek--swollen buds, Bald Eagle--fully expanded, and Moshannon--half expanded.

Table 1—*Environmental conditions and fire behavior at the time of the prescribed fires***.**

Variable	**Bald Eagle**	**Clear Creek**	**Moshannon**
Burn date	23 May 2002	19 April 2002	03 May 2004
Time of burn	13:00 – 15:00	11:00 – 12:00	13:00 –15:00
Burn size (acres)	5	4	6
Aspect	North	East	Northwest
Slope (%)	18	5	2
Slope position	Lower 1/3	Middle 1/3	Upper 1/3
Air temp. (F)	72 – 78	65 – 67	71 – 74
Rel. humidity (%)	23 – 27	35 – 40	42 – 48
Wind direction	West	West	West
Cloud cover (%)	0	0	25
Fuel model (Anderson 1982)	6	8	8
Fuel description	heath shrubs	compact litter	litter, slash
Fuel moisture (%)	10	15	16
Flame length (ft)	4 – 8	<0.5	1 – 4
Rate of spread (ft/min)	3 – 6	1 – 2	1 – 4

Study Design and Sampling Procedures

To determine the establishment timeline of the canopy trees and the sapling layer, increment cores were collected from each site in fall 2004. From the center of 10 systematically selected points in the unburned treatment, all trees intercepted with a 10-factor prism were cored at one foot above the ground on the uphill side. Basal cross sections were cut from an equal number of saplings near, but not in, each control plot. The cores were air dried for several weeks, mounted, and sanded with increasingly finer sandpaper (120, 220, 320, and 400 grit) to expose the annual rings. The cross sections collected from the stands also were dried and sanded. An establishment date for each core and cross section was determined by aging to the innermost ring or pith under a 40-power dissecting microscope. A pith estimator (Villalba and Veblen 1997) was prepared from the cores that intersected the pith and was then used to age the cores that did not intersect the pith. In all, more than 300 cores and 300 cross sections were collected from the three stands.

Because the Bald Eagle and Clear Creek fires occurred with little advance notice to us, collecting pre-burn data was not possible. However, the districts excluded about 50 percent of each stand from the fires as unburned controls and this division was based on visually estimating equivalent densities of striped maple saplings in each half. This provided us with a valid source of data for evaluating the effect of the fires on striped maple survival. Ten to twelve 1/40-acre circular plots were systematically located in each burn and control unit to ensure uniform coverage of the area. In these plots, all saplings (five feet tall to six inches dbh) were identified to species and tallied as alive, i.e., not top-killed by the fires, dead, or sprouting.

Inventories were conducted in fall 2002 and 2004 (one and three growing seasons post-burn) at the Bald Eagle and Clear Creek stands and in spring 2005 (one growing season post-burn) at the Moshannon site.

Statistical Analysis

Because the data set is incomplete at this time, only one year of post-burn inventory for Moshannon, results are preliminary and will be presented as three case studies in this paper. Statistical reporting will be limited to the mean number of living, dead, and sprouting striped maple saplings per acre for the burned and unburned units at each site.

Results

The establishment timeline of oak species at the Bald Eagle site differed considerably from that of the other two sites (*fig.* 1). At Bald Eagle, the oak overstory trees originated on a continuous basis between 1875 and 1950, while oak recruitment ceased after the 1920s at the other two sites. Peak establishment and recruitment were between 1915 and 1950 but no distinct cohorts are discernible. Oak regeneration ceased in the 1950s. Establishment and recruitment of other hardwoods coincides with that of oak but continues on into the 1990s. The present striped maple understory began in the 1960s with maximum recruitment in the late 1970s and 1980s. Striped maple only lives for about 40 years, thus there is no evidence to determine whether striped maple was a component of these stands before the 1960s.

The Clear Creek and Moshannon sites were quite similar to each other (*fig. 1*). In both, the oaks and other hardwoods began as distinct cohorts between 1900 and 1915. Oaks ceased to regenerate in the early 1920s, and other hardwoods did likewise by 1935 at Moshannon, and by 1955 at Clear Creek. The striped maple understories in both stands originated in the 1960s, with peak establishment in the early 1970s at Moshannon and late 1980s at Clear Creek.

The mean density (stems per acre) of the striped maple understories varied among stands but was reasonably equivalent between treatments within each stand (*fig. 2*). Moshannon had the most striped maple, 1422 stems per acre, while Bald Eagle and Clear Creek had 798 and 787 stems per acre, respectively. At all sites, the burned and unburned treatments had similar densities of striped maple. Bald Eagle striped maple densities were 752 and 845 stems per acre in the burned and unburned treatments. At Moshannon, densities of striped maple were 1451 and 1393 stems per acre for the burned and unburned treatments, while Clear Creek had 820 and 754 stems per acre in the burned and unburned treatments.

There were clear differences between the burn and unburned treatments in all stands after the first post-burn growing season (*fig. 2*). In the unburned treatment, virtually all the striped maple saplings were alive. Conversely, the burn treatment, regardless of the stand, consistently had more dead and sprouting striped maples and fewer living ones than the unburned treatment. At Bald Eagle, densities of dead, sprouting, and live striped maple were 436, 282, and 34 stems per acre, respectively, in the burn treatment, while the corresponding unburned densities were 13 dead, 21 sprouting, and 811 live. The densities of living, dead, and sprouting striped maple in the unburned treatment at Clear Creek were 704, 19, and 31 stems per acre, while those of the burned unit were 160, 427, and 233 stems per acre. The unburned treatment at Moshannon contained 1347 living, 15 dead, and 31 sprouting striped maple stems per acre, while the corresponding densities in the burned treatment were 95, 338, and 1018 stems per acre.

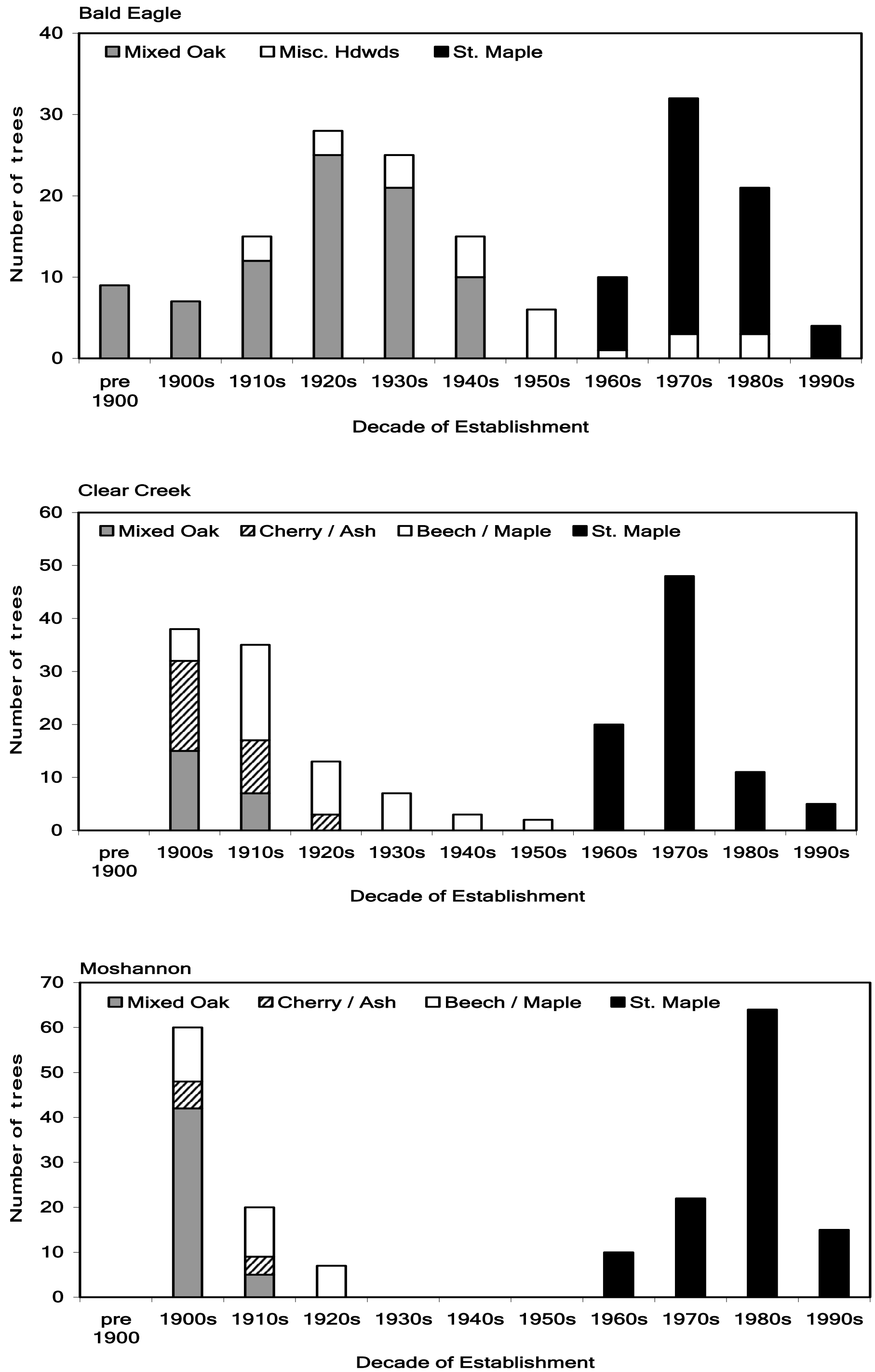
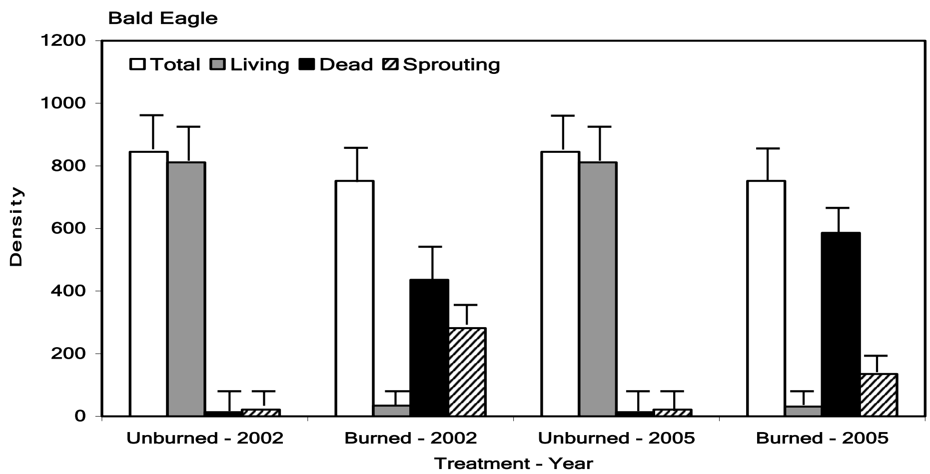
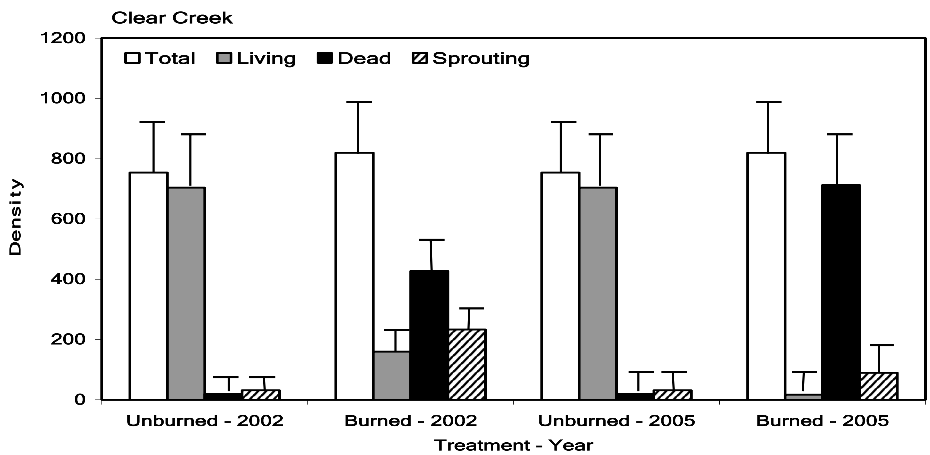


Figure 1—Species establishment timelines for the mixed oak stands at the Bald Eagle, Clear Creek, and Moshannon stands.





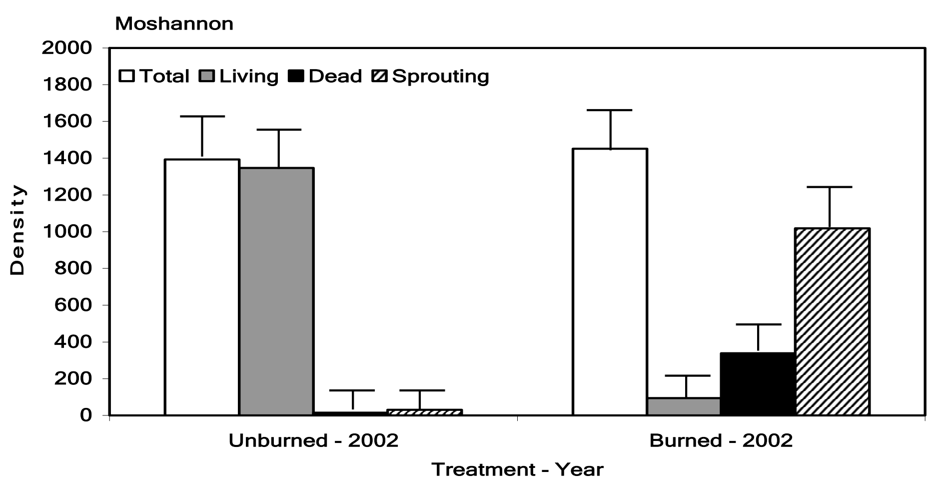


Figure 2—Density (mean number of stems per acre) of striped maple saplings of the Bald Eagle, Clear Creek, and Moshannon stands by type (living, dead, or sprouting), treatment (burned or unburned), and year (2002 or 2005). The bars on top of the columns represent one standard error.

Striped maple density data were available for the third post-burn growing season (2004) from the Bald Eagle and Clear Creek stands. For both stands, the number of dead striped maple in the burned treatments increased to 586 and 712 stems per acre, respectively. At Bald Eagle, this additional mortality appears to have come primarily from previously sprouted stems as those densities declined from 282 in 2002 to 135 in 2004, while the number of living striped maple saplings decreased by only three stems during that period. At Clear Creek, the increase in the number of dead striped maple saplings came from the demise of living and sprouting stems as these decreased from 160 to 17 and 233 to 90, respectively.

Discussion

One of the hindrances to restoring fire to mixed oak forests is a poor understanding of exactly how fire fit into their establishment and development a century ago. The oak establishment timelines of these three stands gives us some indication of the role fire and other disturbances played in their history. The Bald Eagle stand had continuous oak regeneration from 1875 to 1950. This long period of successful oak establishment and recruitment to the canopy was likely the result of several factors. First, the charcoal iron industry and subsistence farming caused repeated disturbances to central Pennsylvania forests. Farmers routinely burned forests to promote good grazing for their livestock and made repeated small-scale timber harvests for fences, tools, and fuel (Whitney 1994). The charcoal iron industry used copious amounts of wood, especially small diameter stems, and was the ignition source for many wildfires. The charcoal iron industry ceased in the 1870s and the 1915 fire exclusion law put an end to woodlot burning by farmers. Also in the early 1910s, the chestnut blight (*Cryphonectria parasitica*) swept through the state, killing virtually all American chestnuts (*Castanea dentata*), the major tree species of central Pennsylvania. Its demise also helped to promote oak. This was also a time when deer populations were quite low and they had little or no effect on forest regeneration. A disturbance regime of periodic surface fires mixed with partial harvests and other canopy disturbances creates and maintains the light environment young oaks need to grow while keeping their competition at bay (Brose et al. 1999).

The Clear Creek and Moshannon stands originated en masse in the early 1900s. This corresponds to the era when this part of the state was extensively logged and subsequently burned (Marquis 1975). Oak regeneration already in place at the time of logging and fire disturbances sprouted and dominated the newly-forming forest. Fire has been excluded from these stands since the formation of the Bureau of Forestry in 1905, allowing succession to proceed along an altered pathway since the 1920s.

The presence of dense striped maple understories in the mixed oak forests of Pennsylvania is coincident with the exclusion of fire and thus likely a result of altered forest succession. The striped maple establishment timelines show these saplings to have originated since the 1960s. This is decades after the last fire in any of these stands. It is unknown if striped maple existed in these stands a century ago when fire was present, or if this species is a relative newcomer moving into these stands since the 1960s.

Another hindrance of restoring fire to the mixed oak forests of the eastern states is the lack of knowledge regarding fire effects on important competing hardwoods. Striped maple certainly falls under that heading as evidenced by the contradictory results reported in the few fire studies that included the species. This study helps clarify the picture, at least in term of spring fires.

Striped maple was extremely sensitive to spring fires, regardless of fireline intensity. It is more sensitive to fire than red maple and may be as sensitive as yellow-poplar--two other common eastern hardwoods (Brose and Van Lear 1998). Its paper-thin bark offers little or no protection against fire, as even the small flames at Clear Creek top-killed more than 80 percent of the saplings. The more intense fires at Moshannon and Bald Eagle increased top-kill to more than 93 and 95 percent, respectively. In fact, striped maples surviving the fire at Bald Eagle were only able to do so if they were growing in protected microsites that precluded burning.

Not only were striped maples easily top-killed, but substantial numbers of rootstocks also were killed by fire. More than 50 percent of the striped maples at Bald Eagle and Clear Creek failed to sprout the first year after the fires. This was apparently due to the fires being able to scorch the root collars, thereby killing the dormant basal buds. While this was not unexpected at Bald Eagle, given its relatively high fireline intensity, it was surprising at Clear Creek where the fire barely burned. In fact, after that fire, all the striped maples expanded their leaves as if there had been no fire. However within a few weeks, they began wilting in large numbers. Apparently the fire was sufficient to girdle these saplings and prevent carbohydrate and water flow through the cambial tissue. Given the sensitivity to fire displayed by striped maple in this study, it probably was not the major component of the understory when periodic surface fires occurred that it is now in the absence of fire.

The delayed mortality at Bald Eagle and Clear Creek is puzzling. Both stands showed an increase in the number of dead striped maples in the burn units from 2002 to 2004. The intervening two growing seasons were exceptionally cool and wet leading to a major outbreak of anthracnose. This foliar pathogen may have caused the additional mortality because many of the dead stems were sprouts close to the ground. *Armillaria mellea*, a root pathogen, may also be the causal agent, as this fungus is ubiquitous in eastern forest soils and routinely attacks trees weakened by a stress. Whatever that mechanism was, between it and the fire, more than 80 percent of the striped maple saplings were dead within 3 years after the burns.

From this study, it appears that prescribed fire is another means to control striped maples when it becomes a silvicultural obstacle. Fire can be used in lieu of herbicides when the latter is not feasible due to policy constraints or site restrictions, i.e. too steep or rocky for equipment, striped maple is too tall, it’s a drought year, etc. Fire and herbicides can also be used in tandem with fire initially removing some stems and spot application of herbicide finishing the job.

Acknowledgments

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Reintroducing Fire in Regenerated Dry Forests Following Stand-Replacing Wildfire[1]

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Abstract:

Prescribed fire use may be effective for increasing fire resilience in young coniferous forests by reducing surface fuels, modifying overstory stand structure, and promoting development of large trees of fire resistant species. Questions remain, however, about when and how to reintroduce fire in regenerated forests, and to what end. We studied the effects of spring prescribed fires on stand structure and canopy fuel properties in 25- to 34-year old ponderosa pine forest that was planted following the Entiat wildfire in 1970. Six adjacent units were ignited over the course of four days within a 256-acre, south-facing management unit in the Preston Creek drainage, near Entiat, Washington. Fire effects were assessed on a grid of 264 small (0.014 acre) plots, of which 219 (83%) contained at least one tree. Fires reduced mean tree density from 426 to 280 trees per acre and reduced mean stand basal area from 47 to 38 ft^2/acre. Fires also modified canopy fuels, raising mean canopy base heights from 1.0 to 6.3 feet and reducing canopy bulk density from 0.0064 to 0.0061 lbs/ft^3. Fire behavior and fire effects were heterogeneous within treatment units, however, and local fire severity was positively correlated with local stand basal area. Tree mortality probabilities declined with increasing tree diameter for all species. For any given diameter, however, mortality probabilities increased with local stand basal area, probably due to higher fuels and local fire intensity. At the median basal area (37 ft^2/acre), fires killed mostly small trees (dbh < 2 inches). In more dense stands, fires also killed larger trees (dbh up to 5-6 inches). Mortality rates varied little among species except for larger trees in patches with high basal area, where survival rates were higher for ponderosa pines than for Douglas-firs and other conifer species. Overall, prescribed fires were effective for thinning stands from below, raising canopy base heights, and, to some extent, favoring ponderosa pine over Douglas-fir and lodgepole pine. Additional fires (and possibly some mechanical thinning) may be needed, however, to maintain low surface fuel loads, further modify canopy fuels, and further increase forest resilience to future wildfires.

Introduction

Modifying wildfire severity through manipulation of forest structure and surface fuels has become an important management objective in many dry coniferous forest types of North America (Graham et al. 2004, Peterson et al. 2005). Decades of fire exclusion have significantly altered forest stand density and species composition, particularly in dry western forests dominated by ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*) that historically supported fire regimes with

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short mean fire return intervals and mostly low severity fires (Cooper 1960, Covington and Moore 1994). These changes in forest structure and surface fuels increase risks for extreme fire behavior and large, stand-replacing wildfires in areas that formerly supported low- and mixed-severity fire regimes (Covington and Moore 1994, Graham et al. 2004).

Efforts to restore fire resiliency (the ability of forests to tolerate fire and recover quickly following wildfire) in dry coniferous forests have focused on reducing surface fire intensity and severity, reducing the probability of crown fire initiation, reducing the extent of crown fire spread, and creating defensible spaces for fire suppression activities (Agee 1996, Peterson et al. 2005). Mechanical thinning and prescribed fire are the tools commonly proposed for achieving these objectives. Mechanical thinning (with subsequent treatment of residual coarse woody debris) is typically recommended for altering stand structure and species composition, while prescribed fire is often recommended for reducing and maintaining acceptable levels of surface fuels (Graham et al. 2004, Peterson et al. 2005).

Young, regenerated forests present a challenge for increasing fire resilience through fuel reduction treatments. Opportunities for modifying stand structure with mechanical thinning in these young forests may be limited due to high unit treatment costs for thinning and pruning, large numbers of acres needing treatment, lack of merchantable timber to help offset treatment costs, reduced management emphasis on timber production, and limited funds for timber stand improvement activities. However, increasing fire resilience in young ponderosa pine and dry Douglas-fir forests is important because such forests (1) are abundant, (2) often have stand structural characteristics that support crown fire behavior, and (3) are likely to experience wildfire prior to reaching maturity.

In this paper, we present early results from an ongoing study of prescribed fire effects on fuels and stand structure in young, dry, ponderosa pine and Douglas-fir forests regenerated after stand-replacing wildfire in 1970. We wanted to know whether spring prescribed burning could be used effectively to increase the fire resiliency of these forests while perhaps also achieving other management objectives, such as thinning stands and increasing structural heterogeneity, and reducing risks of large-scale insect disturbances. Specific goals for increasing fire resiliency in mature forests typically include: (1) reducing surface fuels, (2) increasing live crown base heights, (3) reducing canopy bulk densities, and (4) retaining large trees of fire resistant species (Graham et al. 1999, Agee and Skinner 2005, Peterson et al. 2005). We use the first three of these goals as benchmarks for evaluating prescribed fire treatment efficacy. However, large trees of fire-resistant species are typically scarce or nonexistent following stand-replacing wildfires, so we propose an alternative goal of promoting the rapid development of large, fire-resistant trees as our fourth benchmark. Although in this paper we evaluate treatment efficacy primarily with respect to fire resiliency objectives, we note that promoting rapid development of large trees is also consistent with forest health and productivity objectives.

Methods and Materials

Study Area

The study area was the Preston Creek drainage within the Entiat River Basin, approximately 32 miles north of the town of Wenatchee in north-central Washington

State. Soils are well-drained sandy loams and loamy sands, derived primarily from volcanic ash and pumice deposits that overlay granitic bedrock. The climate features cold, moderately wet winters and warm, dry summers. Annual precipitation is about 23 inches per year (1961-1970), about 70% of which falls as snow (Helvey et al. 1976). The drainage supports a variety of dry forest types dominated by ponderosa pine, Douglas-fir, and lodgepole pine (*Pinus contorta*).

Fire scar records from the lower Entiat River Basin indicate that pre-settlement fire regimes (before 1860) featured fires of generally low severity with mean fire return intervals of about seven years (Everett et al. 2000). Fire frequencies became more spatially variable during the settlement period (1860-1910), but mean fire return intervals increased only slightly to 7-10 years. However, mean fire return intervals increased substantially (to about 40 years) during a subsequent period of active fire suppression beginning around 1910 (Everett et al. 2000).

Since 1970, large, stand-replacing fires have burned large portions of the Basin. The 1970 Entiat Fire burned 61,000 acres within the Basin. The Dinkleman and Tyee fires burned a combined 146,000 additional acres in 1988 and 1994, respectively. Reforestation efforts have produced extensive areas of young, even-aged forests (up to 34 years old) with relatively uniform structure, spatial pattern, and species composition. Post-fire logging and subsequent fuel treatments removed much of the coarse woody debris, so current surface fuels consist primarily of decaying stumps and fine fuels produced by the existing vegetation.

Prescribed Fire Treatment

As part of an overall management strategy for the Entiat Basin, managers have developed the following objectives for these young, regenerated forests: (1) reduce short-term (0-30 years) risks of severe disturbance from fire and insects, (2) take advantage of the current thinning window and attempt to expand it, (3) put the landscape and component stands on a trajectory toward conditions closer to the natural range of variability, and (4) restore fire as an active ecosystem process. To help them achieve these objectives, Entiat Ranger District staff proposed using prescribed fire. The primary benefits of the prescribed fire treatments were expected to be reduced risks of severe wildfire, reduced stand densities (thinning from below) and reintroduction of fire as an ecosystem process.

For this study, prescribed fires were ignited on six contiguous prescribed fire management units totaling 256 acres on four different days between March 23 and April 7, 2004 (*fig. 1*). Fires were ignited using the strip head-fire method. Residual snowpack protected adjacent north-facing slopes and higher elevation stands against fire escape and spotting.

Data Collection and Analysis

A randomized grid sampling approach was used to assess treatment effectiveness for modifying forest stand structure, species composition, and live fuels. Field surveys occurred in late August and September, 2004, at the end of the first growing season following fire. Sample plots were established on a square grid pattern at a density of one sample plot per acre, for a total of 264 sample plots. Plot centers were staked to allow repeated sampling. Variability of fire activity within units was assessed by recording the percent soil surface area burned for each plot.

At each sample plot, prescribed fire effects on forest stand structure were assessed by measuring fire effects on individual trees within a 14-foot radius of the

Figure 1. Management units in Preston Creek drainage burned with prescribed fire between March 23 and April 7, 2004. Photo was taken in late April, 2004.

plot center (616 ft^2 area). Species, diameter at breast height (dbh), height, crown class, and post-fire status (alive or dead) were recorded for each tree deemed to have been living before the fire. Pre-fire and post-fire height to the base of the live crown were also recorded for each tree, with pre-fire crown condition reconstructed based on the presence of scorched needles that were judged to have been alive prior to the fire. This retrospective approach was feasible because of the generally low intensity and severity of the prescribed fires.

Tree density, stand basal area, canopy bulk density, and canopy base height were calculated for each plot before and after fire based on pre-fire and post-fire plot tree lists (living trees only). Canopy bulk densities and base heights were calculated using the CrownMass software (FMAPlus 2003). For plots without trees, tree density, stand basal area, and canopy bulk density were set to zero, while canopy base height was undefined and treated as a missing value.

Fire effects on stand structure and canopy fuels were analyzed using a "before-after" repeated measures design using linear mixed models (Littell et al. 2006). Measurement plots were treated as random factors nested within prescribed fire management units (the latter being the replicated experimental units). Tree mortality was modeled using a similar, but nonlinear, mixed model (logistic regression) with plot-level random effects. The type I error rate for judging statistical significance of treatment effects was set at 10% ($\alpha = 0.10$) for all analyses.

Results

Most of the sample plots were on 20-60% slopes and southerly to southeasterly aspects. Of the 264 sample plots, 45 (17%) contained no trees (individuals taller than 4.5 feet dbh) prior to treatment. The number of plots without trees increased to 67 (+25%) after the prescribed fire treatments. Ponderosa pine was the most abundant tree species across all size classes, but Douglas-fir was also common, particularly in the smallest size classes. Lodgepole pine was present on many plots, but was usually a minor species component of the stand.

The prescribed fire treatments significantly changed stand structural attributes, including mean tree density and stand basal area. Fires reduced mean tree density from 426 to 280 trees per acre (SE = 23 trees/acre) and reduced mean stand basal area from 47 to 38 ft^2/acre (SE = 4.9 ft^2/acre). *Figures 2a, 2b* show that the prescribed fires shifted the frequency distributions of plot-level tree density and basal area estimates toward lower local density and basal area.

Prescribed fires also modified forest canopy fuels by raising canopy base heights and reducing canopy bulk densities. For plots with at least one tree, prescribed fires raised canopy base heights by 5.3 ± 0.4 feet (mean ± SE), from 1.0 to 6.3 feet and increased variability in canopy base heights among plots (*fig. 2c*). The treatments also reduced canopy bulk density somewhat, from 0.0064 to 0.0061 lbs/ft^3 (SE = 0.0004 lbs/ft^3).

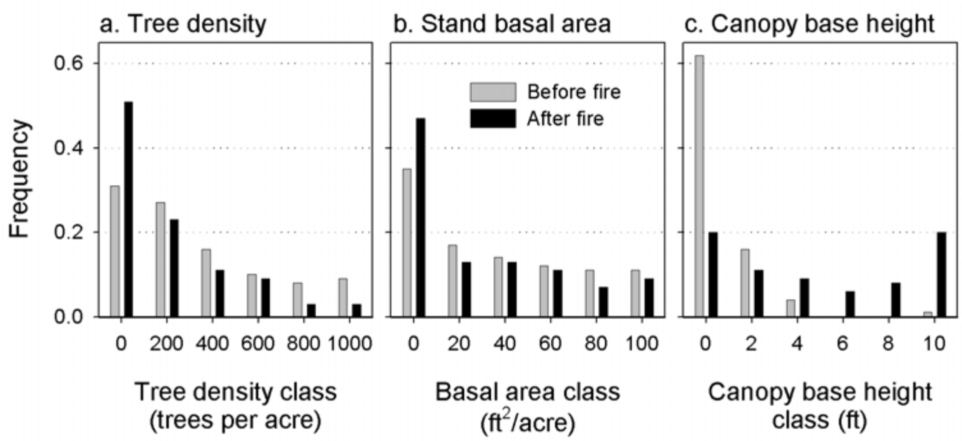

Figure 2—Changes in frequency distributions for tree density, stand basal area, and canopy base height estimates for all plots in the prescribed fire management units. Plots were grouped into classes for presentation purposes, and the horizontal axis tick labels indicate the lower bound for each class. Gray bars show pre-fire frequencies and black bars show post-fire frequencies.

Fire behavior and corresponding fire treatment effects were spatially variable within the prescribed fire management units. Field assessments of percent forest floor charred showed that 45% were completely burned (100% forest floor charred), 27% were partially burned (5-95% charred), and 28% remained unburned (0% charred). Observing that a disproportionately high percentage of plots without trees remained unburned, we tested for and found a significant positive correlation between plot basal area (a proposed surrogate for local productivity and fuels) and percent forest floor charred. This suggested that plot basal area might explain some variance in fire behavior and effects among plots.

Tree mortality/survival responses to prescribed fire varied with tree size and species. Probability of tree death from fire declined with increasing tree diameter for all species. Ponderosa pines had higher expected mortality rates than Douglas-fir and lodgepole pine (the latter two species were analyzed as a group) for trees with dbh less than one inch. However, tree mortality rates also varied significantly with plot basal area. On plots with median basal area, predicted mortality rates were very low (< 10%) for trees over two inches diameter (*fig. 3*). On plots with high basal area

(90th percentile and higher), mortality rates were over 10% for ponderosa pines with dbh less than 3.5 inches and other conifers with dbh less than 5 inches (*fig. 3*).

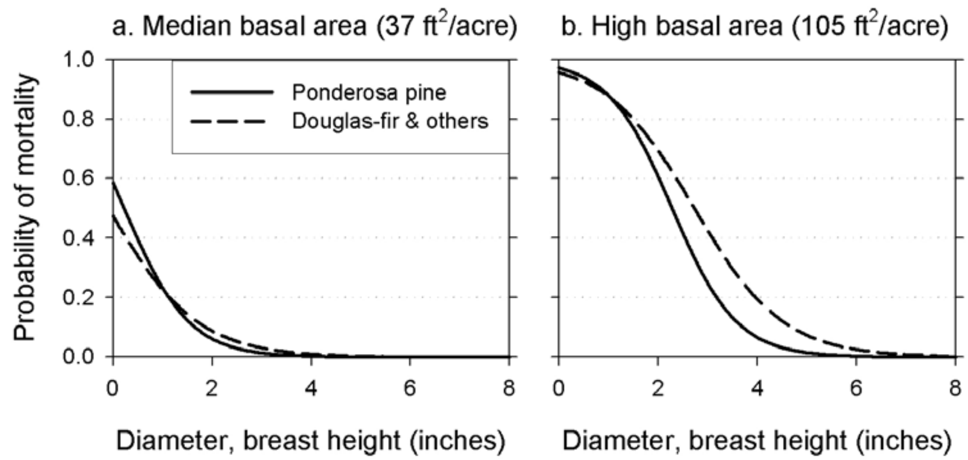

Figure 3—Predicted tree mortality probabilities for ponderosa pine and other conifers as influenced by tree diameter (dbh). Predicted mortality also varies with pre-fire plot basal area, as seen by differences between predicted mortality functions for a) the 50th percentile (median) basal area, and b) the 90th percentile (high) basal area.

Because the fires killed mostly smaller trees, the fire treatments changed tree size distributions. Before prescribed fire, the diameter distribution of trees across all management units was bimodal, with peaks in the sapling (0-2 inch) and small tree (4-6 inch) dbh size classes (*fig. 4*). High mortality rates within the sapling size class produced a post-fire tree size distribution with a single mode in the small tree size class (*fig. 4*). However, sapling densities were still relatively high overall due to their continued abundance on unburned plots within the management units. Omitting unburned plots from the analysis produced the expected unimodal tree size distribution with the highest mean density in the small tree class (*fig. 4*).

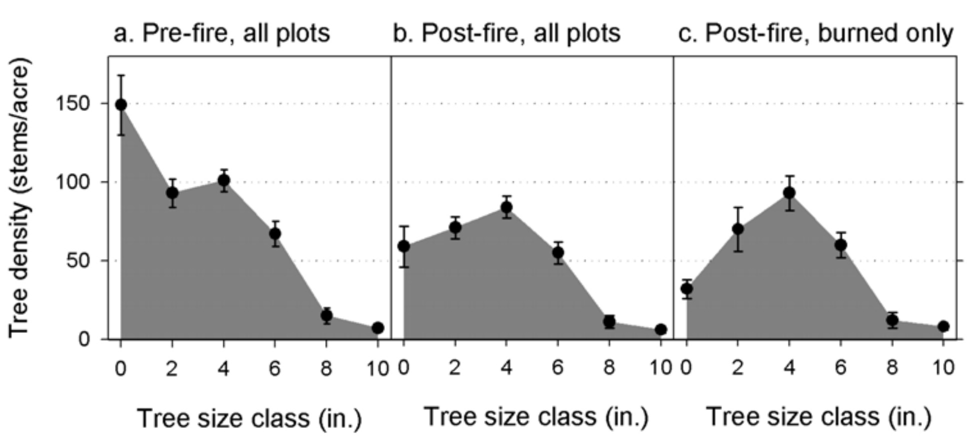

Figure 4—Mean tree size distributions on management units for a) all plots before prescribed fire, b) all plots after prescribed fire, c) all burned plots (minimum 5% of forest floor charred by fire) after prescribed fire.

Discussion

We proposed four treatment goals for assessing the efficacy of prescribed fire for modifying fuels and increasing fire resilience in young coniferous forests: reduce surface fuels, increase height to live crown base, reduce canopy bulk density, and promote development of large trees of fire-resistant species. Based on these goals, spring prescribed fire proved reasonably effective.

Visual inspection of post-fire surface fuels suggested that spring prescribed fires consumed most of the fine surface fuels and most of the decaying stumps from post-fire salvage logging in the completely burned areas. In the short term, surface fuels may not be sufficient to support even surface fires. However, the fires also produced future fine fuels by scorching tree crowns, killing small trees, and topkilling shrubs. As scorched needles and dead branches accumulate on the forest floor, fire risks will increase again and subsequent prescribed fire treatments may be needed to maintain low surface fuel loads and limit potential surface fire intensity.

The fires were also effective at reducing vertical continuity of fuels. By killing most understory tree seedlings and saplings and raising the height to the base of the live crown for surviving trees, the fires significantly reduced ladder fuels, increased canopy base height, and reduced risks of torching. Based on the nomograms provided by Scott and Reinhardt (2001), the prescribed fires increased the torching index by an average of about 15 km/hr for normal summer drought conditions with a surface fuel model 5 (brush, 2 feet) and 100% foliar moisture content. By monitoring surface fuel accumulations, subsequent prescribed fires can be planned to further raise canopy base heights and prevent development of new ladder fuels.

The fires did little to reduced risks of active crown fire, as mean canopy bulk density was reduced by only about 5%. The nomograms provided by Scott and Reinhardt (2001) suggest that such a small change in canopy bulk density would have little effect on the crowning index for the stand overall. However, canopy bulk density and fire effects were both spatially variable within the management units, so one would expect crowning behavior to vary as well. Based on fire behavior during large wildfires in central Washington State in 1994, Agee (1996) established a threshold value of 0.100 kg/m^3 (0.00615 lbs/ft^3) for canopy bulk density in ponderosa pine and Douglas-fir forests, above which crown fire behavior was likely under wildfire condition and below which no crown fire activity occurred. Our mean canopy bulk densities were very close to this threshold both before and after the fires. Future treatments will likely be required to more significantly reduce crowning index and achieve management objectives for increasing forest resiliency to wildfire.

Finally, the prescribed fires promoted the development and dominance of larger trees of fire resistant species. Larger trees had much higher survival rates than smaller. Our analysis also indicated that ponderosa pine trees had lower mortality rates than other conifers for trees with diameters greater than two inches. Once trees recover from reductions in crown volume, we expect that the lower stand densities will enhance growth rates of surviving trees by reducing competition for soil water and other limiting resources. These changes may also serve to reduce risks of large-scale insect damage.

Overall, spring prescribed fire treatments proved to be modestly effective for modifying fuels and increasing fire resilience in these young ponderosa pine forests. Additional prescribed fires will be needed in the next decade to further improve fire resilience and maintain low surface fuel loads. Because costs of treating stands with prescribed fire are considerably less than with mechanical thinning and pruning, multiple fires may be easily justified economically. We also expect fire prescriptions will become broader over time as fire resilience improves, making future fire treatments easier.

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Restoring Fire-Adapted Forested Ecosystems—Research in Longleaf Pine on the Kisatchie National Forest[1]

James D. Haywood[2]

Abstract

Prescribed burning research on the Kisatchie National Forest, Louisiana spanned the last five decades and led to a greater understanding of fire behavior and the importance of burning in longleaf pine (*Pinus palustris* P. Mill.) forests. Early research found that biennial burning in May favored the growth of longleaf pine seedlings. However, burning over several decades more greatly influenced diversity and productivity of herbaceous plant communities than burning affected long-term pine yields. Thinning sustains productive herbaceous plant understories in older stands because herbage yields decrease about 90 kg/ha with each m^2/ha increase in overstory basal area. In recent work, the use of container planting stock and a low incidence of brown-spot needle blight infection (caused by *Mycosphaerella dearnessii* M. E. Barr.) have been important in establishing longleaf pine. Emergence from the grass stage and growth of sapling longleaf pines have been better on recently harvested and prepared sites than on grass-dominated range partly because herbaceous plants are more competitive with longleaf seedlings than small woody plants and prescribed fire intensities are greater on grassy sites than on brushy sites. Differences in fuel types and rapid regrowth of vegetation both influence how prescribed burning affects long-term fuel loads.

Introduction

Longleaf pine (*Pinus palustris* P. Mill.) is a fire dependent forest type that formerly covered 24 to 38 million ha stretching from eastern Texas to southeastern Virginia, occupying wet poorly drained flatwoods to dry mountain ridges (Landers et al. 1995, Outcalt and Sheffield 1996, Brockway et al. 2005). Today, what was once the most extensive forest ecosystem in North America has been reduced to a remnant 1.5 million ha. The recovery of longleaf pine within the historic range is now necessary to arrest the decline of nearly 200 associated taxa of vascular plants and several vertebrate species (Brockway et al. 1998, Hardin and White 1989, Outcalt and Sheffield 1996).

In this challenging recovery effort, the management of longleaf pine regeneration can be difficult partly because of its unique morphology, in which it develops little above ground for the first two to nine years as the root system develops (Harlow and Harrar 1969, Wahlenberg 1946). The bunch of needles at the

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soil surface resembles a clump of grass, hence, the term "grass stage" to describe the juvenile period of growth. Grass-stage longleaf pine seedlings are susceptible to encroachment by brush and seedlings of other pine species, smothering by dead grass and litter, and brown-spot needle blight infection (caused by *Mycosphaerella dearnessii* M. E. Barr.) (Boyer 1975, Croker and Boyer 1975, Kais et al. 1986, Wahlenberg 1946). Longleaf pine seedlings do not respond to cultural practices meant to increase growth unless brown-spot needle blight is controlled (Derr 1957, Kais et al. 1986).

Prescribed burning can relieve longleaf pine seedlings from these stresses and thereby improve seedling survival (Grelen 1983, Smith 1961). Once the seedlings have well developed root collars (about 2.5-cm diameter), they are able to initiate height growth (Wahlenberg 1946). However, established pine-hardwood brush can still outgrow young longleaf pine seedlings, even after emergence from the grass stage, unless action is taken (Haywood 2000, Haywood and Grelen 2000, Haywood et al. 2001).

It is widely accepted that the management of longleaf pine at the landscape level requires an aggressive prescribed burning program not only to establish longleaf pine but also to keep older stands open and favor the myriad of herbaceous plants native to pine-grassland habitats. This understanding developed partly from the study of prescribed burning on the Kisatchie National Forest over the last five decades. This paper summarizes results from this extended period of research.

Early Research

Fire on the Range

The bluestem (*Andropogon* spp. and *Schizachyrium* spp.) range extended from northwestern Florida and southern Alabama to eastern Texas, and occupied primarily the Gulf Coastal portion of the longleaf-slash pine timber type (Grelen 1974). It included about four million ha in 1935. By the 1930s, uncontrolled harvesting had denuded most of the original longleaf pine within the bluestem range in Louisiana. The remaining vegetation was being burned repeatedly, overgrazed by cattle, and foraged by other livestock. In 1930, the Kisatchie National Forest was established in large part through the efforts of naturalist Caroline Dormon and as a response to the prevailing "cut out and get out" attitude of the timber industry (Joy 2005). The Civilian Conservation Corps (CCC) helped to replant these cutover lands on the national forest, but the effort focused on establishing loblolly (*P. taeda* L.) pine and slash pine (*P. elliottii* Engelm.) rather than longleaf pine because foresters mistakenly believed that longleaf pine could not be artificially regenerated (Croker 1989). However, longleaf pine recovered naturally where advanced regeneration and seed trees were present on some forestlands (Haywood et al. 2001), albeit on only a fraction of longleaf's native sites (Landers et al. 1995, Outcalt and Sheffield 1996, Brockway et al. 2005). Because of the history of range use by local people, much of the range within the national forest was placed under livestock management and unrestricted grazing and foraging continued on surrounding private lands.

Research in the bluestem range began on the Kisatchie National Forest during the mid-1940s and originally emphasized the effects of prescribed burning on range resources and herbage quality. Duvall and Whitaker (1964) recommended that range managed for cattle be rotationally burned in winter or early spring every three years to top kill brush, control undesirable herbaceous plants, and remove litter, thereby increasing bluestem grass productivity (Grelen and Epps 1967a). In addition, prescribed burning from midspring through early summer materially increased the protein content of bluestem grasses (Grelen and Epps 1967b).

Triennial burning in tandem with moderate cattle grazing kept the range relatively open and improved herbaceous plant richness. However, heavy grazing exposed mineral soil and was detrimental to water infiltration and percolation, increased soil bulk density, and reduced the percentage of large soil pores (Duvall and Linnartz 1967, Linnartz et al. 1966, Wood et al. 1989). Biennial or triennial prescribed burning did not adversely affect long-term soil sediment yields on flat to gently sloping sites, although sediment yields were higher immediately after burning (Dobrowolski et al. 1987). Spring burns had less effect than winter burns on sediment yields.

Despite apparent benefits in keeping the range open, prescribed burns have to be reapplied regularly to control brush, especially if cattle grazing stops. For example, in a prescribed burning study with the highly flammable shrub, wax myrtle (*Morella cerifera* (L.) Small), biennial or triennial burning reduces shrub height, but the shrubs regain stature between burns (*fig. 1,* left) (Haywood et al. 2000). Annual burning is best. However, general fuel conditions often are unable to support annual burning (Haywood and Grelen 2000), and annual burning is difficult to sustain operationally. Eight years of either biennial or triennial burning result in wax myrtles that are smaller in circumference than on no-burn plots (*fig. 1,* right). Overall, biennial burning is more effective than triennial burning.

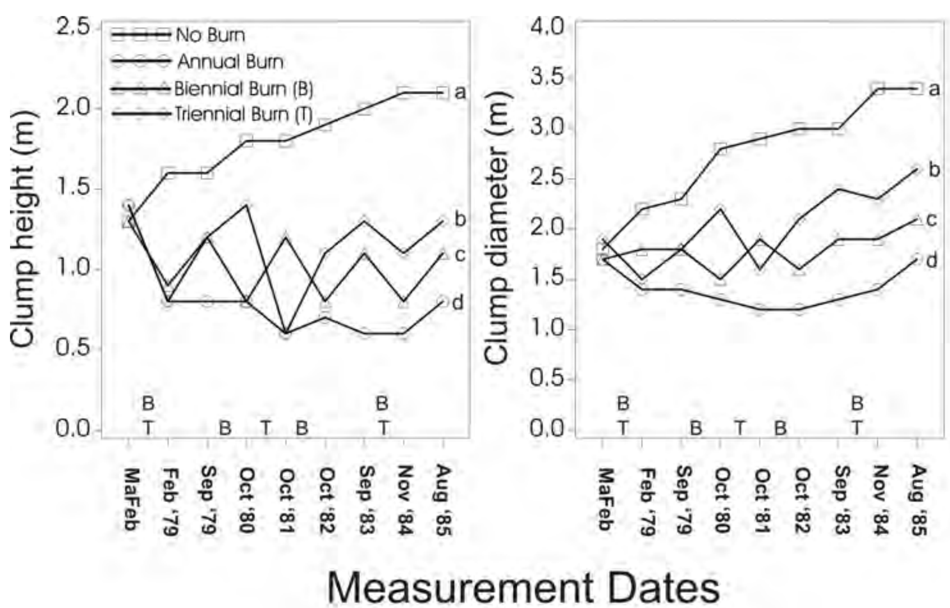

Figure 1—Shrub height (left) and clump diameter (right) of *Morella cerifera* in response to prescribed burning (B-biennial and T-triennial) from March 1978 to August 1985; final measurements followed by a different letter are significantly different based on Duncan's Multiple Range Tests (α=0.05) (Haywood and others 2000).

Fire Favored Longleaf Pine Regeneration

It became apparent in these early studies that longleaf pine regeneration tolerated range burning and became the dominant woody plant (Grelen 1975, 1983). This occurred partly because, during its unique grass-stage period, longleaf pine seedlings growing in full sunlight reach sufficient girth to tolerate high temperatures. Large tuffs of needles protect the terminal bud when fire moves quickly through the grass fuels and the highest temperatures are above the seedlings that are no more than 12 cm tall (Lindenmuth and Byram 1948). Once the longleaf pine seedlings emerge from the grass stage, they are more susceptible to heat injury until about 2m tall (Bruce 1951). Nevertheless, the majority of longleaf pine seedlings survive, while the other woody species are top killed by fire.

Grelen (1975) reported that biennial burning in May resulted in larger longleaf pine saplings than biennial burning in either March or July. Grelen (1983) attributed the better growth following May burns to the morphological characteristics of the new shoots, which in March is usually a silvery “candle” that by May has elongated, hardened, and is surrounded by an insulating sheath of needles.

Another factor influencing the seasonality of tree growth might have been differences in root injury even on repeatedly burned sites. Kuehler et al. (2004) reported that fine root production was less on plots repeatedly burned in July compared to plots burned in May and the rate of starch deletion from roots was lower on July-burned plots than on May-burned plots. In addition, foliar Mg concentration was lower on the July-burned plots than on the May-burned plots. Possibly, fire related injury resulted in less metabolic activity in roots following July burns.

Fuel bed conditions may influence the seasonal effect of fire on tree growth. It has been generally argued that there is an accumulation of dead fine fuels in March with few green fuels to lessen fire intensities resulting in more longleaf pine seedling injury. In May, green fuels are present, which lowers the heat of combustion because heat is lost in drying green fuels (Byram 1959). In July, high ambient temperatures and an accumulation of dried fine fuels also mean higher fire intensities than in May.

Fire in Older Stands

The positive relationship between May burning and greater longleaf pine stature in seedling or small pole stands reported by Grelen (1975, 1983) continued through 20 to 37 growing seasons in two separate studies (*tables 1 and 2*) (Haywood and Grelen 2000, Haywood et al. 2001). However, when both longleaf and loblolly pines were considered in these studies, the unburned plots were similar in pine basal area to the average for prescribed burned plots. Thus, burning in direct seeded or natural stands did not influence long-term pine yields but radically changed overall stand structure and species composition, especially in the herbaceous layer. Fire did this by maintaining longleaf pine grasslands (Bruce 1947, Haywood and Grelen 2000, Haywood et al. 2001), in which fires controlled woody vegetation and removed litter allowing sunlight to reach the forest floor, and common grasses and forbs naturally establish in pine grasslands in the West Gulf Coastal Plain (Haywood and others 1998a).

Table 1—*Stand characteristics 6 and 20 years after prescribed burning began to be monitored in a longleaf pine stand in Louisiana that had been direct seeded 4 years earlier (Grelen 1983, Haywood and Grelen 2000)*

	Fifty best longleaf pine trees per hectare after 6 years	Only longleaf pine trees after 20 years			Loblolly and longleaf pine trees after 20 years		
Treatments	Height (m)	Stocking (stems/ha)	Basal area (m^2/ha)	Total height (m)	Stocking (stems/ha)	Basal area (m^2/ha)	Total height (m)
Unburned	2.7	74	1.3	15.5	1829	34.2	14.1
Biennial March burns*	4.0	1154	9.1	8.5	1154‡	9.1	8.5
Annual-triennial March burns+	1.9	1712	18.0	10.1	1712‡	18.0	10.1
Biennial May burns*	3.9	1278	21.4	12.5	1317	23.3	12.5
Annual-triennial May burns+	4.9	1772	24.4	11.0	1796	24.6	10.7

*Monitoring of burns began in 1973 and continued through 1993, for 11 research burns over a 20-yr period.
+Plots were annually prescribe burned from 1973 through 1980, because of a lack of fine fuels, annual burning ceased and triennial burning began in 1983 and continued through 1992, for 12 burns over a 19-yr period.
‡There were no loblolly pines on the plots.

Table 2─*Stand characteristics 12 and 37 years after prescribed burning began to be monitored in a natural longleaf pine stand in Louisiana; 20 burns were applied over a 37-yr period from 1962 through 1998 (Grelen 1975, Haywood et al. 2001).*

	Hundred best longleaf pine trees per hectare after 12 years	Only longleaf pine trees after 37 years			Loblolly and longleaf pine trees after 37 years		
Treatments	Height (m)	Stocking (stems/ha)	area (m^2/ha)	height (m)	Stocking (stems/ha)	Basal area (m^2/ha)	Total height (m)
Unburned	6.5	136	10.7	24.1	193	18.4	24.7
March burns	8.3	519	22.3	21.3	519*	22.3	21.3
May burns	10.5	482	30.2	24.4	482*	30.2	24.4
July burns	6.9	217	15.1	21.3	217*	15.1	21.3

.* There were no loblolly pines on the plots.

Without the repeated use of fire, however, the longleaf pine grasslands revert to mixed pine-hardwood cover (Bruce 1947). As shown in *table 3*, forest canopy develops with a basal area divided among longleaf pine (32%), other pine species (52%), and hardwood trees (16%) 20 to 37 years after burning ceases (Bruce 1947, Haywood and Grelen 2000, Haywood et al. 2001). Beneath this canopy is a well-developed understory of woody plants and vines, but the deep shade and accumulation of litter nearly eliminates herbaceous vegetation and pine regeneration.

Table 3—*Percentage of basal area among three taxa of vegetation on three ranges in Louisiana in which burning ceased 20 to 37 years earlier (Bruce 1947, Haywood and Grelen 2000, Haywood et al. 2001).*

	No burning for…		
	20 years	32 years	37 years
Taxa	Haywood & Grelen 2000 (pct)	Bruce 1947 (pct)	Haywood et al. 2001 (pct)
Longleaf pine	3	54	40
Other pine species	89	38	29
Hardwoods	8	8	31

Another important management technique in longleaf pine grasslands is the control of stand density through thinning. Thinning, initially recommended to stimulate forage production for cattle (Wolters 1982), is now recognized as an important tool in the restoration of herbaceous plant communities (Haywood and Harris 1999). Once longleaf pine stands reach crown closure, usually within 17 years after planting on cutover range, herbage yields are predicted to decrease about 73 to 102 kg/ha with each m^2/ha increase in basal area (Wolters 1973, 1982). These predicted values compared favorably with Haywood and Harris' (1999) reported yields for longleaf pine stands in central Louisiana, in which longleaf stands with brushy understories lost herbage more quickly than stands with less woody vegetation in the understory (*table 4*). However, predicted herbage values are conservative partly because they do not account for the above average rainfall that fell during 1995 on Haywood and Harris' (1999) sites. Rainfall influences herbage yields by increasing production by about 10 kg/ha per cm of rainfall during the growing season (Wolters 1982), and gains in yield can range from 7 to 11 kg/ha per cm of rainfall as stand basal area decreases from 23 to 14 m^2/ha, respectively (Grelen and Lohrey 1978). In the 1995-growing season, rainfall was 24 cm above average, which could have increased herbage yields by 168 kg/ha for the stands in *table 4*. Regardless, when used together, prescribed burning to remove litter and thinning to reduce overstory basal area can rejuvenate understory herbaceous plant communities that are under stress and in decline (Grelen and Enghardt 1973).

Current Research

Planted Longleaf Pine Seedlings and Fire

Artificial regeneration is necessary when converting pastures and fallow agricultural fields to longleaf pine or when too few longleaf pine seed trees are present on forest sites. Under these circumstances, the best option for reestablishing longleaf pine is removal of the woody vegetation, site preparation, and planting. A key factor in reforesting longleaf pine has been the development of container planting stock (Barnett et al. 2002), which is recommended over bareroot seedlings to ensure better survival under adverse conditions (Barnett 2002).

Table 4—*Influence of woody vegetation on actual and predicted current-year herbaceous plant production on four sites in Louisiana (Haywood and Harris 1999).*

Stands	Woody plants >10 cm in dbh basal area (m^2/ha)	Trees and shrubs <10 cm in dbh		Canopy cover (pct)	Measured current-year herbage production in 1995 (kg/ha)	Predicted herbage production (kg/ha)
		Total (stems/ha)	Height (m)			
Catahoula RD (Brushy sites) [a]						
Compartment 71	24.4	60,146	0.8	77	452	358
Compartment 86	24.4	74,130	.5	57	753	358
Calcasieu RD (Grassy sites) [b]						
Compartment 10	22.5	10,873	.4	56	1640	1051
Compartment 22	28.5	35,008	.4	61	1160	659

[a]Formula for predicting herbage production on brushy sites (kg/ha):
$Y = 2853.58 - 102.25 \cdot (m^2/\text{ha of basal area})$ (Wolters 1982).

[b]Formula for predicting herbage production on grassy sites (kg/ha):
$Y = 2520.78 - 65.33 \cdot (m^2/\text{ha of basal area})$ (Wolters 1973).

In current research, longleaf pine stands were established from container stock in either grass-dominated range (the grassy site) or after mature loblolly pine-hardwood forest was clearcut, followed by chop and burn site preparation (the brushy site) (Haywood 2005). One finding has been that container grown longleaf pine emerges more quickly from the grass stage (*fig. 2*) than natural or direct seeded regeneration has in the past, as reported by Harlow and Harrar (1969) and Wahlenberg (1946). Interestingly, emergence was more rapid on the brushy site than on the grassy site (*fig. 2*). Emergence was almost 100 percent after three growing seasons on the brushy site, regardless of how the vegetation was treated. This level of emergence was not reached on the grassy site until after six growing seasons.

Herbaceous plant control significantly increased the growth of longleaf pine regeneration on both sites (*fig. 3*). Although season-long herbaceous plant control was no better than 50 percent on both sites, woody plants were the primary understory vegetation on the brushy site (Haywood 2005). In addition, longleaf pines on the brushy site were as tall after three growing seasons as longleaf pines on the grassy site were after six growing seasons, partly because of more rapid emergence from the grass stage.

I partly attributed the differences in growth rate between the two sites to

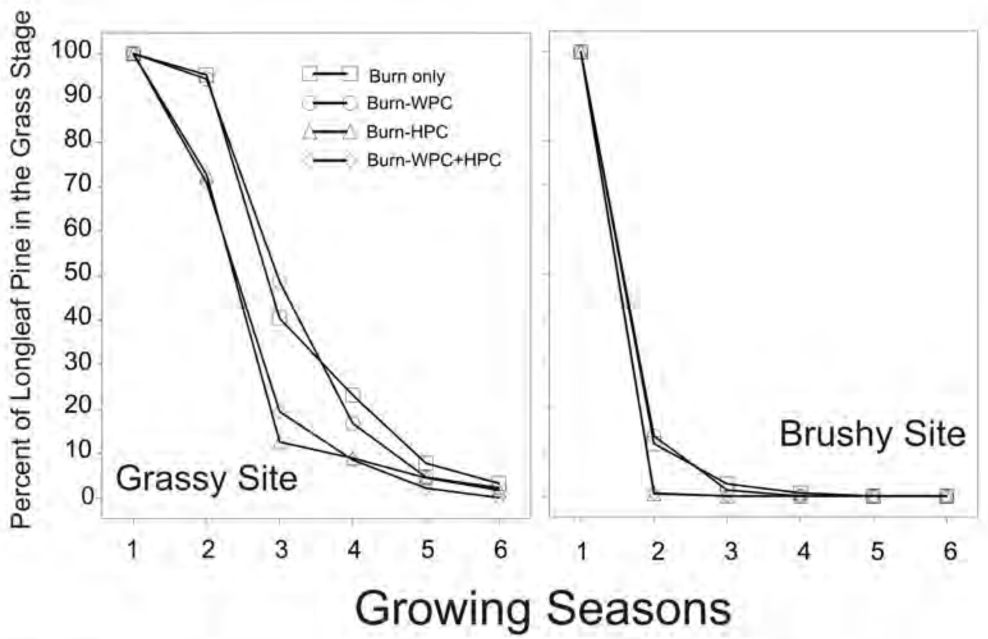

Figure 2—Percent of longleaf pine in the grass stage on two sites in central Louisiana by treatment combination: burn only, burn-herbaceous plant control (HPC), burn-woody plant control (WPC), and burn-WPC+HPC (Haywood 2005).

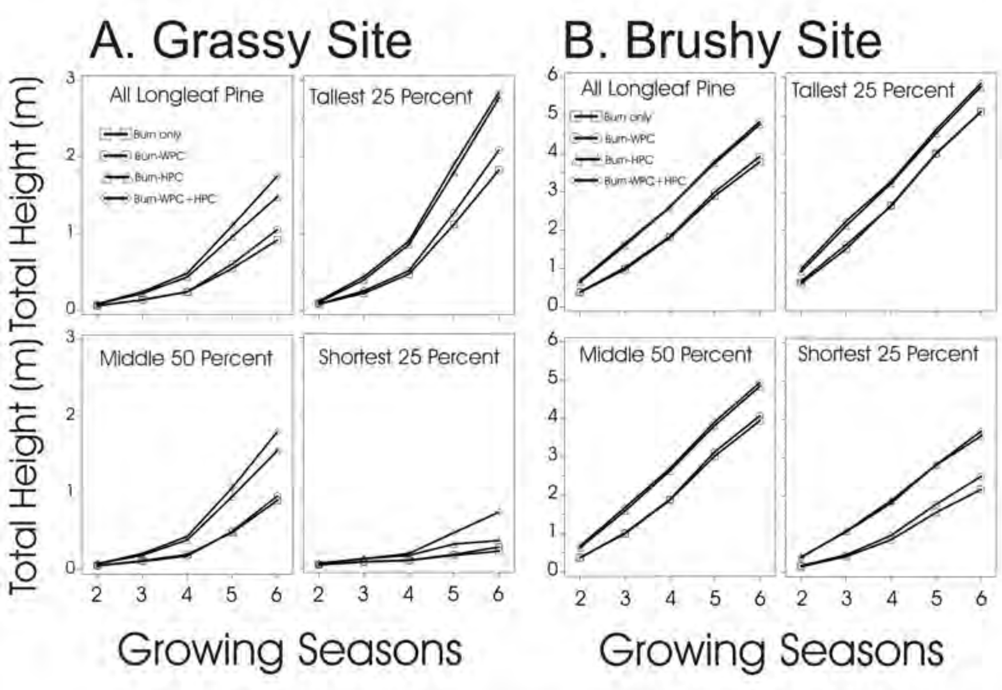

Figure 3—On two sites in central Louisiana, total height of all longleaf pine trees and total height of the tallest 25%, middle 50%, and shortest 25% of the trees by treatment combination: burn only, burn-woody plant control (WPC), burn-herbaceous plant control (HPC), and burn-WPC+HPC (Haywood 2005).

differences in the degree of herbaceous competition, with the grassy site having the most herbaceous vegetation and the least longleaf pine growth. In other work, Brockway and Outcalt (2000) increased development of longleaf pine seedlings by applying herbicide to prescribed burned and grass-dominated cover in Florida, and Haywood (2000) also increased height growth of longleaf pine seedlings with no more than 50 percent herbaceous plant control.

Differences in inherent site quality probably influence growth rate differences between the two sites (*fig. 3*). However, differences in intensity of the prescribed burns may be more important. Grass-dominated fuels carry intense prescribed burns in young longleaf pine stands, which can adversely affect seedling and sapling growth (Haywood 2002). At the brushy site, there was less grass and more erect forbs than at the grassy site, and this non-uniform, sparse, vertical fuel bed kept fire intensities low (Haywood 2005).

Another factor has been a low incidence of brown-spot needle blight. This disease can keep longleaf pine seedlings in the grass stage and nullify benefits from vegetation management treatments (Derr 1957). A low incidence of disease is likely contributed to the timely initiation of height growth (Kais et al. 1986), and the positive response of trees to herbaceous plant control (Derr 1957).

Woody plant control did not affect longleaf pine growth on either site (*fig. 3*). On the grassy site, fires were intense and the combination of burning and herbaceous plant competition may have kept the woody vegetation in check (Haywood 2005). On the brushy site, fires were less intense but woody plant control still did not benefit longleaf pine seedlings.

Some land managers may be willing to allow brush encroachment on sites where intense prescribed burns are not achieved during stand establishment with the opinion that by the time canopy closure is reached needle cast will improve fuel bed conditions and more intense fires will be possible. After stand closure, repeated burning coupled with a chemical or mechanical hardwood release can be used to create open stand conditions. Less intensive management is especially appealing if land managers focus on the tallest 25 percent of the longleaf pine trees and not on the whole population of trees (*fig. 3*). Nevertheless, where vegetation components have shifted away from grassland to brush, an aggressive burning program applied over several decades will eventually be necessary to decrease the number and stature of competing woody plants and favor herbaceous vegetation (Waldrop et al. 1992).

Diverse plant communities developed on both sites. On the grassy site, there was a well-established grassland community before planting the longleaf pines. Eighty-five herbaceous species and 19 woody species were commonly found. Sixteen of these (*table 5*) were indicators of a well-developed understory in longleaf pine forests in Louisiana, as described by Turner et al. (1999). On the brushy site, many plants common in the original mature loblolly pine-hardwood forest were still common after harvesting, site preparation, and longleaf pine establishment. By the fourth growing season after planting, 101 herbaceous species and 34 woody species were widespread on the brushy site. Twenty of these were indicators of a well-developed understory in a longleaf pine forest (*table 5*).

Apparently, the brushy site had a more diverse plant community than the grassy site. Based on visual observation of site conditions, this was surprising. Usually, loblolly pine-hardwood stands established on longleaf pine sites are thought to be degraded, requiring great effort to restore to longleaf pine grasslands. However,

Table 5—*Understory plants inventoried in two sapling size longleaf pine plantations that are indicators of well-established longleaf pine understory in Louisiana (Turner et al. 1999).*

Common names	Scientific names	Common names	Scientific names
Herbaceous plants			
Grassy site		*Brushy site*	
big bluestem	*Andropogon gerardii* Vitman	broomsedge bluestem	
broomsedge bluestem	*Andropogon virginicus* L.	arrowfeather threeawn	
arrowfeather threeawn	*Aristida purpurascens* Poir.	calico aster	*Symphyotrichum lateriflorum* (L.) A. & D. Löve var. *lateriflorum*
needleleaf rosette grass	*Dichanthelium aciculare* [Desv. ex Poir.] Gould & C.A. Clark	Nuttall's wild indigo	*Baptisia nuttalliana* Small
tapered rosette grass	*Dichanthelium acuminatum* [Sw.] Gould & C.A. Clark	spurred butterfly pea	*Centrosema virginianum* (L.) Benth.
flowering spurge	*Euphorbia corollata* L.	needleleaf rosette grass	
bushy goldentop	*Euthamia leptocephala* (Torr. & Gray) Greene	tapered rosette grass	
		flowering spurge	*Euphorbia corollata*
sharp blazing star	*Liatris acidota* Engelm. & Gray	erect milkpea	*Galactia erecta* (Walt.) Vail
beaked panicgrass	*Panicum anceps* Michx.		
blackeyed susan	*Rudbeckia hirta* L.	littleleaf sensitive-briar	*Mimosa microphylla* Dry
little bluestem	*Schizachyrium scoparium* [Michx.] Nash var. *divergens* [Hack.] Gould	dollarleaf	*Rhynchosia reniformis* DC.
slender rosinweed	*Silphium gracile* Gray	blackeyed susan	
sidebeak pencilflower	*Stylosanthes biflora* (L.) B.S.P.	little bluestem	
multibloom hoarypea	*Tephrosia onobrychoides* Nutt.	anisescented goldenrod	*Solidago odora* Ait.
		sidebeak pencilflower	*Stylosanthes biflora* (L.) B.S.P.
		Virginia tephrosia	*Tephrosia virginiana* (L.) Pers.
		Small’s noseburn	*Tragia smallii* Shinners
		nettleleaf noseburn	*Tragia urticifolia* Michx.
Woody plants			
blackjack oak	*Quercus marilandica* Muenchh.	saw greenbrier	*Smilax bona-nox* L.
farkleberry	*Vaccinium arboreum* Marsh.	farkleberry	*Vaccinium arboreum*

Haywood et al. (1998a) found that common native herbaceous plants naturally establish on forestlands if proper management practices, such as prescribed burning, create the open conditions necessary for colonization. Therefore, restoring diverse understory plant communities of common plants may be less difficult than originally believed in the West Gulf Coastal Plain and elsewhere (Smith et al. 2002). However, ecosystem restoration goes beyond re-vegetating forestlands (Covington et al. 1998), and the restoration of rare plants within unique habitats may require more than basic silvicultural practices.

Fire is not a Panacea

Intense fires can reduce the growth rate of longleaf pine saplings, although most of the trees survive (Haywood 2002). For example, a series of prescribed burns initiated after the longleaf saplings were about 2m tall adversely affected height growth (*fig. 4*) because the successive biennial burns were very intense and always scorched most of the pine foliage (*table 6*). The three burns in March and July averaged 630 and 650 kJ/s/m in intensity, respectively, which was nearly four times

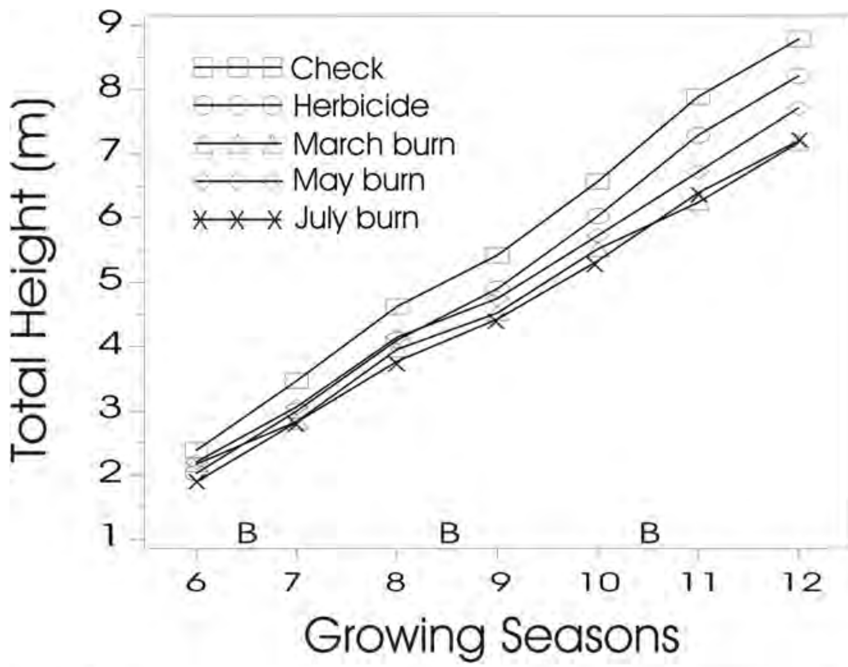

Figure 4—Height growth of longleaf pine saplings from the 6 through 12 growing seasons; either the plots were prescribed burned (B) in March, May, or July in the 7, 9, and 11 growing seasons or the woody vegetation was treated with herbicides (Haywood 2002).

the intensity of an average winter backfire (Haywood 1995). Burns in May averaged 551 kJ/s/m. Fire intensity might have been related to subsequent height growth because total height of the longleaf pine trees was less on the March- and July-burned plots than on the May-burned plots after three burns over a 6-year period.

Regardless, the differences in fire intensity, although always severe and resulting in high percentages of needle scorch, seem to support the argument that the accumulation of dead fine fuels in March, with little green fuel, results in fast rates of fire spread and high fire intensities (*table 6*). Overall, May fires were the least intense, as predicted.

Severe needle scorch can result in growth loss among even large 65-year-old longleaf pine trees in pine grasslands (Haywood et al. 2004b). The diameter at breast height (dbh) growth of longleaf pine was significantly less for scorched trees (0.34 cm/year) than for unscorched trees (0.46 cm/year) over a 5-year period. Likewise, longleaf pine root sucrose and starch concentrations were significantly reduced in response to crown scorch, and, therefore, there were fewer carbohydrates available for pine root metabolism (Sword and Haywood 1999).

Despite high fire intensities, fine fuel loads do not necessarily decrease. For example, repeated burnings in seedling and sapling longleaf pine grasslands maintain conditions for good grass development and continued high fuel loads. As shown in *table 6*, the first burns in 1999 were in a 6-year-old grass rough and the fuel loads averaged 4700 kg/ha. Four years later, fuel loads averaged 5900 kg/ha.

Under forest canopy, most of the fine fuels are litter and the recovery of fuels after prescribed burning may be different from pine grasslands. For example, operational prescribed burns were done in fully stocked loblolly pine (22.9 m^2/ha of basal area) and two mixed pine (23.4 m^2/ha of basal area) stands (Haywood et al.

Table 6—*Fuel loads and fire intensities for prescribed burns conducted on a pine-grassland site in 1999 through 2003.*

Years and treatments	Burning date	Oven-dried fuel load (kg/ha)	Rate of spread (m/s)	Range in fire intensity (kJ/s/m)	Average fire intensity (kJ/s/m) *
1999					
March burn	March 2	3702	0.06	319 – 429	385
May burn	May 14	6003	.03	290 – 378	341
July burn	July 8	4377	.08	400 – 688	590
2001					
March burn	March 13	5287	.06	522 – 561	544
May burn	April 30	6171	.06	548 – 827	734
July burn	July 31	6323	.08	871 – 1026	943
2003					
March burn	March 11	6240	.08	905 – 1035	962
May burn	May 6	6543	.05	504 – 662	579
July burn	July 22	4863	.05	334 – 534	417

*A low intensity winter backfire would be between 0 and 173 kJ/s/m.

2004a). Springtime prescribed burns destroyed most of the live foliage in the understory on these three sites, but the effect was short term and understory vegetation recovered between burns (*fig. 5*). Burning reduced the amount of 1-hour time-lag dead fuels, but the 1-hour fuels accumulated between burns. The 10-hours fuels were significantly reduced by burning. Interestingly, dead fuel loads decreased on unburned plots as well, although the same personnel collected all of the fuel samples. Whither this is a long-term or short-term trend is uncertain.

Southern pine beetle (*Dendroctonus frontalis*) and Ips engraver beetle (*Ips* spp.) are the most destructive insects in the pine forests of the southern United States. Bark beetle abundance often increases following prescribed burning (Haywood et al. 2004a). Additionally, ambrosia beetles (*Platypodidae*) are attracted to fire-stressed stands and are vectors of known and suspected root pathogens. In the mixed pine and loblolly pine stands, prescribed burning resulted in increased numbers of beetles (*fig. 6*). Principally, the beetles trapped were in three guilds--ambrosia beetles, bark beetles (*D. terebrans*, *Hylastes tenuis* and *salebrosus*, *Ips grandicollis* and *arulsus*), and weevils (*Cossonus corticola*, *Hylobius pales*, and *Pachylobius picivorus*). No southern pine beetles were trapped, but the majority of weevils trapped were *C. corticola*, which may be an associate of the southern pine beetle (Goyer et al. 1980). Too few ambrosia beetles were trapped to determine response trends.

Another adverse effect from prescribed burning could be an increase in soil bulk density caused by exposure of the mineral soil to rainfall, with consequent dispersal of aggregates that can clog soil pores (Boyer and Miller 1994). Boyer and Miller (1994) observed this negative effect on soil bulk density after 12 years of repeated burning; bulk density of the surface 15 cm of mineral soil was 1.22 g/cm^3 on the unburned plots and 1.26 g/cm^3 on the burned plots. In a study in Louisiana, however, I did not find a significant difference between burning (1.28 g/cm^3) and no burning

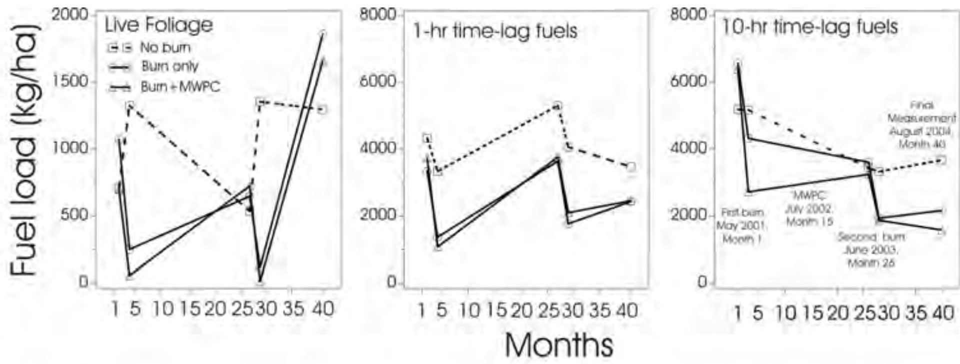

Figure 5—Fuel load changes after two prescribed burns and mechanical woody plant control (MWPC) in three fuel classes (live foliage, 1-hr time-lag dead fuels, and 10-hr time-lag dead fuels) on three sites in central Louisiana; the burns were in May 2001 and June 2003 and the mechanical treatment was in July 2002.

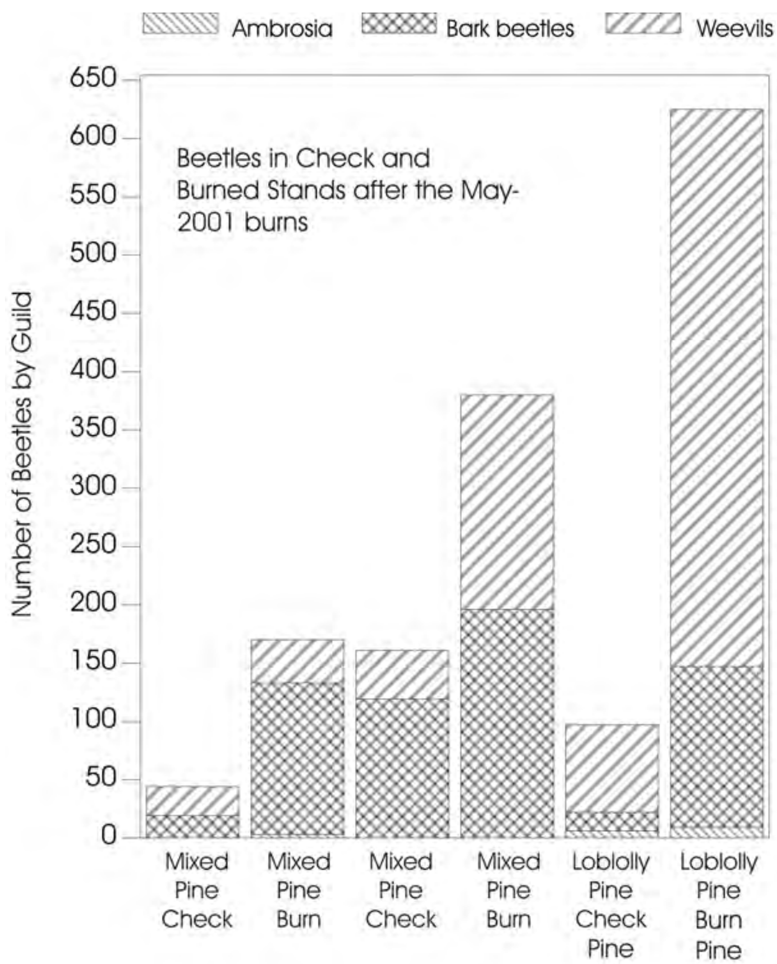

Figure 6—Number of beetles captured by guild after prescribed burning in May 2001 (Haywood and others 2004a).

(1.24 g/cm^3) on soil bulk density in the surface 10 cm of mineral soil, although four prescribed burns were applied over a 12-year period.

Other Brush Control Options

Other brush control treatments besides prescribed burning are available. However, the chemical treatment of brush can be associated with some sublethal injury that stunts longleaf pine growth for several years, although the injured longleaf pine trees appear to recover and begin growing at a rate similar to untreated trees (*fig. 4*). In addition, Patterson et al. (2004) found that the chemical control of woody vegetation resulted in greater soil bulk densities in seedling longleaf pine plantations; untreated plots averaged 1.48 g/cm^3 compared to 1.62 g/cm^3 on woody plant control plots. They attributed the increase in soil bulk density to less root system proliferation and soil agitation.

The mechanical harvesting of pine straw controlled understory vegetation and increased soil bulk densities (Haywood et al. 1998b). Nine mechanical treatments over an 11-year period left the mineral soil exposed to natural weathering processes and raised soil bulk density to 1.44 g/cm^3 in the surface 10 cm of mineral soil, compared to 1.26 g/cm on the untreated plots. At a soil depth of 10 to 20 cm, bulk density was 1.55 g/cm^3 and 1.51 g/cm^3 on the treated and untreated plots, respectively.

For felling brush and midstory vegetation, the Kisatchie National Forest currently uses machine-mounted horizontal-shaft drum shredders (described by Haywood et al. 2004a). However, mechanical treatments are expensive, and in 2004, the cost was nearly $370/ha. In addition, mechanical felling of brush was no more effective at reducing available fine fuels than burning alone (*fig. 5*). Mechanical treatments may control large midstory vegetation, but unless prescribed burning follows mechanical woody plant control, the effectiveness of mechanical treatments will not last long because brush recovers rapidly in the West Gulf Coastal Plain.

Conclusion

Fire research on the Kisatchie National Forest originally focused on improving herbage quality for cattle, and how prescribed burning coupled with control of stand stocking through thinning could increase herbage productivity. However, from a resource perspective, the most important long-term benefit from prescribed burning was the establishment of longleaf pine-grassland forests with rich herbaceous plant communities and the continued maintenance of these pine grasslands by the repeated application of fire. Through the 1990s, the fire research program provided crucial information on how fire influenced forest stand structure and species diversity in the West Gulf Coastal Plain.

Under current funding levels and with the available technologies, a timely fire regime is the only vegetation management practice that is generally applicable over many thousands of acres to reduce fuel loads and control hardwood trees and shrubs. Fire should be applied when woody stems start to become reestablished in the understory. Frequency of burns is dependent upon site productivity and the desired or existing plant community.

The development and use of container stock has been a major advance in the artificial regeneration of longleaf pine. Container stock provides better survival under adverse conditions and more rapid emergence from the grass stage than if either bare-root planting stock or direct seeding is used. A low incidence of brown-spot needle blight, a disease that keeps seedlings from initiating height growth and responding to vegetation management practices, may be as important as planting containers.

Emergence from the grass stage is faster and growth greater on brushy sites than on grassy sites for several reasons: (1) herbaceous plants are more competitive with longleaf seedlings than small woody plants, (2) brushy sites may be inherently more fertile than grassy sites, and (3) prescribed fire intensities are greater on grassy sites than on brushy sites. Although controlling woody plants was not beneficial in terms of longleaf pine seedling development, woody vegetation cannot be allowed to grow unchecked in longleaf pine plantations. Without woody plant control, a mixed pine-hardwood forest will develop, because loblolly pine and hardwood brush will outgrow many of the longleaf pine seedlings. At some point, woody vegetation has to be controlled by fire or other means to establish longleaf pine grasslands.

A long-term burning program successfully started when the longleaf pines are still seedlings can maintain pine grasslands on the Kisatchie National Forest, but often the understory herbaceous plant community is no longer productive once the stands reach 60 percent canopy closure. However, a lack of productivity does not equate to a lack of species richness. Thinning of overstocked stands of longleaf pine and continued burning can maintain productive and rich understory plant communities.

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Silviculture for the 21st Century—Objective and Subjective Standards to Guide Successful Practice[1]

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Abstract

Silviculture is increasingly being applied in ways that go beyond traditional timber management objectives. Across the National Forest System, on other public lands, and increasingly on private lands as well, foresters are working with professional colleagues and landowners to develop innovative silvicultural prescriptions designed to meet diverse resource management objectives. Some of those innovations involve treatments, timing, or intensity that are not supported by published or ongoing research studies. This can lead to problems over time, especially if the treatments fail to achieve their intended goal. To maintain trust and credibility with other resource professionals, as well as with the landowners they serve, silviculturists must act according to a simple philosophy--say what you'll do, do what you said, and watch what you did. A set of ten quantifiable metrics and subjective tools is suggested as a guide to implementing that simple philosophy. Taken collectively, this set of tools and metrics comprise a subjective decision support framework for silviculturists, especially as practices are proposed that go beyond scientific support in the literature. The degree to which these elements should be quantified depends upon the complications that will arise from failure to detect whether a prescription has been properly prescribed and implemented.

Introduction

The turn of the 21st century has seen shifting paradigms in forest management. Standard practices, such as clearcutting, that were widely prescribed as recently as three decades ago, have been critically examined in light of new conceptual approaches to forestry (Franklin et al. 2002). As a result of these changes in strategic thinking about forest management, the tactics by which management decisions are being implemented have changed as well.

For example, three decades ago, silviculture was applied primarily to timber production, so much so that society generally has come to consider the terms as synonymous (e.g., Spurr 1979, Graham and Jain 2004). Even today, some professionals feel that the term silviculture is inappropriate to use in any context other than that of timber production, and that some other term should be developed for manipulative treatments in a forest in which timber production is not the objective. This has even extended into some university curricula, in which classic

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silviculture courses and textbooks are being replaced by "applied forest ecology," and forestry schools are now "schools of the environment." By that logic, the use of the term "silviculture" in a habitat restoration or stand structural context might be viewed by some as oxymoronic.

Similarly, there have been changes in the tactics that agency critics use to stop timber sales. Appeals and litigation of timber sales are still common, especially in western States. But as agency silvicultural prescriptions become more diverse, appeals and litigation have changed from a stance that advocated cessation of clearcutting to one that proposes to stop all logging in National Forests. And, in a clever response to working within the system, some groups now bid on and purchase agency timber sales, with the intent of not proceeding with the harvest.

A more holistic definition of silviculture, advocated here, speaks to the values retained in forest stands after a harvest has occurred, thereby attaining landowner objectives through greater attention to what is retained in the woods, rather than what is removed (Behan 1990, Franklin 1989, Kessler et al. 1992, O'Hara et al. 1994, Swanson & Franklin 1992). We believe that, over time, this perspective will broaden the constrained view of silviculture as a practice appropriate only for timber production to one that is appropriate for all resource values--wildlife habitat, watershed, ecological restoration, and others--that depend upon silvicultural manipulations to advance desired stand conditions. Adoption of this view has the potential to defuse internal disagreements between professionals over silvicultural prescriptions, as well as to weaken the ecological logic of the stance of agency critics acting to stop all logging.

However, as silviculturists rush headlong into forest stands with paint guns that are targeted on structures and habitats to retain rather than on trees to cut for forest products, they often depart from a firm foundation of research findings to support their decisions. This can be exacerbated in situations where field technical crews have not been baptized with the same fervor as their professional counterparts on the transition from timber goals to habitat and restoration goals. And given the diversity of specific management goals and objectives that may change over time, it is likely that the research support for silvicultural innovations will continue to lag behind the practice.

To maintain the trust of not only the other resource professionals and technicians with whom they work, but also the landowners they serve, silviculturists must act according to a simple philosophy--*say what you'll do, do what you said, and watch what you did.* Fellow coworkers and landowners both will generally tolerate a silvicultural prescription that does not go as intended if the silviculturist is honest about the plans that were made and the outcomes that occurred. Moreover, the rationale for such silvicultural activities needs to be articulated and both the risks and uncertainties of the activities disclosed and documented in a silvicultural prescription.

As the adage goes, "the devil is in the details". An important part of a silvicultural prescription is the exact specification of intensity, timing, and tactics that will help to determine whether a given silvicultural practice is likely to meet the intended goal. These details center on the silviculturist knowing the conditions within the stand or landscape prior to the treatment, the context both physically and socially of the proposed treatment, the conditions under which the treatment is conducted, the conditions that result after the treatment has been completed, and the short- and long-term expected vegetative response to the treatment. Some of these details require measurement of conditions in some varying sample intensity. Others are subjective guides that will determine the ease with which proposed treatments can be implemented within a stand or across a sufficient area to make a significant change in the resource attributes of interest.

In this paper, we propose a set of five quantifiable metrics and five subjective considerations to consider when implementing silvicultural practices for any landowner goal on public or private forest land, and which can be applied to encompass diverse goals of ownership--from habitat management, ecological restoration, specific configurations of stand structure, and even timber production.

Quantifiable Metrics

Quantifiable metrics are variables for which measurements can be taken. Those measurements produce data from which simple statistics, such as a sample mean and variation about the mean, can be calculated for the variable of interest. The intensity of the sample used to quantify the variable depends upon the ease with which the variable can be measured, the inherent variation of the variable, and the degree to which measurement of the variable gives biologically meaningful and practical information to the silviculturist. For example, in some situations, a cruise or inventory of acceptable sampling intensity should be conducted. Other situations might be acceptably quantified using a visual estimate, which is itself a subjective determination of sample mean and variation for key variables (such as stem density, basal area, and stand structure) based on practical experience and insight.

In the five quantifiable metrics that follow, the degree to which sampling or visual estimation is sufficient to provide data of suitable rigor will vary. But assessing these metrics themselves provide feedback to the silviculturist about whether silvicultural prescriptions will be or were successful, and if the desired conditions and/or stand development trajectories were achieved.

Pre-Treatment Inventory Information

Pre-treatment inventories are used to quantify the current conditions in the stand or on the landscape, to estimate how the silvicultural prescription will change those conditions, and to guide the imposition of the proposed treatment. Variables typically include stem density, basal area, species composition, the current structural stage of the stand, canopy layers present or absent, presence of pathogens or insects, edaphic and physiographic conditions, forest fuel conditions, and also the larger context of how the stand contributes to the large landowner goal (biologically, socially, economically).

The pre-treatment condition contains all the elements available to attain the post-treatment stand condition, either directly or through stand development over time. An awareness of what exists now and what must be retained for future needs provides the necessary information about that portion of the biomass--for example, overstory trees, understory vegetation that competes with desired species in the understory, or invasive exotics--that is superfluous to the post-treatment condition. The greater the degree to which the silviculturist can understand the pre-treatment condition, the greater the degree to which the post-treatment stand can be described, and the better the planning that can be done to enable the transition from the exiting to the future condition.

The appropriate sampling method for a particular resource relates to the value of that resource relative to the landowner's ownership objectives. At one extreme, a walkthrough with notes might suffice, in another, plots taken at some predetermined intensity would be indicated. In an extreme example, a 100% tally of high-value products, such as black walnut, or endangered wildlife nesting sites, would be recommended. But some quantifiable or even qualitative understanding of stand condition prior to treatment--soil, vegetative spatial and size distributions--helps not only to decide what treatment to do, but whether or not a treatment is commercially feasible.

The context of operations in the stand being entered is increasingly important in contemporary practice. As recently as several decades ago, silvicultural prescriptions on National Forest land were based on an individual compartment as part of a larger landscape, with no specific requirement for entering adjoining compartments. Typically, about 10% of the compartments on a district were examined annually, their management needs determined, and prescriptions written to achieve desired conditions. And, on districts where rangers and professional staff changed relatively frequently, the order of entry was often directed by an extraordinarily valuable human resource--the field technical crews, who often have the longest tenure on the district, and who remembered when silvicultural treatments had been previously conducted in a given compartment.

The biggest differences today derive from the fact that silviculturists now enter and plan treatment prescriptions on landscapes and/or watersheds of thousands of acres in size, rather than individual 1,000-acre compartments, and desired conditions are locally determined in watershed analyses, or Forest Plans, rather than in agency handbooks. In addition, the complexity of proposed treatments often increases, especially in the wildland urban interface. Increasingly, a soundly-developed silvicultural prescription depends on being able to visualize and document the larger ecological goals on the landscape, and how specific silvicultural practices can be implemented to achieve them.

A Detailed Silvicultural Prescription

A silviculturist whose prescription involves the removal of trees, shrubs, perennials, or herbaceous plants--in short, any biomass in excess of that deemed desirable for retention--must describe how, why, and what is to be removed. That description must be done in sufficient detail such that those responsible for the removal can do so in a way that satisfies both the short- and long-term goals or management direction of the landowner.

For trees of commercial size, some sort of inventory of trees being cut is typically prepared. Similarly, an estimate or description of the non-commercial material to be removed may be needed if that removal costs money, such as through site preparation contracts or fuel reduction treatments.

In some situations, it might be better to mark the trees being retained rather than the trees being removed. Examples include, immature overstocked even-aged stands where designation by spacing or diameter make it easier to implement thinning prescription, the seed cut (cf. Smith 1986) in seed-tree and shelterwood stands where it makes sense to mark the trees to leave as seed producers, and in uneven-aged stands with diverse structural goals to retain after marking.

Marking trees to be retained can be more accurate, especially if marking tallies are generated using sampling methods. A pre-harvest timber inventory has an inherent sample error associated with it. If a 100% tally of trees marked for removal is taken during the marking, the sampling error falls in the unmarked component of the stand. This can lead to errors in achieving the desired prescription goal in the residual stand. However, when marking trees to leave in a stand, the 100% tally is taken of the residual trees, and the sampling error falls in the portion of the stand being removed rather than the portion being retained. Such a procedure would require other means of estimating the number, volume, or other descriptor of trees removed to facilitate their selling in the case of commercial products, or paying for their removal in the case of non-commercial trees.

We suggest that a description of what remains after treatment and how it meets silvicultural objectives or management objectives ought to supercede product sale needs and objectives. In contrast to completing a 100% tally of the residual stand, targeting residual basal areas (e.g., by tree class, by stand, by species, or all combinations) and other stand attributes, such as structural stage distribution, species composition, or canopy descriptions, might be a more meaningful way to describe a stand after treatment.

Another reason to favor the marking of residual trees is that it allows field crews to concentrate their attention on trees, structures, and compositions being retained. This can be of special advantage when non-traditional attributes, such as nesting cavities or potential for development of living and dead snags, are sought for retention. The marking tally becomes the primary point of contact between the intentions of the silviculturist and the actions of the field technical crews responsible for implementation of those intentions. In what is often the greatest single fault with modern forestry, the silviculturist is usually not present when the technical crews mark the stand. The silviculturist must thus ensure that the field crews understand the intent of the prescription, and can act independently in the woods to implement that intended silvicultural objective.

For example, in the free selection approach (Graham and Jain 2005), there is little or no explicit development of quantitative standards for marking. But the qualitative standards are quite well developed, and field crews must have a clear understanding about them in order to mark the stand acceptably. As silvicultural prescriptions become even more innovative to meet targets for stand structure or other ecological attributes, that description and vision must be very clearly defined for the marking crews. And if it can be quantified, so much the better. That is especially important so the field crews can react to local variations of density, structure, and attributes within the stand, instead of forcing predetermined and inflexible standards of basal area, species composition, or spacing in portions of the stand that cannot meet them.

Adequacy of Regeneration

Regeneration of the desired species at the stand level following reproduction cutting is the fundamental stand-level indicator of sustainability. The quantity and quality of regeneration must be appropriate for the stand age, habitat requirement, structural attributes, and species composition sought by the silviculturist for meeting both short- and long-term goals. This implies a survey of appropriate statistical rigor, with a defendable sampling implementation having the power to test an explicitly-given size of departure from a target stem density, which would allow a forester to conclude whether regeneration density and distribution are adequate.

For example, in longleaf pine (*Pinus palustris* Mill.) stands in the lower west Gulf coastal plain, regeneration is an episodic event (Wahlenberg 1946), and regeneration surveys that use plot sampling are important to determine whether adequate regeneration is present. On the other hand, in loblolly-shortleaf pine (*P. taeda* L.-*P. echinata* Mill.) stands of the upper west Gulf coastal plain, regeneration is adequate four years in five (Cain and Shelton 2001). If other conditions are right, loblolly and shortleaf pine seedlings are both abundant and visible in the second year after reproduction cutting onward. After five years, regeneration surveys are difficult to implement because the density of saplings impedes one's progress through the woods.

Too much regeneration is a far more desirable situation than too little, because reducing stem density is usually easier and less costly than increasing it. The longleaf situation is one in which a rigorous regeneration survey should be conducted. In the loblolly-shortleaf example, a visual estimate might suffice to establish whether or not a stand has been successfully regenerated, and if not, whether a plot-based regeneration survey is needed. In some circumstances, the spatial juxtaposition of regeneration (e.g., groupiness, patchiness, relation to reserve trees) is also important for sustaining specific structural stages (Long and Smith 2000).

Sensitivity Analyses Through Computer Modeling

Computer models of forest growth and yield are useful but occasionally demanding tools for foresters to test the outcomes of silvicultural prescriptions. In the Forest Vegetation Simulator (FVS) and other computer models, the tools are available to run growth projections for a given prescription and for variations on that prescription (Dixon 2002). Where sufficient data are available, models are quite useful, especially if interpreted as comparative models in the context of sensitivity analysis, or in the relative evaluation of alternatives over time. In addition, they often contain visualization tools that can display stand attributes (e.g., structural stages, fire characteristics, etc) through time. These visualizations can be effective communication tools for displaying silvicultural treatment results and how they will most likely develop through time. They can be used to communicate with other disciplines, as well as with the public at large.

This is especially important when communicating silvicultural activities and their inherent degrees of risk and uncertainty. A rule of thumb is to assess the relative risk and uncertainty associated with given conditions, and act accordingly. For example, igniting a prescribed fire during severe winds falls into the realm of high risk and low uncertainty--essentially, a situation where the treatment is dangerous and there is little doubt that it is dangerous. Variations that combine low risk with low uncertainty could be imposed with less fear of unacceptable consequences. Situations that include high uncertainty should include provisions for observation of the treatment and its effects that account for the uncertainty associated with the treatment. Finally, one may not need detailed data to understand this--disclosure and recognition of the question may be adequate to assess the relative risk and uncertainty.

Post-treatment Assessments

The ultimate success of any silvicultural prescription is best judged by whether the intended outcome was actually achieved. That starts with a post-treatment assessment of residual stand density, basal area, and stand structure, and comparing that residual stand with the target standards originally specified in the prescription before treatment. That can be done quantitatively or qualitatively if the silviculturist has sufficient experience to judge stand metrics by visual estimation. Among the elements to revisit is whether the harvest retained what was intended, whether the predicted vegetation development (growth and yield) is being obtained, and, if a reproduction cutting was imposed, whether regeneration development is acceptable (quantity, quality, juxtaposition). These are simple metrics to judge within acceptable standards for most purposes using simple walkthroughs and visual estimates, especially if one is experienced with fieldcraft in the given forest type. If the target for a given metric is narrow or exacting to meet habitat or silvical requirements, the metric should be sampled, documented and recorded--not only for recording the treatment for the next silviculturist and facilitating the planning of future treatments, but also documented in a manner so that they can withstand a challenge as to the quality of the data.

Too often, silviculturists fail to invest the amount of time needed to determine whether the treatments that were imposed have actually been successful. The dilemma is easy to understand, because follow-up inspections and reviews are often of lower priority on a day to day basis than the preparation of new silvicultural prescriptions for different areas. Thus, not only must silviculturists appreciate the time required to revisit treatments imposed in the past, but their supervisors must also appreciate the need to invest their subordinates' time in such reviews. Moreover, such reviews are excellent learning experiences and, when conducted in an interdisciplinary manner, they can foster learning among disciplines. By doing so, the development of future silvicultural systems that can fulfill a wide variety of objectives may be more readily achieved and possibly accepted in an integrative fashion.

A complicating factor is that the time span with which inspection of past work is meaningful often exceeds the tenure of a silviculturist on the land base, especially on public lands. Consider that a prescription is often written three to five years prior to harvest, and that meaningful evaluation of success can require the passing of five years to a decade or more. Few silviculturists on National Forest lands are in place for that 10-15 year period. It follows that the need to quantify conditions and quantitatively examine the results of previous prescriptions probably is inversely related to longevity on a district. Moreover, part of the training of a silviculturist new to a district should be a review of a handful of prescriptions that the previous incumbent thought were successful, and also (perhaps especially) those that were not.

Repeated exams after implementation of the prescription are better than just one. Repeated visits give field personnel a sense of the rate of change of conditions over time, and whether or not the treatment is doing what was intended over the short-term. Information from those visits can be used with models such as FVS to give an indication of what the long-term prognosis of stand development will be. Such work favors streamlining and preparing better prescriptions for future stand tending. That is a prerequisite for doing what is intended over the long term.

Subjective Assessments

In contrast to the quantifiable assessments discussed above, subjective assessments are also valuable for judging whether silvicultural operations will be successful. These deal primarily with creating opportunities for sale of the surplus biomass, and to reinvest the proceeds from the sale in operational treatments to further the management objectives. In addition, there are two sources upon which to rely to determine whether a treatment achieves its intended objective--the internal element that allows the silviculturist to decide whether a prescription has been properly imposed, and the external review that allows others to certify the same thing.

Availability of Local Timber Markets

The ability to sell trees to a willing buyer who will harvest them, haul them away, and manufacture wood products from them is a terrific advantage for a silviculturist. Local markets are fundamental to modern forestry, arguably more so today than during the era of timber primacy. When clearcutting was the rule, loggers were assured of large volumes per acre harvested. But harvests today are more likely to have lower volumes per acre because part, if not most, of the trees are retained after the harvest. In addition, fuel reduction and restoration prescriptions often target trees of small size, inferior quality, and low value. Thus, the ability to sell small volumes of products (often inferior in quality) removed during partial cutting is essential to the success of those prescriptions.

Two kinds of markets are needed to enable the future implementation of harvests with low volumes per acre. The first, of largest scale but marginal economic reward, is a fiber market for small diameter products harvested during thinning or other intermediate treatments. However, small diameter products will never generate much more financially than a break-even profit for the land manager, which is still an advantage relative to the costs of conducting a similar treatment non-commercially. The second, of smaller potential by area but of far greater financial opportunity, is a market for large sawlogs of high quality and relatively old age, taken during even-aged reproduction cutting, late-rotation thinning, or uneven-aged cutting cycle harvests. Such products are becoming increasingly scarce as a result of the reduction of rotation age and maximum harvested diameter found on forest industry lands. In essence, the potential exists for a niche market to develop in which some manufacturers of high value products increasingly rely on a sustainable supply of large-diameter trees harvested from public lands. This is a utilization approach that in essence maximizes the financial return per tax dollar spent in forest management--a good place for Federal forest managers to be.

We believe it is especially important that the non-silvicultural professionals on the staff understand the value of market opportunities. An unfortunate legacy of the program of timber primacy that existed within the Forest Service two decades ago is the alienation of many of the other professionals in the agency against the timber program. But there are examples within the agency in which all of the resource professionals on a management staff cooperate to fulfill a complicated set of management objectives, and that use a viable timber program selling trees at high value in local markets to fund the achievement of those objectives.

Operable Harvest

The availability of a local market provides an opportunity for removal of the excess biomass, but enough biomass must be available for sale to interest a local buyer. Sufficient superfluous ecological material surplus to the desired condition will allow either for a commercial sale to be feasible, or allow a contract to be written that will attract a bidder. The question is one of efficiency of operations, and whether the surplus biomass is available, either so someone will buy it, or so someone will be willing to be paid to dispose of it.

As silviculturists, a treatment prescription proposes a change in condition, from an existing condition to a different one. The silvicultural goal is the redistribution of biotic influence within the stand, such that conditions after the treatment better reflect the desired conditions than those that existed prior to the treatment. But if a silvicultural goal is to be met, doing the treatment is better than not doing it. As a practical matter, being paid to execute the treatment is better than paying someone to do it. Both, in turn, are better than not having the treatment done, if it is in fact a priority treatment to conduct. This is why the practice of having an environmental group buy stumpage but not cut the trees is a management failure--the desired treatment effect is not being achieved.

This is currently an important issue in the debate about the Sierra Nevada Forest Plan. Questions exist about whether large trees ought to be included in timber sales in order to attract a willing buyer. The answer to this depends upon the degree to which the large trees contribute to the desired ecological condition of the residual stand. The answer to this debate is beyond the scope of this paper, especially in light of the site-specific conditions that must be considered to make a management decision on the question. But there is nothing innately improper about the sale of biomass surplus to the needs of the stand (most often defined ecologically but may also contain social and economic elements), especially if that sale promotes opportunities to conduct additional treatments that would be beneficial to the eventual attainment of the desired stand condition.

Plans to Reserve Proceeds from Harvest to Enable Additional Treatments

When an operable harvest is made on public or private lands, other opportunities become available for a landowner or manager to reinvest some of the proceeds of the sale in paying for supplemental treatments that bring the stand closer to its desired condition. On National Forest lands, the Knutsen-Vandenberg Act of 1933 and its administrative implementation procedures allow for the development of planned activities for improvement of the sale area. These sale area improvement (SAI) plans allow the reserving of funds received for the harvested timber in order to pay for necessary follow-up activities, such as additional silvicultural treatments. Similar opportunities exist through salvage sale collections and stewardship contracts on National Forest lands.

A classic example is the shortleaf pine-bluestem (*Andropogon* spp.) restoration on the Ouachita NF in western Arkansas to restore floral and faunal diversity associated with those open woodland habitats, including nesting and foraging habitat for the endangered red-cockaded woodpecker (*Picoides borealis*) (Guldin et al. 2004). The first step in implementation of the restoration prescription is a commercial timber sale that thins the overstocked pine overstory. SAI plans prepared for the timber sale allow collection of funds from the proceeds of the sale to use in subsequent removal of the hardwood midstory, and the conduct of a program of prescribed burning for the first decade after the timber sale. Relying on sale proceeds rather than appropriated dollars increases the area that can be restored by several orders of magnitude, making this truly a landscape-scale restoration program.

The concept is equally appropriate for private landowners, too. Reinvestment in the stand for activities that might not have been affordable without a timber sale is a hallmark of clever management planning on private forest land. Some landowners are more comfortable with this idea than others, and whether a given landowner has the wherewithal to divert cash proceeds from a sale to pay for additional silvicultural treatments on the land depends on the wisdom of the landowner (short- and long-term views), and on the quality of advice being given to the landowner by the professional with whom he or she is working.

In a nutshell, the argument here is simple--what is retained is more important to future stand conditions than what is cut. But what is cut can help pay for treatments to optimize ecological condition of what is retained. That's a tradeoff that is unwise to ignore, whether on public or private lands. An important characteristic of disclosing the tradeoffs is presenting the risks and uncertainties associated with each scenario considered.

Including Monitoring Standards in the Implementation

Monitoring standards are a tool used by the silviculturist to codify the plans by which successful implementation of the prescription can be judged. The questions about whether a treatment did what it ought to have done fall along three lines, as recognized by the Forest Service and others:

(1) Implementation monitoring—meeting the standards for implementation. In other words, such monitoring verifies whether the standards relevant to the implementation of the prescription were properly imposed.

(2) Effectiveness monitoring relates to what is being done to achieve the intended silvicultural effect. Here, the question is whether maintaining standards as specified in the prescription actually achieved the effect that was intended.

(3) Validation monitoring attempts to quantify the observed effect with respect to testing whether modifications in the standards should be made. This category is where experimentation can occur to evaluate whether the standards as imposed are actually effective in addressing the questions that they were intended to address, but are seldom applied for individual projects.

The value of conducting one or more of these classes of monitoring relates to the opportunity to conduct an internal process check or review of the implementation of the silvicultural prescription. At the very least, the silviculturist would like some assurance that the treatment was imposed as planned, and implementation monitoring provides that. The larger and more interesting questions, such as whether the treatments that are imposed actually work, or should be modified to work better, are equally important, if not more so, over the long term. Even with the knowledge of the importance of monitoring, it is often one of the most neglected functions occurring in forest management. When it does occur, it can be an expensive endeavor using dubious design and improper resolutions of data chasing ill framed questions. Under such circumstances, these monitoring efforts are often the source of litigation and make no one pleased about the outcome.

External Professional Review of Plans and Products

Related to the importance of monitoring and post-treatment assessment is the concept of internal and external review of silvicultural activities. Such reviews evaluate whether the goals and objectives of a particular silvicultural practice have been achieved. Procedures are currently in place to conduct internal agency reviews. For example, the management review process of the National Forest System gives the local Supervisor's Office an excellent opportunity to review practices at the Ranger

District level, in a constructively critical environment. Similarly, Regional Office and Washington Office reviews meet similar goals.

However, when compared to internal agency reviews, the overwhelming advantage of external review is independence and impartiality. It carries a connotation that success in attainment of objective standards is of greater value, especially if there is no implicit benefit to the reviewer. Outside the agency, third party audits of industrial silvicultural treatments under the AF&PA's Sustainable Forestry Initiative, of other private forest management under the Forest Stewardship Council, or the BMP compliance audits available through many state forestry agencies for non-industrial private forest lands, achieve a similar intent.

These reviews bring credibility to programs and can also highlight areas in which improvement is needed. The reviews from outside the organization can be an effective tool to develop, strengthen, and redirect programmatic support within the organization. Silviculturists can thus secure renewed commitment to the objectives of treatment and to the techniques used to make the treatments happen. Reviews also provide an opportunity to learn the strategies and tactics that others might recommend to meet the intended goals as well.

Summary

The key for a silviculturist to maintain the trust of not only fellow resource professionals and technicians, but also the landowner, is to act according to a simple philosophy--say what you'll do, do what you said, and watch what you did. Factors such as short position tenure and pressures to implement new treatments often conspire against living up to this philosophy. A simple set of protocols is presented here to guide silviculturists in regard to careful implementation and observant follow-up activity over time. The greater the degree of experience a silviculturist has with the place identity and the forest types under his or her management, the more comfortable that silviculturist will be in stretching the application of innovative silvicultural practices, and with watching them over time to see how the residual stand responds to the treatment. But even if a silviculturist is brand new to a region or a forest type, attention to the objective and subjective standards presented here will allow for a more rapid assessment of success or failure to achieve the intended silvicultural goal. In essence, these ideas serve as a beta-testable model for an operationally meaningful program of adaptive silviculture, toward the goal of meeting ownership objectives in a sustainable manner.

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Free Selection: A Silvicultural Option[1]

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Abstract

Forest management objectives continue to evolve as the desires and needs of society change. The practice of silviculture has risen to the challenge by supplying silvicultural methods and systems to produce desired stand and forest structures and compositions to meet these changing objectives. For the most part, the practice of silviculture offers a robust set of procedures well suited for the timely and efficient production of timber crops but too often leaves simplified forests that do not necessarily reflect historical conditions, do not provide a full range of wildlife habitats, nor provide a sense of place for many different forest users. To achieve these and similar objectives we propose a silvicultural system that we call "free selection." This multi-entry, uneven-aged system is intended for use in forests in which the remaining structure and composition is paramount. It is well suited for restoring the old-growth character of forests as well as reducing the risk of wildfire within the urban interface. Rather than using precise stand structural guidelines to define the stand treatments, we suggest that a well articulated "vision" of the immediate and desired future conditions is used to guide the planning and to control the marking. This vision accounts for the interaction of all components of a forest from below ground to the high forest canopy. It relies on an integrated ecological view of how forests function. We have applied free selection guided by such a vision to the dry ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.) forests of southern Idaho to restore their old-growth character. We include a 100-year simulation (using the Forest Vegetation Simulator) of free selection and display stand attributes using the Stand Visualization System.

Forestry and Silviculture's Role

Forestry is a highly integrated profession incorporating the core sciences (mathematics, botany, physics and so forth), applied sciences (silviculture, engineering, fire and so forth), and the political, social and economic sciences (law, policy, decision and so forth) (Nyland 2002). Forestry's professional ethic was articulated by Gifford Pinchot in 1905 as "the use of natural resources for the greatest good of the greatest number for the longest time" (Lewis 2005). The significance of this ethic is exemplified by its use in forming Forest Service policy and its use in forestry texts to express the importance of forests to the citizens of the United States (Meyer et al. 1961, USDA Forest Service 1928). Pinchot, and his ethic, not only set the course for the Forest Service, but because he was instrumental in founding the

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Yale Forestry School, the Society of American Foresters, and the Journal of Forestry, Pinchot all but created the forestry profession in the United States (Lewis 2005). Moreover, within the profession and the conservation ethic that guided it, timber harvesting was permitted but not required for forest management (Lewis 2005). As a result, the interpretation of each phrase of the ethic (use of natural resources, for the greatest good, of the greatest number, for the longest time) has been a struggle for the profession and, in particular, the Forest Service for over 100 years.

Early Forest Service harvesting was in general very light and conservative and concentrated upon the least desirable species and individuals in the stand in an effort to improve the forest (Baker 1934). In doing so, the Forest Service by 1942 was only producing two percent of the nation's timber supply. However, this practice failed to provide sufficient quantities of merchantable material to timber operators, leading to increased use of clearcuts, shelterwoods, and so-called selection systems with long cutting cycles. Although the use of these silvicultural systems allowed heavier and financially attractive cuts, the results were sometimes good and sometimes silviculturally unfortunate (Baker 1934, Lewis 2005).

Following World War II the harvest from private lands decreased and the National Forests were a source of timber to supply the booming housing and consumer markets of the Cold War era (Lewis 2005). The President's Materials Policy Commission during the Eisenhower Administration in 1951 called for developing natural resources quickly to defeat communism. By 1952, the pattern of annual increases in timber production from federal lands was firmly established and by 1960, timber management became the focal point of forest management. Silviculture, being the art and science of influencing the establishment, growth, composition, health, and quality of forests to meet the diverse needs and values (management objectives) of landowners and society on a sustainable basis, responded to the challenge (Helms 1998).

Intensive even-aged silvicultural systems which included the use of herbicides, frequent clearcutting, precise thinning regimes, and relatively short (≈ 60 to 100 years) rotations were prescribed in management plans and dominated forest management during the 1960s and 1970s (Lewis 2005). By the late 1990s, because of public attitudes, laws (for example, Endangered Species Act, National Environmental Policy Act, National Forest Management Act), the recovery of private forest lands and their increased contribution to the timber supply, and the failure of a timber famine to develop, the amount of timber harvest from National Forest lands fell to levels reminiscence of those in 1905. Moreover, by 2001 the Forest Service was committed to ecological restoration and the Chief of the Forest Service, Dale Bosworth, articulated, "what remains in a forest after treatment is more important than what is removed" (Lewis 2005, Miller and Staebler 2004).

Although timber and fiber production are still valid management objectives in many forests (Graham et al. 2005), other values such as water quantity and quality, biodiversity, scenery, old-growth, wildfire resilient and resistant forests, and maintaining the spiritual or sense of place in forests are emerging issues. Sense of place is a holistic concept that focuses on the subjective and often shared experience or attachment to the landscape emotionally or symbolically[4]. It involves a subjective

[4] The importance of this concept was shown on September 11, 2001 when the USDA Forest Service and the USDI Park Service waived entrance fees during Veteran's Day weekend to "help Americans find comfort and solace." Forest Service Chief Dale Bosworth stated,

experience or view of place description of the meanings, images, and attachments people give to specific locations. These places reflect the perception people have of a physical area where they interact, whether for a few minutes or a lifetime, giving that area special meaning to them, their community, or culture (Galliano and Loeffler 1999, Schroeder 2002).

Many of these values depend upon the development and maintenance of complex and interacting forest elements such as high forest cover (presence of tall and/or large trees), disturbance, vegetation patchiness, multiple canopies (vertical diversity), old trees and decadence, down logs, and the presence and interspersion of a complete suite of vegetative structural stages inherent to a forest (Franklin et al. 2002, Galliano and Loeffler 1999, Reynolds et al. 1992, Thomas 1979). Moreover, these conditions often are required to occur over landscapes and be sustained through time. Even though traditional silvicultural systems have ultimate flexibility, these emerging values and the complex combination of forest elements can not be readily quantified or translated into traditional stand metrics (Backlund and Williams 2004, Franklin et al. 2002, Oliver and Larson 1990). Also, because of fire exclusion, animal grazing, timber harvesting, and other forest disturbances or lack there of, many of the current forests have higher densities, different vegetation compositions, and different disturbance regimes not apparent in past forests (Covington and Moore 1994, Graham 2003, Graham et al. 2004, Quigley et al. 1996). Additionally, a resilient forest today is most likely different than those occurring in the past because of climate change (cycles) and the introduction of exotic plant and animal species that are now an intrinsic part of the environment. This all poses a unique challenge to silviculturists. Yet, because silviculture is founded on studies of the life history of forests and has been honed by experience from more than 100 years of management, it is well suited for meeting these challenges.

Through time, silviculture has evolved as a consequence of the progression in values and needs of landowners and society beginning as early as the 17th Century (Evelyn 1664). As a highly integrative discipline and an applied science, silviculture is a continuing, informal kind of research in which understanding is sought and new ideas are applied (Smith et al. 1997). In particular, the concepts and methods inherent in traditional even-aged and uneven-aged systems can be used for developing and maintaining forests that meet these new and emerging objectives. This is a perspective that the forestry community needs to better recognize and contemplate, and foresters must become emboldened about using and implementing them in innovative and creative ways in order to meet the challenges of the 21st Century (O'Hara et al. 1994).

The Evolution of Selection Systems

In North American silviculture, four primary silvicultural systems are normally recognized: clearcutting, seed-tree systems, shelterwood systems, and selection systems (Baker 1934, Smith et al. 1997). Baker (1934) suggested that the systems are often confused with rigidity when actually they are as flexible as the silvical conditions require. He went on further to stress that there are not three or four or ten

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"National forests and grasslands can offer peaceful experiences and spiritual renewal." This gesture by public agencies acknowledges the importance of the experiences people have in natural places (Schroeder 2002).

or a hundred separate and discrete silvicultural systems, rather silvicultural systems are more or less arbitrarily named classifications of the almost infinite number of possible combinations under which a forest may be cut. This was very evident early in the 1900s with the formation of silvicultural strategies intended for harvesting and managing forests, in particular those containing irregular structures and complex compositions (Gifford 1902, Schlich 1904). During this time, selection systems were most often used to manage stands and forests to reflect this heterogeneity.

In the 17th Century, John Evelyn (1664) in his presentation to King Charles II, of England, described individual tree treatments that appeared to be early forms of single tree selection to provide timber for England's burgeoning navy. He suggested that for *"vigour and perfection of Trees a Felling should be celebrated; since whiles our Woods are growing it is a pity, and indeed too soon; and when they are decaying, too late."* He also suggested *"for the improvement of the speedy growth of Trees, there is not a more excellent thing than the frequent rubbing of the Boal or Stem, with some piece of hair cloth, or ruder stuff, at the beginning of spring."* He also supplied descriptions of harvesting, drying timber on the stump, and other techniques applicable for managing individual stems in forests.

In the late 1800s, based on his experience in India and England, Schlich (1904) described selection systems, shelterwood selection systems, and two specialized selection systems that were intended to improve game production and domestic animal grazing. In the United States, Gifford (1902) described selection systems as ideal for protecting the soil resource and "an excellent system for the production of park effects where variety is desirable." He went on to say "in this system the best is constantly favored. It is a process of weeding out the poor kinds and favoring the good. It is just the opposite of what has been practiced heretofore in this country."

Krauch (1926) described several different cuttings on Forest Service (Coconino and Tusayan National Forests) and private lands near Flagstaff, Arizona. His study was designed to determine volume increment, and he classified the trees as blackjacks or yellow pines (*Pinus ponderosa* Dougl. ex Laws.) and further divided these classes into thrifty and unthrifty trees. Furthermore, because of the heterogeneity of the stands, he concluded that (based on tree classifications) determining volume increment per tree was far superior than determining volume increment per acre even when sample plots exceeded 450 acres.

Baker (1934) described the selection system as harvesting technically ripe trees, simultaneous thinning or improvement cutting, and the reservation of seed trees where necessary. Also, he described a transition selection system used on National Forest lands in the western United States that utilized cutting cycles of 30 to 40 years. This cutting was a temporary and crude method of harvesting which would later give way to more intensive methods. Hawley (1937), to counter this crude selection, stressed the concept of defined cutting cycles for use with selection systems. He used diagrams to describe the age distributions of trees dispersed in an ideal selection system and the random spatial extent of the 100 different age classes. He also described a maturity selection system used in ponderosa pine forests of the Pacific Northwestern United States in which approximately 40 percent of the volume was removed and another harvest was planned in about 30 years.

These selection systems were designed to reserve 20 to 40 percent of the sound and thrifty trees in a stand. A shortcoming of these prescriptions was when the percentage guidelines (quantification) were strictly followed it resulted in unsatisfactory results (Dunning 1928). Dunning (1928), using European tree classifications, as well as those presented by Krauch (1926), developed a tree classification for use in selection forests of the Sierra Nevada. He demonstrated how seven qualitative tree classes could be used to mark ponderosa pine stands and he quantified the prescriptions by determining the proportion of basal area and trees per acre occurring in each of the tree classifications.

Keen (1936), using a similar approach as Dunning, defined four age classes of ponderosa pine and further divided each age class into four crown-vigor classes for determining a tree's susceptibility to bark beetle attack. He redefined his classification (Keen 1943) for use in the Pacific Northwest and indicated it had been adopted in the Black Hills of South Dakota and Wyoming and in the southwestern United States. Roe (1948) provided tree vigor classifications for western larch (*Larix occidentali* Nutt.) and Douglas-fir (*Pseudotsuga menziesii* Mirb.) Franco var. *glauca* (Beissn.) Franco). Wellner (1952) developed a vigor classification for western white pine (*Pinus monticola* Dougl. ex D. Don).

Throughout the western United States these vigor classes were used to select individual trees to leave or harvest in partial, improvement, high risk, salvage, and other cuttings. Harvesting occurred but a portion of the value (volume) was reserved for later harvest and often these cuttings were repeated (≈ 5 to 20 year intervals) allowing uneven-aged or irregular structures and compositions to develop (Graham et al. 1999).

Meyer (1934) indicated that ponderosa pine responds well to many different silvicultural practices and described the Forest Service ponderosa pine management in Oregon and Washington as approximating heavy grade selection cutting. It had characteristics of tree selection, group selection, and a shelterwood. The system stipulated that the faster growing trees and trees less subject to windfall and insect damage be left. At least 15 to 30 percent of the merchantable volume would be reserved for accelerated increment and insurance of seed supply and for a later cut planned in 40 to 75 years. He illustrated the concept visually in diagrams and photos.

Meyer (1934) also recognized that even-aged yield tables had little value for estimating the yields of uneven-aged stands and, in particular, those treated by selection cutting. He used Dunning's (1928) seven crown classes to represent structure based on the proportion of the basal area or cubic volume occurring in each crown class within a stand. However, he suggested using all seven classes would be too unwieldy and that classes 1, 2, and 3 exert the most powerful influence upon volume growth. Therefore, a 25-50 structure indicated 25 percent of the basal area or cubic volume occurred in crown classes 1 and 2 and 50 percent occurred in crown class 3. This structure information and the correction factors he developed could be used to project the yields of uneven-aged ponderosa pine stands. He illustrated how tree classifications could be used in different cutting methods and showed the effects the different prescriptions had on yield and increment.

Pearson (1950) described several forms of selection silviculture that had been used in the southwestern United States. Group selection systems in which yellow pine groups of large trees were removed leaving blackjack groups were commonly used in the early 1900s. To favor regeneration which was poor in 1913, light selection systems were used in which more mature trees were left between the groups of black jacks. As stated earlier, Keen's (1943) tree classification had been adapted for use in the Southwest and maturity selection was devised using these classifications. Thus on Forest Service lands, prior to 1946 a variety of stand conditions were created using these systems. Pearson (1950) indicated that using tree classifications in the Southwest for selecting trees to leave and harvest had mixed results. Pearson recognized the heterogeneity of the spatial distribution of ponderosa pine, and that a tree's position on the ground was an important determinant of its growth. Keen's (1943) tree classifications did not reflect this heterogeneity. To explain this phenomena, he explored the implications of the groupy nature of ponderosa pine on the resulting root patterns. Taking all of his understanding of ponderosa pine regeneration and development, he devised a selection system called, "improvement." In general, the aim of this selection system was to build up an effective growing stock and this goal would take precedence over immediate timber sale receipts and yield in the near future. This qualitatively described system integrated tree classes, soil moisture, and bole descriptions. The system removed fewer yellow pines and the limbiest blackjacks. Pearson (1950) indicated that the increment borer was a better guide than spacing rules for implementing the system.

Kohm and Franklin (1997) have suggested modifying traditional even-aged systems in particular clearcuts with the addition of reserve trees or retaining green trees. Even though they do not term their work as silvicultural selection, it does reflect some of the characteristics of early selection systems in which a proportion of the stand and forest was retained (Meyer 1934, Pearson 1950). By using this method, the stand and structures they suggest to maintain take on more of a multi-aged condition than an even-aged condition. They suggest such systems for use in the Douglas-fir region of the Pacific Northwest for commercial timber harvest, and they stress maintaining structural and functional legacies such as snags and down-logs as important forest characteristics.

For use in the Pacific Northwest, Camp (1984) described what he termed natural selection, an all-aged and all species management system. The strategy he described was aimed at the small landowner and emphasized the characteristics of selected trees to be removed. He emphasized removing trees that were diseased, broken, suppressed, or those having no ecological value. His system stressed producing a variety of products, including mushrooms, hardwood for furniture, fence posts, huckleberries, and so forth, while at the same time providing an environment of great pleasure.

Timber Management and Selection Systems

Meyer (1934, Meyer et al. 1961) described the classic reversed-j shaped diameter distribution as an approach of forest regulation for obtaining a sustained yield of raw materials for industry and economic support. They based their descriptions of uneven-aged stands or forests on work by the Frenchman De Liocourt in 1898. However, they also emphasized that forests expressing this structure were practically nonexistent in the United States. They went on to indicate that an uneven-aged forest is a forest in which no separate age classes are recognized and even-aged stands may be present but are not treated as permanent units. Under this concept of management, the actual age of trees has little or no practical importance. They indicated that stand/age relations, yield tables based on age, site index, and other even-aged concepts applied to uneven-aged management were misleading, inaccurate, and a waste of time. They also suggested that an entirely different philosophy of management concepts and characteristics must evolve.

Davis (1966) also described the regulation of forests using uneven-aged stands for the production of timber. He stressed that a clear distinction should be made between silvicultural treatments and the general timber management framework. By doing so, he suggested that much confusion could be avoided. In contrast to Meyer et al. (1961), Davis placed less emphasis on diameter distributions and more on volume control for regulating uneven-aged forests. He concluded his discussion by indicating that uneven-aged management is simple when using a general outline but complex when applied.

Davis (1966) and Meyer et al. (1961) eloquently described the concepts and procedures for regulating uneven-aged forests and inferred that the application of uneven-aged regulation was fraught with difficulties. As a result, it is not surprising that there are few examples where uneven-aged management has been planned and implemented in the United States. This was apparent when workshops held in both the eastern and western United States in 1975 and 1976 reviewed the concepts of uneven-aged management, no examples of operational uneven-aged application were offered, but excellent examples of selection systems (uneven-aged management), applied experimentally, were shown (USDA Forest Service 1978). Similarly, in 1997, an international symposium on uneven-aged management affirmed the above observation, but examples of operational uneven-aged management in Europe were presented (Emmingham 1999).

Haight and Monserud (1990a) evaluated optimum any-aged management of mixed species stands. For their optimization they chose a mixed conifer stand represented by the western hemlock (*Tsuga heterophylla* (Raf.) Sarg.) habitat type that occurs in northern Idaho. Grand fir (*Abies grandis* Dougl. ex D. Don), western white pine, Douglas-fir, ponderosa pine, western redcedar (*Thuja plicata* Donn. ex D. Don) along with western hemlock naturally regenerate in these forests in response to canopy opening. They illustrated that commercially thinning all trees between 7 and 18 inches produced the optimum and sustainable management evaluated by present value. Haight and Monserud (1990b) refined this prescription to show that optimal any-aged management during the first 40 years of a stand's life included thinning heavily from above removing all trees greater than 10 inches and precommercially thinning a portion of the trees between 2 and 7 inches. In 60 years and beyond, optimal harvesting approached a steady state by commercially thinning all merchantable trees and controlling the number of younger trees with precommercial thinnings. Using this approach, Haight et al. (1992) showed the unconstrained optimal any-aged management (determined by maximizing present value) for mixed conifer stands occurring on the grand fir habitat type to consist of cutting all merchantable trees every 20 years and precommercially thinning trees between 4 and 7 inches in diameter.

The previous discussion illustrates that timber management and silviculture are two distinct but highly related disciplines involved in the forestry profession (Graham and Jain 2004, Nyland 2002). Meyer (1934) illustrated uneven-aged (60 to 579 years) ponderosa pine stand structures both spatially and by the proportions of trees occurring in different crown classes. He went on to project the annual volume increment 60 years into the future of these qualitatively defined structures. The work of Haight and associates (Haight and Monserud 1990a, 1990b, Haight et al. 1992) illustrate what they termed any-aged management was sustainable but the diameter and/or age distributions were far from the reversed j-shaped curve that Meyer presented for timber management. Even though Meyer (1934) and Davis (1966) described the balanced diameter distribution as a way of regulating uneven-aged forests, or working circles for timber production (in other words, the forest area at which a sustained yield of products was determined), these distributions are frequently associated with selection silviculture. They are often used as targets for stand structure and the presence of a balanced uneven-aged (diameter) distribution(s) is often used as an indication of sustainability even though they were not designed to do so (Graham and Smith 1983, Graham et al. 1999). This discussion of selection systems and the forest management settings in which they have been developed and used illustrates that they take on many forms. Most importantly they have been developed and applied since the 17th Century to meet the objectives of the forest land owner. Within this context we present a selection system that we feel has applicability for meeting many of the challenges of forest management and silviculturists of the 21st Century that is built upon this foundation of selection system development and application (for example, Davis 1966, Keen 1973, Meyer 1934, Pearson 1950).

Free Selection

Selection silvicultural systems (uneven-aged) have a longer, more diverse history of definition and application than any other system. In general they were designed to maintain high forest cover through the use of treatments that ensure the development of desired forest structures and compositions that produce a continual flow of goods and services. These systems are particularly well suited for meeting many of the emerging and current management objectives (Graham and Jain 2004, Marquis 1978, Nyland 2002, Smith et al. 1997). However, silviculturists responding to these management issues have not only applied uneven-aged systems but have modified even-aged systems by specifying reserve tree components, patch size and group metrics, ground level vegetation requirements, and deadwood components to name a few (Camp 1984, Kohm and Franklin 1997, Meyer 1934, Pearson 1950). By doing so, the distinction between even and uneven-aged systems becomes less obvious and such systems, even though they appear to be new, are reminiscent of those applied early in the 20th Century (for example Dunning 1928, Keen 1943, Gifford 1902). Even with the specific quantitative designations of reserve trees and other metrics, such defined structures and compositions will not generally emulate horizontal tree distributions or the juxtaposition of the different structural stages inherent to natural forests that are often the focus of numerous management objectives (Franklin et al. 2002, Reynolds et al. 1992, Thomas 1979).

Elements from both even-aged and uneven-aged silviculture can be integrated to produce diverse stand compositions and structures (Nyland 2002). In contrast to early silvicultural systems, an integrated system might include provisions for maintaining a variety of structural stages, tree densities, patch sizes, compositions, tree sizes and so forth within stands and across landscapes in a pattern reminiscent of those that historically occurred (Long and Smith, 2000). Such a system would provide for snags, decadence, down wood, and other often overlooked forest components (for example, interlocking crowns, interspersion of structural stages, disturbance) that are relevant to many current forests and management objectives. We call such a hybrid system “free selection.” It is a silvicultural system suited for maintaining forests with high cover and heterogeneity both in composition and structure. Because it is a selection system (uneven-aged system), it utilizes multiple tending and regenerating entries at various intervals to develop and maintain the desired forest conditions.

Similar to traditional uneven-aged systems, the full range of silvicultural methods from regeneration to thinning can occur at each entry, if needed (Smith et al. 1997). Successful regeneration (natural or artificial) is required when implementing the system to ensure that future desired forest conditions are developed and maintained. All tree, shrub, forest floor, and other components need to be evaluated and managed to create, develop, and maintain the desired forest compositions and structures. Free selection may also incorporate openings (for example, as done in group selection systems) of sufficient size to regenerate early and mid-seral species (for example, western white pine, western larch) but not necessarily provide them optimum space for long-term development. Because it is a selection system, subsequent treatments would tend to these regenerated cohorts, releasing selected trees while insuring that they contribute to the desired stand and forest composition and structure in the immediate and long-term (Jain et al. 2004).

Because free selection incorporates multiple entries, patience can be exercised in developing the desired forest structures and compositions. The term “free” indicates that the frequency, kind, and intensity of entries are undetermined but will depend on how the stand develops within the context of the biological and physical environment when fulfilling the desired conditions. The system could be viewed as stand or landscape level adaptive management (Franklin et al. 2002). In addition, it is similar in concept to applying an even-aged system in a fine scale mosaic or group selection using area regulation. Even though the practice of silviculture strives to create desirable residual stand conditions, free selection appears to be very appropriate in situations where the condition of the forest after treatment is of paramount importance, such as maintaining conditions for wildlife or providing a feeling of security and wildness in the urban interface (*see footnote 4*). We suggest that free selection is applicable in both the moist and dry forests in the western United States for addressing hazardous wildfire conditions within the wildland-urban interface, for restoring and maintaining old forest structures, or for other objectives that require the maintenance of high forest cover and a diversity of forest structures and compositions at various spatial scales. In contrast to traditional single-tree and group selection systems that depend on precise diameter (age) distributions, we believe that free selection is best applied using a vision that describes a desirable set of forest and stand conditions in both the short- and long-term.

The Free Selection Vision

A vision articulates a comprehensive description of the desired forest conditions both in the short-term (10s of years) and long-term (100s of years) over multiple spatial scales ranging from canopy gaps to landscapes (Long and Smith 2000). The use of a vision encourages collaboration and a common understanding between and among natural resource disciplines as to the forest conditions required to fulfill the management objectives. Moreover, a vision can be an excellent communication tool among and within disciplines and with the public at large. Excellent visualization systems are available to illustrate a vision within stands and across landscapes (for example, Forest Vegetation Simulator, Stand Visualization System) (Dixon 2002, USDA Forest Service no date).

Based on our experience of implementing strict quantitative uneven-aged systems we believe a vision, based on silvics and ecology, is preferred to highly technical stand descriptors that may have limited practical use (Graham et al. 1999). No matter how complex and precise a quantitative silvicultural prescription is, it cannot encompass all of the structures, compositions, processes, and functions inherent in forests, nor can it include all of the forest conditions that are presented when a prescription is applied. We believe a vision can incorporate a diversity of structures, spatial pattern richness, long time periods, and the complex contribution of disturbances that Franklin et al. (2002) indicate are lacking in traditional silviculture[5]. However, stand descriptors and especially their variation (for example, range of tree density in basal area, or range of tree numbers per unit area, variable tree spacing) and ecological thresholds (for example, basal area at which bark beetles become problematic, or canopy openings where one tree species can have a competitive advantage over another) are often beneficial when communicating a vision.

A well expressed vision includes management objectives to insure an appropriate outcome at an appropriate temporal and spatial scale is achieved. Also, it should include the relevant structural features (for example, big trees, patchiness, horizontal diversity) of a forest that fulfill the management objectives. Included in a vision is a well conceived view of forests incorporating complex structures (for example, soils, vegetation, biological legacies), processes (for example, succession, disturbance), appropriate concepts (for example, wildlife habitat connectivity, vegetative structural stages), and the recognition of ecological variation relevant to a particular setting (Franklin et al. 2002). A comprehensive and inclusive description of the sub-stand components, stands, and forests, is suggested as more important than precise and complex quantification. This description would include such things as the desired composition, seral stages, horizontal and vertical structure (mix of structural stages), patchiness, decadence, forest floor conditions, down logs both in the short-term and into the future, and other features as required. The tree species preferences for a given situation can be described, as well as the regeneration requirements of the various species (tree, forb, and shrub) that may occur on a site. Detailed information about each attribute is not necessary; rather an integrative view of the attributes is suggested when describing the vision.

Reference conditions (for example, historical, hypothetical, functional and so forth) can further explain the vision with the understanding that these conditions may not be possible, or necessarily desired. However, reference conditions can be used to provide context or give practitioners and the public with a view, feeling, or concept of what the vision is attempting to express (Franklin et al. 2002). However, the vision must be set in context with the current stand conditions (for example, soils, down wood, ground level vegetation, overstory, wildlife use) and the ongoing disturbances, or lack thereof, thus providing the boundaries that are essential for planning silvicultural activities and ensuring the desired future condition (vision) is fulfilled.

Quantifying Complex Forests

We have found that traditional stand and forest descriptors, such as trees per unit area, basal area, tree spacing, species preference lists, and similar metrics, are deficient in their ability to effectively disclose complex forest structure and

[5] Franklin et al. (2002) suggest that foresters can and must learn to manage stands that sustain biological diversity and a range of essential processes. They describe over 40 complex structures, structural processes, and spatial patterns of structural elements that operate during stand development and they list nine developmental stages of forests.

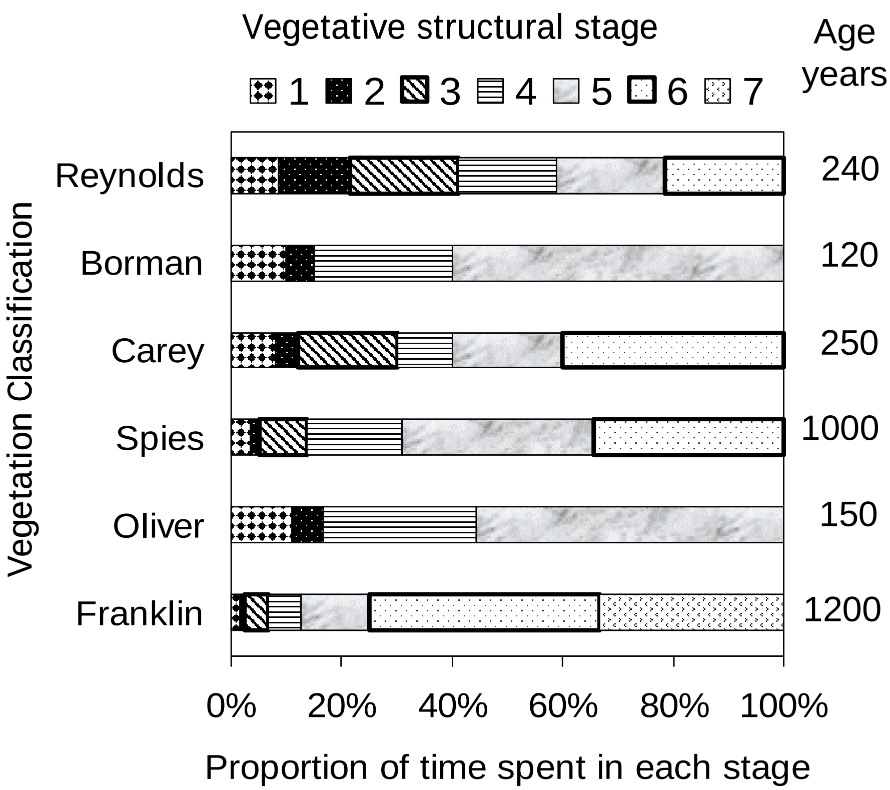
composition. One alternative is to classify relevant forest structure and/or composition and describe the distribution and proportion of the stand or landscape to be created and maintained in the desired vegetation class. Oliver and Larson (1990), Franklin et al. (2002), Thomas et al. (1979), have proposed forest structural classifications that can be related to wildlife habitat, timber production, a functioning forest, sense of place, old-growth, or other forest attributes that society values. In general they have described forest development from vegetation initiation after a disturbance through various stages of maturation. Franklin et al. (2002) and Spies and Franklin (1996) described multiple (six or greater) developmental stages for periods exceeding 1,000 years (*fig. 1*). Reynolds et al. (1992) described six structural stages and related their occurrence to wildlife populations. They went on to suggest that proportions of a landscape should mirror the proportions of the years that each structural stage occurs within the life of a forest.

In general, the amount of time a landscape spends in the early vegetative structural stages tends to be less than the time spent in the older structural stages. However, even in the longest lived forests, a portion of the landscape contains early vegetative structural stages. These stages are often given names such as initiation, cohort establishment, ecosystem initiative, or other terms (*fig. 1*). Most often these structural stages occur across the landscape in a highly intermixed fine scale mosaic, especially forests frequented by low severity, mixed or frequent fire regimes. On these settings, fire, insects, diseases, wind, snow, and ice can facilitate the development of the different structural stages (Franklin et al. 2002, Long and Smith 2000, Reynolds et al. 1992). In contrast, forests frequented by severe lethal crown fires (in other words, lodgepole pine, *Pinus contorta* Dougl. ex. Loud.) tend to have vegetative mosaics with larger patch structure (Fischer et al. 1987).

Vegetative classifications such as those presented by Oliver and Larson (1990), Franklin et al. (2002), Thomas (1979), Reynolds et al. (1992), or others may capture issues of importance, such as fuel condition class or sense of place (*fig. 1*). However, there is difficulty when attempting to identify areas characterizing a particular classification level from traditional stand metrics (for example, height, diameter, density). Therefore, we suggest classifying the vegetation prior to quantifying its attributes. The proportion of each structural stage occurring can be estimated for stands, landscapes, polygons, or other aerial extents (Meyer 1934). A useful analogy for understanding this approach is to consider a room with 100 people. Often in forestry, heights and diameters of trees are estimated in a stand and they are used to classify the stand, for example, as old-growth. This approach would be similar to taking the height and weight of each person in the room and using this information to estimate the number of males and females in the room, a very dubious undertaking at best. A far better approach is to classify each person as either female or male and then describe their weights and heights. Using vegetation classifications in this way may be very appropriate for quantifying free selection prescriptions to determine if the variation within and among aerial extents is being achieved and can be used to determine the location and intensity of subsequent entries (Meyer 1934) (*Appendix A*).

Free Selection in the Dry Forests

We suggest that free selection has the most applicability in forests in which a fine scale mosaic of vegetative structural stages is desired and treatments can be



Examples of names given by authors
to similar structural stages

Structural stage

1. Establishment, initiation, ecosystem initiative, reorganization, grass/forb/shrub
2. Canopy closure, stem exclusion, thinning phase, aggregation phase, seedlings/saplings
3. Biomass accumulation, young forest
4. Maturation, understory re-initiation, mature phase, transition phase
5. Vertical diversification, old-growth, early transition, niche diversification, steady state
6. Mature, horizontal diversification, late transition
7. Old, pioneer cohort loss, shifting gap phase

Figure 1—The proportion and amount of time a forest spends within a structural stage depends on the classification system used, the total forest age represented by the classification system, and the rate at which the vegetative structural stage develops and passes into another stage (Bormann and Likens 1979, Carey and Curtis 1996, Franklin et al. 2002, Oliver and Larson 1990, Reynolds et al. 1992, Spies and Franklin 1996).

applied at relatively frequent intervals to maintain the desired structures and compositions (Reynolds et al. 1992). Beginning in the 1990s, we used free selection to successfully treat stands within the dry forests (for example, those growing on Douglas-fir potential vegetation types) of southern Idaho (*Appendix A*). Our objective was to restore and maintain the old-growth character of ponderosa pine stands and, in particular, decrease the risk of lethal stand replacing fires in the Boise Basin Experimental Forest located near Idaho City, Idaho.

Reports of forest settings prior to European settlement (late 1800s) (Fulé et al. 1997) and those desired by wildlife (Reynolds et al. 1992, Thomas 1979) were used as reference conditions to develop our vision, target stands, and the desired future conditions. Most working hypotheses suggest that dry forests were dominated by ponderosa pine but species composition has changed since the late 1800s (Covington et al. 1994, Everett et al. 1994, Hann et al. 1997). Low intensity, non-lethal surface fires were frequent in the dry forests and endemic populations of insects and diseases interacted with these fires to create a mosaic of forest conditions (Agee 1993, Fulé et al. 1997, Hann et al. 1997, Kaufmann et al. 2000, Sloan 1998, Steele et al. 1986). In general, minimal amounts of shrubs and trees (ladder fuels) occupied the lower vegetative layers (Harrod et al. 1999, Pearson 1950, White 1985) and snags, decadence, grasses and forbs, and down logs were irregularly distributed across landscapes (Hann et al. 1997). Because of frequent fires that occurred prior to 1900, surface organic materials did not usually accumulate and ectomycorrhizae and fine roots tended to develop deep in the mineral soil, thereby protecting them from damage during the frequent surface fires (Harvey et al. 1999).

Using this information, we defined the immediate and desired future conditions for the ponderosa pine stands in southern Idaho as consisting of an aggregation of the forested clumps of structural stages ranging from stand initiation to old forest, like those that existed prior to 1900 (Long and Smith 2000). Grasses and other ground level vegetation are an integral component of the desired setting reflecting the open, park-like appearance. Organic layers fluctuate in depth reminiscent to those maintained by low intensity surface fires. Crown base heights will be high (>30 ft.), and because of the tree patches and low tree density, canopy bulk density will be low. Ladder fuels will vary depending on structural stage, but canopy discontinuity will minimize crown fire risk. The current condition (what was presented) bounded the vision and guided the kind, intensity, and location of treatments that would fulfill the vision (*Appendix A*).

Untreated Stand Conditions

In areas within the Experimental Forest where harvesting had not occurred, large ponderosa pines tended to dominate the ridge tops and side slopes. Because fire had been excluded in the Forest for over 100 years, a mixture of Douglas-fir and ponderosa pine occupied the intermediate and mid-canopy layers (*fig. 2*). These small trees create ladder fuels that allow wildfires or prescribed fires to burn crowns of the large ponderosa pine. The dominant trees, 150 to 450 years-old, occurred as isolated trees and in groups of trees with interlocking crowns (5 to 8 groups per acre) (*fig. 3*).

Figure 2—An example of an untreated stand within the dry forests of southern Idaho. Note the inherent groupiness of the stems and the low crown base heights.

The size of these tree groups ranged from 0.008 to 0.10 acre and tree density within the groups ranged from 22 to 800 trees per acre (*fig. 4*). The mean stand density of live trees averaged 73 trees per acre and the diameters of the dominant trees ranged from 8.0 to 33.9 inches.

At the base of the large ponderosa pine, needle and bark slough had accumulated resulting in deep layers (over 3.5 inches) of organic material. These layers contained over 0.005 grams per cm^3 of fine roots (obtained from soil cores, 4 by 12 inches, extracted from around the base of large ponderosa pine). Because of the presence of fine roots in these layers, the destruction of these layers could stress or even kill these large trees. This observation exemplifies the importance of incorporating the full range of forest components (for example, soil, trees, snags, shrubs) when developing a vision and free selection prescriptions.

Stand Conditions After Treatments

We decreased the ladder fuels and removed as much Douglas-fir as possible while still maintaining the integrity of the stands (*Appendix A).* We wanted to maintain the clumpy nature of the large ponderosa pine, plus increase regeneration of ponderosa pine, grasses, forbs, and shrubs. When marking, we were aware of stand densities (>120 ft^2 basal area pre acre) at which bark beetles (*Dendroctonus* spp.) become problematic (Schmid and Mata 1992) and watched for locations where root disease (*Armillaria* spp.) would likely threaten or kill Douglas-fir. We also expect future mortality from disease, insects, weather, and fire (*Appendix A*). This

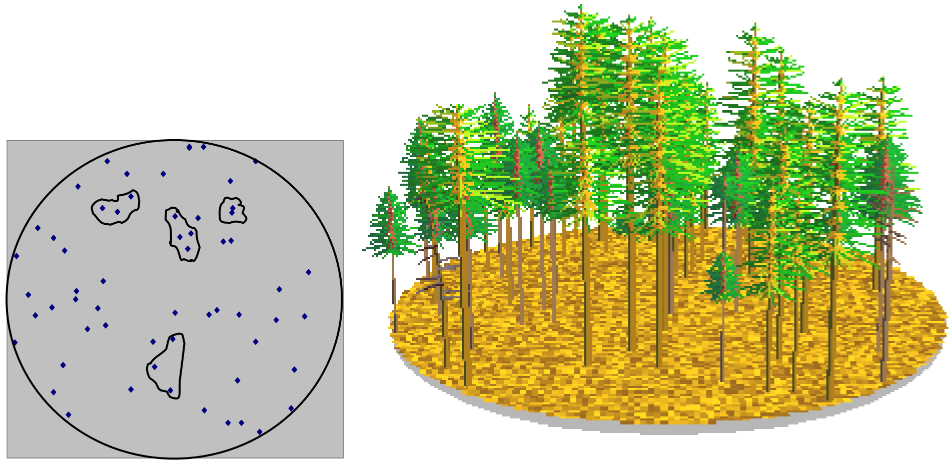


Figure 3—A stand map and visualization of an untreated (plot 10) mature stand of ponderosa pine and Douglas-fir growing on the Boise Basin Experimental Forest in southern Idaho. Four groups of pines were defined as those with touching or overlapping crowns.

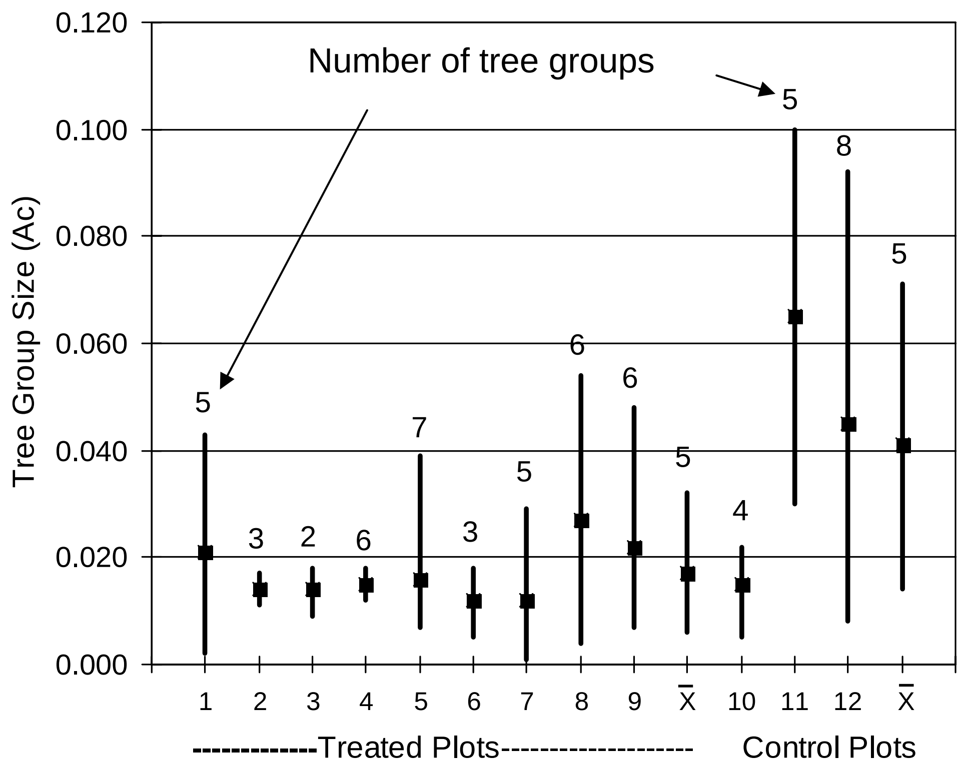


Figure 4—Stand structure within the dry forests of southern Idaho. The maximum, minimum, and mean size of tree groups defined as overstory trees with touching crowns occurring within treated areas (plots 1-9, 1.0 acre in size) and control areas (plots 10-12, 1.0 acre in size) and their respective means ($\bar{x}$). The values above the bars equal the number of tree groups identified per acre.

current and future endemic mortality was included in our vision.

The stands have gentle, sloping (< 35 percent slope) and undulating topography, requiring shifts in tree density and species composition from one place to another. Along ridges we kept large ponderosa pine but often emphasized shrub communities on more northerly exposures and at the base of slopes where tree root diseases tended to occur. On the steeper (> 30 percent), southerly slopes, we created or maintained conditions which encouraged the development of grass and forb communities. By maintaining this pattern of species occurrence and stand structure, we maintained the natural heterogeneity of the site (*Appendix A*).

The cutting and cleaning operations reduced the canopy bulk density and continuity along with reducing the ground level and mid-story ponderosa pine trees (ladder fuels) (*fig. 5, Appendix A*). After treatment, 91 percent of the trees in the stands were ponderosa pine and 9 percent Douglas-fir. The remaining 150 to 450 year-old high canopy was irregularly distributed (*figs. 5, 6, Appendix A*) with up to 7 tree groups per acre ranging in size from 0.001 to 0.048 acres (*fig. 4*). Basal area within some tree groups exceeded 1800 ft^2 per acre (*fig. 7*) but the stand containing this group averaged 64 ft^2 per acre of basal area (*fig. 8*). This density is below the threshold where bark beetles frequently stress or kill trees (Schmid and Mata 1992).

Mechanical methods and/or prescribed fire are being used to reduce the organic layers around large ponderosa pine but in a way that prevents fine root mortality and encourages their development in the deeper mineral soil layers. This includes mixing the organic layers and burning the surface organic material when moisture content of lower organic layers exceeds 100 percent and temperatures at similar depths are below 40^o F (fine root activity is minimal at this temperature). Mixing the surface organic layers allows moisture to more readily penetrate and, because canopy cover was reduced, more heat reaches these layers fostering decomposition. Burning under these conditions allows the surface layers (1 to 3 inches in depth) to be consumed, similar to peeling an onion. These conservative techniques reduce the deep organic layers and encourage fine root development in the mineral soil (*fig. 9*). After we found the fine roots concentrated in the mineral soil, we used a low intensity surface fire to clean the forest floor and to create the desired conditions (*fig. 9*). This is an example of how the intensity and timing of treatments used in free selection are predicated on how forest components (in other words, surface organic layers and fine roots) respond to treatments.

Discussion

The forestry profession in the United States was founded in the conservation ethic articulated by Gifford Pinchot in 1905 (Lewis 2005). Timber production was not a requirement of this ethic; however, it was permitted. By the 1960s, in the United States, timber management was the primary objective of forest management and this objective was firmly associated with the practice of silviculture; nonetheless, they are two distinct disciplines (Meyers et al. 1961, Nyland 2002). Silviculture was described early in the 1900s as applying silvicultural systems within forests to produce desired forest conditions to meet the objectives of the land owner (Gifford 1902, Schlick 1904). In general, this definition is still valid today (Helms 1998). What continues to evolve and change and cause confrontation since the dawn (1900) of the forestry profession in the United States are the objectives for forest management, especially those occurring on public lands (Lewis 2005). In that light,

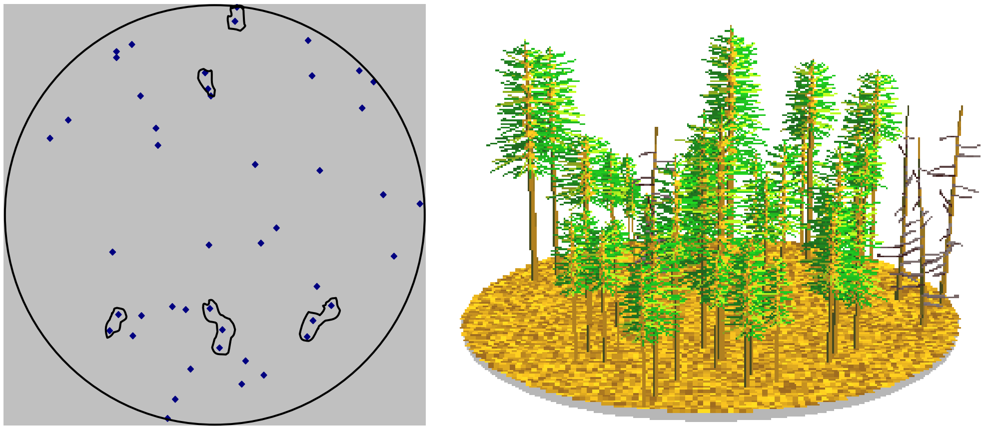


Figure 5—A stand map and visualization of a treated (plot 7) mature stand of ponderosa pine and Douglas-fir growing on the Boise Basin Experimental Forest in southern Idaho. Five groups of pines were defined as those with touching or overlapping crowns. Note the 3 snags showing on both the stem map and the visualization on the right side of the plot. *Figs. 4, 7,* and *8* display the group and stand metrics of plot 7.

Figure 6—An example of a treated stand within the dry forests of southern Idaho. Note the presence of large ponderosa pine with yellow bark. The small trees and surface fuels were masticated.

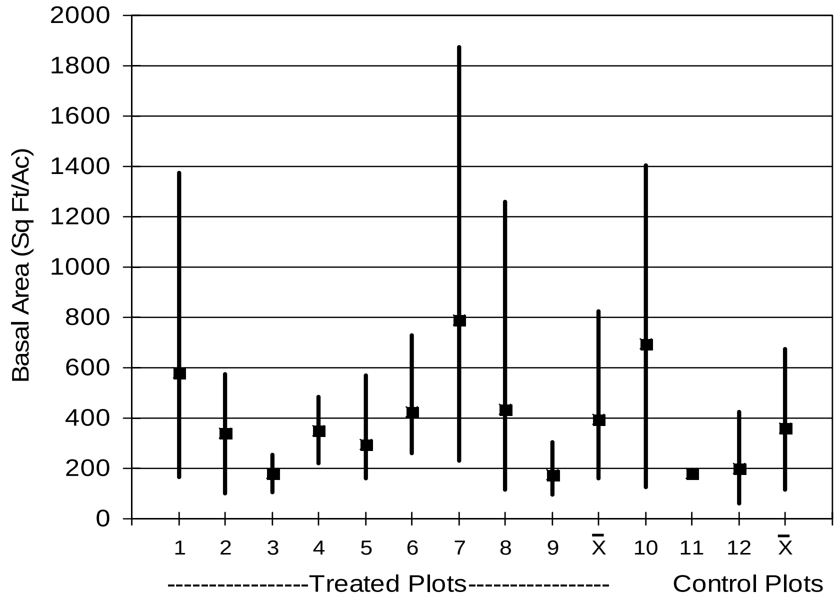


Figure 7—Stand structure within the dry forests of southern Idaho. The maximum, mean, and minimum basal area of tree groups defined as overstory trees with touching crowns occurring within treated areas (plots 1-9, 1.0 acre in size) and control areas (plots 10-12, 1.0 acre in size) and their respective means ($\bar{x}$). Note the extreme variation in basal area showing in plot 7.

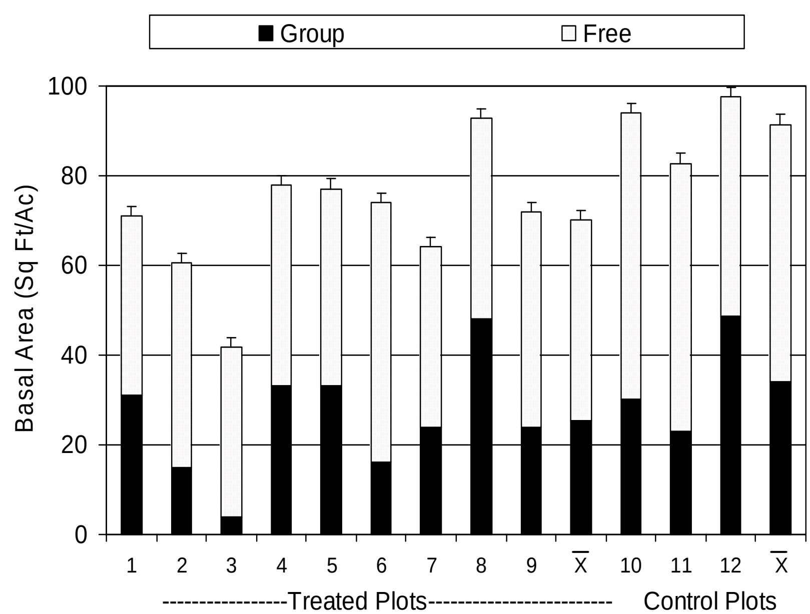


Figure 8—The stand basal area occurring within groups (group) of trees and basal area occurring within trees not associated with groups (free). Treated areas are (plots 1-9, 1.0 acre in size) and control areas are (plots 10-12, 1.0 acre in size) and their respective means ($\bar{x}$) are displayed. Note the stand basal area shown in plot 7 compared to the group basal area for plot 7 shown in *fig. 7*.

Figure 9—Fire being used to decrease the amount of organic material that developed at the base of this large ponderosa pine most likely because of fire exclusion. Fire was applied early in the spring when the temperature of the lower organic layers was below 40^{o} F (when fine root activity is minimal) and when their moisture contents exceeded 100%. Lower photo is the application of a low intensity prescribed fire treating the entire area after three spring “snow well” treatments to reduce the organic layers at the base of the trees.

we offer the free selection silvicultural system as an option for many of the emerging management objectives of the 21st Century.

There are numerous examples of which selection systems have been developed and used to create a variety of stand and forest conditions over the last century. Free selection is grounded in forest ecology and draws upon proven silvicultural practices while building on past knowledge. As early as 1524, group selection systems were used in Europe to enhance natural regeneration and protect seed trees from damaging winds (Fernow 1907). Numerous examples of selection systems used in the United States early in the 20th Century were based on tree and/or stand classifications (Dunning 1928, Meyer 1934, Pearson 1950). Moreover, the classic reversed j-shaped diameter distributions commonly associated with selection silvicultural systems were developed for timber management and were suggested to be applied at the working circle or forest scale (Davis 1966, Meyer et al. 1961). The sustainability of all silvicultural systems is predicated on how they are implemented and this assertion is no different for free selection. Haight and Monserud showed that a simple silvicultural prescription of removing commercial material and precommercial thinning was sustainable and would produce a continual flow of products (Haight and Monserud 1990a, 1990b). Even though Marquis (1978) eloquently showed how reversed j-shaped diameter distributions could be used to apply selection systems, he also showed how patch cuttings could be used to create multi-aged stands and favor the regeneration of shade-intolerant species (Marquis 1965). Building on this information, the concept of patch-selection system was introduced by Leak and Filip (1977). This hybrid selection system combined the cutting of fixed-area patches with single-tree selection system designed to regenerate both shade-intolerant and shade-tolerant species. We suggest that many of these hybrid systems have been planned, applied, and monitored qualitatively rather than using complex and strict quantification (Baker 1934, Dunning 1928, Gifford 1902, Meyer 1934, Pearson 1950).

By no means do we suggest that traditional even-aged and uneven-aged silvicultural systems not be developed and presented quantitatively in prescriptions to address many emerging forest management issues. What we are offering is an alternative to traditional even-aged and uneven-aged systems for those situations in which the quantification and/or decision making rule-sets required are so complex that they become unwieldy and/or impossible to implement. In addition, by using a comprehensive description of the short- and long-term desired conditions of a forest presented in a vision, it may be more readily communicated to disciplines outside of forestry (for example, law, social, recreation, wildlife) and to the public at large (*Appendix A*). These disciplines and the public may respond more favorably to a comprehensive and well thought out forest description than a list of technical forest descriptors (such as, crown competition factor, species preference rules, stand density index, torching index, or canopy bulk density). However, we suggest prescriptions can be quantified using vegetative classifications (*fig. 1*) and displayed geographically by using visualization systems (*Appendix A*).

Rarely have silvicultural systems and/or methods been recognized as a means for addressing objectives like sustaining the sense of place in forests (see *footnote 4*), emulating natural stand development, or for maintaining ectomycorrhizae habitat (important for the habitat of goshawk [*Accipiter gentilis* Linnaeus] prey). Free selection, and using a vision to guide it, is well suited to these objectives that are not readily quantifiable (see Franklin et al. 2002, *footnote 5*). Forest products would also be produced, albeit in uncertain quantities and at indeterminate intervals. In southern Idaho, the ponderosa pine restoration project yielded approximately 500 ft^3 per acre of commercial products and an undetermined amount of domestic firewood. In addition, projecting the system for 100 years would produce over 14,000 board feet per acre (*Appendix A*).

As we implemented the free selection system, we found it initially challenging but exciting. Nevertheless, within a couple of days, the implementation of our vision became effortless. Rather than choosing trees for removal in the treatments, we concentrated on the forest components that were to be left (for example, soil, trees, shrubs, disease), and projected how they would respond in both the short- and long-term (*Appendix A*). Moreover, the process of implementing the treatments necessitated continued discussion among the people doing the marking. That helped to channel their collective silvicultural knowledge into an integrated vision when making on-the-ground decisions. The vision of naturally occurring clumps and groups of vegetation in the stands served as a reference point for decisions on where to remove trees and in what numbers (*figs. 2-7, Appendix A*). A shared concept of maintaining a functioning forest guided the treatments even while we made the stands more resilient and resistant to crown fire. We did this by decreasing the overall stand density, decreasing surface fuels, and raising crown base heights. We created openings for regeneration (for example, tree, shrub, grass), thereby meeting a prerequisite for long-term success of the selection system (*figs. 5, 6, 9, Appendix A*). However, we recognize that subsequent treatments (for example, canopy removal, prescribed fire, cleanings, thinning, site preparation, planting) must occur to further promote the development of the desired forest structures and compositions as disclosed in the vision. These follow-up treatments are critical to the success of any selection system and many will provide commercial products (*Appendix A*).

The demands on forests by society are ever changing, as are the forest management objectives that guide our stewardship of the forests in our care. This is exemplified by the passing of the Healthy Forest Restoration Act of 2003 which includes provisions to reduce hazardous fuels and restore healthy forest conditions on lands of all ownerships (USDA 2004). However, it will take 10s to 100s of years before management will create forest conditions that fulfill these goals (vision). The free selection system we propose, and the kind of vision statements that we suggest for guiding its implementation, will serve as additional tools for future forest management. Its successful application requires a strong appreciation of the art and science of silviculture (*Appendix A*). Smith (1972) predicted “Silviculture fitted to demonstratable realities of nature and human need will call forth the evolution of methods or treatments more varied than our wildest present imagination can encompass.” Our concept of free selection may help to bring that prophecy to reality.

Acknowledgements

We would like to thank Jim Guldin for his thoughtful discussions and interactions on the concepts and realities of selection systems. The help of Bob Denner, Dan Frigard, Jim Hedgecock, Gail Fullerton, and Don Shipton was invaluable in applying the treatments, collecting the data, and providing continual discussions on the concepts and application of free selection.

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Appendix A: Free Selection Illustrated Using the Forest Vegetation Simulator and the Stand Visualization System

Approximately 100 acres within the Boise Basin Experimental Forest located in southern Idaho were treated using free selection. The vision informing the immediate and future stand treatments was to maintain and sustain the old growth character of a mature to old (150 to 400+ year) ponderosa pine stand. This vision included all forest elements including snags, down logs, vertical and horizontal heterogeneity, tree group dynamics, shrub, grass, and forb conditions, forest floor characteristics (duff and needle layers), and the sense of place that is inherent to big, old, and yellow-barked ponderosa pines. This vision also wanted to create forest conditions that reduced the risk of uncharacteristically severe wildfires.

The mature ponderosa pine stand grew at an elevation of 4800 feet on a northerly aspect with a slope of 35 percent, representing a Douglas-fir/ninebark (*Physocarpus malvaceus* (Greene) Kuntze) habitat type. Large ponderosa pine dominated the site; however, numerous small Douglas-fir and ponderosa pine occupied the understory as a result of fire exclusion. The initial entry of the free selection system consisted of removing the majority of the small trees ≤ 12 inches diameter breast height (dbh) that currently did not or were not likely to develop into essential elements fulfilling the vision in the future. In addition the majority of the seedlings and samplings were removed as commercial fire wood and through the use of prescribed fire.

We used nine, circular one-acre plots randomly located to describe the stand after the treatments were complete (*figs. 4, 5*). Diameter, height, crown ratio, and location of each tree (≥ 8 inches) were recorded. However, for illustrating the free selection silvicultural system for 100 years, we chose plot 7 (*figs. 5 and A1*). The central Idaho variant of the Forest Vegetation Simulator (FVS) was locally adjusted using habitat type, slope, aspect, and elevation of the stand (Stage 1973, Dixon 2002). Because we had the location of each tree, the Stand Visualization System (SVS) attached to FVS reflected the actual horizontal and vertical distributions of the trees (USDA Forest Service no date).

FVS is capable of projecting stands (plots) through different time horizons using a variety of intervals. Our concept of free selection indicates that the interval between entries is predicated on how the stand develops to fulfill the vision. To simplify our illustration, we chose to use only 10-year intervals in FVS; however, multiple simulations with different time intervals could have been used. To project growth and mortality of plot 7 (*figs. 4, 5, 7, 8*), a 100-year simulation was used and the TIMEINT keyword set the number of cycles at 10 and the length of each cycle to 10 years. The Regeneration Establishment Model was used to estimate regeneration for each cycle throughout the projection. One hundred percent of the plot was burned every 10 years and the BURNPREP keyword was used to simulate this treatment.

With the initial inventory (2005) of plot 7, we could identify each tree used in the simulation and give it a unique code (integer 2-99) in the Prsc.code (IPRSC) field. We used integers 2-6 to identify trees that we would subsequently remove in a specific simulation cycle using the THINPRSC keyword. The THINDBH key word was used to target trees within specific dbh ranges for removal. All decisions of removing trees that did not fulfill the immediate or future forest elements of the vision were chosen using SVS. We used the “Marking and Treatment” window in

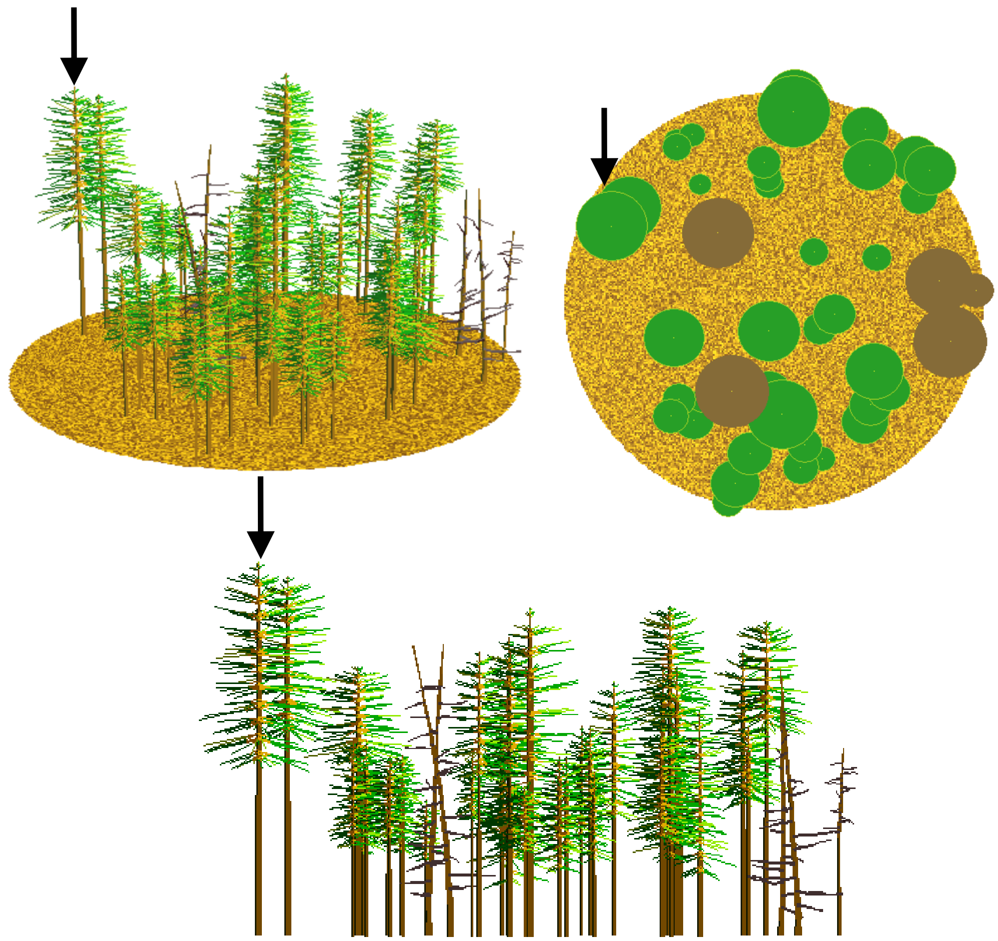


Figure A1—Ponderosa pine stand located on the Boise Basin Experimental Forest in southern Idaho after the initial entry (2005) using free selection. Tree locations in the visualization reflect the actual tree locations occurring on the one-acre plot. Note the snags, groups of trees and the diverse horizontal and vertical structure. This visualization reflects conditions that fulfill the goals of the vision. Trees marked with the arrows lived the entire simulation (see *fig. A11*).

SVS in conjunction with the overhead view of the plot to move a paint gun pointer across the view, displaying the characteristics of each tree (*fig. A2*). This process determined the fate of each tree and established the parameters associated with the THINPRSC and THINDBH keywords. Because Douglas-fir was not a preferred forest element in the vision, all Douglas-firs were removed each cycle. In practice this would occur through prescribed fire and/or precommercial thinning. The Fire and Fuels Extension (FFE) was used to simulate these fires (keyword SIMFIRE). However, we turned off the FFE tree mortality and preferred to more precisely control mortality through our management actions.

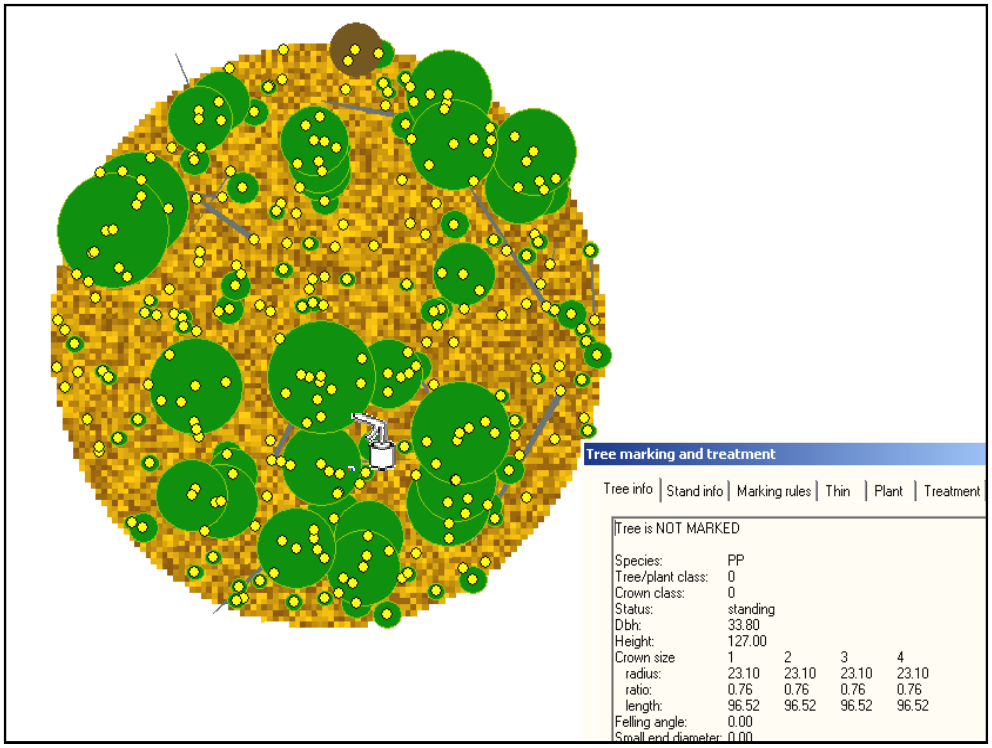


Figure A2—An example of the ponderosa pine stand projected using the Forest Vegetation Simulator and displayed using the Stand Visualizations System. The window in the lower right shows the characteristics of each tree as a paint gun pointer moves across the plot. Using this information, individual trees or groups of trees were chosen to leave or remove every 10 years ensuring that the stand fulfilled the vision of sustaining old-growth character.

Listed below is a summary of our management activities for plot 7 for the 100 year projection. Note that they are not consistent, nor do they reflect how the stand developed. Moreover, this is only one of many series of treatments that could be used in a free selection silvicultural system to fulfill the vision of maintaining the old forest character of this ponderosa pine stand. As Baker (1934) suggested, there are a multitude of forest treatments that can be assembled into silvicultural systems to meet management objectives.

2015: All Douglas-fir seedlings were removed.
All ponderosa pines between 0.0 and 1.0 inches dbh were removed.

2025: All Douglas-fir seedlings were removed.
All ponderosa pines between 0.0 and 2.5 inches dbh were removed.

2035: All Douglas-fir seedlings were removed.
Individual ponderosa pines indicated by a “2” in the IPRSC field were removed using the THINPRSC keyword.

2045: All Douglas-fir seedlings were removed.
All ponderosa pines between 5.3 and 5.8 inches dbh were removed.

2055: All Douglas-fir seedlings were removed.
All ponderosa pines between 2.0 and 6.0 inches dbh were removed.
Individual ponderosa pines were removed using the THINPRSC keyword.

2065: All Douglas-fir seedlings were removed.
All ponderosa pines between 9.4 and 18.0 inches dbh were removed.
All ponderosa pines between 0.0 and 4.0 inches dbh were removed.

2075: All Douglas-fir seedlings were removed.

2085: All Douglas-fir seedlings were removed.
All ponderosa pines between 0.0 and 2.5 inches dbh were removed.
All ponderosa pines between 11.5 and 12.5 inches dbh were removed.
All ponderosa pines between 20.0 and 22.0 inches dbh were removed.
Individual ponderosa pines were removed using the THINPRSC keyword.

2095: All Douglas fir seedlings were removed.
All ponderosa pines between 0.0 and 3.5 inches dbh were removed.
All ponderosa pines between 26.0 and 26.5 inches dbh were removed.

At the beginning of the simulation (2005), the diameters of the old-growth ponderosa pine stand ranged from 8 inches to over 32 inches with considerable vertical and horizontal diversity meeting the goals set forth in the vision. (*figs. A1, A3*). Through the 100-year simulation, precommercial thinning and prescribed fire were intended to remove all Douglas-fir regeneration and a portion of the ponderosa pine regeneration in most cycles (*fig. A4*). In the simulation, tree numbers less than 6 inches dbh peaked in 2055. In 2065, we removed all regeneration less than 4.0 inches dbh. These data show how diverse the treatments meeting the free selection vision can be, but also show the constant need for treating ground level vegetation in order to keep in check the risk of severe wildfire.

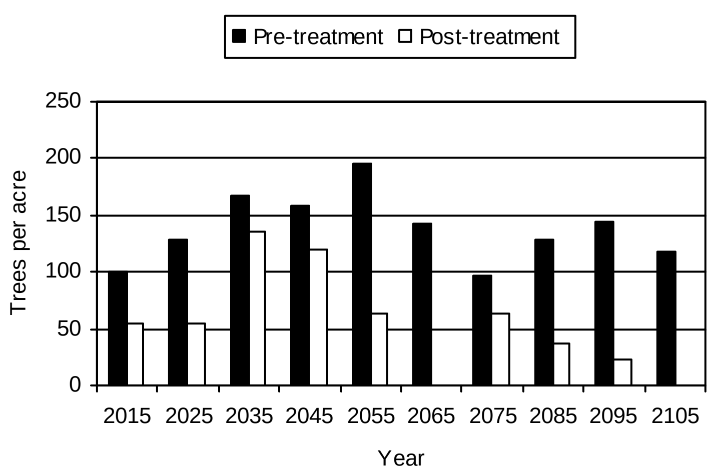


Figure A3—Distribution of tree diameters after the first (2005) free selection entry in a ponderosa pine stand located in southern Idaho.

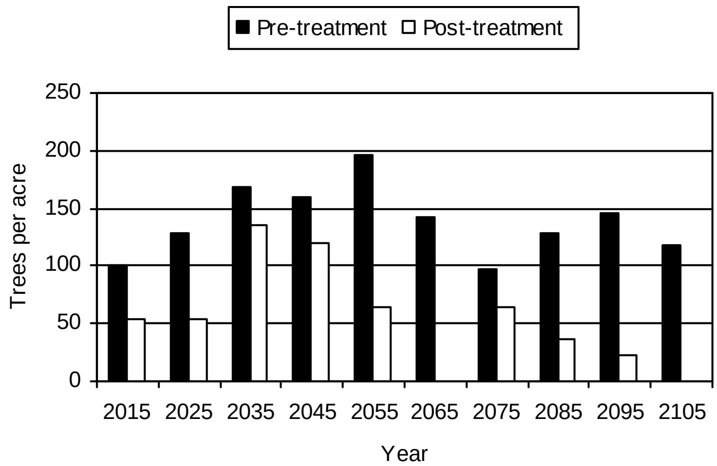


Figure A4—Amount of Douglas-fir and ponderosa pine regeneration (≤ 6 inches) estimated using the Forest Vegetation Simulator. The regeneration was treated using prescribed fire and precommercial thinning. In 2065 we removed all regeneration less than 4.0 inches dbh.

When locating trees in the field, a group was identified when tree crowns were touching and/or overlapping. A similar procedure was used in the simulation. To determine the distribution of tree groups for each cycle during the projection, a 20 point grid was randomly placed on the overhead view of the stand and the presence of a tree group, snag, or down log was determined. At the beginning of the simulation in 2005, approximately 20 ft^2 per acre of basal area occurred in groups (*fig. 8* plot 7, *A5*). The distribution of groups of trees and trees not associated with groups was dynamic and explained by different factors. Decreases in basal area associated with tree groups may occur when large trees in a group fall or become snags. For example, at the beginning of the 2015 cycle, a 28-inch (dbh) tree associated with a tree group

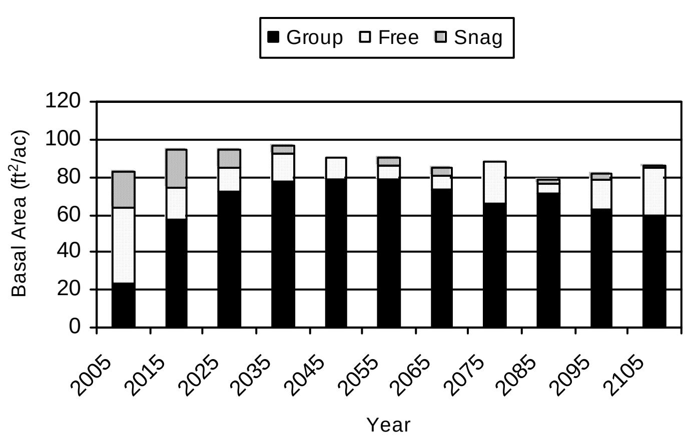


Figure A5—Stand basal area apportioned among group trees, free trees, and snags for the initial free selection treatment (2005) and subsequent treatments projected through 2105 using the Forest Vegetation Simulator. Group trees are defined as trees with touching or overlapping crowns and free trees are defined as those not associated with groups. See appendix *figs. 1, 2, 9, 10* for illustrations of the overhead view of the stand showing the groups and free trees.

fell. Decreases in basal area associated with tree groups also occurred during the middle of the projection, when some co-dominant trees were removed. Increases in basal area associated with groups occurred when group trees increased in size, or when free trees become associated with a group when their crowns touched or overlapped with crowns of nearby trees. Also during the simulation we were able to regenerate ponderosa pines and have them develop into tree groups containing trees exceeding 16 inches (dbh) in diameter and over 100 feet tall. Two of the groups at the end of the projection consisted entirely of trees established after 2005. As a result of these dynamics inherent to the application of free selection, we were able to increase the amount of stand basal area occurring in groups to the vicinity of 80 ft^2 in 2045. By the end of the simulation, we had approximately 60 ft^2 of basal area per acre occurring in groups of trees with a total stand basal area of 85 ft^2 per acre (*fig. A5*).

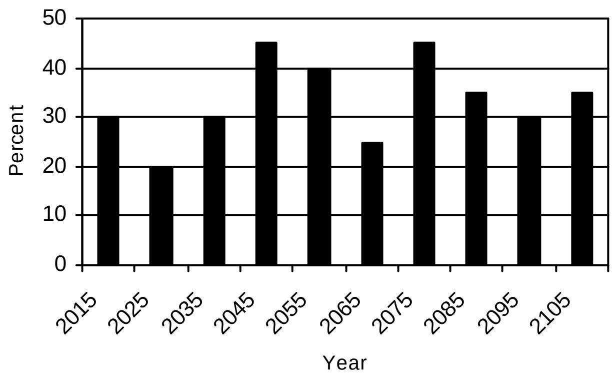


Figure A6—Proportion of the stand occupied by groups of trees determined by locating 20 random points on the overhead views as projected by the Forest Vegetation Simulator through 2105 and displayed by the Stand Visualization System. See appendix *figs. 1, 2, 9, 10* for overhead stand views.

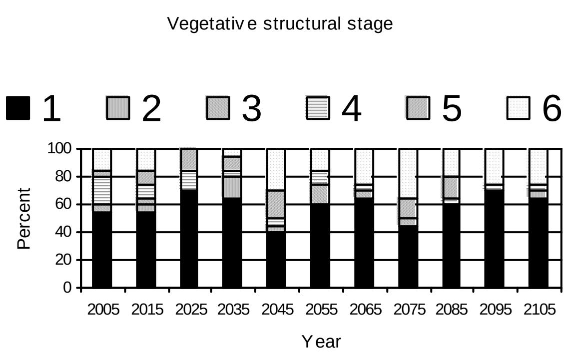


Figure A7—The estimated proportion of the stand occupied by different vegetative structural stages (VSS) determined by 20 points randomly located on the overhead views. VSS 1=grass, forb, seedling, 2=sapling, 3=young, 4=mid-aged, 5=mature, 6=old (Reynolds et al. 1992).

Figure A8—Distribution of tree diameters after 100 years (2105) of treatment using free selection as projected with the Forest Vegetation Simulator.

In fulfilling the vision, the presence and distribution of tree groups were important considerations. Through the simulation, we were able to maintain a large proportion of the aerial extent of the stand in groups of trees. The proportion of the stand occupied by tree groups peaked in 2045 and 2075 at 45 percent and, by the end of the simulation in 2105, 35 percent of the stand was occupied by tree groups (*figs. A1 A6, A9, A11*). The vegetative structural stage (VSS) that each tree group represents was also quite variable.

The per acre values and averages are not as telling of the success of the free selection system as are the views available from the FVS projections, using SVS. After treatment, mean stand diameter ranged from 10.0 to 20.7 inches during the 100-year simulation and the stand density index remained below 170 during the simulation (*table A1*). Through the 100-year simulation, 14,347 board feet per acre were removed and fewer than 150 trees per acre were removed during each of the precommercial thinnings and prescribed fires every 10 years (*table A1*). In most years, at least one snag per acre existed and over 10 down logs per acre existed. What this stand summary does not show is the high amount of horizontal and vertical diversity that occurred in the stand and the maintenance of the old-growth character that was paramount for fulfilling the essence of the vision.

Free selection, as we describe it, follows the development of silvicultural systems used in forestry for over 100 years. It is adaptive management at the stand and landscape level and is preferably guided by a vision rather than rigid marking guides trying to describe the highly diverse stand structures of which free selection is most suited for developing and sustaining. By using FVS, and by displaying the results of the simulation with SVS, we have shown how free selection can be applied in maintaining the old-growth character of a ponderosa pine stand located in southern Idaho. We suggest that free selection, directed by a well conceived and articulated vision, can be used to address many forest management objectives in which high forest cover is required, and when what is left after treatment is of paramount importance. This example of free selection shows that such an approach is sustainable and that the Forest Vegetation Simulator and the Stand Visualization System are excellent tools for displaying such a silvicultural system.

Table A1—*Summary of the stand characteristics for the 100 year simulation using free selection in a stand of ponderosa pine located on the Boise Basin Experimental Forest in southern Idaho. The central Idaho variant of the Forest Vegetation Simulator was used and adjusted for the Douglas-fir habitat type, 4,800 feet elevation, and a northern aspect.*

	Pre-treatment						Removals		Post treatment		
	Dead		Live				Live		Live		
Year	Sg[1]	Lg[2]	Tr[3]	BA[4]	CF[5]	BF[6]	BF[6]	Tr[3]	BA[4]	SDI[7]	QMD[8]
	----------Per acre----------										inch
2005	5	0	41	64	2130	11601	0	0	64	95	16.9
2015	6	1	140	74	2590	14527	0	46	74	126	12.0
2025	3	6	165	85	3092	17697	0	72	85	141	13.0
2035	2	13	204	97	3606	20994	866	36	92	168	10.0
2045	0	16	205	105	3929	23050	3408	43	90	164	10.1
2055	1	19	250	104	3868	22464	2727	134	86	148	11.7
2065	1	15	191	98	3884	23211	2200	157	81	111	20.7
2075	0	14	128	88	3827	23789	0	30	88	147	12.8
2085	2	14	161	98	4248	26531	4443	106	76	116	15.8
2095	3	14	164	82	3783	24493	703	122	79	113	18.5
2105	1	18	135	85	3984	23083	0	0	85	151	10.7

[1]Snags, [2]down logs, [3]trees, [4]basal area, [5]stand volume in cubic feet, [6]stand volume in board feet, [7]stand density index, [8]quadratic mean diameter.

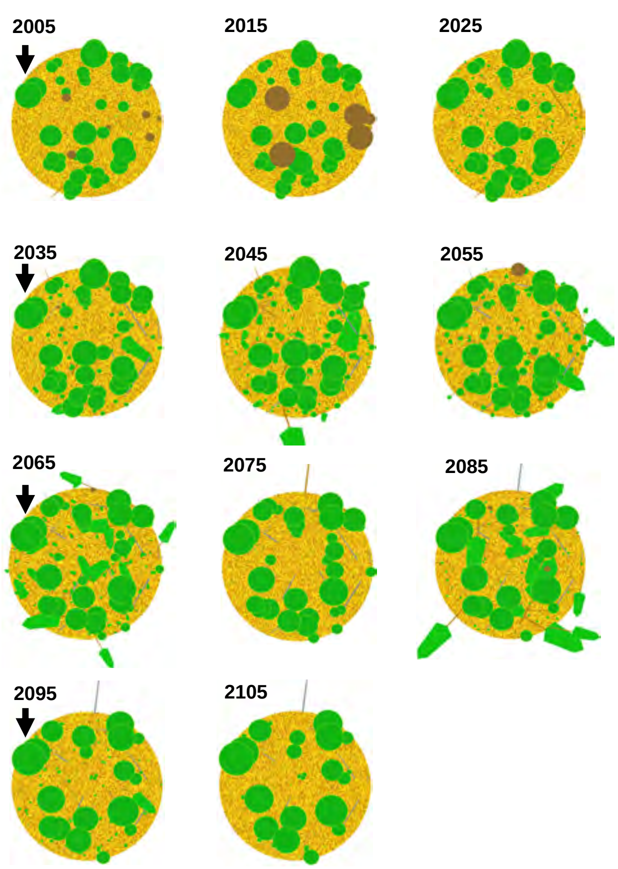
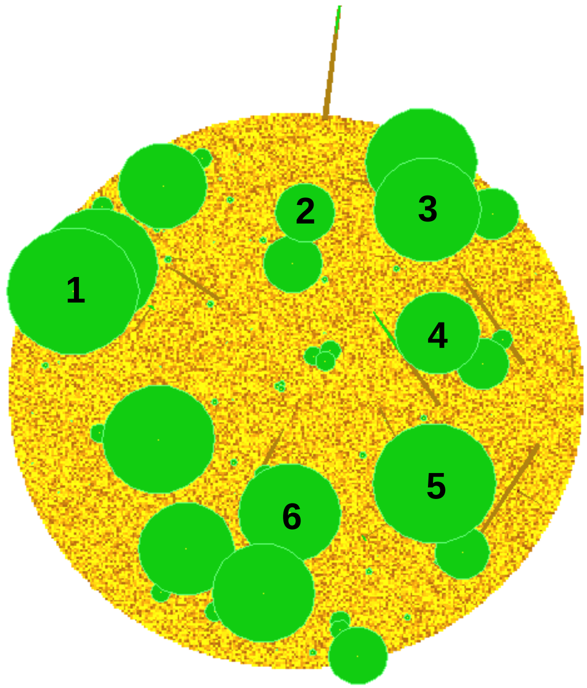


Figure A9 — Overhead views for each year of the simulation after a treatment (for example, prescribed fire, precommercial thinning, commercial harvest). Note the crown expansion of the trees (marked by the arrows) in the upper left of each view.



Group	Number of trees	Size in acres	Basal Area in ft^2/ acre
1	2	0.064	251
2	2	0.020	149
3	3	0.050	260
4	2	0.026	172
5	2	0.047	190
6	3	0.082	179

Figure A10—Overhead view of a ponderosa pine stand after 100 years of applying free selection. As a result 6 groups of trees were created and/or maintained.

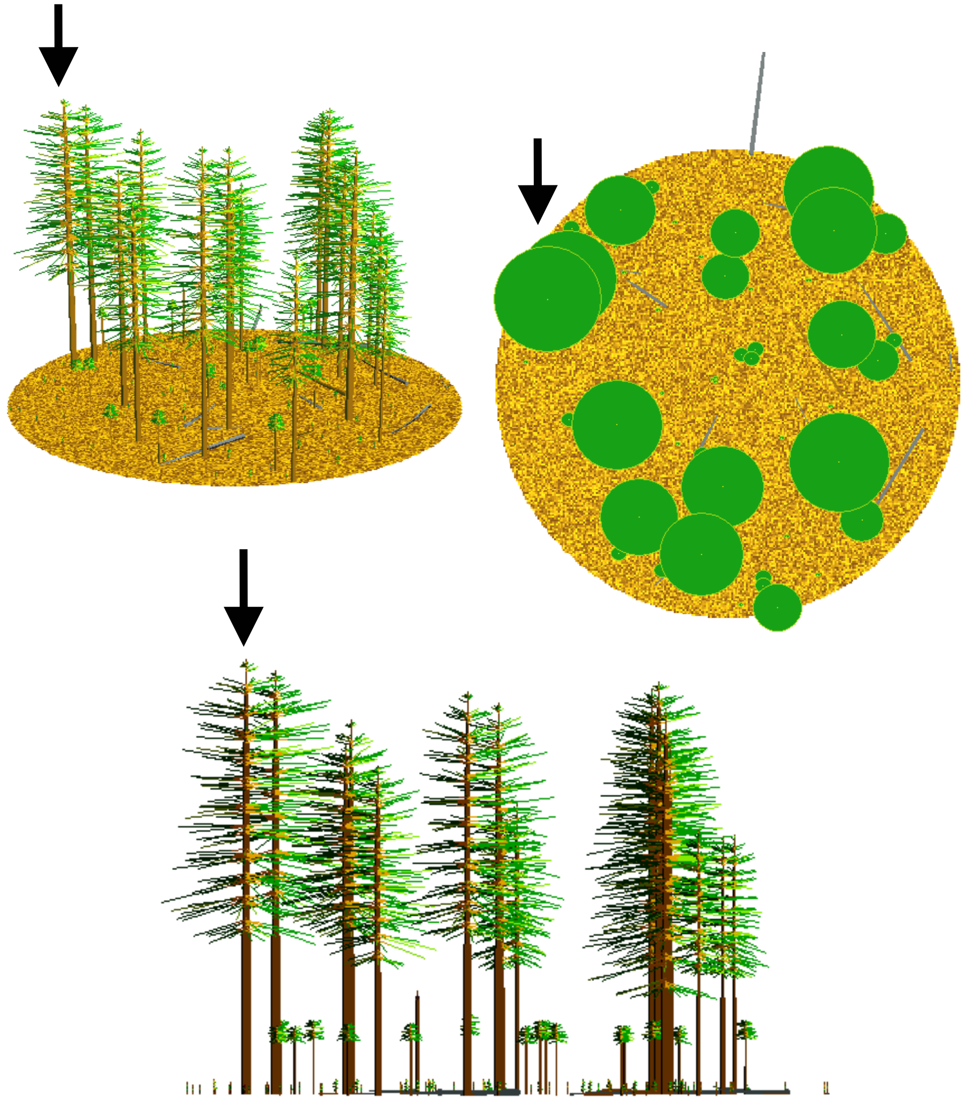


Figure A11—Ponderosa pine stand located on the Boise Basin Experimental Forest in southern Idaho after 100 (2005-2105) years of applying free selection as projected using the Forest Vegetation Simulator and displayed by the Stand Visualization System. Note the down logs and the horizontal and vertical diversity. At least 5 cohorts of trees are noticeable. Trees marked with the arrows lived the entire simulation (see *fig. A1*).

Landscape Silviculture for Late-Successional Reserve Management[1]

S. Hummel and R.J. Barbour[2]

Abstract

The effects of different combinations of multiple, variable-intensity silvicultural treatments on fire and habitat management objectives were evaluated for a ±6,000 ha forest reserve using simulation models and optimization techniques. Our methods help identify areas within the reserve where opportunities exist to minimize conflict between the dual landscape objectives. Results suggest that most of the trees removed by silvicultural treatments designed to support fire and habitat objectives, while generating enough revenue to break-even, would be medium-sized (17-40 cm), shade-tolerant conifers. The study produced information that was used by a planning team on the Gifford Pinchot National Forest to develop stand-level treatments based on mid-scale landscape patterns. New contracting authorities give the Forest Service ways to offer sales that support landscape management objectives in the reserve, but the contracts are time-consuming to prepare and award. Implementation of a stewardship contract associated with the study reserve is scheduled to begin in summer 2006.

Introduction

The Northwest Forest Plan (Plan) designated late-successional forest reserves on some federal lands in Oregon, Washington, and California. One goal of the reserve network is to sustain habitat for the northern spotted owl (*Strix occidentalis caurina*) and other species associated with older, late seral forests (USDA and USDI 1994). Plan guidelines require land managers to protect these reserves, or LSR, from large-scale natural and human disturbances. An ongoing challenge for managers of LSR in drier Plan provinces is to conserve and develop older forests in ways that support their resilience in fire-adapted ecosystems. In such places, LSR managers are concerned about effects to late seral forest habitat structures associated both with severe wildfires and with silvicultural treatments done to reduce fire severity. The problem of potentially conflicting effects from forest management is not confined to LSR, however. Throughout fire-adapted forest ecosystems of the western US, federal land managers seek ways to promote forest structures and processes that are consistent with pre-fire exclusion conditions, while preserving some of the attributes of existing conditions that people have come to value.

Our interest lies in developing methods to quantify tradeoffs among various forest management objectives. A need for analytical methods of this type occurs everywhere that multiple resource objectives exist but is acute in the West, because of extensive areas of public land combined with an increasing human population.

[1] A version of this paper was presented at the National Silviculture Workshop, June 6-10, 2005, Tahoe City, California.

[2] Research Forester and Program Manager, respectively, Pacific Northwest Research Station, USDA Forest Service, P.O. Box 3890, Portland, OR 97208.

Many people are interested in land management issues. At times, their preferences and the best ways to provide them appear to be in conflict.

In this paper, our objective is to describe a method for identifying silvicultural solutions to potentially conflicting landscape management objectives. We summarize results from a study in which we investigated how treatments to moderate fire behavior could impact late seral forest structure in one LSR, if treatment expenses might be offset by revenue generated from harvest activities, and the dimensions of the trees removed. We use the term "landscape silviculture" for treatments applied to a stand but evaluated collectively according to objectives for an entire reserve.

Site Description

The Gotchen LSR lies on the eastern flank of the Cascade Range in Washington State, covering about 6,070 ha of the Mount Adams Ranger District on the Gifford Pinchot National Forest. Like many other reserves in the drier provinces of the Plan area, the Gotchen LSR includes a mix of older, mixed-conifer forests and plantations. Tree species include Douglas-fir (*Pseudotsuga menziesii*), grand fir (*Abies grandis*), subalpine fir (*Abies lasiocarpa*), ponderosa pine (*Pinus ponderosa*), western larch (*Larix occidentalis*), and lodgepole pine (*Pinus contorta*). Six documented spotted owl nest sites exist (Mendez-Treneman 2002). Defoliation of true firs (*Abies*) associated with an outbreak of western spruce budworm is contributing to increasing fuel loads and to declining crown cover, which affects owl habitat quality (Hummel and Agee 2003). Managers responsible for the Gotchen LSR seek ways to moderate ongoing risks to owl habitat associated with potential stand-replacement severity fire, while retaining older forest structures within the reserve landscape (Hummel and Holmson 2003).

Methods and Analysis

Characterize Landscape Conditions

At the outset of the Gotchen LSR study, we considered it important to use a simulation model that could recognize the contribution of individual trees to forest structure at both within-stand and among-stand (landscape) scales. We needed a model that could track residual stand structure and forest dynamics following a silvicultural treatment, and account for any trees cut during the treatment both by size and species. In addition, because wildfire can affect multiple stands, we wanted the model to have spatial database capabilities, so that the influence of conditions in neighboring stands on fire behavior and effects within a stand (and vice-versa) could be simulated. We ultimately selected the Forest Vegetation Simulator East Cascades variant (FVS) (Stage 1973, Johnson 1990, Crookston and Havis 2002).

We began by using aerial resource photos to identify vegetation patches in the Gotchen LSR, and then spatially described them in a geographic information system (GIS) database (*fig. 1*). The patches were stratified into a summary matrix of stand types based on structure class and potential vegetation type (details in Hummel et al. 2001). By selecting patches within stand types using probability proportional to size, field samples made in 2000 and 2001 covered the range of existing conditions. The data were used to create "tree lists" for sampled patches following FVS procedures (Dixon 2003). We randomly assigned a FVS tree list to any unsampled patches within the same stand type, our assumption being that within-stratum variation in forest structure is lower than among-strata variation.

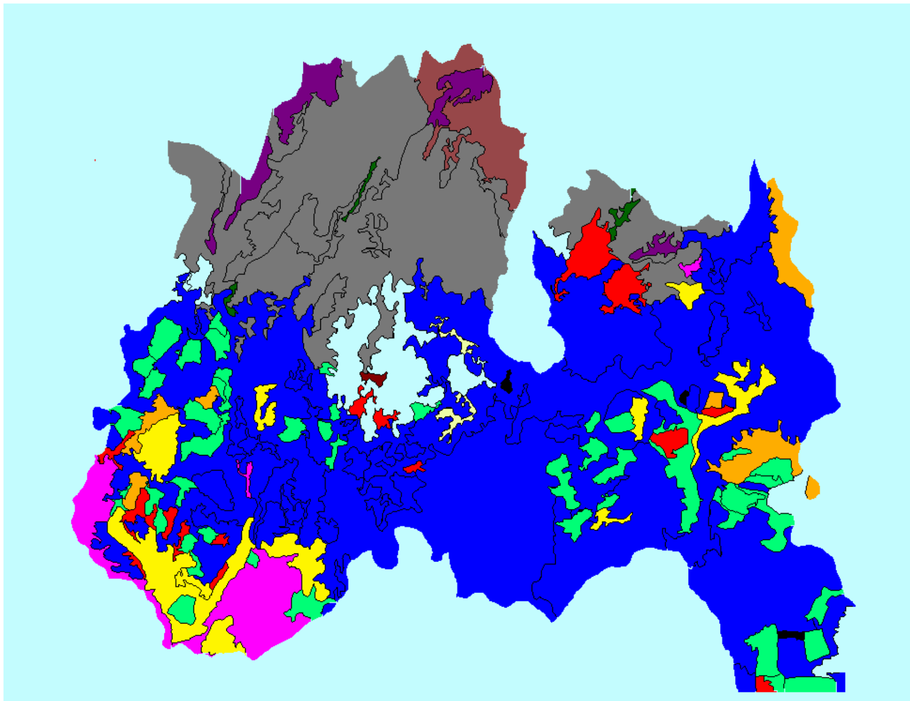


Figure 1–Landscape patches in the Gotchen LSR identified by using aerial photographs. Each colored patch is a different combination of forest structural stage and vegetation cover type (Source: Hummel et al. 2001).

We considered it vital that vegetation patterns be able to change with time in our analysis and not be constrained by existing landscape geometry. Some of the large patches in the southern part of the Gotchen LSR, for example, result from previous logging and fire suppression activities, and tend to be bigger than regional studies of disturbance ecology would suggest for areas like this one with mixed-severity fire regimes. We therefore introduced the ability for new patterns to emerge by sub-stratifying the original patches into smaller "projection units." These units represent the smallest area to which a treatment could be applied. Each unit received the FVS tree list associated with its original patch, but individual unit growth trajectories could differ based on stochastic variation within the model, and on the treatment schedules ultimately selected for each one (details in Hummel et al. 2002, Calkin et al. 2005).

Use Forest Structure to Describe Fire and Habitat Objectives

Once we had a database representing existing forest vegetation, we turned our attention to how its structural dynamics related to fire and to owl habitat. We focus on structure, or the arrangement and variety of living and dead forest vegetation, because it can be measured and it is physically and biologically relevant to fire behavior and to owl habitat. We selected a 30-year analysis period by considering both fire return intervals for mixed-conifer forests in the region and model capabilities. For fire threat (FT), we used three variables: flame length, crown fire initiation, and crown fire spread, which we estimated for each unit using the Fire and Fuels Extension to FVS (FVS-FFE) (Reinhardt and Crookston 2003). A unit's FT index (low, moderate, or high) was a weighted combination of these variables within a unit and its adjacent units (details in Calkin et al. 2005):

$$Threat10i=FLi+w1*(Torchi+Crowni)+(w2*\sum j[Edgej*\{Torchj+Crownj\}] / \sum jEdgej)$$

Where,

i is the reference unit for which fire threat is being calculated,

j references adjacent units to unit i,

FL_i = 1 if flame length for reference unit i < .92m,

= 2 if flame length is between .92 and 1.22m,

= 3 if flame length is between 1.23 and 1.51m,

= 4 if flame length is between 1.52 and 1.82m,

= 5 if flame length is ≥ 1.83m,

Torch = 1 if torching potential wind speed < 95% local wind speed,

= 0 else,

Crown = 1 if crowning potential wind speed < 95% local wind speed and Torch = 1,

= 0 else,

$Edge_j$ is the amount of perimeter accounted for by adjacent unit j, and

w_1 and w_2 are relative weighting variables.

The fire threat index is on a continuous scale ranging from 1 to 10. However, Calkin et al. (2005) collapsed it into a three point scale to make it easier to express results consistent with area in reduced threat categories after treatment:

Threat

= 1 if Threat10*i* < 3 (low threat, control likely, fair survival of residual trees),

= 2 if Threat10*i* = 3-5.99 (moderate threat, control problematical, some residuals survive),

= 3 if Threat10*i* ≥ 6, (high threat, control unlikely, high mortality likely).

A landscape FT was computed from the proportion of the reserve in each of the low, moderate, and high FT categories for each decade (details in Calkin et al. 2005).

For late seral forest (LSF) structure, we developed a definition using the basal area of trees in specific diameter classes that incorporated eastside owl habitat requirements (Hummel and Calkin 2005) (*table 1*). By using FVS, the Western Root Disease Model (Frankel 1998), and FFE-FVS, we evaluated LSF structure and assigned a FT index to each unit.

Model Forest Structure with and without Treatment

We first projected trends in LSF structure and in high FT (*fig. 2*) in the Gotchen LSR over three decades by simulating all units without any treatment (NoRx).

For comparison with the NoRx baseline, we also applied multiple, variable-intensity silvicultural treatments to each unit and simulated forest development over

Table 1--*Structural definition of late successional forest (LSF).*

Basal area (BA) at least 55.2 m^2/ha
BA of trees greater than 61.0 cm dbh $\geq$ 8.3 m^2/ha
BA of trees greater than 35.6 cm dbh $\geq$ 33.1 m^2/ha
BA of trees less than 35.6 cm dbh $\geq$ 8.3 m^2/ha

Source: Hummel and Calkin 2005.

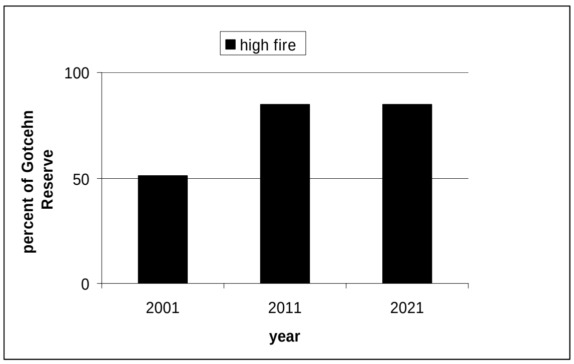


Figure 2–Percent of Gotchen Late-Successional Reserve (LSR) projected to be in high fire threat (high fire) in each of three decades without any treatment (NoRx). (Source: Hummel and Calkin 2005)

the same analysis period. Vegetation structure in each unit in each decade associated with each treatment was evaluated according to our FT and LSF structure definitions. The treatments differed in the type of thinning, species removed, maximum diameter limit, residual basal area target, and residual fuel loads) (*table 2*) (details in Hummel and Calkin 2005). All FVS simulation results were saved and linked to the GIS database. The list of trees cut by FVS following any active treatment was saved in a format compatible with the Financial Evaluation of Ecosystem Management Activities (FEEMA) model (Fight and Chmelik 1998), with which we also kept track of treatment costs, i.e., reforestation and fuels treatments.

We entered harvest costs, fuel treatment costs, and wood product prices into the FEEMA model to calculate net revenue per unit per treatment per decade. Harvest costs, including hauling, road maintenance, contractual requirements, reforestation, slashing, and piling and burning were obtained from reserve managers. We estimated defect by log size class by using previous timber sale records for the area and recommendations from Pacific Northwest (PNW) personnel responsible for scaling and cruising. Product prices represent a stable market in the PNW. We assumed that all harvest and operational costs remain constant over time, and discounted future costs and revenues by four percent.

We used results from the NoRx baseline simulation to identify the total area of LSF structure to be maintained over 30 years, subject to reducing FT in the reserve landscape. This objective was then written in the form of an algorithm. By using a simulation model, we could apply each treatment to each unit subject to a rule set and then let the algorithm select the one that, together with conditions in neighboring units, best met the dual landscape objectives over time (details in Calkin et al. 2005). Because we kept track of treatment costs and revenues on a per unit basis, we could evaluate the net revenues (negative, break-even, positive) earned collectively by any set of treatments. This feature became invaluable when we specified financial requirements to be met by landscape silviculture treatments.

Table 2--*Silvicultural treatments.*

Treatment (Rx)	Rx Objective	Silvicultural treatment applied in FVS	Citation for Rx targets
No Rx	Minimize human disturbance	No activity scheduled	
Reduce Rx	Reduce crown fire potential in historically low fire-severity forest ecosystems	40% canopy cover Thin from below to 50.8 cm dbh Preferentially remove ABGR Pile and burn Plant PSME, PIPO, LAOC	Agee 1996 Agee et al. 2000 Graham et al. 1999
Restore Rx	Reduce density of shade-tolerant true fir trees (Abies) that have established since the 1920s	Thin from below to 38.1 cm dbh Keep at least 23 m2/ha basal area Retain PSME, PIPO, LAOC, PIEN Pile and burn	Hummel et al. 2002
Protect Rx	Protect large trees (>53.3 cm dbh) and retain sufficient basal area to meet LSF definition	If more than 55.2 m2/ha in unit then thin trees 0-35.6cm dbh to 8.3 m2/ha Pile and burn	Johnson and O'Neil 2001 Mendez-Treneman 2002
Diameter Rx	Reduce density of shade-tolerant understory true fir trees (Abies)	Thin from below to 25.4 cm dbh Keep at least 23 m2/ha basal area Retain PSME, PIPO, LAOC, PIEN Pile and burn	
Accel Rx	Accelerate the development of LSF structure	Thin to 247 trees/ha Keep PSME, PIPO, LAOC Plant 247 PSME/ha	

ABGR = grand fir (*Abies grandis*)
PSME = Douglas-fir (*Pseudotsuga menziesii*)
PICO = lodgepole pine (*Pinus contorta*)
LAOC = western larch (*Larix occidentalis*)
PIPO = ponderosa pine (*Pinus ponderosa*)
PIEN=Englemann spruce (*Picea englemannii*)
Source: Hummel and Calkin 2005.

Develop Production Curves by Varying Silvicultural Treatments and Area Treated

Using the GIS data layers linked with the FVS treatment results and costs, Calkin et al. (2005) constructed a set of production possibility (PP) curves for the Gotchen LSR by using a simulated annealing (SA) algorithm. The SA algorithm was written to maximize reducing landscape FT subject to constraints on the total area of LSF structure maintained and on the amount of area that could be treated in any decade. Each point on the PP curves represents the results of different combinations of silvicultural treatments in terms of FT reduction and LSF structure, while each curve represents the relative tradeoffs between FT reduction and LSF structure subject to a given area constraint. The LSF constraint was varied, from allowing any unit that qualified as LSF to be treated (unconstrained), to allowing no unit that qualified as LSF to be treated (strict). Intermediate constraints included 6,678, 6,780, and 6,880 ha of LSF structure maintained over the 30-year analysis period. The effectiveness of treatments was assessed by identifying if existing FT levels were reduced for treated units and their overall effect on landscape FT. The silvicultural treatment scheduling problem is defined in Calkin et al. 2005:

Maximize $\Sigma_t \Sigma_i$ (Threat$_{i,t}$ (no treatment) – Threat$_{i,t}$ (j, adj$_i$(j)))* Area$_i$
- B_1* LSF Penalty – B_2*Total Area Penalty

If $\Sigma_t \Sigma_i$ Area$_i$ * LSF $_{i,t}$ < X

LSF Penalty = X-$\Sigma_t \Sigma_i$ Area$_i$ * LSF $_{i,t}$

Else LSF Penalty = 0

If Σ_i Area$_i$ * Period$_{i,t}$ >Y for t = 1,2,3

Area Penalty$_t$ = Σ_i Area$_i$ * Period$_{i,t}$ – Y,

Else Area Penalty = 0

Total Area Penalty = Σ_t Area Penalty$_t$

Where,

i indexes the individual projection units,

j indexes the set of treatment alternatives including no treatment,

t indexes the planning horizon periods 1 to 3 (three decade planning horizon),

Threat$_{i,t}$ is the fire threat class of unit i in period t,

adj$_i$(j) is the set of treatment selected for adjacent units that affect the threat index for unit i,

LSF Penalty is the penalty for violating the minimum area required to meet the LSF definition, Total Area Penalty is the aggregate penalty for violating the maximum area treated in each period;

B_1 is the weighting factor for the LSF penalty

B_2 is the weighting factor for the area penalty

Area Penalty$_t$ is the periodic penalty for violating the maximum area treated,

Area$_i$ is the size of unit i in hectares,

LSF $_{i,t}$ = 1 if Unit i meets the LSF definition in period t, otherwise= 0,

Period $_{i,t}$ = 1 if Unit i is scheduled for an active treatment in period t, otherwise=0, X is the minimum amount of LSF structure required, aggregated for all three periods, and

Y is the maximum amount of area that could be treated in a single period.

Identify Marginal Costs of Silvicultural Treatments

The PP curves helped identify a range within which silvicultural treatments could achieve relatively high FT reduction at relatively low cost to LSF structure. We then examined how FT reduction levels within this range would be affected by different financial requirements for the treatments. To develop a three- dimensional production function in terms of net present value (NPV), LSF structure, and FT, we held LSF structure and treated area constant. The objective function was to reduce landscape FT, while setting as constraints the area of LSF structure (6,880 ha), the maximum treated area (≤10% of the reserve), and NPV. The NPV constraint levels were USD $0 (break-even), $0.5 million, $1 million, and $1.5 million in addition to an unconstrained baseline. We also refer to the three cases in excess of $0 as positive financial constraints or +NPV (details in Hummel and Calkin 2005).

Results

Costs of Landscape Silviculture Treatments

The levels of FT reduction and LSF structure in the reserve decreased with increasing financial requirements (*fig. 3*). A positive financial constraint imposed costs on landscape objectives because the net revenue from silvicultural treatments at a unit scale could be either positive or negative, depending on the existing stand conditions and the treatment applied. Treatments that lost money at the scale of an individual unit were rarely selected by the algorithm when a high NPV constraint was imposed even though they reduced landscape FT at low cost to LSF structure (details in Hummel and Calkin 2005).

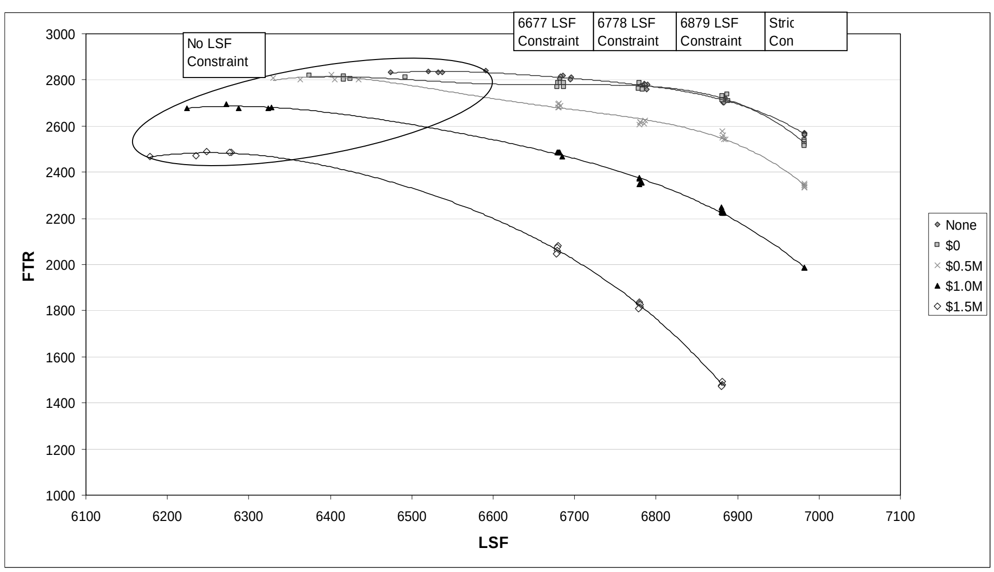


Figure 3–Fire threat reduction (FTR) (weighted index values) vs. late-seral forest (LSF) structure (hectares) for a range of net present value (NPV) requirements over 30 years in the Gotchen LSR. The ellipse highlights the unconstrained option (Source: Hummel and Calkin 2005).

With or without a financial requirement to break-even, treatments accomplished about the same level of FT reduction and LSF structure over the 30-year analysis period (*fig. 3*). Although the treatment results were similar, the net revenues were not. For similar levels of FT reduction and LSF structure, net revenues for various mixtures of treatments ranged from -$1,000,000 to $3,000 over thirty years (details in Hummel and Calkin 2005).

Characteristics of Trees Removed in the Break-Even Case

The mixture of treatments that met landscape objectives for fire and habitat management, and also generated enough revenue to break-even, included wood volume from trees <17.8-55.9 cm. The largest component came from the 18-40.6 cm size class (*fig.4*). In each decade, the main species harvested in this diameter class was grand fir (*fig.5*).

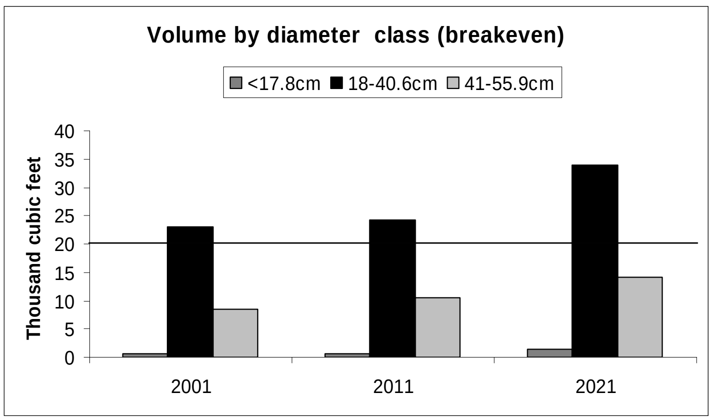


Figure 4–Volume of wood removed by diameter class in simulated treatments that broke even over 30 years in the Gotchen LSR landscape and met fire and habitat management objectives**.**

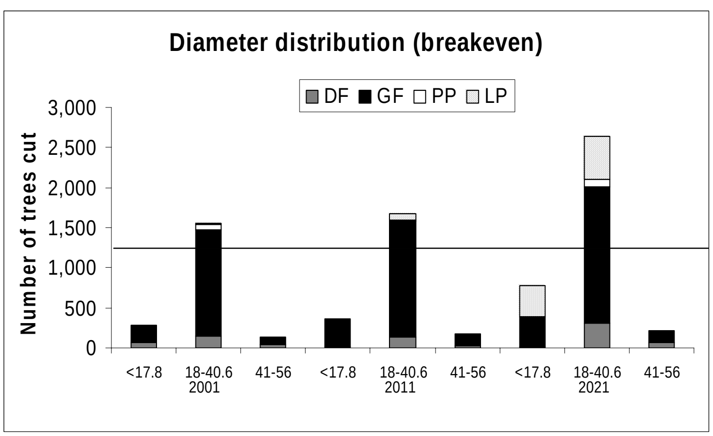


Figure 5–Number of trees cut by diameter in simulated treatments that broke even financially over 30 years in the Gotchen LSR landscape and met fire and habitat management objectives (DF=Douglas-fir, GF= grand fir, PP=ponderosa pine, and LP=lodgepole pine).

Discussion

The predominance of medium-sized, shade-tolerant conifers in the break-even case contributes to contemporary discussions about conserving and developing older forests in ways that support their resilience in fire-adapted ecosystems of the West. Such trees are unlikely to be legacies remaining from pre-settlement, pre-fire exclusion days. For example, a 38.1 cm grand fir tree in the Gotchen LSR is, on average, about 80 years old (Hummel et al. 2002). This means that a focus on removing medium-sized, shade tolerant trees to support fire and habitat management objectives in the reserve would be consistent with reducing the number of trees that have established since federal fire exclusion policies began in the early 1900s. While this result may seem intuitive with respect to fire, by using simulation and optimization techniques, we gain additional information about tradeoffs with other landscape objectives. Namely, an emphasis on removing medium-sized trees may not directly conflict with owl habitat objectives for late-seral forest at the among-stand scale. This is informative, given that questions exist about the compatibility of fire and habitat objectives in the drier provinces of the Plan area (e.g. Spies et al. 2006). While results from the Gotchen LSR case study are applicable only in the study reserve, the method we used--linking landscape dynamics and patterns of forest structure to stand level silvicultural treatments by considering the treatments collectively rather than on a stand-by-stand basis--could be used anywhere that multiple management objectives share a common basis in forest structure (e.g. wildfire and home sites, recreation opportunities, and wildlife habitat).

Medium-sized grand fir trees are merchantable in the mills located in the geographic area of the Gotchen LSR and should be readily accepted in local markets (Barbour et al. 2001). Available data (1991 to 2001) for lumber from grand fir indicates that true firs (hem-fir lumber) has consistently been priced about 12 to 20 percent (average 13.5 percent) lower than Douglas-fir in coastal markets and about 65 percent below ponderosa pine in interior markets (Warren 2004).

Possibly a more important question than whether the harvested wood is merchantable is whether stewardship contracts, such as the one associated with the Gotchen LSR planning area (Stray Cat), will find bidders. The issue is not solely one of merchantability but also one of risk, including how it is calculated, what it costs, and who bears it. Stewardship contracts are for services (such as fuel reduction and pre-commercial thinning) in which some of the implementation costs can be offset by the value of the resources that are removed. Revenues from stewardship contracts might not be returned to the US Treasury. Because of concerns about decay in true fir trees (Aho 1977), potential bidders on Stray Cat preferred a scaled sale, whereas the Forest Service tends to prefer lump sum sales.

Other challenges related to designing and awarding stewardship contracts relate to the length of contractual obligations for services. Obligations that impose contractual burdens into the future (like controlling invasive herbaceous plants) carry uncertainty for bidders. Such an obligation can seem very risky to bidders, particularly when experience is limited with which to evaluate probable costs. For the Forest Service, such obligations carry a different challenge. While collections processes exist for contractual requirements, like reforestation and fuels treatments, similar processes are not yet in place for other, newer requirements related to landscape restoration activities and community needs. In addition, budget reductions and reduced agency personnel limit the marking of individual trees within sale units. Designation by description (DxD) is often used instead. Bidders on stewardship contracts can also be concerned about the costs associated with deciding which trees to remove during sale volume determination and logging activities. From a different perspective, interest groups express concern whether residual forest conditions will be consistent with the intended prescription. Stray Cat, which includes DxD units, is lump sum and due to be implemented, beginning in 2006. Almost two years elapsed between the signed project decision and the award of a stewardship contract.

Summary

When the Gotchen LSR study began, forest growth models were relatively limited in their ability to link a landscape shape file with the multiple stands and the trees comprising it. FVS was no exception (e.g. Hummel et al. 2001). In the years since then, model developers have devoted considerable time to link stand attributes with landscape models (e.g. FARSITE [Finney 1998], LMS [McCarter 1997], and FVS). Continued progress in this area will help to automate and standardize the approach we took in this study, namely, to link a landscape to its stands and then to individual trees. Our intent in taking this approach was to expand silvicultural decision-making beyond a unit-by-unit approach, and instead to consider adjacent units and landscape objectives explicitly in such decisions. In landscape silviculture (as we define it), post-treatment conditions in a given unit can only be evaluated within the context of objectives for an LSR, because what appears to support landscape objectives in isolation may change when considered in total. Indeed, study results suggest that the potential for conflict or compatibility among landscape objectives for fire and habitat management is scale-dependent. There are questions raised or unanswered by the Gotchen LSR study that seem important, if additional progress is to be made on designing and evaluating silvicultural treatments to accomplish multiple landscape objectives. With respect to wildfire in particular, these include understanding what mean size and shape of treated units is most effective in reducing severity, and whether the spatial pattern of treatments matters more than the total biomass removed, or vice versa.

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SDI-Flex: A New Technique of Allocating Growing Stock for Developing Treatment Prescriptions in Uneven-Aged Forest Stands [1]

Wayne D. Shepperd[2]

Abstract

One of the difficulties of apportioning growing stock across diameter classes in multi- or uneven-aged forests is estimating how closely the target stocking value compares to the maximum stocking that could occur in a particular forest type and eco-region. Although the BDQ method had been used to develop uneven-aged prescriptions, it is not inherently related to any maximum stocking guide. Adapting Stand Density Index (SDI) to uneven-aged silviculture by apportioning stocking equally in all diameter classes has been proposed. SDI_{max} is the maximum stocking possible for a given species and region and provides a consistent and reliable benchmark on which to base silvicultural prescriptions, which can be expressed as a percentage of maximum SDI ($\%SDI_{max}$). However, allocating a consistent percentage of maximum SDI desired after thinning across all diameter classes raises or lowers the resulting stocking curve, but does not change its shape, as could be done by changing the Q value in BDQ stocking control. This results in high numbers of small trees being retained, regardless of the $\%SDI_{max}$ prescribed, and has been a major criticism of using SDI-based stocking control for fuels treatment biomass estimates. The SDI-Flex procedure presented here combines the flexibility of the BDQ method with an SDI-based stocking guide. In this method, a Flex Factor can is used to proportionally reduce the amount of SDI assigned to successively smaller dbh classes. A simple spreadsheet program can be used to iteratively manipulate both the Flex Factor and $\%SDI_{max}$ values to arrive at a desired stocking configuration for a particular silvicultural objective.

Introduction

The science of silviculture as practiced on Federal lands has undergone a metamorphosis, from a discipline primarily concerned with the efficient production of forest products to one in which the emphasis is now on manipulating forest composition and stocking to produce conditions that will meet other natural resource needs. However, in spite of the need to maintain some forest structures and conditions through time, the adoption of uneven-aged silviculture has been largely avoided by practicing silviculturists. I believe there are several reasons for this. Many silviculturists view uneven-aged silviculture as being too complex and too hard to readily apply on the ground. The body of published knowledge and guidelines describing the use of uneven-aged silviculture in various forest types is limited.

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[1] A version of this paper was presented at the National Silviculture Workshop, June 6-10, 2005, Tahoe City, California.

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Specific standards for using BDQ stocking guide curves have not been developed for all forest types. No equivalents to Gingrich curves (Gingrich 1967), stand density management diagrams, or other stocking guides exist for uneven-aged silviculture. Silviculturists are forced to rely on intuitive measures and personal experience to develop uneven-aged prescriptions and, therefore, may be reluctant to prescribe vegetation treatments that cannot be substantiated by scientific literature. This paper proposes to remedy this situation by introducing a new modification of a long-established even-aged stocking guide that can be used to develop a wide variety of uneven-aged prescriptions in all forest types.

A Brief Review of Stand Density Index

A thorough and complete discussion of the history of Stand Density Index (SDI) and an evaluation of the various ways of calculating it was presented by Ducey and Larson (2003) and will not be duplicated here. Basically, SDI was conceived by Reineke (1933) to describe the empirical relationship between quadratic mean stand diameter (D_q) and tree density in even-aged forests. Reineke noticed that a consistent pattern existed when average tree size and stand density data from numerous stands were plotted on a log/log scale (*fig. 1*). He chose to express this relationship mathematically as an index equivalent to the number of 10-inch trees that might occur in a forest of a given density and derived the following equation to describe it:

$$SDI = N(D_q/10)^{1.605}$$

Where: N = trees ac^{-1}

And D_q = quadratic mean stand diameter

Since D_q can be other than 10 inches, SDI is really an index most of the time. The pattern shown in *Figure 1* occurred among all species that Reineke investigated, although the height of the data swarm varied by species. The use of a 10-inch size standard in the SDI equation allows the maximum SDI value for a tree species in a locality to be calculated by dividing the maximum stand basal area found in the defined population by the basal area of a 10-inch tree (0.5454 ft^2):

$$SDI_{max} = (BA_{max}/0.5454)$$

For example, if the maximum average stand basal area for a tree species in an eco-region is 270 $ft^2\,ac^{-1}$, then:

$$SDI_{max} = (270/0.5454) = 495$$

Stocking in any given stand can therefore be expressed as a percentage of SDI_{max}. These percentages are roughly equivalent to the percent maximum density lines that appear on Gingrich stocking charts (*fig. 2*) (I say roughly, because some Gingrich curves were sometimes re-fit to individual species data using modern non-linear regression techniques, and were not derived using Reineke's SDI equation). However, the similarities are sufficient to provide a direct link between even-aged stocking guides that appear in Forest Service Silviculture handbooks and Reineke's SDI.

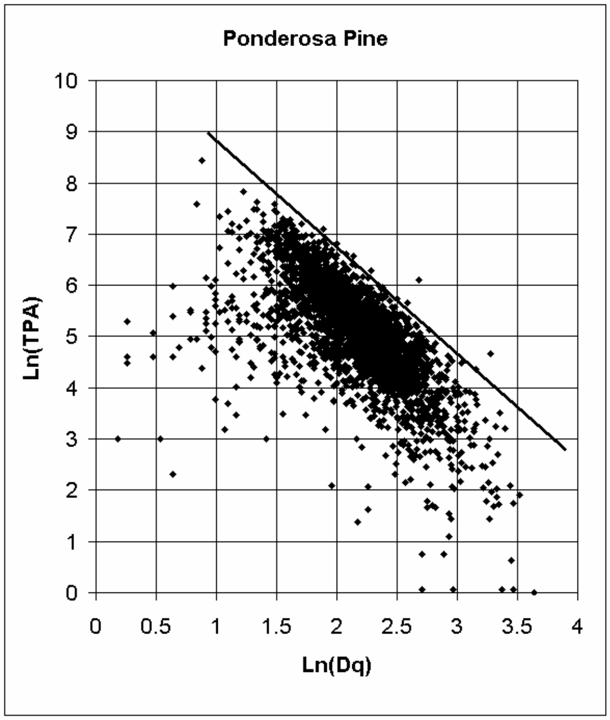

Figure 1--Quadratic mean stand diameter (Dq) versus tree density (TPA) for ponderosa pine in Colorado, plotted on log normal scales with SDI_{max} = 450 line (from FIA data).

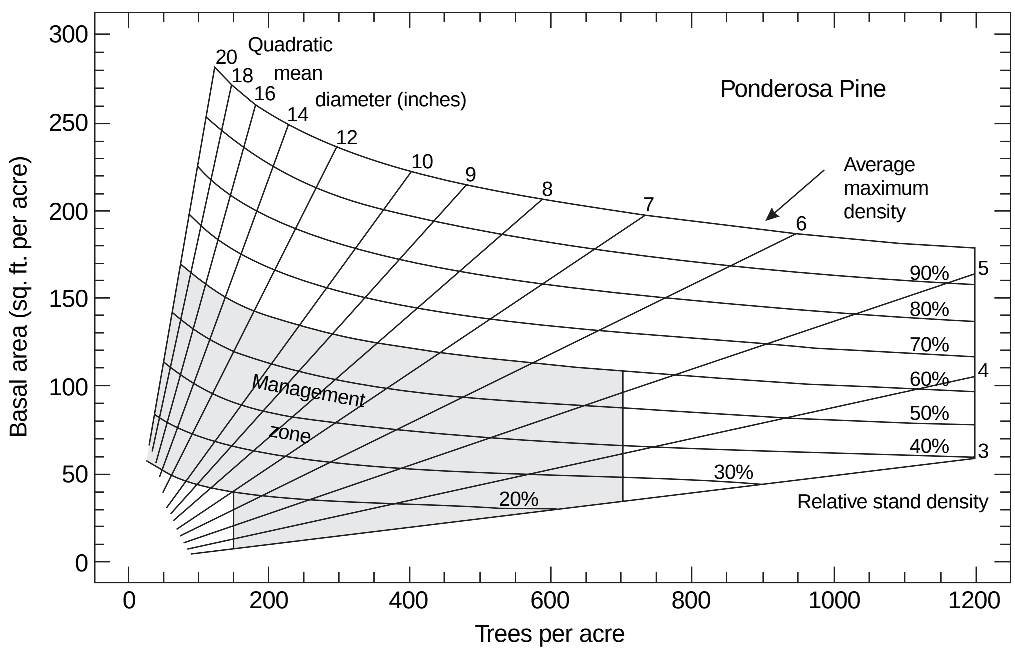

Figure 2--Gingrich stocking chart for even-aged ponderosa pine in Colorado. Average maximum density corresponds to SDI_{max}.

Modifying SDI for Use in Uneven-Aged Silviculture

Long and Daniel (1990) presented a modification of SDI for use in uneven-aged stands. They proposed calculating a partial SDI for each diameter class, and then summing the values to obtain an overall stand SDI in the following manner:

$$SDI = \sum N_i (D_i/10)^{1.605}$$

Where: N_i = trees ac^{-1} in diameter class i
And D_i = mid-point of diameter class i

This equation can be utilized to proportion desired post-harvest target stocking evenly among diameter classes by calculating a target stocking density for each diameter class (D_i) in the stand. First, a target SDI (SDI_t) is calculated by dividing the percentage of maximum SDI desired for the residual stand by the number of dbh classes in the stand. SDI_t is then substituted for SDI in the above equation and solved for N_i. giving:

$$N_i = SDI_t/(D_i/10)^{1.605}$$

This procedure results in tree ac^{-1} stocking values across diameter classes that typify the "reverse J" target stocking curve associated with uneven-aged forests. In this case, each dbh class contains an equal proportion of the overall desired percentage of maximum SDI for the stand. While this procedure allows the overall target stocking to be adjusted to any percent of maximum SDI, the even apportionment of SDI over all diameter classes does not provide the ability to adjust the slope of the resulting stocking curve, as could be done by changing the Q value in BDQ stocking control. The result is a high number of small trees being retained regardless of the percentage of maximum SDI prescribed. This shortcoming hinders the flexibility and usefulness of the Long and Daniel approach, especially when open understories are needed (e.g. prescribing uneven-aged treatments to reduce crownfire risk).

The SDI-FLEX Procedure

This paper presents an alternative method of calculating a target stocking distribution for uneven-aged forest stands that allows the shape of the SDI target stocking curve to be changed, as well as its height (*fig. 3*), and permits the development of an infinite variety of uneven-aged stocking curves. This is accomplished by utilizing a "Flex Factor" to systematically reduce the portion of SDI assigned to smaller dbh classes. Stocking can then be adjusted to alter stand characteristics to meet multiple resource needs, while still meeting the percentage of maximum SDI stocking desired for the stand. For example, denser understories can be provided to stimulate the development of good tree form, provide hiding cover, or discourage the development of undesirable shrubs. Or, conversely, sparse, open understories can be created to favor forage growth, or reduce live ladder fuels where crownfire risk is a concern. *Figure 4* illustrates the dramatic effect that flexing SDI can have on the appearance of a forest stand. The only limitation that should be placed on the development of uneven-aged stocking curves using this technique is that sufficient numbers of trees should be retained in smaller diameter classes to grow into and replace trees in larger classes (e.g. the stocking curve should always slope slightly downward to the right to retain the "reverse J" shape).

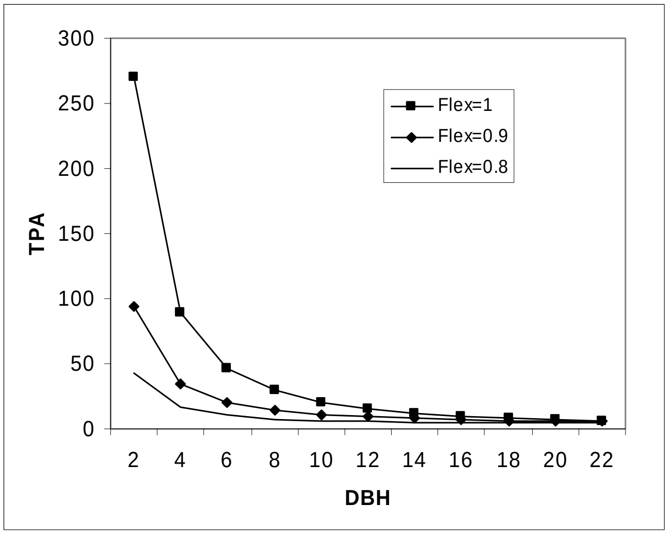

Figure 3--Flexing the distribution of SDI allows both the height and shape of a desired stocking curve to be changed (Note that the %SDI_{max} is not the same under these three curves).

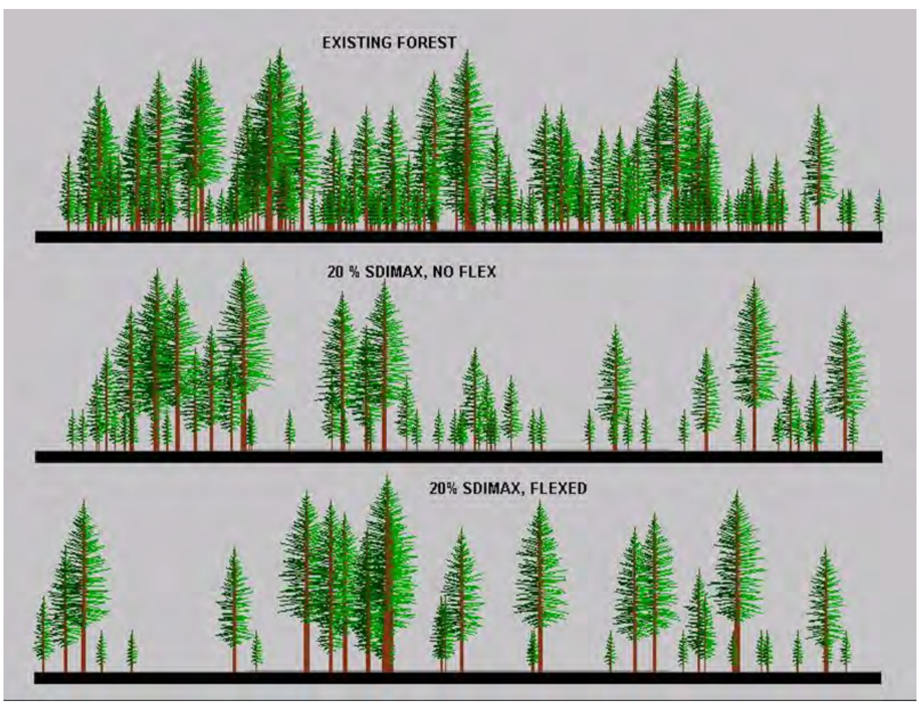

Figure 4–The stand at the top was thinned to 20% SDI_{max} using an even apportionment of SDI to all diameter classes (center) and a Flexed apportionment that kept overall stocking at 20% SDI_{max}, but reduced stocking in smaller diameter classes (lower). Graphs were produced using Stand Visualization System (SVS) software (McGaughey 1997).

SDI-FLEX Calculations

An example of a spreadsheet that can be constructed to develop prescriptions using the SDI-Flex approach is shown in *table 1*. Dbh class midpoints are listed in Column A. Trees per acre (TPA) stocking values obtained from inventory data for each dbh class are listed in Column B. The user must also specify the maximum SDI value used for the forest type and locality and enter the number of dbh classes that contain trees in appropriate cells. Basal areas are calculated in Column C using the following formula:

$$BA_i = TPA_i * (dbh_i^2 * 0.0054542)$$

SDI values for each diameter class in the existing stand are calculated in Column D, using the formula given earlier. Succeeding columns are used to develop a prescription and marking guide to treat the stand by first calculating a desired target stocking curve, using user supplied values for SDI_{max} and two control parameters called the Seed and Flex factor, which control the percentage of SDI_{max} under the target stocking curve and the shape of the curve. This is done by placing a “1” in the largest dbh class desired for the stand, and sequentially reducing that value by the Flex Factor listed at the bottom of Column E in cells above for each successively smaller dbh class (e.g. $FLEX_{20} = 0.9 * 1$, $FLEX_{18} = 0.9 * 0.9$, $FLEX_{16} = 0.81 * 0.9$, etc.). Target SDI values in Column F are calculated using the following formula:

$$\text{Target } SDI_i = (SDI_{max} / \text{No. dbh classes} * \text{SDI Seed}) * FLEX_i$$

(Note: the number of dbh classes specified should be either the number of stocked dbh classes, or the number of dbh class desired for the managed stand.)

The SDI Seed parameter is used to adjust stocking uniformly over all dbh classes to adjust the stocking to a desired percentage of maximum SDI. The values in Column F represent the desired stocking in each dbh class following the geometric stocking curve specified by the Flex Factor, which sequentially reduces the portion of SDI in each successively smaller dbh class. Setting the Flex Factor to 1.0 results in an even apportionment of SDI over all dbh classes, as with the Long and Daniel (1990) procedure. Using a value less than 1.0 flattens the stocking curve, reducing the number of trees in smaller dbh classes. Target TPA values for each diameter class are calculated in Column G by substituting each SDI_i value into the formula presented earlier:

$$TPA_i = SDI_i / (D_i/10)^{1.605}$$

Setting the SDI Seed to a desired percentage of SDI_{max} will result in an actual percent SDI stocking target that is less than the desired value, since the portion of SDI assigned to smaller db classes is sequentially reduced. Therefore, both the Flex Factor and SDI Seed values must be iteratively manipulated until the desired actual $\%SDI_{max}$ is obtained. The smaller the Flex Factor, the larger SDI Seed must be to achieve the desired actual $\%SDI_{max}$. In practice, the user should first specify a Flex Factor of 1.0 and the $\%SDI_{max}$ as the SDI Seed, then iteratively decrease the Flex Factor until stocking in smaller diameter classes is reasonable, and finally raise the specified SDI Seed until the actual $\%SDI_{max}$ approaches the desired level for the residual stand.

Table 1--*Example of uneven-aged prescription developed in a spreadsheet using the SDIFlex procedure. Calculations are explained in detail in the text.*

SDIFLEX PRESCRIPTION SPREADSHEET - PONDEROSA PINE, COLORADO FRONT RANGE												
A	**B**	**C**	**D**	**E**	**F**	**G**	**H**	**I**	**J**	**K**	**L**	**M**
MaxSDI =		450	Existing	dbh_i	Target	Target	Target	Cut	Residual	Residual	Residual	Residual
DBH	**TPA**	**BAac^{-1}**	**SDI**	**FACTOR**	**SDI**	**TPA**	**BAac^{-1}**	**TPA**	**TPA**	**BAac^{-1}**	**SDI**	**Cut%**
2	200	4.36	15.11	0.35	7.13	94.42	2.06	105.58	94.42	2.06	7.13	0.53
4	100	8.73	22.98	0.39	7.92	34.49	3.01	65.51	34.49	3.01	7.92	0.66
6	60	11.78	26.43	0.43	8.81	19.99	3.92	40.01	19.99	3.92	8.81	0.67
8	40	13.96	27.96	0.48	9.78	14.00	4.89	26.00	14.00	4.89	9.78	0.65
10	20	10.91	20.00	0.53	10.87	10.87	5.93	9.13	10.87	5.93	10.87	0.46
12	30	23.56	40.20	0.59	12.08	9.01	7.08	20.99	9.01	7.08	12.08	0.70
14	40	42.76	68.64	0.66	13.42	7.82	8.36	32.18	7.82	8.36	13.42	0.80
16	30	41.89	63.79	0.73	14.91	7.01	9.79	22.99	7.01	9.79	14.91	0.77
18	4	7.07	10.27	0.81	16.57	6.45	11.40	0.00	4.00	7.07	10.27	0.00
20	2	4.36	6.08	0.90	18.41	6.05	13.20	0.00	2.00	4.36	6.08	0.00
22	1	2.64	3.54	1.00	20.45	5.77	15.23	0.00	1.00	2.64	3.54	0.00
24	0	0.00	0.00									
26	0	0.00	0.00									
Classes	527	172.03	305.00	**FlexFac**	140.36	215.88	84.88	322.39	204.61	59.11	104.83	
11			0.68	**0.9**	0.31						**0.23**	
				SDI seed								
				0.5	**%MaxSDI of Target Stand**							
	%MaxSDI of Existing Stand				**(If all dbh classes stocked)**						**%MaxSDI of Residual Stand**	

Since these calculations derive a theoretical desired stocking curve, they have to be adjusted to reflect actual stocking if any dbh classes are unoccupied, or are stocked at less than desired target values. This is accomplished using a MIN spreadsheet function in Column I to choose the minimum of either the target (Column G), or existing (Column B) stocking and subtracting that value from the existing stocking to calculate the number of trees to be cut.

Residual TPA, BA, and SDI values after the harvest are calculated in Columns J, K, and L, respectively. The actual residual $\%SDI_{max}$ left after harvest is shown at the bottom of Column L. If local stem volume conversion factors are available the spreadsheet can be further modified to calculate per acre volume yields by multiplying the cut TPA values for each diameter class by the associated conversion factor and summing the results.

To help develop marking guides for the prescription, cuts for each dbh class are also expressed in Column M as a proportion of original stocking. For example, a cut proportion of 0.8 would mean to mark eight out of every 10 trees encountered in that dbh class. Per acre summaries of all columns and $\%SDI_{max}$ values are displayed for existing, target, and residual stand conditions to aid the user in evaluating prescription alternatives and allow the user to see the relationship between tree density, basal area, and SDI for a given prescription. The user may also desire to set up a spreadsheet graph that plots actual, target, and residual stocking across dbh classes to aid in prescription development.

Discussion

This procedure provides an easy-to-use and highly effective means of developing uneven-aged silviculture prescriptions that are based on a long-accepted empirically-derived stocking guide that is unique to each tree species. Residual stocking can be adjusted to meet a variety of silviculture needs, while retaining an expectation of the forests potential for future production and growth.

The SDI-flex approach must be applied with an eye toward achieving the desired residual stand condition configured in a way that meets the diverse structural attributes in the stand. The best approach is for the user to decide in advance what the target residual basal area and residual SDI should be. Then, adjustments in flex and seed should be made iteratively until those target attributes are attained. It’s tempting to merely look at the volume to be removed and work the program until a desired volume target is obtained, and this may be valuable to help decide whether a proposed treatment will be commercially operable. But future growth of the stand depends on retaining acceptable basal area after the harvest, distributed in the proper diameter classes to meet the structural needs of the stand. Most growth models show that volume increment is highly correlative with the starting basal area. Future growth in stands managed using the SDI-flex will depend upon retaining acceptable residual basal area and $\%SDI_{max}$ after the proposed cut. There are no safeguards other than user attentiveness to be certain that the seed and flex variables are properly applied, and result in an appropriate residual stand condition. But if these cautions are observed, the model is a very quick and creative way to configure SDI among diameter classes toward the goal of meeting the desired structural goal.

The actual percentage of maximum SDI retained after harvest is roughly equivalent to the percent of maximum stocking lines shown on even-aged Gingrich stocking curves and can serve a similar purpose in guiding management. As in even-aged silviculture, trade-offs exist between optimizing the productive capabilities of a site and meeting alternate resource demands for the forest. For example, a forest stocked at levels above 50% maximum SDI is likely to be subject to density-dependent mortality and be susceptible to insect attack (Oliver 1995). Conversely, a forest stocked at 15–20% maximum SDI is probably stocked at less than full occupancy, is not producing optimal yields, and may contain open-grown trees that are limby and of poor form.

Several observations have become apparent from using the SDI-Flex procedure that should be mentioned here. Initial users have had questions as to what maximum SDI value to use for mixed species stands. I suggest using a maximum SDI associated with the tree species that is most likely to regenerate, or alternatively, using the max SDI value for the most intolerant species for which regeneration is sought. If attempting to restore a ponderosa pine forest from a mixed conifer condition, I recommend using the maximum SDI for ponderosa pine to produce favorable conditions for regeneration and growth of that species. Higher residual stocking levels will tend to favor regeneration of shade tolerant species while lower stocking will favor intolerant species. Similarly, "Steep" stocking curves with numerous small trees in the understory will favor shade tolerant species, while "shallow" curves will favor intolerant species. Low SDI values in residual stands will generally encourage abundant natural regeneration, as well as development of associated understory species. This could be desirable in vegetation associations that provide forage to wildlife or livestock, or undesirable if the understory consists of aggressive shrubby species.

When developing prescriptions for reducing crownfire risk, even a few low-crowned residual trees will affect average crown height and thus result in a low torching index. Adverse effects can be avoided in these cases by managing for grouped or clumped regeneration, or marking the stand to isolate low-crowned trees from larger trees. Even so, managers must accept that partial tree removal alone cannot change fire behavior in all cases and may result in abundant regeneration, regardless of whether even- or uneven-aged silviculture is used. Mechanical pruning or prescribed burning under safe conditions may be needed to raise crown base heights and reduce torching potential.

One reviewer suggested using the "Solver" spreadsheet function to automatically do the iterative calculations to find the SDI Flex and Seed factors that would yield the $\%SDI_{max}$ desired for the stand. While that is certainly possible to do, the intent here is to give the user maximum flexibility to seek a stocking configuration that will meet the overall management objectives for the stand. Controlling the iterative process manually allows the user to examine alternative approaches to developing a prescription, and to weigh the trade-offs between yields and residual stocking across all diameter classes. Hopefully, this approach will help achieve a workable compromise among sometimes conflicting management goals.

In conclusion, I believe that using the SDI-Flex procedure provides silviculturists a quick and easy method of evaluating uneven-aged silviculture prescription alternatives. The procedure uses a well-established scientifically-based, stocking guideline familiar to even-aged silviculture practitioners that can be adjusted to specific tree species and local growing conditions. The spreadsheet example presented here allows easy calculation of existing, target, and residual stocking and easy manipulation of parameters to develop a wide variety of uneven-aged prescriptions for today's diverse forest management needs. Example SDI-Flex spreadsheets are at: http://www.fs.fed.us/rm/landscapes/Solutions/SDI_Flex.shtml

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Gap-Based Silviculture in a Sierran Mixed-Conifer Forest: Effects of Gap Size on Early Survival and 7-year Seedling Growth [1]

Robert A. York,[2] John J. Battles,[3] and Robert C. Heald[2]

Abstract

Experimental canopy gaps ranging in size from 0.1 to 1.0 ha (0.25 to 2.5 acres) were created in a mature mixed conifer forest at Blodgett Forest Research Station, California. Following gap creation, six species were planted in a wagon-wheel design and assessed for survival after two growing seasons. Study trees were measured after seven years to describe the effect of gap size on early growth of planted trees. Giant sequoia had the lowest mortality (2.4 – 5.0 percent), sugar pine, incense-cedar, ponderosa pine, and Douglas-fir all had comparable levels of mortality (5.8 - 18.9 percent), and white fir had the highest level of mortality (35.7 – 47.2 percent). To rank candidate models according to goodness of fit while penalizing for model complexity, we used an information-theoretic approach using Akaike Information Criteria. An asymptotic fit of height growth to gap size was most commonly selected as the best model among a set of feasible a priori candidate models, although there was some model parity. As gap size increased, height gains tended to diminish between 0.3 and 0.6 ha (0.75 to 1.5 acres). Shade tolerance classifications did not predict relative mortality levels or functional responses of height growth to gap size.

Introduction

Society places great demands on forests, managing them through the application of silviculture, to provide highly valued financial and conservation assets. As Kimmins (2002) notes, however, the rate of change in society's expectation of forests outpaces the scientific foundation to accommodate these new demands. For example, in the American West, social, political, and ecological concerns about single-cohort silvicultural systems have motivated demands for multi-cohort systems, which more closely approximate natural forest dynamics (O'Hara 2001), before methods for sound implementation have been developed, or their effects assessed.

Gap based silviculture, i.e., group selection, is one multi-cohort system in particular that has been proposed as a promising regeneration method, capable of achieving the multiple expectations of forest management. In theory, gap based silviculture mimics the structural diversity created by fine scale natural disturbances, resulting in canopy gaps (Smith et al. 1997). In practice, it is often perceived as a

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compromise approach for landowners aiming to avoid alleged environmental degradation associated with clearcuts and assumed limited productivity associated with single-tree selection (Bliss 2000). Management practices incorporating gap based silviculture have been proposed as a means for achieving a wide variety of objectives, including ecological restoration (Storer et al. 2001), maintenance of species diversity (Hamer et al. 2003, Lahde et al. 1999, Schutz 1999), and management of endangered species habitat (USDA Forest Service 1995). Gap based silviculture has recently been included in proposals for managing forests across regional scales for use within a wider framework of management, where the objective is maintaining fire-adapted forests (e.g. Herger-Feinstein 1998, USDA Forest Service 2003, USDA Forest Service 2002). However, scientific information to support the management decisions to implement these proposals is often limited.

A major source of uncertainty rests with the details of implementing a gap harvesting regime (Webster and Lorimer 2002). Of primary concern is the cost in terms of reduced growth productivity associated with the high edge-to-interior ratio of smaller openings (Bradshaw 1992, Dale et al. 1995, Laacke and Fiske 1983, Leak and Filip 1977). To address this concern, much of the research involving artificially created gaps has focused on the appropriate (often minimum) opening size that meets management objectives, particularly successful regeneration and growth of desired species within openings (Coates 2000, Gray and Spies 1996, Leak and Filip 1977, Malcolm et al. 2001, McDonald and Abbott 1994, McGuire et al. 2001,Van Der Meer et al. 1999, York et al. 2003, York et al. 2004). Still, the question of what is the “best” opening size, one that achieves the multiple promises of gap based silviculture, remains largely unanswered for even well-studied forest ecosystems.

To demonstrate an experimental approach for guiding local adaptive management decisions, and to provide specific insight into the capacity for gap-based silvicultural regimes to promote regeneration and growth in a western conifer forest, we established a manipulative experiment using artificial gaps and planted seedlings. Using treatments that remove competition between trees and from non-tree vegetation, we track the survival and early growth of planted trees, as it varies by species, within-gap position, and gap size. Here, we present results quantifying relative species performances in terms of seedling survival after two years and height growth through the first seven years after gap creation.

Methods

Study Site

Blodgett Forest Research Station (BFRS) is located on the western slope of the Sierra Nevada mountain range in California (38°52’N; 120°40’W). The study area lies within BFRS at an elevation between 1220 and 1310m. The climate is Mediterranean with dry, warm summers (14 to 17 °C) and mild winters (0 to 9 °C). Annual precipitation averages 166 cm, most of it coming from rainfall during fall and spring months, while snowfall typically occurs between December and March. The soil developed from granodiorite parent material and is highly productive for the region. Heights of codominant canopy trees typically reach 27 to 34m in 50 to 60 years (BFRS 2003). Olson and Helms (1996) provided a detailed description of BFRS, its management, and trends in forest growth and yield.

Vegetation at BFRS is dominated by a mixed conifer forest type, composed of variable proportions of five coniferous and one hardwood tree species (Laacke and

Fiske 1983, Tappeiner 1980). Native tree species include white fir (*Abies concolor* [Gord. & Glend.] Lindl. Ex Hildebr.), incense-cedar (*Calocedrus decurrens* Torr.), Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco var. *menziesii*), sugar pine (*Pinus lambertiana* Dougl.), ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.), and California black oak (Quercus kelloggii Newb.). In harvested openings, BFRS also plants giant sequoia (*Sequoiadendron giganteum* [Lindl.] Buchholz), a species that is not present in the study area, but in the past had an expanded range that encompassed BFRS (Harvey 1985). Treatments for this study were all located on the same, north-facing slope (10-25 percent). Like much of the mixed conifer forests in the Sierra Nevada range (Beesley 1996), the study area was clearfell harvested for timber extraction in the early 1900's and allowed to regenerate from sparse residual trees. Nearly a century following this disturbance, young-growth stands at BFRS have developed to form a continuous mixed species canopy, averaging 35m in height and 83 m^2/ha in basal area (BFRS 2003).

Experimental Design

Experimental treatments were designed to isolate the factor of opening size and the potential influence it may have on seedling growth within the openings, as well as on the mature forest (i.e. matrix) between the canopy openings over time. Care was taken to ensure a balanced design, so that each species was represented with the same density evenly throughout the gaps. The experiment involved a regeneration treatment (the harvest of the gaps and planting of seedlings), and a series of maintenance treatments designed to minimize confounding factors of non-tree and inter-tree competition. Clearing the harvested areas of debris and then planting seedlings reduced the degree of micro-site heterogeneity that could obscure gap size and position effects (Gray and Spies 1997). Four circular opening sizes (0.1, 0.3, 0.6, and 1.0 ha), each replicated three times, were harvested in 1996 and planted with an even mix of six species (Douglas-fir, giant sequoia, incense-cedar, ponderosa pine, sugar pine, and white fir). The ratios of gap diameters to the surrounding canopy heights for these sizes from smallest to largest are 1, 1.8, 2.6, and 3.2. Seedlings were planted along rows with a wagon wheel design, each species planted along "spokes" extending from the drip lines into the opening centers in cardinal and inter-cardinal directions. Douglas-fir, incense-cedar, white fir, and ponderosa pine were planted from bare root stock. Sugar pine and giant sequoia were from container stock. Seedling sizes were similar for all species at the time of planting. Non-tree vegetation was suppressed throughout the openings through the first three years after planting, and study trees were thinned from 3m spacing to an average of 4.5m spacing after the 7th growing season to avoid inter-tree competition. More information on findings of within-gap spatial variation in seedling growth, as well as further details of the treatments and layout design, were given by York et al. (2004).

Measurements

To assess the capacity for successful artificial regeneration within this gap regime, a mortality survey of all planted seedlings was conducted after the second growing season. For describing height growth responses to gap size, the experimental unit is each gap (n = 12). All study trees within the gaps were measured for height after the 7th growing season (n = 2440). In one of the 0.1 hectare gaps, Douglas-fir and white fir seedlings experienced high mortality where two seedling rows overlapped with a high water table near the edge of the gap. Typically, swampy areas like this would not be converted to gaps artificially and are therefore considered unrepresentative of our intended study domain. Mean heights for Douglas-fir and white fir from this gap were therefore not included in the analysis.

Data Analysis

In accordance with our whole-gap level of inference, percent mortality was calculated for each gap (n = 12) for each of the six species. To explore gap size effects on mortality, standard linear regressions were used, with gap size as the independent variable, and percent mortality per gap as the dependent variable. Evidence of a gap size effect on mortality is confirmed by the probability of the regression line's slope differing from zero ($\alpha = 0.05$). To compare how overall survival differed between species, given the gap sizes used in this gap regime, mean percent mortality and 95 percent confidence intervals of the means are used for species comparisons. The gap size range used here represents the range used to define group selection regeneration method in the California Forest Practice Rules.

To assess the fine-scale relationship between opening size and within-gap tree height growth at year seven, we relied upon an information-theoretic approach to select an appropriate model of the data from a set of a priori candidate models. Candidate models (*table 1*) were selected to represent distinct and feasible biological realities. Their justifications are derived from either expectations generated by results from previous measurements or other studies, or from expected growth responses to the environmental gradients generated by the range of gap sizes, i.e., growth responses to changes in light and soil moisture availability. Candidate models were also both relatively parsimonious and had an implication for management. In other words, bona fide models (*sensu* Johnson and Omland 2004) had few parameters in order to maximize their application elsewhere, and they had realistic potential for guiding management decisions about appropriate opening sizes for meeting given objectives. Our inference, therefore, directly corresponds to the ranking of models and their associated strengths of evidence, given the data and set of models considered. Because of the small sample size (n=12), the number of candidate models was limited to four (Burnham and Anderson 2002).

We use the concept of shade tolerance to build a priori expectations for the model selections, and thereby assess the practical value of shade tolerance in predicting species' growth responses within gaps. The shade tolerance concept is widely used in categorizations of species into successional niches, but its significance has been criticized because the concept fails to incorporate drought tolerances that are usually not correlated with shade tolerances (Coomes and Grubb 2000). Those species considered to be highly responsive to varying magnitudes of light levels with respect to growth (giant sequoia and ponderosa pine [McDonald 1976, Schubert 1962]) are expected to conform to a power function, responding steeply and monotonically to the increased light levels across the range of gap sizes. The species less sensitive to light availability (white fir and incense-cedar [Minore 1988]) are expected to fit the more parsimonious models (power or asymptotic) with flat curves. The intermediate species (Douglas-fir and sugar pine [Oliver and Dolph 1992]) are expected to be relatively sensitive to gap size around a narrow range, corresponding to a logistic fit.

To rank the models according to goodness of fit, while penalizing for model complexity, we used a modified Akaike's information criterion (AIC), derived by Sugiura (1978). The modified AIC incorporates a bias-correction term to account for small sample:parameter ratios among the alternatives. Our model alternatives have ratios of 12:2 for the asymptotic and power functions, and 12:3 for the quadratic

Table 1 - *A priori model alternatives and their implications for the relationship between mean tree height within openings and opening size.*

Model alternative	No. of parameters	Biological implication	Management implication
1. Asymptotic (Michaelis-Menten)	2	Heights increase with opening size and then level off above a certain opening size.	Above a threshold, increases in opening size return comparatively little in terms of increased height growth.
2. Quadratic	3	Heights increase with opening size and then decrease in the larger opening sizes.	Larger opening sizes can have a negative effect on height growth.
3. Logistic	3	Heights rapidly increase above an opening size threshold and then level off.	Height is very sensitive to opening size around a narrow range. Below a threshold, severe height suppression occurs.
4. Power	2	Heights increase monotonically, but the rate of increase diminishes across the range of opening sizes.	Seedling height is maximized in the largest opening size, although returns in height are diminishing.

and sigmoidal functions. The differences in AIC values are used to assess the level of empirical support, where differences of less than two are considered to have substantial support (Burnham and Anderson 2002). To evaluate candidate models in relation to the highest ranked model, the AIC values are transformed to Akaike weights and normalized to sum to one. The weights are interpreted as the likelihood that within the limits of the data and the set of alternatives, the given model is the most appropriate choice. The application of AIC for statistical inference in ecological studies is described in detail by Anderson et al. (2000) and Johnson and Omland (2004).

Results

No effect of gap size on % mortality at the gap level was detected for any of the species. High variability in survival between the 12 gaps, however, caused power of detection to be low (<0.27). Despite gap-to-gap variability, there were clear differences in mean % mortality per gap between the species when comparing means and confidence intervals. Giant sequoia had the overall lowest mortality (CI95 percent = 2.4 - 5.0 percent). Sugar pine (CI95 percent = 6.6 - 12.4 percent), incense-cedar (5.8 - 14.2 percent), ponderosa pine (8.9 - 18.4 percent), and Douglas-fir (9.1 - 18.9 percent) all had comparable mortality levels. White fir had considerably higher mortality (35.7 - 47.2 percent).

For all species, height increased with gap size (*fig. 1*). An asymptotic fit was the highest ranked model for every species except Douglas-fir, which had the most support for a power function and a low level of support for an asymptotic model (*table 2*). For four of the six species, there was parity among the candidate models

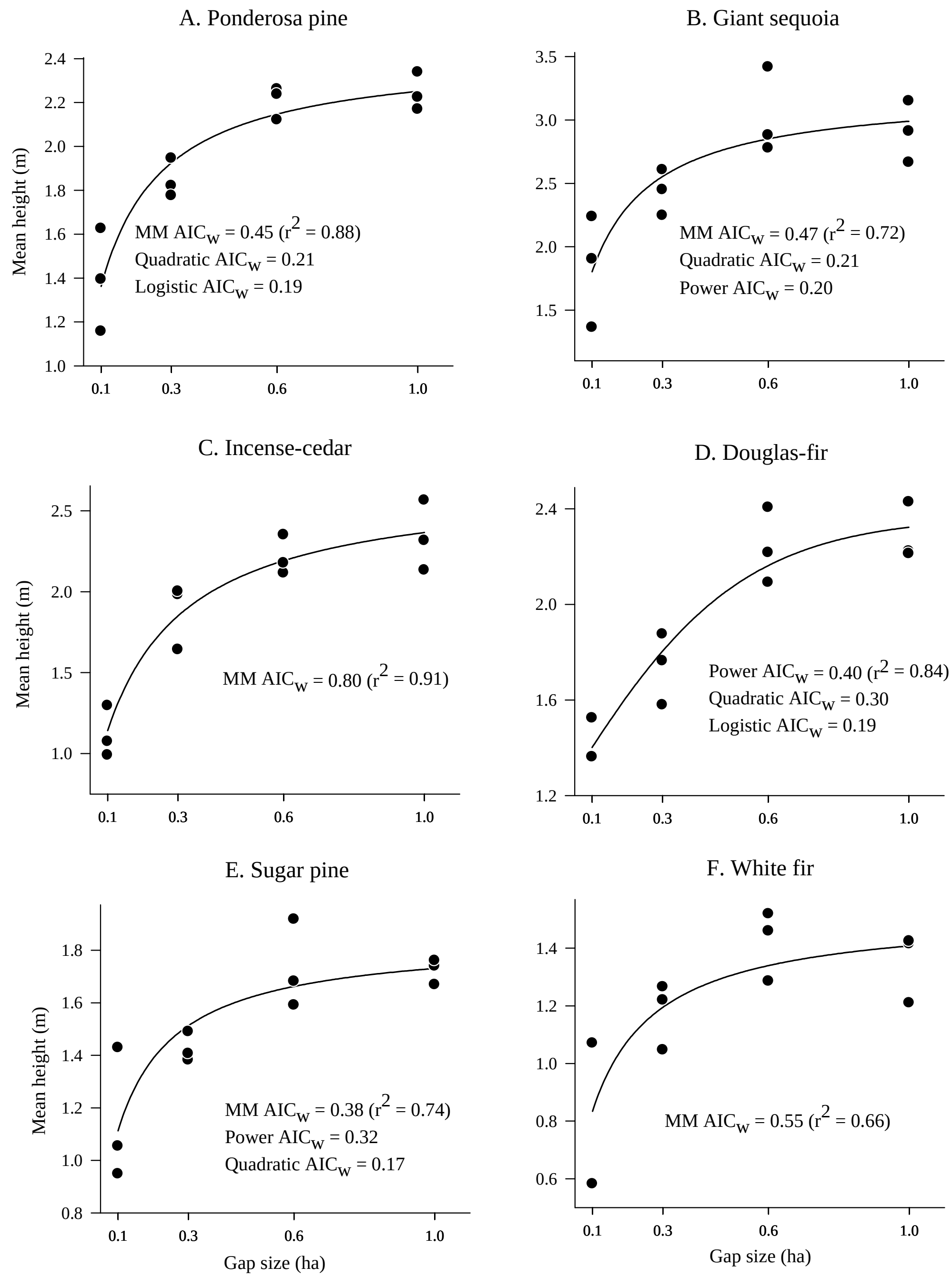

Figure 1 - Year 7 functional height responses to gap size in experimental gaps at Blodgett Forest, CA. Curves are shown for the highest ranked models. Akaike Information Criteria weights (AICw) are given for models with substantial empirical support (MM = Michaelis-Menton asymptotic curve). Y-axes are presented at different scales to display species-specific patterns.

for selecting a discriminating pattern of influence of gap size on growth, and no single model was consistently ruled out as a plausible alternative across all species. Douglas-fir, giant sequoia, ponderosa pine, and sugar pine growth patterns each had substantial support for three of the four models considered (*table 2*). Incense-cedar and white fir model selections had relatively strong support for an asymptotic fit.

Table 2--*Model ranks using Akaike Information Criteria. K_i= number of parameters in model ranked i; AIC_i= Akaike Information Criteria; Δ_i = absolute difference (AIC_i – $AIC_{i=1}$); w_i = Akaike weight (relative likelihood of model given the data and other candidate models.)*

Model ranks	K_i	AIC_i	Δ_i	w_i	Ratio of ranks, w_1/w_i
Douglas-fir					
1. Power	2	63.7	0.0	0.40	
2. Quadratic	3	64.3	0.6	0.30	1.3
3. Logistic	3	65.2	1.5	0.19	2.1
4. Asymptotic	2	66.4	2.7	0.10	4.0
Giant sequoia					
1. Asymptotic	2	85.6	0.0	0.47	
2. Quadratic	3	87.2	1.6	0.21	2.2
3. Power	2	87.3	1.7	0.20	2.4
4. Logistic	3	88.3	2.7	0.12	3.9
Incense-cedar					
1. Asymptotic	2	70.0	0.0	0.80	
2. Logistic	3	73.7	3.7	0.13	6.2
3. Quadratic	3	76.0	6.0	0.04	20.0
4. Power	2	76.3	6.3	0.03	26.6
Ponderosa pine					
1. Asymptotic	2	65.6	0.0	0.45	
2. Quadratic	3	67.1	1.5	0.22	2.1
3. Logistic	3	67.4	1.8	0.19	2.4
4. Power	2	68.0	2.4	0.14	3.2
Sugar pine					
1. Asymptotic	2	68.7	0.0	0.38	
2. Power	2	69.0	0.3	0.32	1.2
3. Quadratic	3	70.2	1.5	0.17	2.2
4. Logistic	3	70.9	2.2	0.13	2.9
White fir					
1. Asymptotic	2	64.5	0.0	0.55	
2. Power	2	66.9	2.4	0.17	3.2
3. Quadratic	3	67.0	2.5	0.17	3.2
4. Logistic	3	67.7	3.2	0.11	5.0

Discussion

Ostensibly, gap based silviculture creates steep resource gradients within gaps that can successfully regenerate a wide variety of tree species. Reluctance by managers to accept this concept, however, arises mainly from uncertainty with the ability of shade intolerant species to survive in gaps that are partially shaded by the surrounding matrix forest. In this study, there was no alignment of overall survival with shade tolerance rankings. In fact, the shade-intolerant giant sequoia had the best survival, while shade tolerant white fir had the poorest survival. With some extra planting effort, we indeed successfully regenerated all six species. By planting two individuals at each planting spot, and by transplanting seedlings from nearby reserve areas where both seedlings had died, nearly every planting spot (>95 percent) had a live individual by year three when the first measurements were taken.

The lack of detectable effect of gap size on mortality contrasts with the marked effect of gap size on growth. Shade tolerance rankings helped little in predicting survival by gap size. The mortality levels that we found in these gaps is similar to our observations in larger plantations at Blodgett Forest Research Station. Hence, planted seedling mortality appears to be a species-specific trait, rather than an effect of gap size in gaps above 0.1 ha in this study area.

Height growth responses to gap size consistently diminished as gap size increased, typically leveling off or decreasing in rate beyond a size range from 0.3 to 0.5 ha. Early height growth suppressions could have therefore been avoided in this case with a gap regime consisting of gap sizes above this size range. Because the absolute differences in AIC values were not large (*table 2*), the choice of models can generally be judged to be appropriate. Burnham and Anderson (2002) subjectively suggest an absolute difference of less than two as providing "substantial" empirical evidence for an appropriate model. Using this threshold, every model was appropriate at least twice across the six species. At the same time, the near-unanimity of the asymptotic fit as the selected model (when testing the models against each other and the ruling out of other models in incense-cedar and white fir) give support to the primacy of the asymptotic pattern. Despite testing of the asymptotic fit against "better" models compared to the year-5 analysis, the asymptotic fit continues to be an appropriate quantitative description of the effect of gap size on tree growth.

The resulting model rankings and their strengths of evidence run largely counter to expectations derived solely from tolerance rankings. Primacy of the asymptotic fit over other candidate models was evident for both white fir and incense-cedar. Some ambivalence between quadratic and asymptotic models was expressed after the fifth year for white fir (York et al. 2004), but a resolution of pattern appears to be occurring by the seventh year. Although the asymptotic fits were expected according to their shade tolerance classifications, both species were surprisingly sensitive to gap size in terms of absolute growth. White fir and incense-cedar were the most sensitive to gap size in terms of relative change in stature between the smallest and largest gap sizes. In other words, the functional role of gap size (suppression in small gap sizes, followed by a saturating effect in larger sizes) was consistent with expectations for shade tolerant species. But the magnitudes of the observed pattern's parameters (steep slope and large asymptote compared to y-intercept) were not expected for the shade tolerant species.

For giant sequoia, there was twice as much strength of evidence for an asymptotic model than the next closest model (quadratic). Nevertheless, the quadratic and power models had enough strength of evidence to make it difficult to rule out their plausibility in contributing to the observed pattern, especially in the larger gap sizes. Whether the 0.6 ha size is a leveling-off point (asymptotic), maximum (quadratic), or mid-point (power) would be more clear with incorporation of larger gap sizes. Competition from surrounding gap border trees is certainly an influence on overall giant sequoia height, as its sensitivity to reductions in both soil moisture and direct light availability effectively reduce a large portion of the gap area where maximum growth occurs (York et al. 2003). Within-gap edge zones reduce giant sequoia growth on both pole-facing (light and water limiting) and equator facing (water limiting) edges. This co-limitation in giant sequoia is in contrast to ponderosa pine, which tends to partition growth along a single light gradient. The area where rapid growth occurs for ponderosa pine is expanded closer to equator-facing edges of gaps where direct light tends to be relatively abundant in small gaps of temperate forests (Canham et al. 1990, Minkler et al. 1973).

The power function, predicted to fit well for ponderosa pine because of rapid increases in growth responding to higher levels of light, was instead the lowest ranked model. The predicted model for sugar pine, a logistic fit, was likewise ranked last. Given the data, and because of the high degree of plausibility of each candidate model, a best model could not easily be distinguished for the two pine species. Functionally, however, they are similar in terms of biological and management implications. For both species, height growth diminishes considerably near 0.3 ha and does not increase monotonically. Unlike the other species, the least amount of support was given to an asymptotic fit for Douglas-fir. Height increased sharply between 0.3 and 0.6 ha, dividing the gap sizes into two classes and most clearly defining a size threshold where significant height suppression can be avoided. However, more data from smaller gap sizes is necessary to test whether the threshold is distinct enough to result in a logistic fit as an appropriate model.

Many considerations besides maximization of growth will contribute to the decision of gap size and density in a gap-based silvicultural regime. A diversity of practice in gap size creation should indeed be central in achieving the purpose of structural diversity across forests. The patterns that we found are expected to vary by latitude, gap shape, matrix disturbance history, and time since gap creation. Tracking growth in this study over time and implementing similar studies elsewhere that incorporate both seedling and matrix growth may help in describing commonalities and differences in patterns across various forest types.

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Effects of Alternative Treatments on Canopy Fuel Characteristics in Five Conifer Stands[1]

Joe H. Scott[2] and Elizabeth D. Reinhardt[3]

Abstract

A detailed study of canopy fuel characteristics in five different forest types provided a unique dataset for simulating the effects of various stand manipulation treatments on canopy fuels. Low thinning, low thinning with commercial dbh limit, and crown thinning had similar effects on canopy bulk density (*CBD*) and canopy fuel load (*CFL*), but only the strict low thinning significantly affected canopy base height (*CBH*). In four of five sampled stands, *CBD* and *CFL* responded linearly to increasing treatment intensity in those three thinning treatments. The ponderosa-pine/Douglas-fir stand, with its significant understory component, showed little change in *CBD* with the commercial limit and crown thinning treatments. The diameter-limit harvest exhibited little consistency among sites and, hence, it is not a good silvicultural tool for creating canopy fuel reduction prescriptions. Due to fire-induced mortality, crown scorch (from prescribed fire) was more effective than mechanical pruning (to an equivalent height) at modifying canopy fuel characteristics. At achievable scorch and pruning heights, neither treatment had a significant effect on *CBD* or *CFL*.

Introduction

Silviculturists are frequently asked to manipulate stand structure to meet fire and fuel management objectives, including mitigation of crown fire potential. As summarized by Graham et al. (2004), effective strategies for reducing crown fire occurrence and severity include reducing surface fuels (Biswell 1960, Pollet and Omi 2002), increasing canopy base height (Agee 2002, Schmidt and Wakimoto 1988), and reducing canopy bulk density (Agee 1996, Scott 1998).

The wide variety of available treatment types and intensities, coupled with a wide array of initial stand structures, makes development of a single, uniformly effective treatment impractical (Graham et al. 1999). This paper summarizes detailed canopy biomass measurements in various ways to simulate of the effects of several possible silvicultural treatments (thinning, pruning, and prescribed fire) on canopy fuel characteristics.

Method

Scott and Reinhardt (2002, 2005) measured individual-tree and plot-level canopy fuel characteristics in five coniferous stands in the western U.S., each in a different forest type:

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[1] A version of this paper was presented at the National Silviculture Workshop, June 6-10, 2005, Tahoe City, California.

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- ponderosa pine/Douglas-fir (PPDF)
- ponderosa pine (PP)
- Douglas-fir (DF)
- lodgepole pine (LP)
- Sierra Nevada mixed conifer (SNMC)

The previous canopy fuel study publications report only stand-level summaries. For this analysis, we utilized unpublished tree-level summaries gathered during the same field study. We used the following tree characteristics data to simulate various treatments at each site:

- Species
- Diameter at breast height
- Canopy fuel mass by 1-m height increments
- Number of trees per acre represented by each sample tree

In addition, crown ratio and crown class (crown position) were used in the simulation of crown thinning.

Canopy fuel mass is the oven-dry mass of fuel available to burn in the flaming phase of a crown fire. Only very fine fuel is consumed in the short duration of a crown fire. Van Wagner (1977) assumed foliage was the only available canopy fuel component when computing mass flow rate on an experimental fire. Fine branches may also be consumed in the flaming portion of a crown fire. Others add a portion of the fine branch mass to the foliage (Brown and Bradshaw 1994, Brown and Reinhardt 1991, Call and Albini 1997). In this analysis, canopy fuel is assumed to include the foliage, 0 to 3 mm diameter live branchwood, and 0 to 6 mm diameter dead branchwood. Canopy fuel mass by 1m height layer for each tree was estimated by (1) removing and measuring individual branches in 1m height increments, (2) sorting, drying, and weighing a sub-sample of those branches, (3) developing regression equations for estimating branch biomass by size class and component, and (4) applying those regressions to every measured branch (Scott and Reinhardt 2005). Vertical profiles of canopy fuel mass in each 1m height layer (canopy bulk density) illustrate differences in initial stand condition among the five sites (*fig. 1*).

All study plots were fixed-radius (either 10 or 15m radius), so the number of trees per acre represented by each sample tree in the plot is simply the inverse of plot area.

Canopy Characteristics

Three plot-level canopy fuel characteristics were estimated from the dataset: canopy fuel load, canopy bulk density, and canopy base height.

Canopy fuel load (*CFL*) is the canopy fuel mass per unit ground area. We estimated *CFL* by dividing the sum of canopy fuel mass over all trees (all height increments) by the horizontal plot area. Canopy fuel load is not currently used to predict the occurrence of crown fire, but is used to predict the intensity of a crown fire in some fire behavior simulation systems (for example, Finney 1998, Scott 1999).

Canopy bulk density (*CBD*) is the canopy fuel mass per unit canopy volume (Scott and Reinhardt 2001). In this analysis, *CBD* is estimated as the maximum 3m deep running mean from the *CBD* profile (Scott and Reinhardt 2005). Canopy bulk density is important in modeling the occurrence of active crown fires (Wan Wagner

1977), and, in some fire models, crown fire spread rate (Albini 1996, Butler et al. 2004, Cruz et al. 2005).

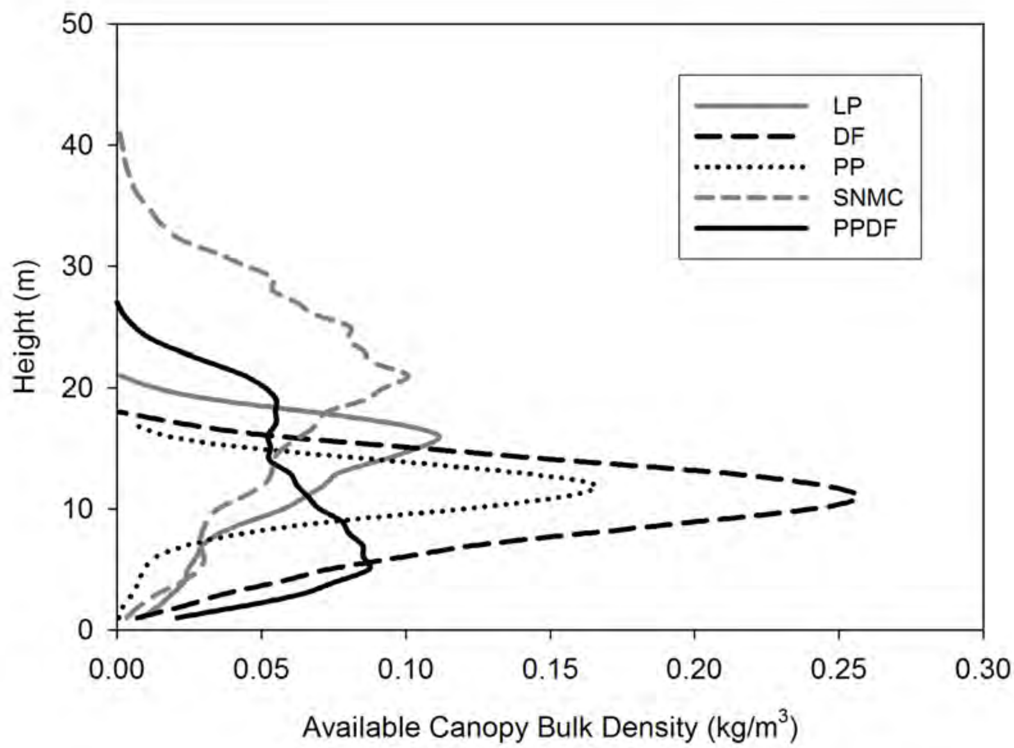

Figure 1—Pre-treatment vertical canopy fuel profiles (3-m running mean) for five conifer stands. Available canopy bulk density (*CBD*) includes the mass of foliage, 0 to 6 mm diameter dead branch material, and 0 to 3 mm live branch material. A single effective value of *CBD* for each stand is defined as the maximum 3-m running mean. Canopy base height is defined as the height at which *CBD* exceeds 0.011 kg/m^3. Canopy fuel load is represented by the area "under" (to the left of) each curve. Stands are labeled: LP = lodgepole pine; DF = Douglas-fir; PP = ponderosa pine; SNMC = Sierra Nevada Mixed Conifer; PPDF = ponderosa pine.

Canopy base height (*CBH*) is defined here as the lowest height above the ground at which there is sufficient available canopy fuel to propagate fire vertically through the canopy (Scott and Reinhardt 2001). Using a method adapted from Sando and Wick (1972), *CBH* is calculated as the lowest height above the ground at which at least 0.011 kg/m^3 of available canopy fuel was present (Reinhardt and Crookston 2003), using a 3m deep running mean to smooth observed values. Canopy base height is important in modeling the transition from surface fire to some kind of crown fire (Cruz et al. 2004, Van Wagner 1977).

Alternative Silvicultural Treatments

Six silvicultural treatments were simulated in each of the five stands:

- Low thinning to a target residual basal area (BA)
- Low thinning to a target residual BA, with a commercial dbh limit
- Crown thinning (high thinning) to a target residual BA

- Diameter-limit cutting to specified dbh
- Mechanical pruning
- Scorch from prescribed fire with resulting mortality

Low thinning is removal of trees from the lower crown classes to favor those in the upper crown classes. We simulated low thinning by removing trees strictly by dbh, with no consideration for crown class, crown ratio, or spacing. The first low thinning treatment is applied to all trees in the plot without regard for a tree's commercial value, thus, all small trees, regardless of commercial value, are removed. In the second low thinning treatment, we applied a commercial (merchantable) diameter limit (dbh below which the direct costs of harvesting exceeds the commercial value of merchantable material). We varied commercial limit among stands to reflect differences in species composition and associated markets: 10” dbh in the SNMC stand, 7” dbh in the PPDF, DF, and PP stands, and 5” dbh in the LP stand. Removal of all merchantable trees from a stand is an “economic clearcut”, not a thinning. We simulate the full range of treatment intensity for academic curiosity, not because it is a suggested or common practice.

Crown thinning is the removal of trees from the dominant and codominant crown classes in order to favor the *best* trees of those same classes. We simulated a crown thinning by removing dominant and codominant trees in order of increasing live crown ratio. That is, dominant and codominant trees with low live crown ratios (poor quality) were removed first. A commercial limit was not applied to the crown thinning, because even poor quality dominant and codominant trees are usually of merchantable size. We simulated crown thinning through to its endpoint--removal of *all* dominant and co-dominant trees--even though the result (leaving only suppressed and intermediate trees) is not a crown thinning at all.

Diameter-limit cutting is the removal of all trees below a specified dbh. (Diameter-limit cutting can also be applied as the removal of trees *above* a specified dbh, and often is restricted to removal of only merchantable trees.) We simulated diameter-limit cutting by removing all trees *smaller* than a specified dbh, without regard for a commercial limit.

Pruning is the removal of live or dead branches from a standing tree. In timber applications, pruning is done to improve wood quality; in urban forest applications, pruning is done to improve aesthetics or tree health; in wildland fire applications, pruning is done to separate surface and canopy fuels (increase *CBH*). We simulated pruning by progressively removing from the dataset the canopy fuel mass in the lowest layers of the canopy fuel profile, but otherwise we left the trees in the treelist. No regard was given to leaving a minimum crown length on pruned trees--each tree is pruned until its crown is gone. Using the *CBD* calculation method used in this analysis, pruning can only affect canopy bulk density if pruning height exceeds the height of maximum bulk density (Scott and Reinhardt 2005). The fuel mass of any given branch was assigned to the 1m layer in which it is attached to the bole, with no accounting for branch angle. Because most branches near the bottom of the crown tend to angle downward, this analysis tends to over-predict the effect of pruning on *CBH*, especially in open stands containing large-crowned trees with branches near the ground. All five of the stands used here were closed-canopy, so this potential for over-prediction is minimized.

Crown scorch is needle death due to convective heat above a wildland fire. Height of crown scorch is a function of in-stand wind speed and intensity of the surface fire (Van Wagner 1973). By controlling fireline intensity in relation to in-stand wind speed, prescribed fire managers can control scorch height. Scorched branches are assumed to contribute to canopy fuel mass. In reality, several months to a few years may pass before scorched foliage and fine branches fall to the ground and are no longer available for a crown fire. However, the time period during which we may overestimate the effects of scorch on canopy fuels corresponds to a period of little potential for surface fire. Scorch from prescribed fire was simulated by progressively removing from the dataset the biomass in the lowest layers of the canopy fuel profile, just as we did for mechanical pruning. In addition, we simulated fire-caused tree mortality by computing the probability of tree mortality based on scorch height in relation to tree height, crown length, and bark thickness (Ryan and Reinhardt 1988).

The equations for predicting tree mortality are logistic. The result they give is a probability of mortality. To use the probability of mortality in our analysis, we simulated mortality in a manner similar to that used in FFE-FVS (Reinhardt and Crookston 2003). Canopy fuel mass for each tree (at each 1m height layer) was reduced by the tree's probability of mortality. The resulting simulations represent the expected value of canopy fuel.

Results and discussion

Results are presented as a series of charts that show the effect of treatment intensity on canopy characteristics for each of the three canopy fuel characteristics. Each figure displays results for all five sample stands and all three canopy fuel characteristics; there is one figure for each treatment.

Low Thinning to Target Residual Basal Area

Canopy bulk density (*CBD*) was linearly related to residual basal area (*fig. 2a*). In fact, despite the wide range of initial stand structure and composition, four of the five sites exhibited similar *CBD* at a given level of residual basal area. For example, with 100 ft^2/ac of basal area remaining after removing trees from below, *CBD* at all but the DF site ranged from 0.06 to 0.07 kg/m^3. *CBD* is strongly linearly related to residual BA, even at the DF site, but the value of *CBD* was much different (*CBD* at the DF site was 0.18 kg/m^3 with 100 ft^2/ac of basal area remaining). The reason for this difference is not clear, but the dominance of Douglas-fir is likely a contributing factor. The fine branching and shade tolerance of Douglas-fir apparently contributed to higher canopy fuel mass per unit canopy volume.

Most stands showed an initial period during which reduction of BA from below had no effect on *CBD*. This occurred because the small trees, which were removed first, had little or no fuel mass in the critical dense canopy layer that determines *CBD*. The PPDF stand, however, exhibited a rapid initial drop in *CBD* with BA, because the critical canopy layer occurred in the predominantly Douglas-fir under- and middle-stories, whereas in the other stands it occurred higher in the canopy (*fig. 1*).

Canopy fuel load (*CFL*) also responded linearly to residual basal area, but, unlike with *CBD*, there was no other similarity in relationship among sites (*fig. 2b*). With 100 ft^2/ac of basal area remaining after removing trees from below, *CFL* varied almost four-fold, from 0.4 to 1.4 kg/m^2. The steepest initial drop in *CFL* with BA again occurred in the PPDF stand. This steep drop corresponded to removal of the under- and middle stories of this stand; slight reduction in BA in such a canopy layer had a significant effect on available canopy fuel.

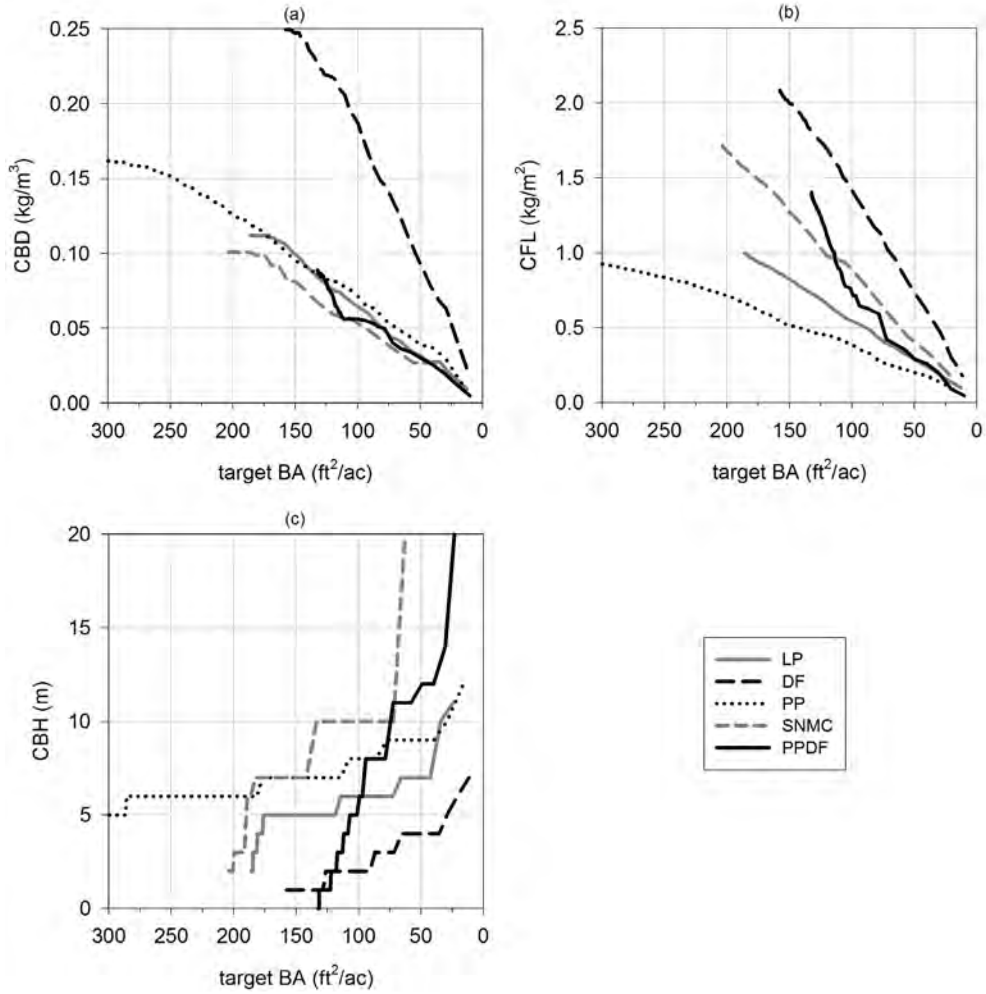

Figure 2—Response of (a) canopy bulk density (*CBD*), (b) canopy fuel load (*CFL*), and (c) canopy base height (*CBH*) to a variable-intensity **low thinning**. Stands are labeled: LP = lodgepole pine; DF = Douglas-fir; PP = ponderosa pine; SNMC = Sierra Nevada Mixed Conifer; PPDF = ponderosa pine.

The relationship between canopy base height (*CBH*) and low thinning residual BA bears none of the consistency of that for *CBD* and *CFL* (*fig. 2c*). First, *CBH* appeared as a step function, because the method we used only estimates *CBH* to the nearest meter. When *CBD* in the critical 1m layer fell below the critical value (0.011 kg/m^3), *CBH* changed abruptly to a higher layer. Nonetheless, meaningful trends emerged. The stands containing a shade-tolerant understory (PPDF and SNMC) showed an initial strong increase in *CBH* with decreasing residual BA, because removal of the understory in those stands greatly increased *CBH*. Once the understory was removed, *CBH* was determined mostly by the overstory, and the response of *CBH* was flat with decreasing residual BA until many overstory trees were removed. The PPDF stand, because it is multi-storied, showed consistent increase in *CBH* with decreasing residual BA. The PP stand, a single-cohort without an understory of any kind, showed almost no change in *CBH* with even large reductions in BA from below.

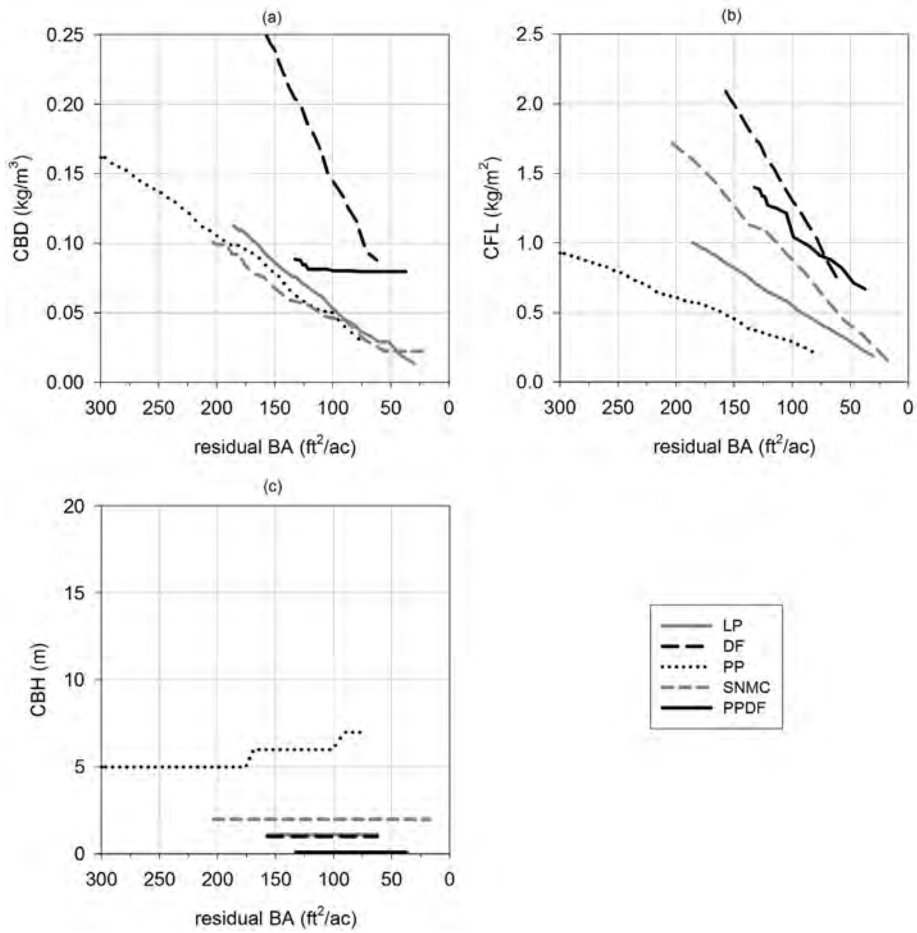

Figure 3—Response of (a) canopy bulk density (*CBD*), (b) canopy fuel load (*CFL*), and (c) canopy base height (*CBH*) to a variable-intensity **low thinning with commercial limit**. Stands are labeled: LP = lodgepole pine; DF = Douglas-fir; PP = ponderosa pine; SNMC = Sierra Nevada Mixed Conifer; PPDF = ponderosa pine. Commercial d.b.h. limit varied among stands to reflect local conditions: 10" dbh in the SNMC stand, 7" dbh in the PPDF, DF, and PP stands, and 5" in the LP stand. Trees smaller in diameter than the commercial limit were retained.

Low thinning to target residual basal area, with commercial limit

The line for each stand begins at the BA and canopy fuel characteristic corresponding to the initial condition, and ends at the BA and canopy fuel characteristic corresponding to removal of all merchantable trees. In four of the five stands, applying a commercial limit did not significantly change the response of *CBD* to residual BA. All showed linear response and similar slope as the strict low thinning (*fig. 3a*). In the PPDF stand, however, *CBD* was nearly unchanged, even after the entire overstory was removed, because the critical dense layer occurred in the layer comprised of the sub-merchantable trees.

Response of *CFL* to residual BA was similar with and without the commercial limit in all stands. The biggest change again occurred in the PPDF stand, whose response to BA was less steep with the addition of the commercial limit. This occurred because the non-commercial trees are composed almost exclusively of

Douglas-fir, whereas the commercial-sized trees are a mixture of Douglas-fir and ponderosa pine. The foliage and fine branching of Douglas-fir apparently give it a higher canopy fuel mass per unit of BA than ponderosa pine.

The largest effect of adding a commercial limit to the low thinning occurred for *CBH*. Almost no amount of thinning increased *CBH* if the non-commercial trees were left (*fig. 3c*). In fact, four of the five stands showed no change in *CBH* even after all merchantable trees were removed, because those trees had enough canopy fuel mass to maintain the critical density that determines *CBH*. Removing larger trees did not remove canopy fuel mass from the low canopy layers. Only the PP stand showed any increase in *CBH* with this treatment, but less so than without the commercial limit.

Crown Thinning

Crown thinning had a similar effect on *CBD* as the low thinning with commercial limit—a linear response of *CBD* with respect to residual BA in all but the PPDF stand, which exhibited no change in *CBD* even with removal of all dominant and co-dominant trees (*fig. 4a*). Just as for the commercial low thinning treatment, this result occurred because the critical dense canopy layers occurred in the Douglas-fir under- and middle-stories, which was composed of suppressed and intermediate crown classes, and because the dominant and codominant trees did not have significant canopy fuel mass in those critical layers.

Crown thinning also had a similar effect on *CFL* as low thinning with commercial limit--only the PPDF stand was different than strict low thinning. With a strict low thinning, the PPDF stand exhibited strong sensitivity of *CFL* to initial reduction in BA (*fig. 4b*), because the smallest trees in the stand contained a large proportion of the total canopy fuel mass. In contrast, crown thinning removed the larger trees from the stand--primarily ponderosa pine--which contained a smaller portion of the total *CFL* than the small understory trees. Therefore, the response of *CFL* to BA reduction in the crown thinning was initially weak, and strengthened only as trees with longer crowns--Douglas-fir in the PPDF stand--were eventually removed.

As with low thinning with a commercial limit, crown thinning had very little effect on *CBH* (*fig. 4c*). In the DF, SNMC, and PPDF stands, CBH did not change even after all dominant and codominant trees were removed. In the PP and LP stands, CBH increased only after nearly all of the dominant and codominant trees had been removed, and then only slightly.

Diameter Limit Cutting

Results of the diameter-limit cutting simulation showed none of the consistency of low thinning in its effect on *CBD* (*fig. 5a*). In contrast to low and crown thinning, diameter-limit cutting did not show a linear relationship between maximum dbh of harvested tree and *CBD*. All sites except PPDF showed no reduction in *CBD* until trees greater than about 5 in. dbh had been removed. In contrast, *CBD* was reduced with removal of even small diameter trees in the PPDF stand, because those trees contributed to the critical dense layers of the canopy. Once *CBD* began to drop in relation to diameter, it did so quickly in all but the PPDF and SNMC sites, for which reducing *CBD* required removal of the large-diameter overstory trees.

Results of the diameter-limit cutting simulations on *CFL* were also quite different than low and crown thinning, showing no clear linear trends or consistencies

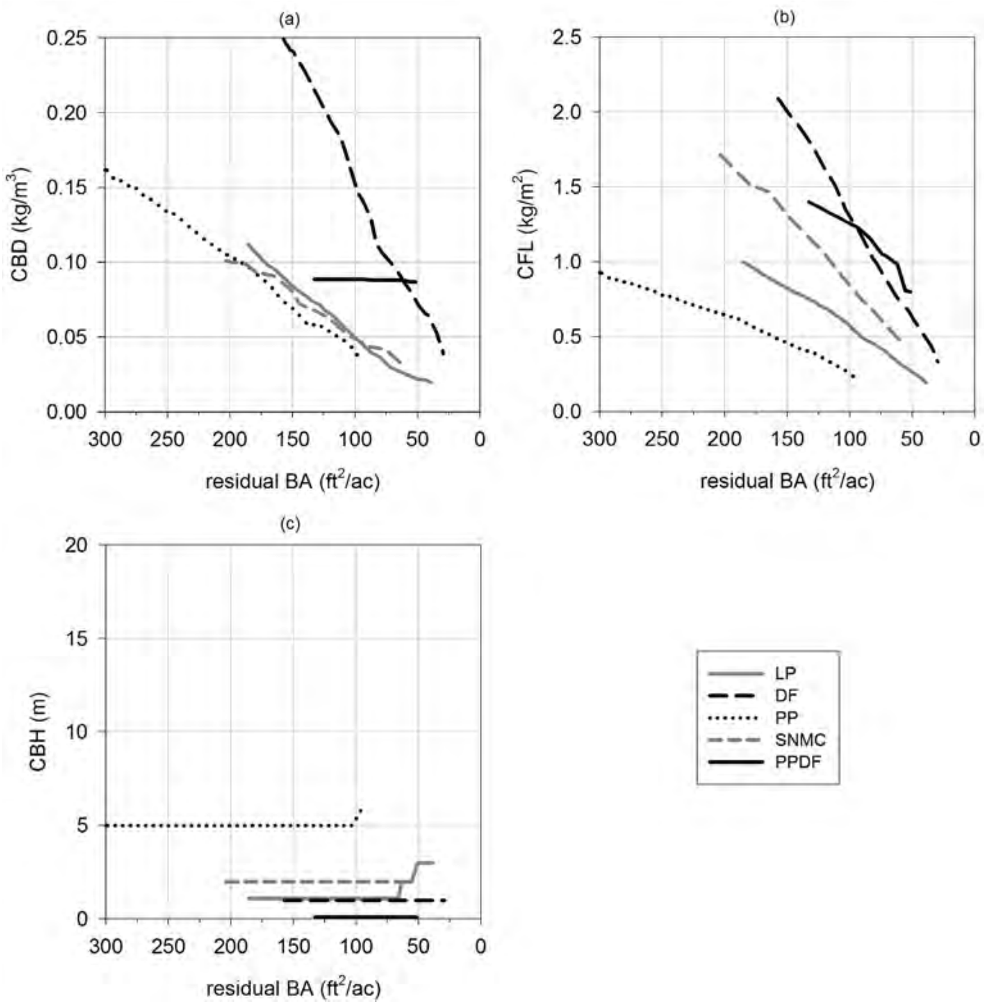

Figure 4—Response of (a) canopy bulk density (*CBD*), (b) canopy fuel load (*CFL*), and (c) canopy base height (*CBH*) to a variable-intensity **crown thinning**. Stands are labeled: LP = lodgepole pine; DF = Douglas-fir; PP = ponderosa pine; SNMC = Sierra Nevada Mixed Conifer; PPDF = ponderosa pine. Crown thinning is removal of trees in the dominant and codominant crown classes in order to favor the best trees of those same classes.

(*fig. 5b*). Just as for *CBD*, *CFL* dropped most quickly in the PPDF stand, indicating the influence of small diameter trees at that site and their significant contribution to *CFL*. Other sites showed little effect of removing small-diameter trees, because at those sites the small trees did not contain a significant fraction of the total canopy fuel mass in the stand.

The effect of diameter limit on *CBH* displayed more apparent consistency than low thinning--as the diameter limit increased, so too did the resulting *CBH* (*fig. 5c*). Nonetheless, the wide range of *CBH* at a given diameter limit makes generalization impractical. At a diameter limit of 5 inches dbh, *CBH* varied from 1 to 6m among the sites; at 10 inches dbh, *CBH* ranged from 4 to 8 m; at 15 inches, *CBH* ranged from 8 to 12m.

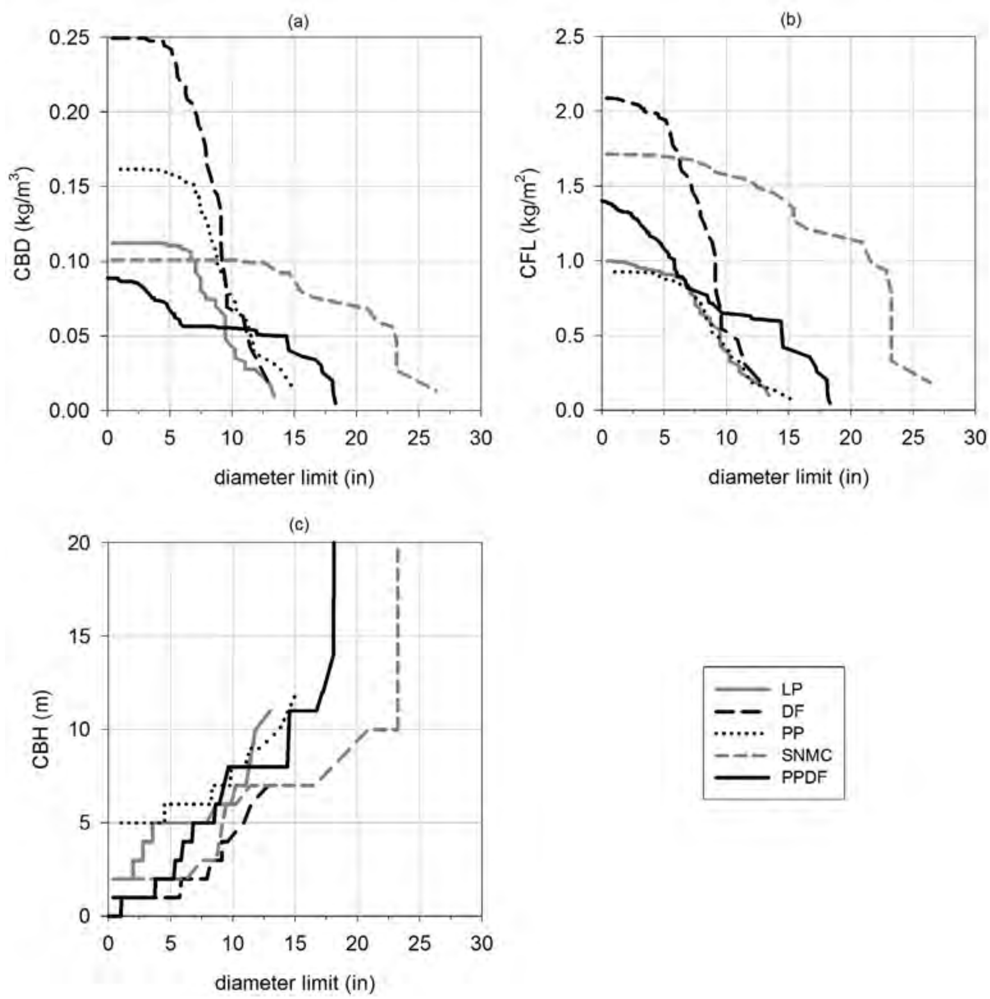

Figure 5—Response of (a) canopy bulk density (*CBD*), (b) canopy fuel load (*CFL*), and (c) canopy base height (*CBH*) to a variable-intensity **diameter-limit harvest**. Stands are labeled: LP = lodgepole pine; DF = Douglas-fir; PP = ponderosa pine; SNMC = Sierra Nevada Mixed Conifer; PPDF = ponderosa pine. All trees less than the diameter limit were removed.

Mechanical Pruning

Pruning to a prescribed height had predictably little effect on *CBD* (*fig. 6a*). Pruning did not have an effect on *CBD* unless pruning height approached the critical dense layers of the canopy. Therefore, the PPDF site, for which the densest canopy layers occurred nearest the ground due to the Douglas-fir under- and middle-stories, showed the earliest effect of pruning on *CBD* (that is, at the lowest pruning height values). Pruning heights above 3m are impractical to apply to all trees in a stand, therefore, mechanical pruning has no practical effect on *CBD*.

Any amount of pruning removes fuel mass from the canopy. However, the lowest layers of the canopy do not generally contain a significant portion of the total canopy fuel load (*fig. 6b*), the exception being stands with significant understories. For example, the greatest reduction in *CFL* at a pruning height of 3m occurs at the PPDF site because of its significant Douglas-fir understory.

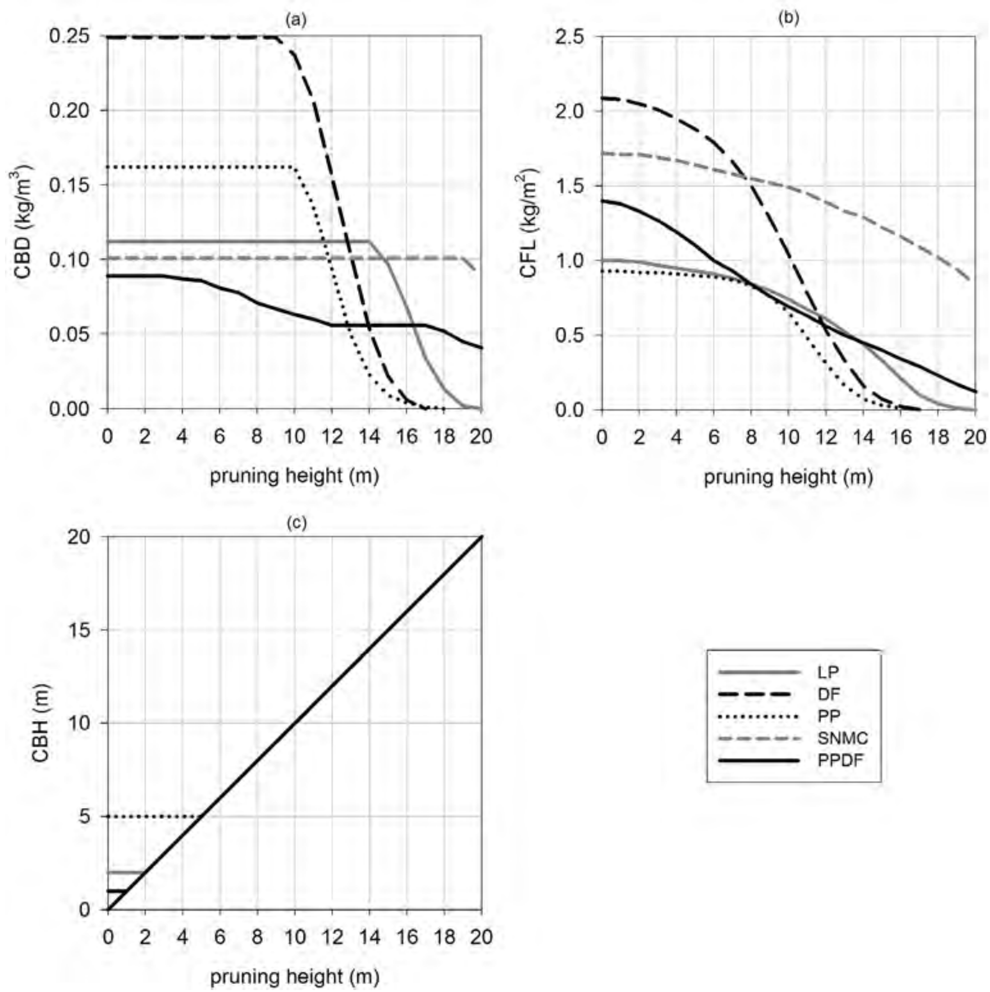

Figure 6—Response of (a) canopy bulk density (*CBD*), (b) canopy fuel load (*CFL*), and (c) canopy base height (*CBH*) to a variable-intensity **mechanical pruning**. Stands are labeled: LP = lodgepole pine; DF = Douglas-fir; PP = ponderosa pine; SNMC = Sierra Nevada Mixed Conifer; PPDF = ponderosa pine.

The effect of pruning on *CBH* was quite predictable (*fig. 6c*). Because pruning by definition removes all available canopy fuel below the pruning height, *CBH* must always be greater than or equal to pruning height. Pruning had no effect if *CBH* already exceeded pruning height. For example, initial *CBH* in the PP stand was 5m because the high stand density caused crowns to recede and prevented understory trees from establishing, so that there was not enough canopy fuel to meet the 0.011 kg/m^3 *CBH* threshold until that height. Pruning to heights below 5m therefore had no effect on *CBH* in the PP stand, while pruning to heights above 5m increased *CBH* directly.

Scorch from Prescribed Fire with Resulting Mortality

The effect of scorch height on canopy fuel characteristics is expected to be similar to that of mechanical pruning, but with one important difference: crown scorch and related fire influences of the fire can cause tree death, leading to a further reduction of canopy. If there were no resulting mortality, scorch and pruning would have the same effect as simulated in this analysis. The difference between the crown scorch and mechanical pruning simulations is entirely due to mortality.

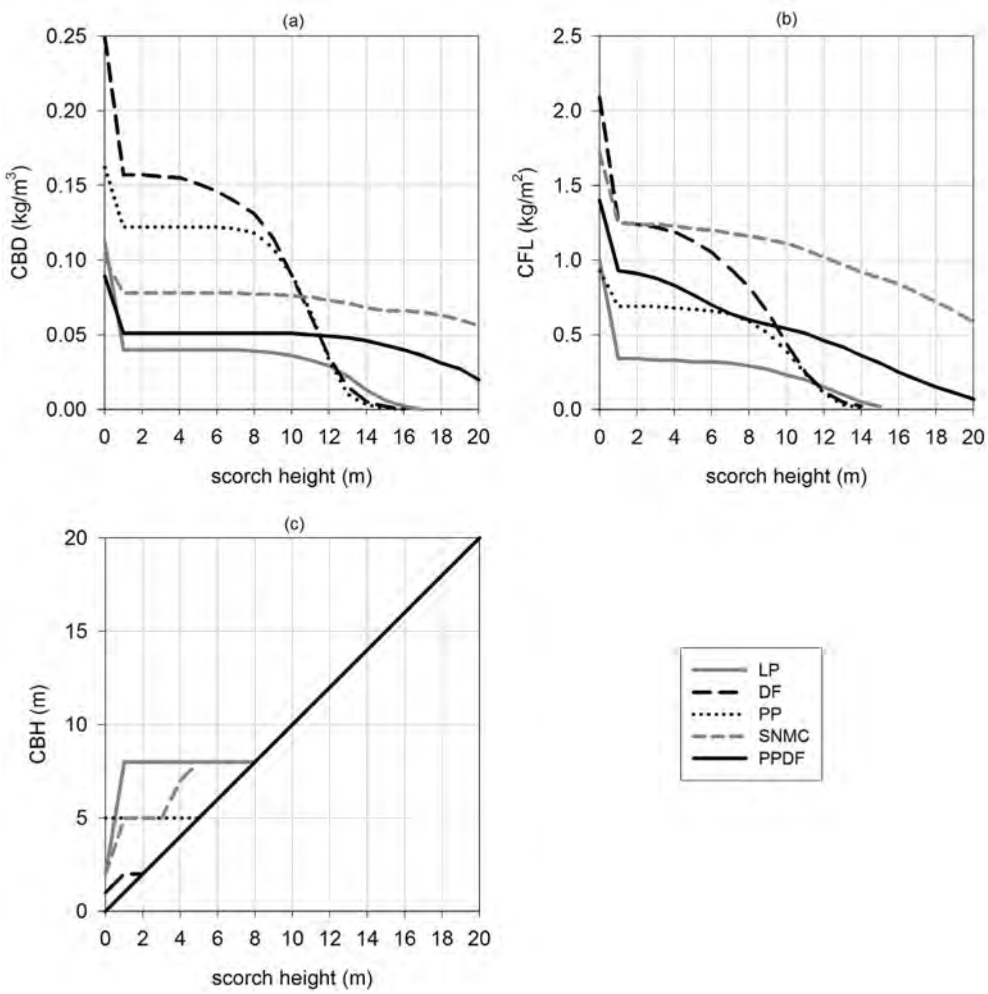

Figure 7— Response of (a) canopy bulk density (*CBD*), (b) canopy fuel load (*CFL*), and (c) canopy base height (*CBH*) to various heights of **crown scorch**. Stands are labeled: LP = lodgepole pine; DF = Douglas-fir; PP = ponderosa pine; SNMC = Sierra Nevada Mixed Conifer; PPDF = ponderosa pine. Response includes scorch-induced mortality.

All stands exhibited a drop in *CBD* with crown scorch height of just 1m, followed by a range (up to about 8m scorch height) in which increasing scorch height did not significantly reduce *CBD* (*fig. 7a*). This pattern is a direct result of the probability of mortality equations used in the analysis, which predict non-zero probability of mortality. Even if a tree experiences a fire that does not scorch its crown, then there is a small increase in probability of mortality as scorch height increases. In the PPDF stand, fire-caused mortality causes a drop in *CBD* even with little crown scorch, because of high probability of mortality in the Douglas-fir under- and middle-stories. However, *CBD* then changes little with increasing scorch height because the remaining trees are not as susceptible to fire-caused mortality. The PP and SNMC sites show the least difference between mechanical pruning and scorch (aside from the initial drop), indicating their structure and composition resists fire-caused mortality.

Again, all stands exhibited an initial drop in *CFL* with a crown scorch height of just 1m, corresponding to mortality of the most susceptible trees in each stand. The initial drop was again smallest in the PP and SNMC sites due to their relatively resistant structure and composition.

The differences in effect on *CBH* between scorch and pruning were generally minor (*fig. 7c*). The exception to that rule is the SNMC site, where even 1m of scorch caused enough mortality to raise *CBH* to 8m. However, increasing scorch height to 8m had no additional effect because the remaining trees were more resistant to fire-caused mortality. The PPDF and SNMC sites show "blips" in the response of *CBH* to scorch height at higher levels of crown scorch. These deviations are minor, and correspond to levels of scorch that cause mortality in trees that, in addition to the biomass removed through scorch alone, reduce *CBD* in the layer(s) just above scorch height enough that the 0.011 kg/m^3 cannot be met.

Conclusion

This paper reports an analysis of a limited but accurate canopy fuel mass dataset. A similar analysis could be performed on a more extensive dataset that includes hundreds of stands in a given forest type. However, such an analysis would require making *estimates* of canopy fuel mass using allometric equations. Because such equations were generally built for dominant and co-dominant trees, some of the trends seen in this dataset might be masked by poor estimates of canopy fuel mass in sub-dominant trees. Improved individual-tree canopy fuel mass and available-fuel prediction models would greatly improve our ability to estimate stand-level canopy fuel characteristics.

In the single cohort stands, low thinning, low thinning with a commercial limit, and crown thinning all had a similar effect on *CBD* and *CFL*: reducing BA reduced *CBD* and *CFL* proportionally (*fig. 8*). This result means that reducing BA by some fraction of the initial condition would reduce *CBD* and *CFL* by roughly the same fraction, *regardless of how the BA was reduced*. That is, a tree's contribution to *CFL* (and therefore *CBD*) is proportional to its contribution to BA. This result is consistent with allometry that relates canopy biomass to the square of dbh (Brown 1978). The exception to this proportionality rule is the PPDF stand, whose strong understory of Douglas-fir resulted in a bi-modal canopy fuel profile that was unique among the study stands. In the PPDF stand, commercial thinning with a commercial limit and crown thinning were resulted in drastically different *CBD* values than the strict low thinning, because it was the dense understory layer that dominated the *CBD* estimates. The rule of proportionality, therefore, only applies to the uni-modal stands in the study.

In stands without an understory, the initial reduction of BA has little effect on reducing *CBD* or *CFL* because the small trees removed first in a low thinning have little biomass, and what biomass they do have occurs below the layers of maximum density. In contrast, *CBD* and *CFL* may be reduced with only small reductions in BA in stands with a substantial under- or middle-story, because in those stands, the layers of maximum density occur lower in the canopy.

The proportionality rule described above does not apply to *CBH*--whether the small trees in a stand remain or are removed has a significant effect on *CBH*. In fact,

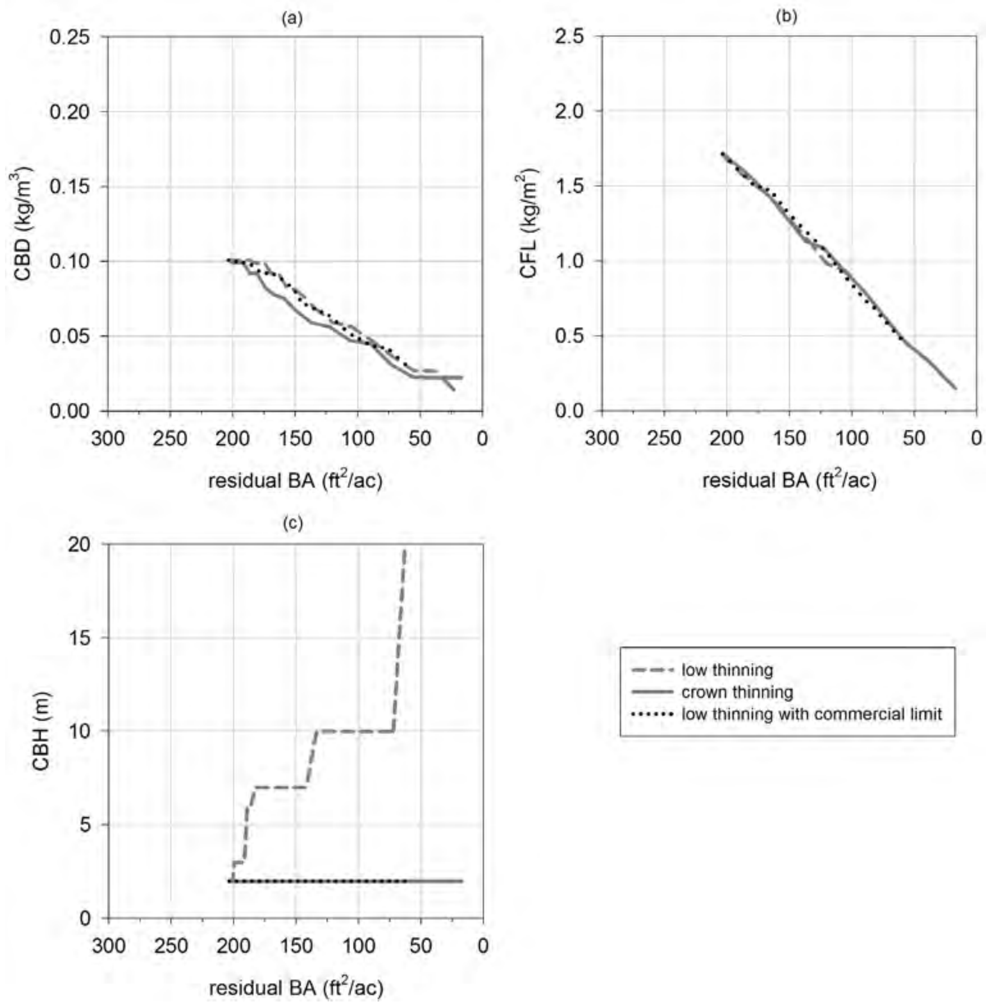

Figure 8—Response of (a) canopy bulk density (*CBD*), (b) canopy fuel load (*CFL*), and (c) canopy base height (*CBH*) to variable-intensity low thinning, low thinning with commercial limit, and crown thinning in the Sierra Nevada Mixed Conifer stand.

CBH remained essentially unchanged in the crown thinning and low thinning with commercial limit treatments, whereas a strict low thinning increased *CBH* significantly.

Due to the wide range of initial stand structures among stands, removing trees to a prescribed diameter limit has no predictable effects on canopy characteristics. Diameter-limit cutting is a convenient marking guideline but a poor prescription variable. Therefore, a canopy fuel treatment analysis should not specify a diameter-limit harvest. However, a low thinning can be marked as a diameter-limit cut if stand-specific structure is taken into account.

Mechanical pruning has little effect on *CBD* and *CFL* in the practical range of manual application (up to about 3m). The layers of maximum bulk density occur higher in the canopy, so little of the total canopy fuel load occurs within reach of mechanical pruning. Pruning can only affect *CBD* if pruning height reaches into the layers of maximum density, which is practically not possible if pruning is accomplished manually. Pruning has a predictable (linear) effect on *CBH:* it is always raised to pruning height unless it already exceeds pruning height.

Scorch from a prescribed fire has two effects on canopy fuel. First, scorch simulates the mechanical pruning by removing available fuel from scorched branches. Second, fire-caused mortality may further affect canopy fuel characteristics by removing available fuel from the canopy above the scorch height. Because fire-caused mortality is a function of species and tree diameter, the strength of this secondary effect depends in part on initial stand structure and composition.

Although we cannot draw general conclusions regarding the effects of these treatments on potential fire behavior, we can make inferences regarding their effects on canopy fuels. Stands with shade-tolerant understories, like the PPDF stand in this study, must be treated differently than single-cohort stands. Canopy base height will often be very near the ground, and will usually result directly from the contribution of the understory rather than the overstory. Therefore, any treatment that does not remove or drastically reduce this canopy layer (for example, crown thinning and commercial thinning) cannot raise *CBH*. Also, this understory layer may often contain the dense canopy layers that determine *CBD*, as the PPDF stand did. In that case, crown and commercial thinning will not decrease *CBD*. Low thinning (including removal of non-commercial trees) and prescribed burning would both be effective at reducing *CBD* and raising *CBH* in such stands. Crown and commercial thinning could be effective if coupled with a prescribed fire aimed at removal of the understory through fire-caused mortality.

More silvicultural tools may be appropriate for management of canopy fuels in single-cohort stands. Crown thinning and commercial low thinning are both effective at reducing *CBD* and *CFL* in these stands, but do not reduce *CBH*. Crown and commercial low thinning are less costly than strict low thinning, so more land area could be treated for the same investment, a potential advantage over strict low thinning. The lack of increase in *CBH* may be tolerable if *CBH* is already high enough, or if the thinning is combined with either prescribed burning or mechanical pruning to raise *CBH*.

This analysis focuses on effects of alternative treatments on canopy fuels, not their effects on fire potential. Each treatment may also affect (positively or negatively) other factors affecting potential fire behavior, including surface fuel characteristics, dead surface fuel moisture content, and wind adjustment factor (ratio of eye-level wind speed to 20-ft wind speed). Thinning a forest canopy generally results in lower dead surface fuel moisture and increased eye-level wind speed. Also, activity fuel from thinning or pruning may result in increased fuel load, unless mitigated as an integral part of the treatment. These side-effects of canopy fuel treatments must be considered when determining the overall effect of a treatment on potential fire behavior. Also, this analysis does not address the potential cost efficiency of each treatment. Because funding for fuel treatment is limited, land managers would presumably choose treatments that offer the most benefit for their cost. Quantifying the net cost of fuel treatment is a relatively simple task. Quantifying the benefit, however, is a much more difficult and abstract endeavor. Clearly, this analysis of the effects of alternative treatments on canopy fuel characteristics is but a small first step toward that goal.

Acknowledgements

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The Relation Between Tree Burn Severity and Forest Structure in the Rocky Mountains[1]

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Abstract

Many wildfire events have burned thousands of hectares across the western United States, such as the Bitterroot (Montana), Rodeo-Chediski (Arizona), Hayman (Colorado), and Biscuit (Oregon) fires. These events led to Congress enacting the Healthy Forest Restoration Act of 2003, which, with other policies, encourages federal and state agencies to decrease wildfire risks by evaluating, prioritizing, and implementing vegetation treatments across large landscapes. Land management agencies, and society, have high expectations that vegetation (fuel) treatments and forest restoration activities will moderate fire behavior (intensity) and its effects, resulting in the enrichment of forest values. However, the uncertainty of these relations is unknown, preventing forest managers from communicating their confidence in the effectiveness of fuel treatments in reducing risk of wildfires. To address this uncertainty, we observed the relation between pre-wildfire forest structure and burn severity across cold, moist, and dry forest types. We used a combination of collaborative studies and field data from 73 wildfire events in Idaho, Oregon, Montana, Colorado, Arizona, and Utah (which burned between 2000 and 2003) to obtain over 900 observations. We used a multiple spatial scale approach to provide insight into how physical setting, weather, and site-specific forest structures relate to tree burn severity, with conditional probabilities that provide an estimate of uncertainty. The burn severity classification we developed integrates fire intensity, fire severity, and the forest's response to wildfire. Forest and wildfire characteristics that determine tree burn severity are: a particular wildfire group, tree canopy base height, total forest cover, surface fuel amount, forest type, tree crown ratio, and tree diameter. Because of the study's wide breadth, results from it are applicable throughout the Rocky Mountains.

Introduction

In recent years, the Bitterroot (Montana), Rodeo-Chediski (Arizona), Hayman (Colorado), Biscuit (Oregon), and numerous other wildfire events have burned thousands of hectares (acres) across the western United States (Bitterroot National Forest 2000, Graham 2003, Graham et al. 2004). These events directed forest management activities towards developing and maintaining forests resilient and/or resistant to wildfire (Stephens and Ruth 2005). For example, the Healthy Forest Restoration Act of 2003, and the National Fire Plan, encouraged federal and state

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agencies to evaluate, prioritize, and implement vegetation treatments across large landscapes, in order to decrease the risk of wildfires (USDA Forest Service 2004). The focus of these vegetation treatments will most likely occur in the wildland urban interface, municipal watersheds, habitats of threatened and/or endangered species, and other places that contain values important to forest users and stakeholders. Land management agencies and society have high expectations that vegetation (fuel) treatments and forest restoration activities will moderate fire behavior (intensity), and its effects, resulting in sustaining many cherished forest values.

Although canopy bulk density, fuel models, canopy base height, and other forest metrics have been related to fire behavior using physical laws, controlled experiments, and models (Graham et al. 2004, Peterson et al. 2005, Scott 1998, Scott and Reinhardt 2001), there is limited information to indicate how forest structure is related to fire behavior and burn severity (what is left and its condition) during a wildfire event (Broncano and Retana 2004, Loehle 2004, Weatherspoon and Skinner 1995). Moreover, the uncertainty of these relations is unknown, preventing forest managers from communicating their confidence in the effectiveness of fuel treatments in reducing the risk of wildfires and effects on forest values. Without these estimates, managers and forest stakeholders could have a false sense of security and a belief that if a wildfire occurs after a fuel treatment, the values they cherish (for example, homes, wildlife habitat, community water sources, sense of place) will be protected and maintained both in the short- (months) and long- (10s of years) term.

Our objective is to define and quantify the relation between forest structure and burn severity, and to determine the uncertainty of the relations (Jain and Graham 2004). Although other studies have quantified this relationship, they often were limited in scope and applicability (Carey and Schumann 2003, Martinson and Omi 2003). To avoid these shortcomings, we designed our study to sample many wildfires (73) that burned in different years throughout the inland western United States. Because of the study's scope, it incorporated a large amount of variation in forest structure as well as disparity in burn severity after extreme wildfires. The data we collected came from wildfires that burned in the moist, cold, and dry forests between 2000 and 2003. By studying wildfires that burned throughout the inland western United States (and in different years), we were able to include a variety of weather, which occurred during the fires, and physical settings in our sampling. The relations between forest structure and burn severity and the uncertainty of these associations after intense and severe wildfires will provide information that could be used in evaluating fuel management decisions throughout the moist, cold, and dry forests of the inland western United States.

Methods

Using intensive, extensive, and focused watershed sampling, we visited 73 wildfire events that burned between 2000 and 2003 in Montana, Idaho, Colorado, Oregon, Utah, and Arizona (*tables 1, 2, 3, fig. 1*). These wildfires occurred in the dry (ponderosa pine, *Pinus ponderosa* Dougl. ex Laws and Douglas-fir, *Pseudotsuga menziesii* [Mirb.] Franco), moist (western hemlock, *Tsuga heterophylla* [Raf.] Sarg., western redcedar, *Thuja plicata*, Donn ex D. Don grand fir, *Abies grandis* [Dougl. ex D. Don] Lindl., white fir, *Abies concolor* [Gord. & Glend.] Lindl. ex Hildebr.), and cold (lodgepole pine, *Pinus contorta* Dougl. ex Loud., and subalpine fir, *Abies lasiocarpa*, [hook.] Nutt.) forests throughout the inland western United States. Since not all forest classifications burned in a single year, we included multiple years in our

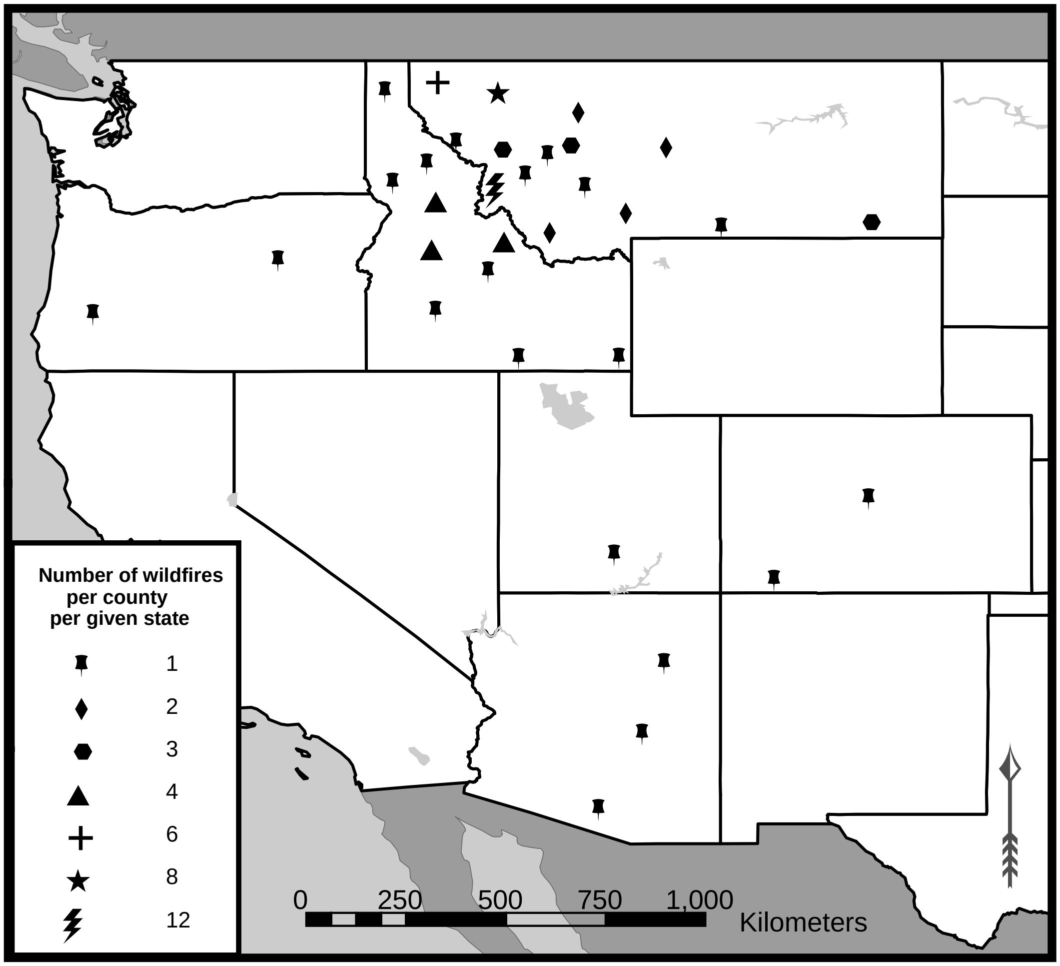

Figure 1—Distribution of the 73 fires that burned between 2001 and 2003. The symbol indicates the number of fires within a state's county. Counties and names of fires appear in *tables 1, 2, and 3*.

data collection. This enabled us to incorporate moist forest wildfires in our study, which tend to burn less frequently when compared to other forests. All areas were sampled the summer after they burned, except areas in Flathead and Lincoln counties in Montana and the Diamond Peak complex of fires in Idaho, which burned in 2000. These were sampled the second summer after they burned (*tables 1, 3*).

Sampling Designs

Fires were selected based on whether they occurred in moist, cold, or dry forests. Initially, all fires that burned in Idaho and Montana during 2000 and 2001 were sampled. We concentrated on wildfires in Colorado that burned in dry forests in 2002 to increase observations in these forest types. In 2004, we focused on wildfires that occurred only in moist forests that burned in 2003. We used three sampling designs to capture the variation in burn severity occurring at different spatial scales. The intensive sampling occurred in wildfires that burned between 2000 and 2003 and was led by Theresa Jain (US Forest Service, Rocky Mountain Research Station) (*table 1*). This extensive sampling revisited previously established Forest Inventory and Analysis (FIA) plots that burned in Montana and Idaho in 2000, in Montana in 2001, and in Arizona and Utah in 2002 (*table 2*). Using the FIA plots, we were able

Table 1—*The intensive sampling involved selecting a specific set of wildfires. The table describes the county and state where the fire occurred. For each fire, we included the fire name and number of observations (no. of obs.). We obtained daily weather for each fire, beginning with the fire weather start date (month/day/year) and continuing through to the end date. We also included fire start date, fire control date, the date the fire was out, and the estimated number of hectares each fire burned. In some places, we were unable to obtain specific dates (no date).*

			Fire weather		Wildfire			
County	Fire name	No. of obs.	Start date	End date	Start date	Control date	Date out	Size (ha)
				Colorado				
La Plata	Missionary Ridge	33	6/9/02	7/19/02	6/9/02	7/19/02	No date	29,591
Park	Hayman	62	6/8/02	6/28/02	6/8/02	6/28/02	7/7/02	55,749
				Idaho				
Bonner	Myrtle Creek	20	8/16/03	8/28/03	8/16/03	8/26/03	8/28/03	1,396
				Montana				
Beaverhead	Mussigbrod/ Maynard	5	7/31/00	10/6/00	7/31/00	10/6/00	11/6/00	18,891
Flathead	Fan Creek	7	8/10/00	8/16/00	8/10/00	8/16/00	8/20/00	318
Flathead	Moose	50	8/14/01	10/15/01	8/14/01	10/15/01	11/5/01	28,733
Flathead	Roberts	19	7/23/03	10/29/03	7/23/03	10/29/03	No date	23,178
Flathead	Taylor	4	8/10/00	10/31/00	8/10/00	9/20/00	10/31/00	531
Flathead	Young J	4	8/10/00	9/1/00	8/10/00	9/1/00	10/15/00	354
Lincoln	Cliff Point/ Lydia/Kelsey	26	8/11/00	9/13/00	8/11/00	9/13/00	10/30/00	5915
Lincoln	Stone Hill	29	8/11/00	9/13/00	8/11/00	9/13/00	10/30/00	4,498
Lincoln	Upper Beaver	31	8/11/00	9/25/00	8/11/00	9/25/00	10/30/00	3651
Mineral	Alpine Divide	16	8/3/00	9/22/00	8/3/00	9/22/00	10/27/00	1,503
Mineral	Landowner	1	8/11/00	9/12/00	8/11/00	9/12/00	No date	2,319
Missoula	Crazy Horse	20	8/6/03	10/17/03	8/6/03	10/17/03	11/21/03	4,573
Missoula	Ninemile	41	8/3/00	9/22/00	8/3/00	9/22/00	10/27/00	7,073
Missoula	Flat Creek	16	8/4/00	9/12/00	8/3/00	9/12/00	11/20/00	4,047
Ravalli	Bear	159	7/31/00	10/30/00	7/31/00	10/30/00	No date	58,696
Ravalli	Blodget	4	7/31/00	10/31/00	7/31/00	11/1/00	11/9/00	4,649
Ravalli	Coyote	8	7/31/00	9/2/00	7/31/00	9/2/00	12/1/00	8,903
Ravalli	Razor	14	8/5/00	10/23/00	8/5/00	10/23/00	11/6/00	5,342
Ravalli	Taylor Springs	2	7/31/00	10/23/00	7/31/00	10/23/00	11/6/00	8,696
Valley	Little Pistol	10	8/10/00	10/12/00	8/10/00	10/20/00	11/1/00	25,803
				Oregon				
Grant	Flagtail	45	7/15/02	9/4/02	7/15/02	9/4/02	No date	3,296

to sample several fires, but with few observations per fire (*table 2*). David Atkins (US Forest Service, Northern Region), Mike Wilson (Interior West Forest Inventory and Analysis Program, Rocky Mountain Research Station) and Theresa Jain led this effort. The focused watershed sampling quantified forest structure and burn severity within watersheds (142 ha to 6,475 ha, 350 to 16,000 ac) using remotely sensed data corroborated with ground-truth data (*table 3*). This sampling was led by David S. Pilliod (California Polytechnic State University), in collaboration with Theresa Jain.

Table 2—*The extensive sampling involved revisiting forest inventory and analysis (FIA) plots that burned during the 2000 (Idaho and Montana) and 2001 (Montana) wildfires. The table describes the state and county where the fire occurred, the fire name, and number of observations (no. of obs.). We obtained daily weather for each fire, beginning with the fire weather start date (month/day/year) and continuing through to the end date. We also included the fire start date, fire control date, the date the fire was out, and the estimated number of hectares each fire burned. In some places, we were unable to obtain specific dates or estimates of size (no date, no est.). For the fires in Arizona, we did not obtain weather data.*

			Fire weather		Wildfire			
County	**Fire name**	**No. of obs**	**Start date**	**End date**	**Start date**	**Control date**	**Date out**	**Size (ha)**
			Arizona					
Gila	Packrat complex	1	No date	No date	8/15/02	9/2/02	9/2/02	1,404
Navajo	Rodeo/ Chediski	2	No date	No date	6/18/02	7/2/02	7/7/02	189,651
Pima	Bullock	1	No date	No date	5/21/02	6/2/02	6/10/02	12,368
			Idaho					
Cassia	STF Assist 5	3	7/15/00	10/10/00	7/15/00	10/15/00	No date	No est.
Clearwater	Elizabeth	1	8/3/00	10/10/00	8/3/00	10/10/00	10/13/00	1,318
Custer	Rankin	1	8/10/00	9/2/00	8/10/00	9/2/00	11/6/00	2,715
Elmore	Trail Creek	5	8/15/00	10/11/00	8/15/00	10/13/00	No date	14,081
Idaho	Burnt Flats	2	8/10/00	9/8/00	8/10/00	9/8/00	10/25/00	9,116
Idaho	Butts	2	7/31/00	10/14/00	7/31/00	11/1/00	11/27/00	10,538
Idaho	Fitz	1	7/15/00	10/15/00	7/15/00	10/15/00	No date	445
Idaho	Hamilton	3	7/15/00	10/15/00	7/15/00	10/15/00	No date	No est.
Idaho	Lonely	5	7/30/00	10/22/00	7/30/00	10/23/00	11/1/00	7,874
Idaho	Papoose	1	8/10/00	10/1/00	8/10/00	11/1/00	11/22/00	1,207
Idaho	Thirty	1	7/15/00	10/15/00	7/15/00	10/15/00	No date	No est.
Idaho	Three Bears	1	7/31/00	10/30/00	7/31/00	10/30/00	10/30/00	6,086
Lemhi	Clear Creek	3	7/8/00	11/01/00	7/8/00	12/1/00	12/11/00	69,661
Lemhi	Morse	1	8/10/00	10/9/00	8/10/00	10/10/00	10/16/00	2,329
Lemhi	Packer Meadow	1	8/6/00	11/1/00	8/5/00	11/1/00	11/27/00	2,226
Lemhi	Shellrock	5	8/10/00	10/31/00	8/10/00	11/1/00	11/27/00	30,042
Lewis	Maloney Creek	1	7/15/00	10/15/00	7/15/00	10/15/00	No date	No est.
Valley	Diamond Peak	9	8/10/00	10/31/00	8/10/00	11/1/00	11/27/00	30,042
Valley	Indian Creek	1	7/15/00	10/12/00	7/15/00	10/12/00	No date	1,133
			Montana					
Beaver-head	Bear/Maynard	2	7/31/00	10/30/00	7/31/00	10/30/00	No date	18,891
Beaver-head	Mussigbrod/ Maynard	7	7/31/00	10/6/00	7/31/00	10/6/00	11/6/00	18,891
Carbon	Willie	1	8/27/00	9/6/00	8/27/00	9/6/00	9/6/00	608
Flathead	Bald Hill	2	8/12/00	8/20/00	8/12/00	8/20/00	No date	No est.
Flathead	Chipmunk	1	8/11/00	10/1/00	8/11/00	10/1/00	10/1/00	1,267
Flathead	Helen Creek	2	7/23/00	10/31/00	7/23/00	10/31/00	12/6/00	666
Gallatin	Beaver Creek	2	8/10/00	9/2/00	8/10/00	9/2/00	10/16/00	4,371

Table 2 Continued—*The table describes the county and state where the fire occurred. For each fire, we included the fire name and number of observations (no. of obs.). We obtained daily weather for each fire, beginning with the fire weather start date (month/day/year) and continuing through to the end date. We also included the fire start date, fire control date, the date the fire was out, and the estimated number of hectares each fire burned. In some places, we were unable to obtain fire name, specific dates, or estimates of size (no date, no est.). For the fires in Utah, we did not obtain weather data.*

			Fire weather		Wildfire			
County	**Fire name**	**No. of obs.**	**Start date**	**End date**	**Start date**	**Control date**	**Date out**	**Size (ha)**
				Montana				
Gallatin	Maudlow/ Toston	6	7/15/00	10/15/00	7/15/00	10/15/00	No date	No est.
Granite	Alder	1	8/24/00	9/25/00	8/24/00	9/25/00	10/10/00	2,226
Granite	Cougar	1	7/23/00	9/25/00	7/23/00	9/25/00	No date	1,942
Granite	Ryan Gulch	3	7/23/00	10/15/00	7/15/00	10/15/00	No date	No est.
Jefferson	High Ore	1	7/15/00	8/19/00	7/15/00	10/15/00	No date	No est.
Judith Basin	Lost Fork Ridge	2	8/1/00	10/6/00	8/1/00	10/6/00	12/4/00	526
Lewis & Clark	Bunyan	1	9/15/00	11/10/00	9/15/00	11/10/00	11/10/00	479
Lewis & Clark	Cave Gulch	4	7/23/00	8/23/00	7/23/00	8/23/00	9/26/00	12,141
Lincoln	Cliff Point	1	8/11/00	9/13/00	8/11/00	9/13/00	10/30/00	No est.
Lincoln	Grambauer Face	1	8/11/00	8/20/00	8/11/00	8/20/00	10/30/00	321
Lincoln	Northwest Peaks	1	8/10/00	8/25/00	8/10/00	8/25/00	10/13/00	12
Lincoln	Stone Hill	2	8/11/00	9/13/00	8/11/00	9/13/00	10/30/00	4,498
Mineral	Alpine Divide	1	8/3/00	9/22/00	8/3/00	9/22/00	10/27/00	1,503
Mineral	Landowner	6	8/11/00	9/12/00	8/11/00	9/12/00	1/22/00	2,319
Missoula	Flat Creek	3	8/4/00	9/12/00	8/3/00	9/12/00	11/20/00	4,047
Missoula	Ninemile	2	8/3/00	9/22/00	8/3/00	9/22/00	10/27/00	7,073
Powder River	Stag	5	7/26/00	8/12/00	7/26/00	8/12/00	9/5/00	24,948
Powell	Monture/Spread	7	7/13/00	10/31/00	7/13/00	11/1/00	12/30/00	9,632
Ravalli	Bear	27	7/31/00	10/30/00	7/31/00	10/30/00	No date	58,696
Ravalli	Blodget	1	7/31/00	10/31/00	7/31/00	11/1/00	11/9/00	4,648
Ravalli	Boundary	1	7/15/00	10/13/00	7/15/00	10/15/00	No date	No est.
Ravalli	Coyote	3	7/31/00	9/2/00	7/31/00	9/2/00	12/1/00	8,902
Ravalli	Mink	1	7/31/00	8/30/00	7/31/00	8/30/00	11/6/00	271
Ravalli	Razor	1	8/5/00	10/23/00	8/5/00	10/23/00	11/6/00	5,342
Ravalli	Taylor Springs	4	7/31/00	10/23/00	7/31/00	10/23/00	11/6/00	8,695
Teton	Clear	8	7/15/00	10/15/00	7/15/00	10/15/00	No date	No est.
Teton	McDonald 2	1	7/21/00	7/30/00	7/21/00	7/30/00	11/10/00	1,758
Teton, Park	Unknown	3	No date	No date	No date	No date	No date	No est.
Flathead	Unknown	7	No date	No date	No date	No date	No date	No est.
Gallatin	Unknown	2	No date	No date	No date	No date	No date	No est.
Powell	Unknown	1	No date	No date	No date	No date	No date	No est.
				Utah				
Garfield	Sanford	1	No date	No date	6/1/02	7/1/02	No date	26,268

Table 3—*The focused watershed sampling design occurred within the Quartz fire and Diamond Peak complex. The table describes the county and state where the fire occurred. For each fire, we included the fire name and number of observations (no. of obs.). We obtained daily weather for each fire, beginning with the fire weather start date (month/day/year) and continuing through to the end date. We also included the fire start date, fire control date, the date the fire was out, and the estimated number of hectares each fire burned.*

County	Fire name	No. of obs.	Fire weather		Wildfire			
			Start date	End date	Start date	Control date	Date out	Size (ha)
				Oregon				
Douglas	Quartz	50	8/9/01	9/26/01	8/9/01	9/26/01	10/31/01	2,494
				Idaho				
Lemhi	Diamond Peak	79	8/10/00	10/31/00	8/10/00	11/1/00	11/27/00	30,042

Intensive Sampling

For each selected wildfire, we used stratified random sampling to represent the variation in forest structure, physical setting, and weather (*table 4*). In establishing the sampling frame, forest cover type (dry, moist, or cold) described the broad-scale vegetation. The stands burned within each wildfire were stratified first by forest cover type and then further stratified by high and low burning index (split at the median burning index for all stands burned by a particular wildfire). Fire progression maps were used to estimate the day a particular stand burned, and then weather data for that day was acquired from the closest weather station (*tables 1, 2, 3*). Using these weather data and the most applicable fuel model for each stand within a fire perimeter, we calculated the burning index[3] using Fire Family Plus for each stand (Bradshaw and Britton 2000). This stratification insured the stands we sampled were burned during the range of weather conditions that occurred throughout the wildfire event.

Within each burning index class (high and low), the physical settings of the stands were placed into two strata: those with slope angles less than or equal to 35 percent and those with slope angles greater than 35 percent (*table 4*). In the Northern Rocky Mountains, settings with slope angles less than 35 percent usually occur on benches, within riparian areas, or along ridge tops. Settings with slope angles greater than 35 percent tend to occur on side slopes. On the Hayman fire in Colorado and Flagtail fire in Oregon, we used a 25 percent slope angle to differentiate the two slope classes because the rolling topography burned by these fires tended to be moderately steep. Within a given slope class, the stands were divided into those containing short, sapling to medium sized trees (≤ 12.2m, 40 ft), and those containing tall, mature to old trees (> 12.2m, 40 ft). Within these structural classes, stands were divided into two density strata, those with canopy cover less than or equal to 35 percent and those with canopy cover greater than 35 percent. This stratification insured that stands selected for sampling would have a broad range of horizontal

[3] Burning index describes the effort needed to contain a single fire within a particular fuel type within a given area. The index is a function of the spread component (SC) and available energy release component (ERC) of a fire, which in turn are used to estimate flame length from which the burning index is computed (Bradshaw et al. 1983, Bradshaw and Britton 2000). Wind speed, slope, fuel (including the effects of green herbaceous plants), and the moisture content of the fuels are used to determine the SC and ERC. The difference between the two components is that SC is determined on the moisture levels of the fine fuels while ERC requires moisture levels from the entire fuel complex.

structures. Therefore, the final sampling stratification contained forest cover (three classes), burning index (two classes), slope angle (two classes), canopy height (two classes), and stand density (two classes) (*table 4*). Each area where a stand existed within a particular stratum and fire perimeter had an equal probability of being selected.

From the sampling frame (approximately 100s to 1000s of stands) for each wildfire, we randomly selected 15 stands. Each stand was evaluated (in selection order) to determine if (1) it met the sampling criteria, (2) had an opportunity to burn (in some cases, stands near the fire perimeters had control lines preventing them from burning), (3) did not have any confounding factors that may have influenced their burning (for example, evidence of fire retardant or other suppression activities), and (4) measured at least 100m by 100m (328 ft by 328 ft) in size (large enough to establish the sample points).

Table 4— *This sampling matrix was used to sample the 2000 Bitterroot National Forest fires for the dry forest type. Within each forest type, stands were stratified by burning index (two classes), slope angle (two classes), canopy height (two classes), and stand density (two classes L=low, H=high). This matrix was replicated between six to nine times. Similar matrices were created for each fire we sampled in the dry, moist, and cold forest types.*

Dry forest type																
Burning index	≤ 75								> 75							
Slope	≤ 35%				> 35%				< 35%				> 35%			
Height (ft)	≤ 40		> 40		≤ 40		> 40		≤ 40		> 40		≤ 40		> 40	
Density (cover) L= ≤ 35% H= > 35%	L	H	L	H	L	H	L	H	L	H	L	H	L	H	L	H

The purpose of our intensive sampling was to quantify the relation between pre-wildfire forest structure and burn severity, not to characterize the variation of burn severity and forest structure within stands. Therefore, to maximize the number of stands sampled (including the full breadth of burn severity), only one plot was placed in each randomly selected stand. An aerial photograph or topographic map was used to obtain an azimuth of a line intersecting the approximate center of the stand. In stands two hectares (5 ac) and larger in size, a minimal slope distance of 100m (328 ft) from the stand edge along this azimuth and a random number between one and six was selected using a dice. This number was multiplied by 16, and additional distance (meters) equaling this value was traversed along the azimuth before plot installation. In stands less than two hectares (5 ac) in size, the plot was located 50m (164 ft) from the stand edge along the line intersecting the center of the stand. The plot was permanently located using a metal stake, and the distance from the stand edge was recorded, as were the global positioning system (GPS) coordinates.

Extensive Sampling

Interior West Forest Inventory and Analysis staff randomly located permanent forest sample points on a grid throughout the forests of the western United States (Interior West Forest Inventory and Analysis 2006). By chance, a number of the plots established by FIA burned in 2000 and 2001 wildfires. After the 2000 wildfires, all plots that burned in Idaho and Montana had burn severity quantified. After the 2001 wildfires, all fires that burned in Montana had burn severity quantified. Wildfires that burned in Utah and Arizona in 2002 were visited and burn severity was quantified as part of the annual FIA sampling (*table 2*). The FIA plots were established on different spatial grids and burned areas varied in size and location. Therefore, the number of FIA plots we could visit after a wildfire varied considerably depending on the wildfire and the sampling design established by FIA. Nevertheless, we visited all previously established FIA plots that burned in 2000 and 2001. As a result, some burned areas had multiple FIA plots sampled after a wildfire, while other areas only had one plot revisited.

Focused Watershed Sampling

The focused watershed sampling occurred within forests burned by the Quartz and Diamond Peak fire complexes in Idaho and Oregon in 2000 and 2001 (*table 3*). In contrast to other post-wildfire sampling we completed, this sampling was designed to ensure that the structure and burn severity observations we collected occurred equally in both upland and riparian areas. Using maps (GIS based), we delineated the watersheds burned by these two wildfire events and subsequently defined a 60m (197 ft) riparian zone along each side of the stream reaches. Areas outside the riparian zone within each watershed were defined as the upland zone. A minimum of twenty-five plots were randomly located within both the upland and riparian zones using a complete spatial randomness (CSR) Poisson process (Diggle 2003). By using this sampling approach, we avoided spatial autocorrelation among the plots and insured their spatial independence (Cressie 1991).

Data Collection

Intensive and focused data collection

For each randomly located plot, physical setting descriptors (aspect, slope angle, topographic position, elevation), a general stand description (species composition, number of stories, horizontal spacing), and stand origin (past harvest evidence, regeneration treatment) were recorded. Our intention was to develop a continuous variable or post-classify burn severity for both the vegetation and the forest floor. To do so, a variety of fine resolution descriptors of soil and vegetation burn severity were used or developed from past burn severity characterizations (DeBano et al. 1998, Key and Benson 2001, Ryan and Noste 1985, Wells et al. 1979) (*tables 5*, *6*). However, in contrast to these classifications, our characterization concentrated on what was left after the wildfire and not on what was consumed. The characterization and description of soils and vegetation were accomplished using four strata: (1) soil surface, (2) grass, forbs, shrubs, and seedlings, (3) saplings and large trees, and (4) woody debris.

Forest floor (soil surface) characterization included total cover and the proportion of total cover dominated by the different char classes on a 1/741 ha (1/300 ac) fixed radius plot. These included new litter (deposition since the fire), old litter (present previous to the fire), humus, brown cubical rotten wood (at or above soil surface), woody debris less than or equal to 7.6 cm (3.0 in) in diameter, woody debris greater than 7.6 cm (3.0 in) in diameter, rock, and exposed mineral soil. The amount of char occurring in each of these cover characterizations was estimated using color (unburned, black, grey, or orange) (*table 5*).

Using a fixed radius plot (1/741 ha, 1/300 ac), the proportion of grass and forbs, the number of new seedlings (species recorded, if identifiable) regenerated since the

Table 5—*Surface components (strata) and char classes for quantifying burn severity are displayed. In addition to proportion of cover and char class, depths (cm) were measured for litter fallen since fire, litter prior to fire, and humus. All measurements were conducted on a 1/741 ha circular plot. Trees were less than <12.7 cm diameter breast height (dbh).*

Strata	Unburned (%)	Light char (%)	Moderate char (%)	Deep char (%)
		Surface		
Litter fallen onto surface since fire	Litter type (fir or pine, leaves) with no char classes			
Litter present prior to fire	No sign of char	Blackened but present	Not present	Not present
Humus (decomposed organic matter)	No sign of char	Blackened but present	Not present	Not present
Bare mineral soil	No sign of char	Blackened	Grey color	Orange color
Rock	No sign of char	No sign of char	Black edges	White residue
Brown cubical rotten wood	No sign of char	Burned on surface	Charred but still present	Imprint on surface
Woody debris ≤ 7.6 cm diameter	No sign of char	Burned on surface	Charred but still present	Not present
Woody debris > 7.6 cm diameter	No sign of char	Burned on surface	Charred but still present	Imprint on surface
Stumps	No sign of char	Burned on surface	Charred but still charred	Stump hole
		Ground level vegetation and small trees		
Shrubs – low 0 - 0.5 cm basal stem diameter	Stems intact	Stems present but charred	Base of stem present	Stump hole
Shrubs – medium 0.51 - 2 cm stem diameter	Stems intact	Stems present but charred	Base of stem present	Stump hole
Shrubs – tall 2.1 - 5 cm stem diameter	Stems intact	Stems present but charred	Base of stem present	Stump hole
Forbs and grasses	Growing on unburned litter	Growing on blackened litter	Growing on grey charred soil	Growing on orange charred soil
New seedlings since fire	Growing on unburned litter	Growing on blackened litter	Growing on grey charred soil	Growing on orange charred soil
Trees present prior to fire < 12.7 cm dbh	No sign of char	Live trees needles present	No or brown needles	Stump hole

fire, and both proportion and number of basal stem diameters for shrubs were estimated. Shrubs were placed into three size classes. Low shrubs were defined as those less than 0.5 cm (0.2 in) basal stem diameters, medium shrubs from 0.51 cm to 2 cm (0.2 to 0.8 in), and tall shrubs from 2.1 to 5 cm (0.8 to 1.9 in) (Brown 1976). For grass, forbs, and new (post-fire) seedlings, the proportion growing on a specific charred surface was recorded, while the char class was defined by their condition (*table 5*).

Small trees (saplings), those less than 12.7 cm (5.0 in) diameter breast height (1.4m, 4.5 ft), were quantified using a 1/741 ha (1/300 ac) circular plot. The total number, species, and height were recorded and classified as to burn severity. Char class was defined by the condition of the saplings (*table 5*). To quantify large tree burn severity, we used a combination of fixed and variable radius plots. A 1/59 ha (1/24 ac) fixed plot was used for trees 12.7 cm (5.0 in) and greater. However, fixed plots tend to insufficiently quantify very large trees and in these situations a variable radius plot based on tree size is preferred (Avery 1967). To insure we quantified large trees, we used a variable radius plot where plot size is proportional to tree size. On the Missionary Ridge, Hayman, and Flagtail wildfires, we used a 4 m^2/ha (20 ft^2/ac) angle gauge. In these places all trees greater than 30.5 cm (12.0 in) dbh were sampled within this variable plot. On the rest of the wildfires a 9 m^2/ha (40 ft^2/ac) angle gauge was used and all trees greater than 45 cm (18.0 in) were sampled (*table 6*). Species, height, diameter, and uncompacted crown ratio (*fig. 2*) were recorded for each large tree. The proportion of the total crown containing green needles, brown needles, no needles, or black stem was determined for each large tree. Scorch height (low and high) on the stem was recorded and the circumference of scorch at the base of the stem was estimated (*table 6*).

Table 6—*Burn severity data taken on large trees (≥ 12.7 cm diameter breast height (dbh) using a fixed (1/59 ha, 1/24 ac) and variable plot (9 m^2/ha or 4 m^2/ha). Trees greater than 45 cm (18 in) dbh were measured on 9 m^2/ha (40 ft^2/ac) variable plot. Trees greater than 30.5 cm (12 in) dbh were measured on 4 m^2/ha (20 ft^2/ac) variable plot on the Hayman and Missionary Ridge fires in Colorado and Flagtail fire in Oregon. Trees with diameters less than these were measured on the fixed plot.*

Strata	Uncompacted crown ratio	Green crown (%)	Brown crown (%)	Black crown (%)	Bole scorch height (cm) and direction (azimuth) scorch is facing		Scorch at tree base (%)
					Low	High	
Trees ≥ 12.7 cm dbh	Total crown ratio	Green needles	Brown needles	Black stems, no needles	Lowest extent of scorch	Highest extent of scorch	Circumference

The amount of woody debris on the site and proportion in each decay class (no decay, decayed wood present, majority decayed wood, and completely decayed) was determined using three 37m (120 ft) linear transects radiating from the plot center at 0, 120, and 240 degree azimuths (Brown 1974, Maser et al. 1979).

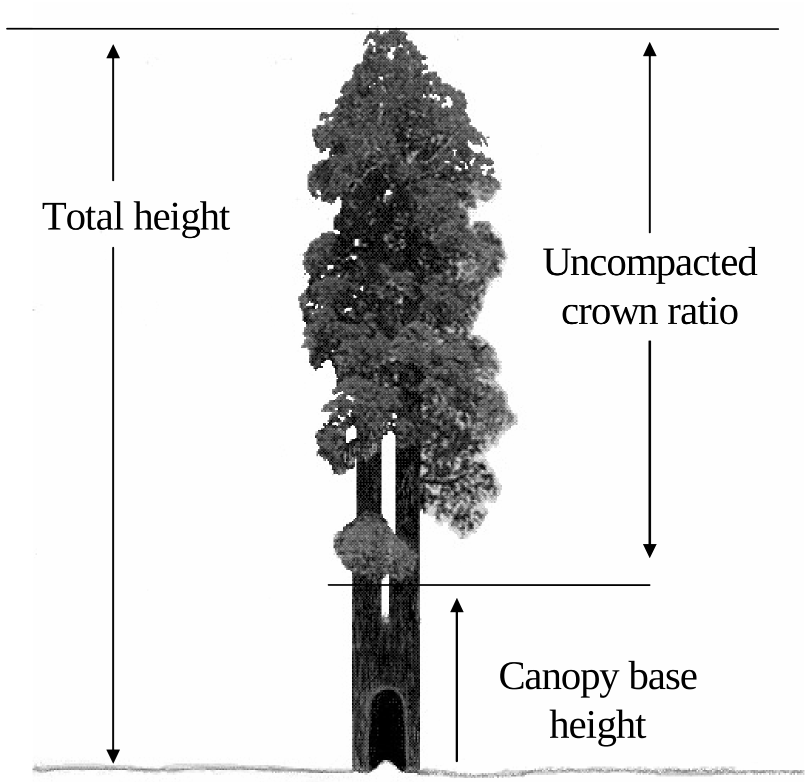

Figure 2—Illustration of how we measured uncompacted crown ratio and canopy base height (total height minus length of uncompacted crown ratio).

Extensive data collection

The extensive sampling occurred on previously established FIA plots that burned in wildfires. The plot design depended on when the plot was established (*table 7, fig. 3*). There were five different plot designs used for the extensive sampling: a single-plot, four-plot, six-plot, seven-plot, and ten-plot design. A fixed, variable, or a combination of fixed and variable plots (1/59 ha fixed circular and 9 m^2/ha variable), often of different sizes (1/59 or 1/741 ha fixed circular), were used for collecting post-wildfire data (*table 7, fig. 3*).

The aspect, slope, topographic position, and elevation of each plot were recorded at the time the FIA plot was established. Although different plot designs were used, the burn severity estimates and forest structure characterizations were similar to those obtained by the intensive and focused watershed designs (*tables 5, 6*). However, for small trees, shrubs, forbs, and grass, cover was quantified by species and the number of shrub stems was not recorded. All trees, including saplings and large trees, were tallied and burn severity was recorded using the proportion of crown containing green, brown, or black stems with no needles (*table 6*).

Physical setting, fire weather, and forest structure

Fire behavior and burn severity, for the most part, are determined by physical setting (location, topography, juxtaposition, and so forth), fuels (live and dead vegetation), and weather (both short- and long-term). We included these factors into

Table 7—*FIA plot designs varied depending upon when the plot was established (Interior West Forest Inventory and Analysis 2006). This table provides the plot design, establishment date for each fire, and shows whether it was a woodland plot (oak, juniper, or pinyon) or forested plot. Variable radius plots used a 9 m^2/ha (40 ft^2/ac) basal area factor, fixed radius plot number 1 (No. 1) were 1/59 ha (1/24 ac), fixed radius plot number 2 (No. 2) were 1/741 ha (1/300 ac), and woodland fixed radius plots were 1/25 ha (1/10 ac).*

				Number of plots			
County	Fire	Date established	Plot design	Variable	Fixed no. 1	Fixed no. 2	Woodland fixed
			Arizona				
Gila	Packrat complex	Unknown	4-plot woodland	-	4	4	-
Navaho	Rodeo/ Chediski	Unknown	4-plot woodland	-	4	4	-
Pima	Bullock	Unknown	4-plot woodland	-	4	4	-
			Idaho				
Cassia	STF Assist 5	1990-1997	4-plot woodland	-	4	4	-
Cassia	STF Assist 5	1980-1981	1-plot woodland	-	-	1	1
Clearwater	Elizabeth	1997-Present	4-plot forest	-	4	4	-
Custer	Rankin	1997-Present	7-plot forest	7	-	7	-
Elmore	Trail Creek	1997-Present	5-plot forest	5	-	5	-
Idaho	Butts	1997-Present	4-plot forest	-	4	4	-
Idaho	Papoose	1997-Present	4-plot forest	-	4	4	-
Idaho	Burnt Flats	1997-Present	5-plot forest	5	-	5	-
Idaho	Fitz	1997-Present	5-plot forest	5	-	5	-
Idaho	Hamilton	1997-Present	5-plot forest	5	-	5	-
Idaho	Lonely	1997-Present	5-plot forest	5	-	5	-
Idaho	Thirty	1997-Present	5-plot forest	5	-	5	-
Idaho	Three Bears	1997-Present	5-plot forest	5	-	5	-
Lemhi	Shellrock	1997-Present	4-plot forest	-	4	4	-
Lemhi	Clear Creek	1997-Present	4-plot forest	-	4	4	-
Lemhi	Clear Creek	1988-1989	10-plot forest	10	-	10	-
Lemhi	Morse	1997-Present	7-plot forest	7	-	7	-
Lewis	Maloney Ck	1997-Present	5-plot forest	5	-	5	-
Valley	Diamond Peak	1997-Present	4-plot forest	-	4	4	-
Valley	Indian Ck	1997-Present	4-plot forest	-	4	4	-
			Montana				
Beaverhead	Bear/Maynard	1993-1998	5-plot forest	5	-	5	-
Beaverhead	Mussigbrod/ Maynard	1993-1998	5-plot forest	5	-	5	-
Carbon	Willie	1993-1998	5-plot forest	5	-	5	-
Flathead	Bald Hill	1988-1989	10-plot forest	10	-	10	-
Flathead	Chipmunk	1993-1998	7-plot forest	7	-	7	-
Flathead	Helen Creek	1993-1998	7-plot forest	7	-	7	-
Missoula	Flat Creek	1993-1998	5-plot forest	5	-	5	-
Gallatin	Beaver Creek	1988-1989	10-plot forest	10	-	10	-
Gallatin	Beaver Creek	1993-1998	5-plot forest	5	-	5	-

Table 7 Continued—*FIA plot designs varied depending upon when the plot was established (Interior West Forest Inventory and Analysis 2006). This table provides the plot design, establishment date for each fire, and shows whether it was a woodland plot (oak, juniper, or pinyon) or forested plot. Variable radius plots used a 9 m^2/ha (40 ft^2/ac) basal area factor, fixed radius plot number 1 (No. 1) were 1/59 ha (1/24 ac), fixed radius plot number 2 (No.2) were 1/741 ha (1/300 ac), and woodland fixed radius plots were 1/25 ha (1/10 ac).*

				Number of plots			
County	**Fire**	**Date established**	**Plot design**	**Variable**	**Fixed no. 1**	**Fixed no. 2**	**Woodland fixed**
			Montana				
Gallatin	Maudlow/Toston	1988-1989	4-plot woodland	-	-	4	4
Gallatin	Maudlow/Toston	1988-1989	10-plot forest	10	-	10	-
Gallatin	Maudlow/Toston	1993-1998	5-plot forest	5	-	5	-
Gallatin	Maudlow/Toston	1993-1998	4-plot woodland	-	4	4	-
Granite	Alder	1993-1998	5-plot forest	5	-	5	-
Granite	Cougar	1993-1998	5-plot forest	5	-	5	-
Granite	Ryan Gulch	1988-1989	10-plot forest	10	-	10	-
Jefferson	High Ore	1988-1989	10-plot forest	10	-	10	-
Judith Basin	Lost Fork Ridge	1988-1989	10-plot forest	10	-	10	-
Lewis & Clark	Bunyan	1993-1998	5-plot forest	5	-	5	-
Lewis & Clark	Cave Gulch	1993-1998	5-plot forest	5	-	5	-
Lincoln	Cliff Point	1993-1998	7-plot forest	7	-	7	-
Lincoln	Grambauer Face	1993-1998	7-plot forest	7	-	7	-
Lincoln	Northwest Peaks	1993-1998	7-plot forest	7	-	7	-
Lincoln	Stone Hill	1993-1998	7-plot forest	7	-	7	-
Mineral	Alpine Divide	1993-1998	5-plot forest	5	-	5	-
Mineral	Landowner	1993-1998	5-plot forest	5	-	5	-
Missoula	Ninemile	1993-1998	5-plot forest	5	-	5	-
Powder River	Stag	1993-1998	5-plot forest	5	-	5	-
Powell	Monture/Spread	1993-1998	7-plot forest	7	-	7	-
Powell	Monture/Spread	1993-1998	5-plot forest	5	-	5	-
Ravalli	Bear	1988-1989	10-plot forest	10	-	10	-
Ravalli	Bear	1993-1998	5-plot forest	5	-	5	-
Ravalli	Bear	1993-1998	4-plot woodland	-	4	4	-
Ravalli	Blodget	1993-1998	5-plot forest	5	-	5	-
Ravalli	Boundary	1993-1998	5-plot forest	5	-	5	-
Ravalli	Coyote	1993-1998	5-plot forest	5	-	5	-
Ravalli	Mink	1993-1998	5-plot forest	5	-	5	-
Ravalli	Razor	1993-1998	5-plot forest	5	-	5	-
Ravalli	Taylor Spring	1993-1998	5-plot forest	7	-	5	-
Teton	McDonald 2	1993-1998	5-plot forest	5	-	5	-
Flathead	Unknown	1988-1989	10-plot forest	10	-	10	-
Flathead, Park	Unknown	1993-1998	7-plot forest	7	-	7	-
Gallatin	Unknown	1988-1989	10-plot forest	10	-	10	-
Gallatin	Unknown	1993-1998	5-plot forest	5	-	5	-
Teton	Unknown	1993-1998	5-plot forest	5	-	5	-
			Utah				
Garfield	Sanford	unknown	4-plot woodland	-	-	4	4

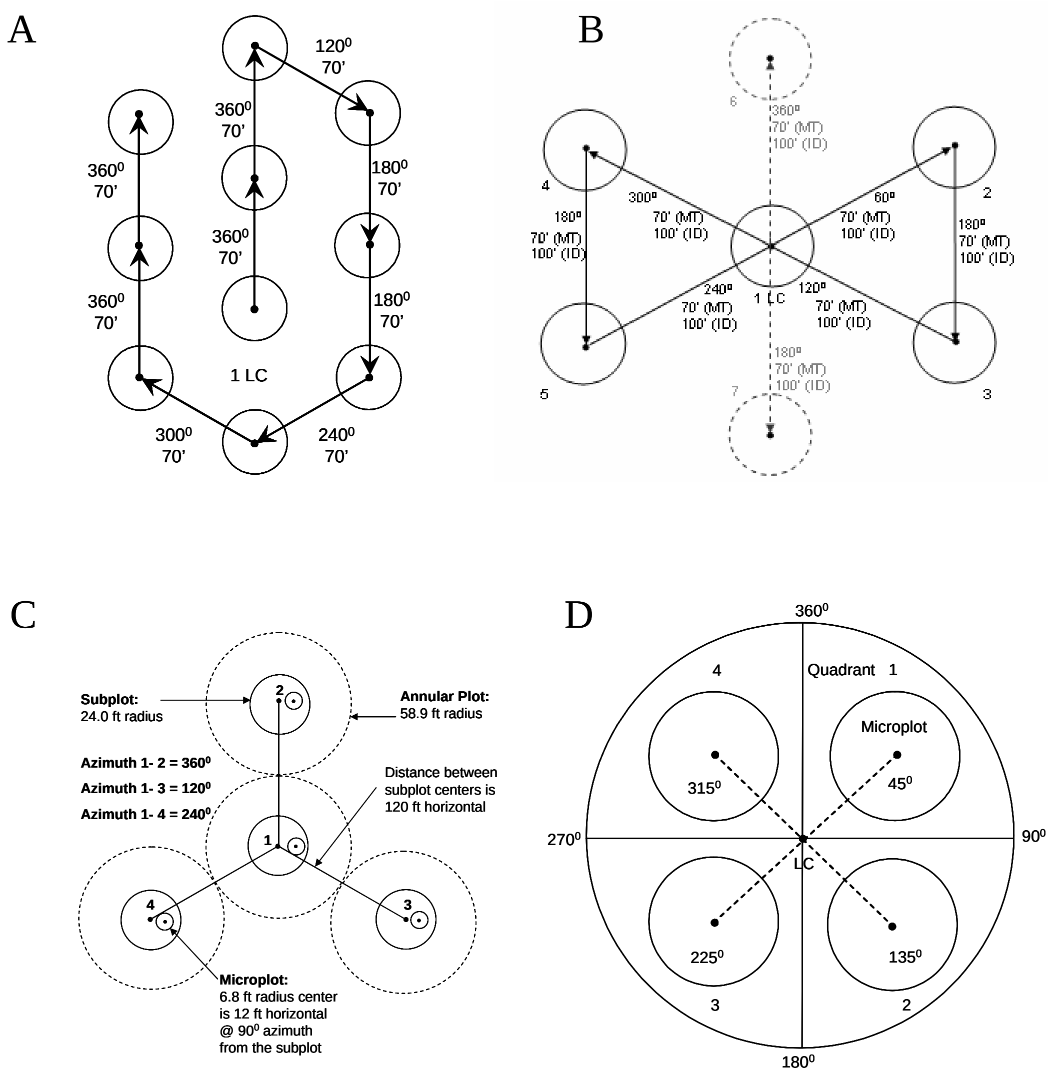

Figure 3—Illustrations showing different plot designs for the forest inventory and analysis (FIA) plots (Interior West Forest Inventory and Analysis 2006). Depending upon when a plot was established, FIA used a ten-plot (A), seven-plot (B where plot 6 and 7 are shown above and below the bowtie), five-plot (B without plots 6 and 7), 4-plot (C), and the one-plot woodland (D).

our study in addition to quantifying burn severity of the different vegetative strata. To describe the physical setting, we used the location of each plot in combination with a digital elevation model to develop several physical setting indices. Common attributes, such as aspect, slope angle, and elevation of each sample point, were obtained along with other descriptors, including slope curvature, compound topographic index (steady-state wetness index) (Gessler et al. 1995), landform index (McNab 1993), and topographic solar index (McCune and Keon 2002).

For each burned area we visited, we obtained hourly weather observations of the conditions under which the wildfire burned (*tables 1, 2, 3*). Data from remote automatic weather stations (RAWS) located in the county where each wildfire burned were summarized into daily reports using Fire Family Plus 2.0 (Bradshaw and McCormick 2000) (*table 8*). Because the exact day and time a specific plot burned is unknown, we summarized the weather data to the specific fire. In limited circumstances, we did not know the fire name and therefore were unable to obtain weather data for that particular fire.

Table 8—*Weather data were obtained from the nearest remote automated weather station (RAWS) in the county where the fire was located. Burning index is the effort needed to contain a single fire within a particular fuel type (Bradshaw et al. 1983, Bradshaw and Britton 2000). The index is a function of the spread component and energy release component of a fire. Wind speed, slope angle, fuel (including the effects of green herbaceous plants), and the moisture content of the fuels are used to determine the spread component and energy release component. The spread component is determined by the moisture levels of fine fuels while energy release component requires moisture levels from the entire fuel complex. We used Fire Family Plus 2.0 to summarize the weather into daily reports (Bradshaw and McCormick 2000). The Keetch-Byram drought index is a soil drought index that ranges from 0 (no drought) to 800 (extreme drought) and is based on soil capacity of 20.3 cm (8 in) of water. Factors in the index are maximum daily temperature, daily precipitation, antecedent precipitation, and annual precipitation (Burgan 1993). The Haines index (HI) was obtained from the Wildland Fire Assessment System (2006), where we selected for the particular day and location. The index is composed of a stability term and a moisture term. The stability term is derived from the temperature difference at two atmosphere levels. The moisture term is derived from the dew point depression at a single atmosphere level (Haines 1988). The indices range from 2 to 6, indicating potential for large fire growth.*

Weather variable definition	Units of measurement or range of index
Date of occurrence	Month, day, year
Maximum temperature	F ^{0}
Minimum relative humidity	Percent
Maximum relative humidity	Percent
Wind speed	Miles per hour
Wind direction	One of eight cardinal points
Precipitation	Inches
One hour fuel moisture	Percent
Ten hour fuel moisture	Percent
One thousand hour fuel moisture	Percent
Energy release component	British thermal units per square foot
Burning index	0-100
Keetch-Byram drought index	0-800
Haines index	2-6

We used the Forest Vegetation Simulator (FVS) and its Fire and Fuels Extension (FFE) to characterize pre-wildfire forest structure (Dixon 2004, Reinhardt and Crookston 2003, Wykoff et al. 1982). FFE-FVS is an excellent tool for forest structure characterization, as it can summarize data from a variety of plot designs and the metrics it produces can be adjusted using model variants reflecting regional forest conditions. For example, data from sites within northern Idaho and western Montana were summarized using the Inland Empire Variant. The Central Rockies Variant was used to summarize data collected in Colorado and Utah. In addition, FFE-FVS produces a variety of forest metrics associated with fire behavior, wildlife habitat, and forest development, and is supported by the U.S. Forest Service, Forest Management Service Center (Dixon 2004). The system is used by federal, state, and private entities throughout the western United States to summarize forest data, thereby making our data compatible, repeatable, and understandable by many forest managers and researchers of the western United States.

Forest structure characteristics derived from FFE-FVS included stand density indices (basal area per unit area, stand density index, trees per unit area, and so forth), characteristics associated with fire behavior (canopy bulk density and canopy base height) (*fig. 4*), and other miscellaneous stand characteristics (number of canopy layers, dominant species, and so forth) (Reinhardt and Crookston 2003) (*table 9*). In addition to these FFE-FVS derived forest characteristics, we estimated canopy base height directly from our data and described total cover, which included canopy overlap as suggested by Crookston and Stage (1999). Also, rather than using quadratic mean diameter (QMD) to describe stem dimensions, we used stem diameter weighted by basal area because it gives a better representation of tree diameters, especially when abundant small trees are present (*table 9*).

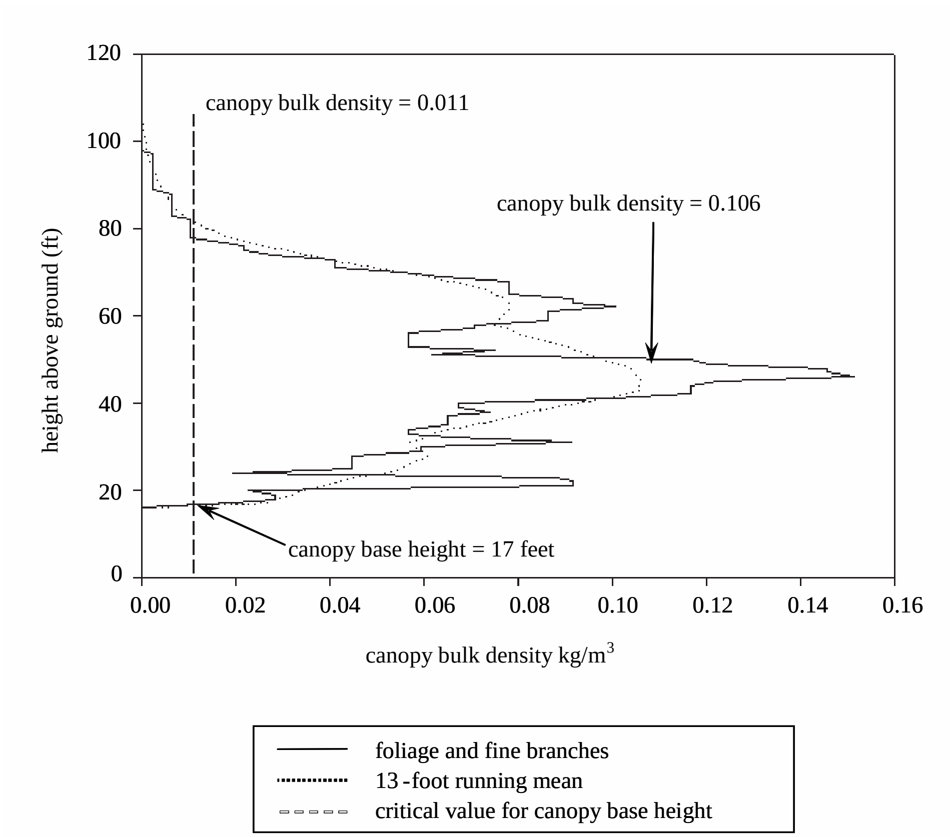

Figure 4—An illustration of how canopy bulk density and canopy base height are calculated in Fire and Fuels Extension of the Forest Vegetation Simulator (FFE-FVS) (Reinhardt and Crookston 2003). FFE-FVS does not include trees two meters and under. In the calculation, they are considered surface fuels.

FFE-FVS provides a suite of characteristics based on our data that describes different elements of forest structure. For example, there are several ways to characterize overstory density, such as basal area per unit area, trees per unit area, percent cover, canopy bulk density, relative stand density index, total cubic volume per unit area, and total standing biomass (*table 9*). We wanted to avoid using multiple correlated variables as predictors. Therefore, we used canonical correlation for data mining and used our expertise to determine which of these variables had promise for identifying the relation between forest structure and burn severity. This process was well-suited, as it decreased the number of variables that we used to characterize forest structure. For density, we used total canopy cover with overlap, for tree size we used basal area weighted diameter and average height, and we used dry, moist, and cold forests to reflect broad variation in species composition. To describe the forest canopy, we used canopy base height (total height minus uncompacted crown length, averaged for plot) and uncompacted crown ratio (*fig. 2*). Because the amount of surface fuel available for burning is frequently used in predicting fire behavior, we included the amount of biomass of these fuels using FFE-FVS algorithms in our analysis.

Classifying Burn Severity

When we started the study, we wrongly assumed an established burn severity classification existed. However, it became obvious that burn severity was variable in application and inconsistently used and defined (Jain et al. 2004). Although there were clearly defined burn severity classes in several publications, the rationale supporting the classes was not provided. Upon comparing many definitions of burn severity, we discovered severity classes were either “lumped” or “split” and most often the classification focused on a “selected” severity condition. As a result, there appears to be no consistent way to communicate burn severity to the scientific community, managers, or to society at large. In fact, both in the scientific literature and lay publications, fire severity, burn severity, fire behavior, and fire intensity are often used interchangeably and inconsistently, leading to confusion and misinformation as to the impact wildfires have on forests and elements important to society. Yet, forest stakeholders are asking managers and policy makers to make decisions on manipulating vegetation to alter “wildfire severity” in forest ecosystems (USDA Forest Service 2004).

In our attempt to alleviate some of the inconsistency in severity definitions and classifications, we investigated and synthesized the literature to develop a burn severity classification with specific objectives. The classification needed to be useful and applicable to managers, scientists, and society. Also, the classes used in the system needed sufficient flexibility as to whether they could be grouped or used individually, depending upon the need or interest of the person or persons using the classification.

To develop a soil burn severity classification, we synthesized fire intensity, fire severity, and the response literature (*fig. 5*). Fire science has provided the knowledge on fire intensity by describing the variation in heat pulse into the soil (Baker 1929, Debano et al. 1998, Hare 1961, Hungerford et al. 1991, Levitt 1980, Lyon et al. 1978, Wells et al. 1979, White et al. 1996, Wright and Bailey 1982). However, in many circumstances, it is important to understand the amount of fuel consumed by a fire event. Therefore, we also incorporated fire severity into our rationale (Debano et al. 1998, Dyrness et al. 1989, Key and Benson 2001, Morgan and Neuenschwander 1988, Ryan and Noste 1985, White et al. 1996).

Finally, we included what responses might be important to society and provided a link in the burn severity classes (what is left) to management and ecological values (for example, wildlife, soil productivity, erosion) (Debano et al. 1998, Neary et al. 1999) (*fig. 5*).

Table 9—*Forest structural characteristics derived from the Fire and Fuels Extension-Forest Vegetation Simulator (FFE-FVS) (Reinhardt and Crookston 2003) and directly from our data.*

Density characteristics	Characteristics related to fire behavior	Biomass characteristics (Mg/ha)	Miscellaneous characteristics
Trees/ha	Canopy base height from FFE-FVS	Foliage	Average top height
Basal area (m^2/ha)	Canopy bulk density	Live branch < 7.6 cm	Number of stories
Stand density index	Canopy base height direct measure (CBH)[1]	Live branches > 7.6 cm	Species composition
Crown competition factor		Surface	Dominant species
Total canopy cover (TCC) (%)[2]		Total	Quadratic mean diameter
Cubic volume (m^3/ha)			Dry, cold, or moist forest
Average canopy cover (ACC)(%)[3]			Uncompacted crown ratio
			Basal area weighted diameter[4]

[1] CBH is total height minus uncompacted crown length.
[2] TCC is $C' = 100(p_i a_i)A^{-1}$ where: C' = percent canopy cover without accounting for overlap, p_i = trees per acre for the *ith* sample tree, a_i = projected crown area for the *ith* tree in ft^2/acre, and A = ft^2/acre (43560) (Crookston and Stage 1999).
[3] ACC is $C = 100\,[1 - \exp(-.01\,C')]$ where: C = percent canopy cover that accounts for overlap, and C' from TCC (Crookston and Stage 1999).
[4] Basal area weighted diameter breast height (dbh-in) is $\sum$ ((dbh*individual tree basal area (ft^2) * number of trees for each dbh class) divided by ($\sum$ (number of trees * individual tree basal area (ft^2)).

The classification included six levels of soil burn severity based on factors that link fire intensity, fire severity, and the response (*fig. 6*). The factors in the soil burn severity include proportion of litter, mineral soil, and exposed rock present after a fire and the dominant char class, defined as unburned, black, grey, and orange char specific to mineral soil (Debano et al. 1998, Ryan and Noste 1985, Wells et al. 1979).

Level 1 describes places where there is evidence of fire, but not enough to consume litter. Thus, there is greater than 85 percent litter cover for all char classes. Level 2 describes places that have between 40 and 85 percent litter cover for all char classes. Places with less than 40 percent litter cover, with mineral soil exhibiting black char, are represented by level 3, while level 4 represents places with less than 40 percent litter cover and the exposed mineral soil is dominated by grey or white char. Levels 5 and 6 reflect very little litter cover (0 to 5 percent), with level 5 characterized by exposed mineral soil dominated by black char and level 6 characterized by exposed mineral soil dominated by either grey or white char.

For defining tree burn severity, we used an approach similar to the one we used when developing the soil burn severity levels. However, instead of using temperature

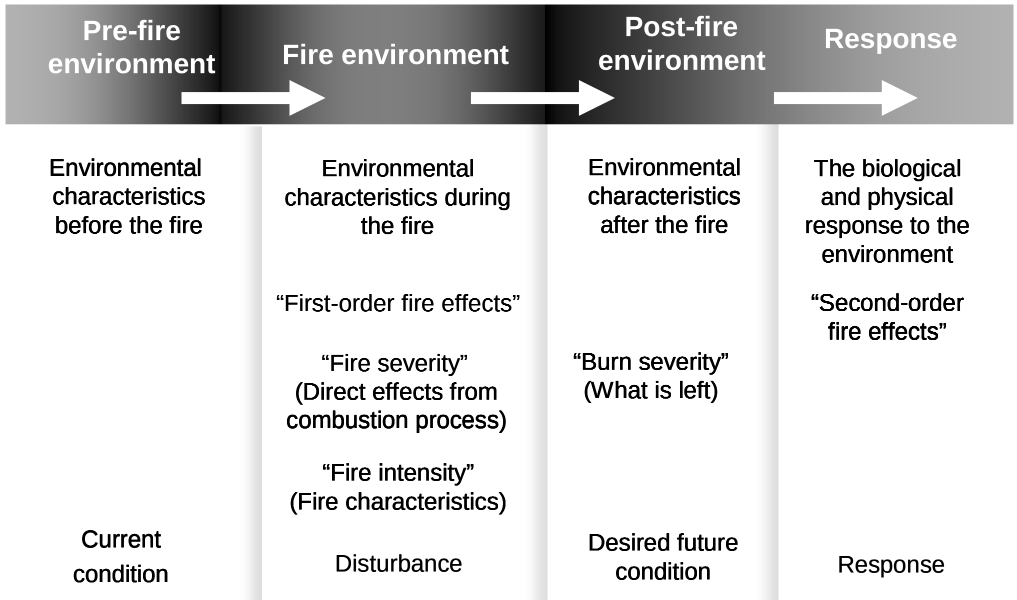

Figure 5—The fire disturbance continuum, of which there are four components, describes the interpretation of different factors involved in fires (Jain et al. 2004). The first component the pre-fire environment, includes forest vegetation and state of the environment (moisture levels, amount of biomass, and species composition). This can also be referred to as the condition just prior to the fire event. The second component, the fire environment, is the environment during the fire event, where fire intensity and fire behavior are characterized in addition to fire severity. Changes to forest components from the fire are also referred to as first-order fire effects. The third component is the environment after the fire is out, referred to as the post-fire environment. This is the environment created by the fire but is also a function of the pre-fire environment and is characterized by what is left after the fire. We refer to this as burn severity. In some cases when fuel treatments are being applied to create a more resilient forest, this could be referred to as the desired condition. The last component is the response, often referred to as second-order fire effects.

to guide the classification, we used flame length to represent fire intensity (Ryan and Noste 1985, VanWagner 1973). Levels of fire severity are dependent upon the amount of tree bole killed or the amount of tree crown scorched or burned by the fire (Peterson and Arbaugh 1986, Ryan and Reinhardt 1988, Weatherspoon and Skinner 1996, Wyant et al. 1986). Tree burn severity is dependent upon the condition of the tree after a fire and, in particular, the portion of the crown and the amount of bole left alive after the fire (*fig. 7*).

The perceived “goodness” of burn severity, or lack there of, depends on the values at risk, the biophysical setting, and/or the management objectives. Therefore, levels of both soil and tree burn severity do not depict a value but rather describe a continuum from a totally unburned forest to a forest in which fire has appreciably altered its pre-fire condition (soil, forest floor, ground level vegetation, trees, and so forth).

Soil and Tree Burn Severity

We combined our six levels of soil burn severity into three levels because we have very few observations of soil burn severity in levels 1 and 6. Level 2 burn sever-

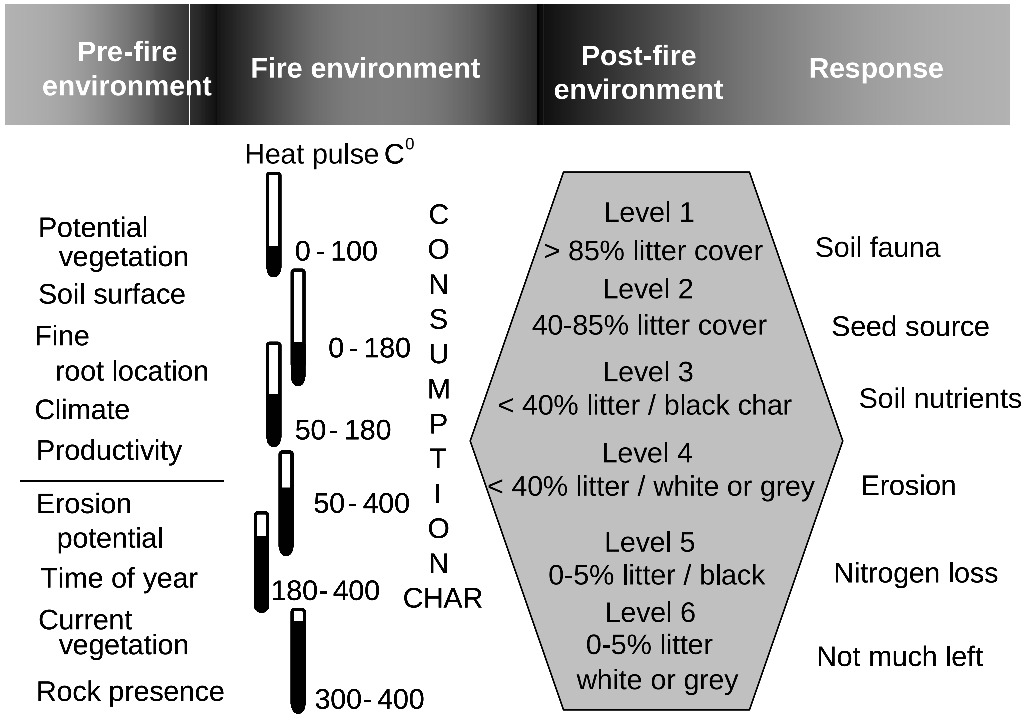

Figure 6—Within the post-fire environment, the soil burn severity classification includes six levels. Going from left to right, a range of temperatures associated with the fire event correspond to the probable indicator of what is left after a fire. For example, to maintain litter cover, the heat pulse into the ground had to be between 0 and 100°C. When surface litter remains, soil fauna are often still alive (level 1). A fire severity description would assume 15 percent litter is consumed. By level 6, the heat pulse into the ground had to exceed 300°C in order to create white ash or a grey charred soil appearance (Hungerford et al. 1991). In a fire severity description, surface nutrients would no longer be present. The char in each burn severity level refers to the dominant char present after the fire.

ity (combined levels 1 and 2, *fig. 6*) consisted of areas with greater than 40 percent litter cover. The forest floor could vary from unburned to areas exhibiting black char, although abundant litter cover existed. Level 4 soil burn severity (combined levels 3 and 4, *fig. 6*) described areas where less than 40 percent litter cover existed and the exposed mineral soil was either black or grey in color. Level 6 soil burn severity (combined levels 5 and 6, *fig. 6*) described sites where there was 0 to 5 percent litter cover and the exposed mineral soil was black, grey, and/or orange colored, or there was an abundance of exposed rock.

We combined our five burn severity levels into four levels to describe trees post-wildfire because we had only a few observations in level 3 tree burn severity (*fig. 7*). The lowest tree burn severity described burned settings in which the trees contained dominantly green crowns (level 1 referred to as containing green crowns, *fig. 7*). The mixed-green tree burn class typified settings in which the trees had greater than 30 percent residual green crown ratio (level 2 referred to as containing mixed green crowns). The mixed brown tree class described stands where all trees had less than 30 percent residual green crown ratio (level 3) and a brown tree class for stands with scorched crowns (level 4). In this study, we combined levels 3 and 4 and referred to these observations as containing brown crowns (*fig. 7*). When black stems and branches were the only tree components left after a wildfire, we used a level 5 tree burn severity to describe these conditions (referred to as containing black crowns (*fig. 7*).

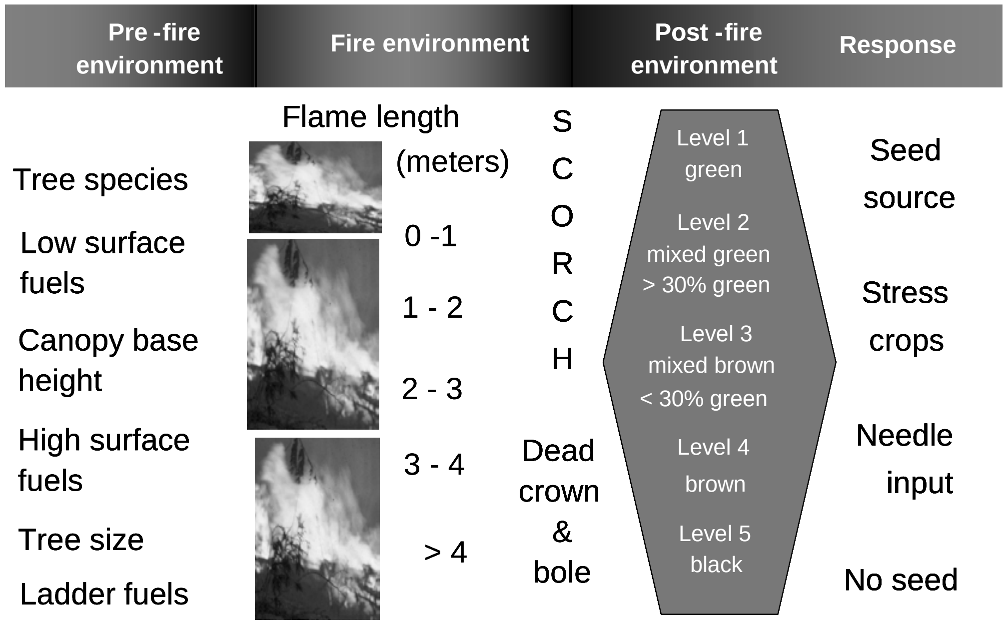

Figure 7—The tree burn severity classification links flame length and amount of crown scorch to burn severity, which indicates the portion of the tree left alive. Ryan and Noste (1985) discussed a conceptual model that described the relation between flame length and crown scorch. We used this model to develop our tree burn severity classes. The lowest tree burn severity class describes settings in which the trees contained dominantly green crowns (level 1). To distinguish between mixed green (level 2) and mixed brown (level 3), we used the proportion of residual crown left alive as an indicator. Greater than 30 percent green indicates this portion of the crown is alive. Trees with a crown ratio greater than 30 percent have a high chance of survival and respond with increased growth after the disturbance (Ryan and Reinhardt 1988, Smith 1986). In contrast, with trees with less than 30 percent of the crown left alive, there is a chance the tree will not survive after the fire. Only a portion of the remaining trees had to contain green crowns to be placed either into the mixed green or mixed brown classes. Brown indicates all trees contained brown needles and no green needles remained (level 4). Black indicates no needles were left on the tree and only black stems and branches remained (level 5).

Analysis and Interpreting Results

The sampling stratification we used was intended to insure the variation in burn severity and forest structure was obtained. The stratification was not used in the analysis, rather, individual fires (categories) and forest structure characteristics (continuous values) were used to predict tree burn severity (categories). A nonparametric classification tree technique (CART) (Breiman et al. 1984, Steinberg and Colla 1997) was used to identify the relation between the predictors and tree burn severity. CART does not require normalizing data through transformations making the results readily interpretable. It identifies interactions, maximizes homogeneity within a particular classification, and can conduct internal cross-validation among classes (a measure of overall performance). The forest structure data were continuous and the burn severity data were categorical, which can be problematic for many analytical techniques that attempt to relate the two (for example, linear regression and analysis of variance). In addition, neither of these techniques identifies thresholds of performance for a given variable.

CART partitions the data using a binary decision process, making it appropriate for both categorical and continuous data. CART produces trees with "nodes" showing where splits (differentiation of the values of a variable into two classes) in the classifications occurred. Based on decision rules, CART classifies observations until all observations are placed in one class, all observations in the node are the same, the node contains equal proportions in the classes, or, as with this analysis, there were 10 observations left to be classified. *Figure 8* shows a 16-outcome classification tree predicting tree burn severity as a function of pre-wildfire forest structure. Outcomes 1 through 16 (shaded) show number of observations correctly classified, total number of observations, and probability of certainty.

Forest characteristics occurring at the top of a classification tree provide an indication that they were clearly related to burn severity compared to characteristics that appear later in the tree. For example, in the classification tree used to predict tree burn severity, wildfire groups (groups of individual fires) were commonly used in the splits, followed by canopy base height, forest type (cold, dry, or moist), and/or total cover and weighted basal area dbh (*fig. 8*). In addition, it identified thresholds of forest structure characteristics that have the strongest relation to a burn severity level. For example, in predicting outcome 1, trees with canopy base height ≤ 1.7m (5.6 ft) split to the left in the classification tree and trees with canopy base heights > 1.7m (5.6 ft) split to the right and went to internode 3.

The value given for a probability of certainty in the CART analysis is a conditional probability (*fig. 8*). An example of a conditional probability is demonstrated by asking the question: what are the chances of a person visiting a particular tire store? Under normal driving situations, the probability of visiting a particular store when four are available is approximately 25 percent. Having a flat tire, however, can dramatically change this probability. If the flat occurs in the neighborhood of a particular store, the probability of visiting that store will likely increase. If the flat tire occurs in the home driveway, the probability of patronizing a store that provides timely home repair will likely increase. These probabilities are conditional upon whether a flat tire has occurred (condition A) and upon the location (condition B) where the flat tire occurred. The CART analysis we performed displays such conditional probabilities of an event happening predicated on a particular situation. For example, if canopy base height in a particular plot averaged less than 1.7m (5.6 ft) (condition B) and occurred in fire group 1 (condition A), there is a 0.52 probability the trees would have green crowns (tree burn severity level 1) (outcome 1, *fig. 8*).

Results and Discussion

Our results suggest that soil burn severity and tree burn severity resulting from wildfires are independent. All three of the soil burn severity levels we identified occurred with all four of the tree burn severities (*fig. 9*). These results indicate that when wildfires burn, there are different pre-fire conditions and fire environments (for example, intensity or behavior) that result in particular soil and tree burn severities.

For example, a low intensity surface fire (slow rate of spread and short flame lengths) can create a level 6 soil burn severity (consume all of the organic forest floor components and change mineral soil color) if a large amount of heat is transferred to the mineral soil for an extended period of time (approximately 10s of minutes to hours). In these situations, because of the short flames (10s of cm, 10s of inches), little crown or bole scorch may occur on the standing trees. An example of such burning could occur in ponderosa pine forests accustomed to frequent low intensity surface fires where, because of fire exclusion, large amounts of surface fuels may have accumulated (Graham 2003, Graham et al. 2004). In contrast, an intense wildfire burning tree crowns, combined with moist soil conditions (for example, lower duff moisture content exceeding approximately 100%), can lead to a level 2 soil burn severity (surface organic layers charred but a large portion of them intact), but leave only blackened stems and branches (level 5 tree burn severity) (*fig. 9*). Fires burning in the boreal forests often typify these burning conditions resulting in different tree and soil burn severities (Dahlberg 2002, Debano et al. 1998). These findings indicate that a composite burn severity integrating both soil and tree burn severity would be difficult. Such a composite could contain many combinations of soil and tree burn severities.

As no two forests in the western United States are identical, the wildfires that burn in them are highly variable in both behavior and burn severity. Nevertheless, we were able to identify seven groups of fires related to tree burn severity (*tables 10, 11*). The grouping of fires in the analysis most likely reflected broad scale attributes such as vegetation type, locale, geography, weather, or other physical setting attributes. Fire group 1 contained the largest number of fires showing similar relations as to how forest structure influenced burn severity. As canopy base height and total cover became relevant to classifying tree burn severity, fire group 1 broke into two additional fire groups (groups 2 and 3) (*table 10, fig. 8*).

The Missionary Ridge wildfire near Durango, Colorado and the Hayman wildfire near Colorado Springs, Colorado occurred in relatively the same geographic area and under similar weather conditions. However, they expressed uniqueness as they classified into separate fire groups early in the CART analysis (*tables 1, 10, 11, fig. 8*). The area burned by the Hayman wildfire (*table 11*) contained rolling topography and was primarily characterized by Douglas-fir/common juniper (*Juniperus communis* L.) or other dry vegetation types (average precipitation 25 cm, 10 in), and was located on the Colorado Rocky Mountain Front Range. In contrast, the area burned by the Missionary Ridge wildfire (*table 10*), located in the San Juan Mountains in southwest Colorado, contained highly variable topography, and tended to be dominated by mixed conifer and/or ponderosa pine, Douglas-fir, and/or oak (*Quercus gambelli* Nutt.) woodlands (average precipitation 48 cm, 19 in) (Casey et al. 1996). Also, these classifications of the wildfires most likely reflected the weather during the fire event. For example, the Keetch-Byram drought index (Keetch and Byram 1988) for the Hayman wildfire averaged 272 while the index for the Missionary Ridge wildfire averaged 382. However, further analysis is needed to evaluate and determine which factor or combinations of factors reflect the different fire groups. These findings indicated that the most telling wildfire characteristic affecting tree burn severity is the wildfire itself and summation of the attributes that determine its occurrence and propagation. These results emphasize the importance of observing burn severity in many different wildfires occurring in different years (weather), forest types (species, potential vegetation), and across geographical areas (for example, northern Rocky Mountains, central Rocky Mountains) (van Mantgem et al. 2001). Our analysis indicated a set of wildfires more than likely had similar characteristics, such as duration, heat produced, physical setting, and geographic location.

Canopy base height, uncompacted crown ratio, and surface fuel conditions are important forest structure characteristics that determine whether a fire will transition from a surface fire to a crown fire (Graham et al. 2004, Peterson et al. 2005, Scott and Reinhardt 2001). Our study indicated that canopy base height was the most important forest characteristic associated with tree burn severity within individual fire groups. However, high canopy base heights, as we surmised, did not always result in green crowns after a wildfire. In fact, we discovered that relatively low canopy base heights of 1.1m (3.5 ft) in fire group 7 (outcome 15), 2.0m (6.5 ft) in fire group 4 (outcome 5), and 1.7m (5.5 ft), in fire group 1 (outcome 1) were important break points in determining tree burn severity (*figs. 8 and 10a*). For example, green tree burn severity (level 1) occurred with a conditional probability of 0.52 for stands occurring in fire group 1, even if they had low canopy base heights ($\leq$ 1.7m, 5.6 ft) (*fig. 8*, outcome 1). With a comparable probability (0.55), a similar green tree burn severity occurred in fire group 4 when canopy base heights were$\leq$ 2.0m (6.6 ft) (*fig 8*, outcome 5). Stands exhibiting these burn characteristics tended to be relatively dense (2100 trees/ha, 850 trees/ac) and relatively short (<12m, 39 ft) compared to many stands we sampled (*figs. 10 b, c*).

In both of these fire groups, thinned stands, plantations, and other stands exhibiting management typified this outcome. The forest floor conditions exhibited in these fire groups could be associated with stand initiation structural stages which frequently contain moist and robust layers of ground-level vegetation. Because these stands were managed, the surface fuel matrix was modified through slash disposal and site preparation activities resulting in a discontinuous fuel bed. Crown fires would burn around these areas and most often there was evidence that firebrands landed in these stands. However, surface fuel conditions prevented sufficient fire from developing that could burn or scorch the tree crowns. These results indicate that high stand densities and low canopy base heights do not necessarily lead to a crown fire or black stems.

The previous examples, because they show that canopy base height impacts tree burn severity at relatively low heights (< 2.0m, 6.6 ft), contradict to some degree what we would expect (Cruz et al. 2002, Graham et al. 1999, Graham et al. 2004, Scott and Reinhardt 2001, Van Wagner 1977). Nevertheless, outcome 6 in our present study reflects the more common notion that high canopy base heights result in low burn severity (*fig. 8*). This outcome illustrates that relatively high canopy base heights (> 6m, 19 ft), occurring on tall trees (22m, 70 ft), with greater than 62 percent cover, results in green tree burn severity (*figs.10a, b, 11*). Although outcome 6 had high tree density (3500 trees/ha, 7413 trees/ac), there was substantial variation. This result may indicate that high overstory tree density shaded out the ground-level vegetation and the high canopy base height prevented the fire from transitioning into a crown fire. This outcome was relegated to one fire group, and it had a high (0.81) conditional probability of occurring. Outcome 7 also illustrates that tall trees with high canopy base heights and very low canopy cover (10 percent), with very low amounts of surface biomass, can result in green tree burn severity (*figs. 10, 11*). This outcome had a high (0.70) conditional probability of occurring and typified the common view that low density forests with high canopy base heights and very little

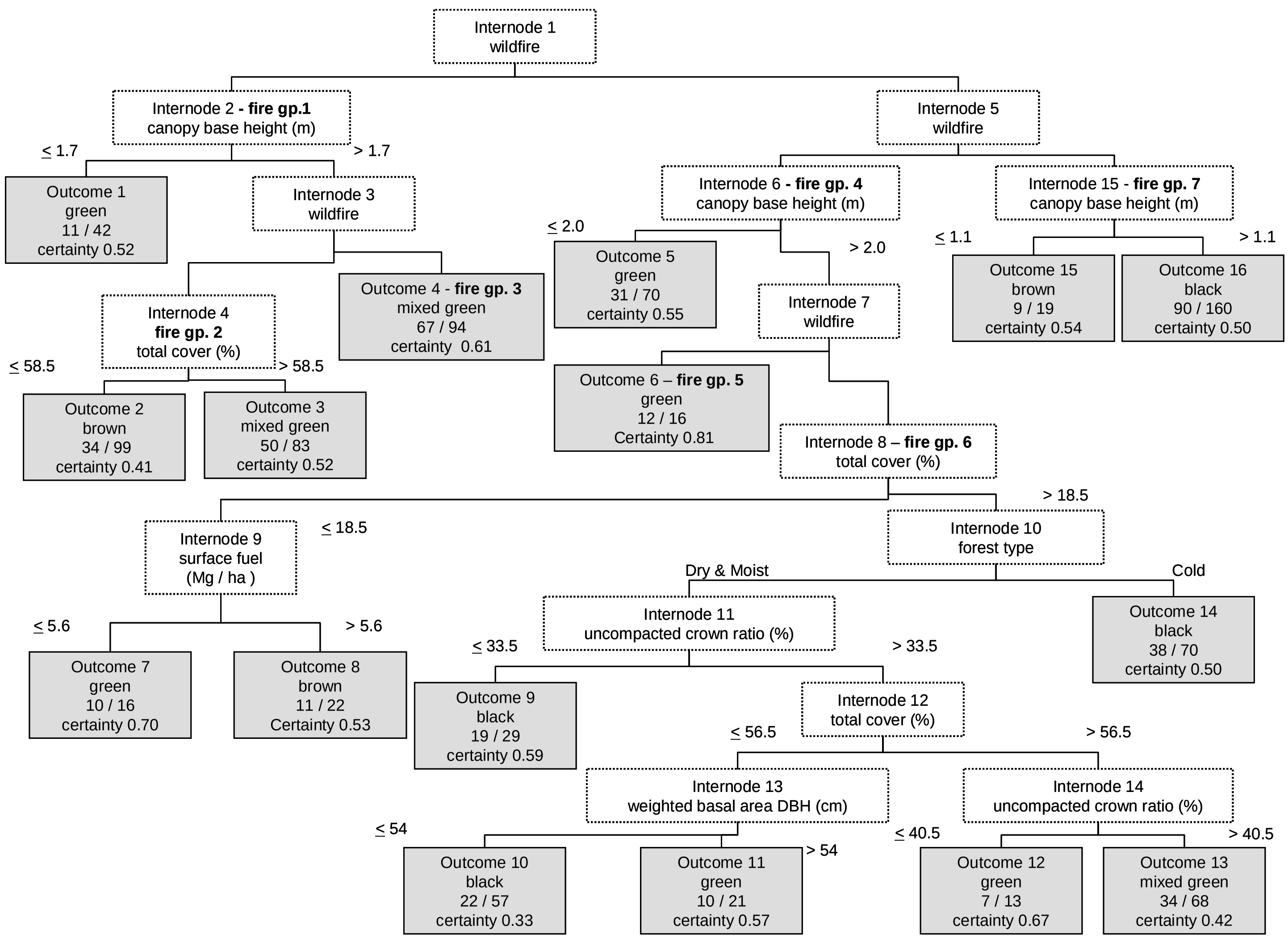

Figure 8—Classification tree for predicting tree burn severity resulting from CART analysis. Shaded areas reflect different predicted outcomes. Each outcome contains the tree burn severity, the number of correctly classified observations versus the total number of observations in the outcome, and a conditional probability referred to as "certainty." The internodes identify the fire group or the forest structural threshold used in predicting a particular outcome.

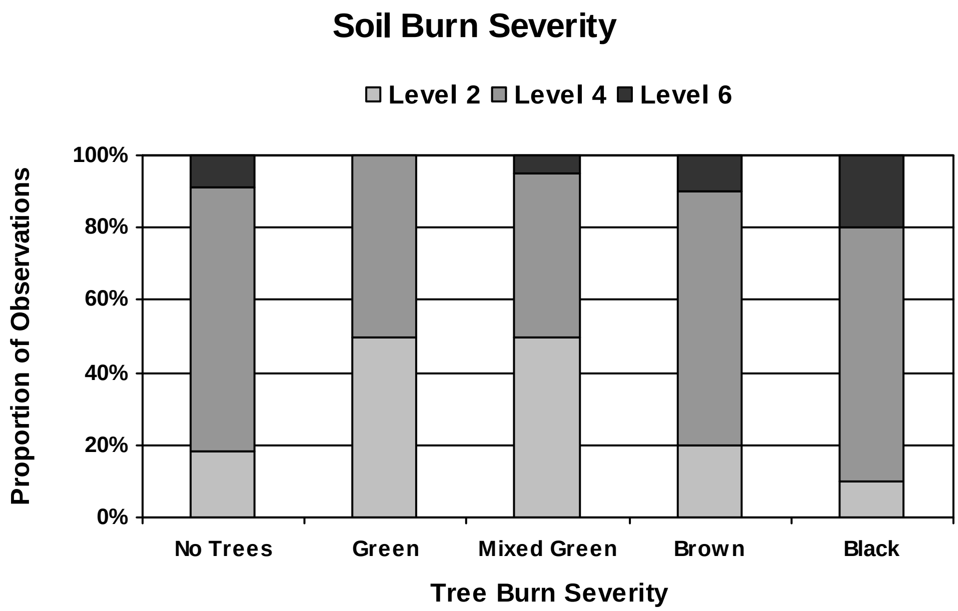

Figure 9—The relation between tree burn severity and soil burn severity is relatively independent. All soil burn severities can occur beneath all tree burn severity classes.

surface fuels are highly resistant to crown fire (Cruz et al. 2002, Graham et al. 1999, Graham et al. 2004, Scott and Reinhardt 2001, Van Wagner 1977).

The winds driving fires in group 7 had the highest minimum and median wind speeds of the wildfires we examined (*fig. 12*). In this fire group, canopy base height was related to tree burn severity, especially within wildfires that tended to burn under extreme conditions (for example, high air temperatures, strong winds, low humidity), such as with the Hayman fire in Colorado (Graham 2003). In this fire group, there was a 0.54 probability of classifying plots with brown tree severity when trees within the plots had mean canopy base heights ≤ 1.1m (3.5 ft.) (*fig. 8*, outcome 15, *fig. 10a*). Within this outcome (15), the tree density was relatively high (1929 trees/ha ± 180 trees/ha, 780 trees/ac ± 73 trees/ac), but there was also considerable variation. Most likely because of this variation and the burning conditions that typified fire group 7, the classified tree burn severity resulted in brown rather than green, which occurred with similar canopy base heights in fire groups 1 and 3. However, in group 7 fires, stands containing trees with a mean canopy base height of > 1.1m (3.6 ft) were classified as having black tree burn severity (probability 0.50) (*fig. 8*, outcome 16). Most likely the relatively high (5m, 16 ft) canopy base heights occurring in these stands allowed sufficient (63.6 Mg/ha, 28.4 tons/ac) live and dead surface fuels to accumulate. These aspects, combined with other factors associated with this group of fires, led to the creation of conditions favoring a crown fire, resulting in black crowns.

Another outcome typifying black tree burn severity occurred in the cold forests, where total cover exceeded 18.5 percent (*fig. 8*, outcome 14). In the burned plots, the trees were relatively tall (15m plus, 50 ft) with canopy base heights exceeding 8m (26.2 ft) (*fig. 10a,b)*. In such dense subalpine fir dominated forests (cold), tree crowns tend to intercept precipitation and evapotranspiration tends to deplete forest

Table 10—*CART uses a hierarchical classification. For predicting tree burn severity, individual fires were placed into seven fire groups. This table shows which fires were placed into fire groups 1 through 3, the forest types that dominated that particular fire group, and the outcome where observations occurred for a particular fire. Within these fire groups, individual forest structure characteristics were identified that related to a tree burn severity.*

Fire-group	Out-come	Forest type C=cold D=dry M=moist	Fire-group	Out-come	Forest type C=cold D=dry M=moist	Out-come	Forest type C=cold D=dry M=moist
1	-	-	2	2	C	3	C
1	-	-	2	2	D	3	M
1	-	-	2	2	C	-	-
1	1	D	2	2	D	-	-
1	-	-	2	2	D, C	3	D, C
1	-	-	2	2	M, C	3	C
1	1	D, M	2	2	D, M, C	3	D, M
1	-	-	2	2	C	-	-
1	-	-	2	2	D	3	D
1	1	D, C	2	2	D, C	3	D, C
1	1	D	2	2	D, M	3	D, M
1	-	-	2	-	-	3	D
1	-	-	2	2	D	-	-
1	-	-	2	-	-	3	C
1	1	C	2	2	D, M, C	3	C
1	-	-	2	2	M	3	M
1	-	-	3	4	C	-	-
1	1	C	3	4	D, C	-	-
1	-	-	3	4	C	-	-
1	-	-	3	4	C	-	-
1	-	-	3	4	C	-	-
1	-	-	3	4	C	-	-
1	1	C	3	4	M, C	-	-
1	-	-	3	4	C	-	-
1	-	-	3	4	D	-	-
1	1	C	3	4	C	-	-
1	-	-	3	4	D	-	-
1	1	M	3	4	D	-	-
1	1	C	3	4	D, C	-	-
1	-	-	3	4	D, C	-	-
1	-	-	3	4	D	-	-
1	1	C	3	4	D	-	-
1	-	-	3	4	M	-	-
1	1	C	3	4	M, C	-	-
1	-	-	3	4	C	-	-
1	-	-	3	4	C	-	-
1	1	D	3	4	D	-	-
1	-	-	3	4	D	-	-
1	1	M	3	4	D, M	-	-
1	1	D	3	4	D	-	-
1	-	-	3	4	D	-	-
1	1	M	3	4	M, C	-	-

Table 11—*CART uses a hierarchical classification. For predicting tree burn severity, individual fires were placed into seven fire groups. This table shows which fires were placed into fire groups 4 through 7, the forest types that dominated that particular fire group, and the outcome where observations occurred for a particular fire. Within these fire groups, individual forest structural characteristics were identified that related to a tree burn severity.*

Fire name	Fire Group	Outcome	Forest type C=cold D=dry M=moist	Fire group	Outcome	Forest type C=cold D=dry M=moist	Outcome `	Forest type C=cold D=dry M=moist	Outcome	Forest type C=cold D=dry M=moist	Outome	Forest type C=cold D=dry M=moist
Bald Hill	4	-	-	5	6	D	-	-	-	-	-	-
Grambauer Face	4	-	-	5	6	D	-	-	-	-	-	-
Little Pistol	4	5	D, C	5	6	D, C	-	-	-	-	-	-
Maloney	4	-	-	5	6	M	-	-	-	-	-	-
McDonald 2	4	-	-	5	6	C	-	-	-	-	-	-
Mink	4	-	-	5	6	D	-	-	-	-	-	-
Thirty	4	-	-	5	6	D	-	-	-	-	-	-
Young J	4	-	-	5	6	M, C	-	-	-	-	-	-
Unknown	4	-	-	6	-	-	8	C	9, 10, 12, 13	D	14	C
Bear	4	5	C	6	7	D	8	D, C	9, 10, 11, 12, 13	D	14	C
Coyote	4	5	C	6	-	-	-		13	D	14	C
Flagtail	4	5	D, M	6	7	D	9, 10, 11, 13	D, M	12	M	14	C
Landowner	4	-	-	6	-	-	8	D, C	9	D	14	C
Maudlow/Toston	4	5	D	6	-	-	-		10, 13	D, M	-	-
Myrtle	4	-	-	6	7	D, M	8, 9	D, M	10, 12	D	11, 13	M
Shellrock	4	5	D, C	6	7	D	8, 9, 11, 12	D	13	D, M	14	C
Taylor Springs	4	-	-	6	-	-	8	D	10, 13	D	14	C
Trail Creek	4	5	C	6	-	-	-	-	13	D	14	C
Blodget	7	15	D	-	16	D, C	-	-	-	-	-	-
Buckskin, Cave G	7	-	-	-	16	D	-	-	-	-	-	-
Clear Creek	7	15	C	-	16	D, C	-	-	-	-	-	-
Crazy H	7	15	M	-	16	M	-	-	-	-	-	-
Diamond Peak	7	-	-	-	16	D, C	-	-	-	-	-	-
Hayman, Moose	7	15	D	-	16	D	-	-	-	-	-	-
Morse,	7	-	-	-	16	C	-	-	-	-	-	-
Mussigbrod, Willie	7	-	-	-	16	C	-	-	-	-	-	-
Razor	7	-	-	-	16	D, C	-	-	-	-	-	-

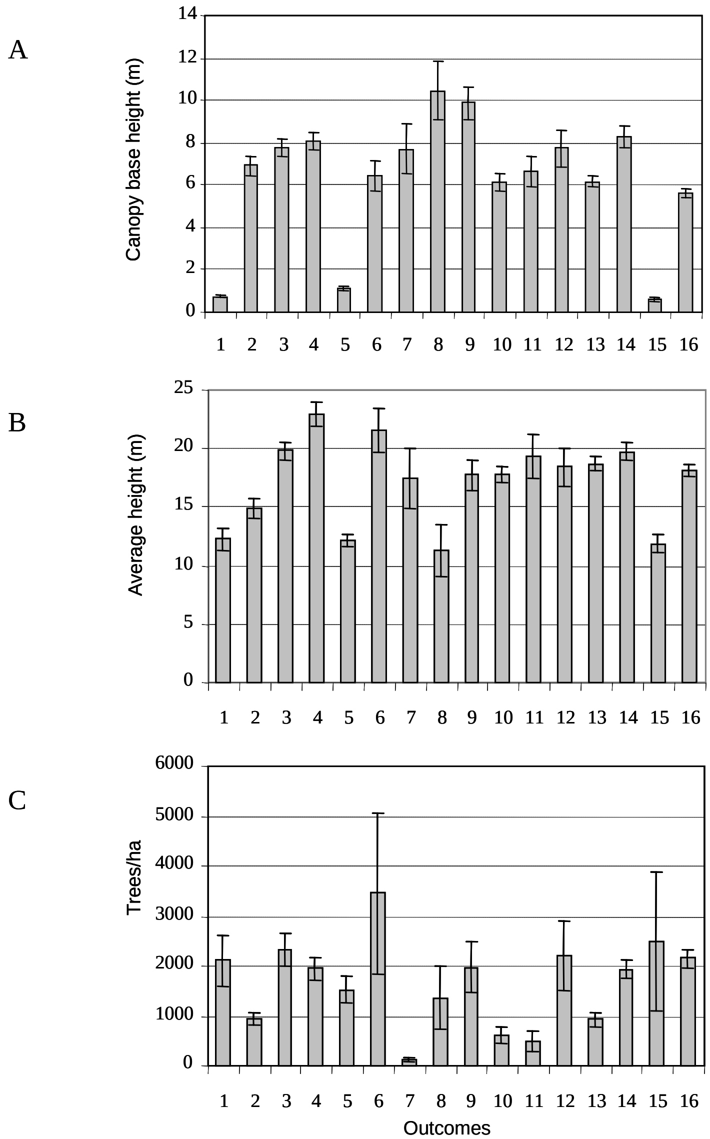

Figure 10—Sixteen outcomes resulted from predicting tree burn severity as a function of forest structure and wildfires. Average canopy base height (A), height (B), and trees/ha (C) are associated with each outcome. Standard error bars are presented to illustrate the variation within and among outcomes.

floor moisture, which can result in dry forest floor conditions (Rutter 1968). These dry surface conditions, coupled with our estimated pre-fire surface fuel loadings exceeding 70.6 Mg/ha (31.5 tons/ac), were probably prime contributors to facilitating surface fire ignitions and the development of sufficient fire intensities to create black crowns. These results indicate that although canopy base height is very important in determining tree burn severity, high canopy base heights may not always protect the needles from being consumed during a fire.

As stated earlier, the forests of the inland western United States are rather complex, both in composition and structure, and the wildfires that burn them are highly variable (Agee 1993, Burns and Honkala 1990, Graham et al. 2004, Hann et al. 1997). Even with this complexity, we were able to show that hierarchal relations exist among forest structure and tree burn severity (*fig. 8*). In this hierarchy (CART tree), the probability of a given forest characteristic influencing a particular tree burn severity is conditional on the previous characteristics occurring in the CART tree. In addition, the characteristics occurring earlier in the classification indicate they are more important in predicting tree burn severity than those listed later. These characteristics are: a particular wildfire group, tree canopy base height, total forest cover, surface fuel amount, forest type, uncompacted tree crown ratio, and tree diameter.

These variables were not only hierarchically related to tree burn severity, but together they predicted green, mixed green, and black tree burn severities very readily. Because we identified four levels of tree burn severity, a random probability of a given severity occurring would be 0.25. Therefore, any probability exceeding 0.25 indicates the additions of forest structural characteristics within a fire group were significantly related to tree burn severity in the cross-validation matrix (*table 12*). The variables, in order of importance, and the relations we identified, classified green crowns with a 0.46 probability, mixed green crowns with a 0.42 probability, and black crowns with a 0.55 probability. However, this same model only predicted brown tree severity with a 0.19 probability (*table 12*).

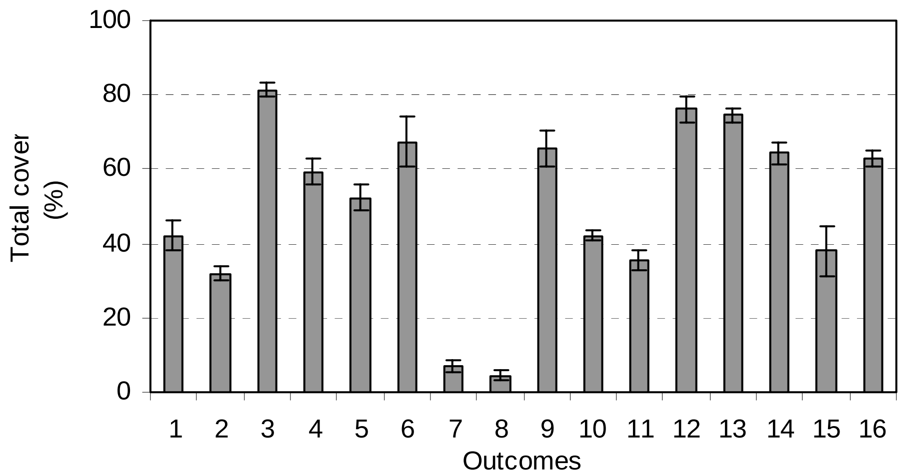

Figure 11—Average total cover in percent for the sixteen tree burn severity outcomes resulting from the classification tree (CART) analysis. Standard error bars are presented to illustrate the variation in total cover within and among outcomes.

These results indicate that wildfire and fuel conditions that create green or mixed green crowns and black crowns tended to be somewhat simpler than those creating brown crowns. For brown crowns to occur, a set of specific conditions needed to exist, such as in outcome 2 and outcome 8 (*fig. 8*). In both these outcomes, observations contained low overstory densities, with less than 35 percent cover for outcome 2 and 10 percent or less cover for outcome 8 (*fig. 11*). Moreover, the difference between outcome 7 (green) and outcome 8 (brown) was a result of very low surface fuels (*fig. 7*). The combination of these conditions could be relatively rare, or there was simply substantial variation when these conditions occurred. This was exemplified in outcome 2, where the probability of certainty was 0.41 (*fig. 8*).

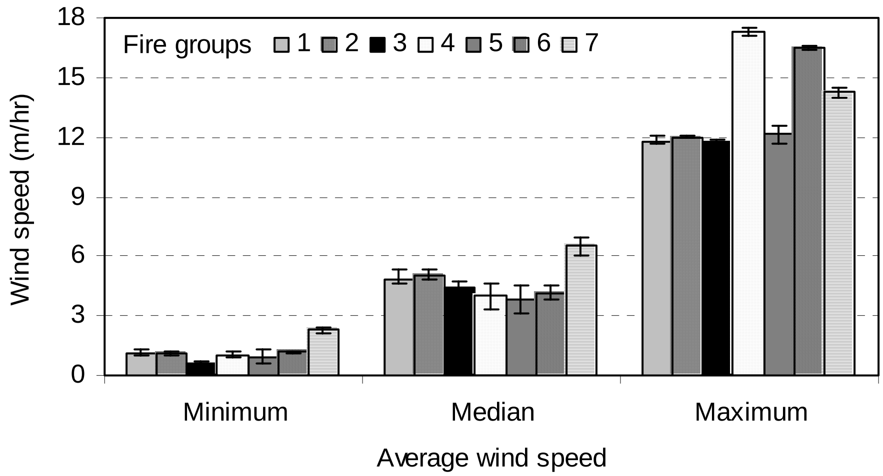

Figure 12—Average wind speeds for three classes: minimum, median, and maximum among fire groups. Standard error bars are presented to illustrate the variation in wind speed within and among the fire groups.

Table 12—*A cross-validation matrix showing how the overall model correctly classified tree burn severity. The highlighted values on the diagonal provide the probability of correctly classifying the actual burn severity given the forest structure characteristics and wildfires used in the classification. Standard errors are presented in parenthesis.*

Actual class	Predicted class			
	Green crowns	Mixed green crowns	Mixed brown & brown crowns	Black crowns
Green crowns	**0.46 (0.04)**	0.14	0.13	0.27
Mixed green crowns	0.20	**0.42 (0.03)**	0.13	0.25
Mixed brown and brown	0.25	0.20	**0.19 (0.03)**	0.36
Black crowns	0.21	0.13	0.10	**0.55 (0.03)**

Conclusion

There are several factors (for example, weather, types of vegetation, fuel moisture, atmospheric stability, physical setting, ladder fuels, surface fuels) that influence fire behavior and burn severity. Forest structure is but one factor (Agee 1996, Graham et al. 2004). Therefore, we did not expect forest structure characteristics to fully explain all of the variation present in burn severity after a wildfire. However, through our study and subsequent analysis, we were able to predict tree burn severity as a function of pre-wildfire forest structure with probabilities far greater than what would have occurred randomly (*table 11*). Throughout the literature, canopy base height has always strongly been associated with fire behavior and with burn severity (Agee 1996, Graham et al. 1999, Graham et al. 2004, Peterson et al. 2005, Scott and Reinhardt 2001). What surprised us was the strong association that canopy base height had with tree burn severity at heights less than 2m (6.4 ft). This is far lower than we expected and, most likely, these low canopy base heights reflect surface fuel moistures, stand structural stages, and past forest management activities. This finding also shows that canopy base height is a forest structure element related to many different forest characteristics. Thus, it relates to fire behavior and tree burn severity in many different ways.

Undoubtedly, intense fire behavior is a primary concern for forest management throughout the western United States. Consequently, fuel treatment to modify this fire behavior becomes a primary consideration (Graham et al. 2004). However, in most circumstances, what a fire leaves behind in terms of soils, homes, and trees is as important, if not more so, than fire behavior. Therefore, fuel treatments need to be designed and implemented to modify burn severity, and the traditional thinned forest with high canopy base heights may not result in the desired burn severity. In fact, the stands with the highest canopy base heights we sampled (10m, 32 ft) had brown or black crowns after a wildfire (*figs. 8, 10*). Stands with canopy base heights less than 1.7m (5.5 ft) had green crowns.

One size does not fit all. Therefore, we would suggest that fuel treatments be designed to consider burn severity as well as fire behavior. In particular, physical setting (forest type, locale, potential vegetation type, and so forth) needs to provide context for planned fuel treatments. Secondly, although high canopy base heights do not always result in reduced burn severity, tree canopy base height needs to be considered when designing fuel treatments. Similarly, reducing total forest cover does not necessarily reduce burn severity. Instead, its interactions with the biophysical setting, canopy base height, and surface fuel amounts and conditions most likely determine burn severity. The last characteristics that we identified as having a relation with tree burn severity, subsidiary to those already mentioned, were forest type, tree crown ratio, and tree diameter. Wildfires burning in the cold forests (subalpine fir) exemplify that high canopy base heights can result in black crowns, especially if the crowns intercept rain and snow, resulting in relatively dry forest floor conditions.

The robust data we accumulated from wildfires that burned throughout the western United States in recent years did not greatly simplify our understanding of the relations between forest structure and burn severity. Nevertheless, we did identify several interactions between forest characteristics and burn severity that have fuel treatment management applications. A significant factor of this work is the estimate of the certainty a forest structure (fuel treatment) will have in modifying burn severity. In addition, the approach we took in identifying the relations between forest structure and burn severity, and the level of certainty we provided, was conditional on the circumstances in which the forest characteristic occurred. This kind of information will be of value when communicating the importance forest structure (fuel treatments) has on determining the aftermath of wildfires. This paper and the analysis and results we reported are a continuation of our work in understanding how forest structure interacts with wildfires, their physical setting, and burning conditions to create a particular burn severity.

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Fire Performance in Traditional Silvicultural and Fire and Fire Surrogate Treatments in Sierran Mixed-Conifer Forests: A Brief Summary[1]

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Abstract

Mixed conifer forests cover 7.9 million acres of California's total land base. Forest structure in these forests has been influenced by harvest practices and silvicultural systems implemented since the beginning of the California Gold Rush in 1849. Today, the role of fire in coniferous forests, both in shaping past stand structure and its ability to shape future structure, is a central force driving both the direction and political debate around forest management on public lands. The purpose of this paper is to demonstrate stand structures which contribute to effective fuel treatments and to provide data which will help managers design desired conditions for future fuel treatment projects. Dr. Jim Agee and Carl Skinner have outlined four principles of fuel treatments which should be integrated when implementing treatments with a goal of enhancing fire resiliency. Stand structures which performed the best with respect to potential fire behavior incorporated most or all of the four principles of fuel reduction. Modification of fire behavior and severity will likely continue to be a driving force in forest management. In most cases, this goal will have to be integrated with multiple forest values and uses, particularly on public lands.

Introduction

Mixed conifer forests cover 7.9 million acres (7.8 %) of California's total land base (CDF 2003a). Forest structure in these forests has been influenced by harvest practices and silvicultural systems implemented since the beginning of the California Gold Rush in 1849. These management practices were partially a reflection of land use, economic trends, and societal values of forestlands during different periods of development of the Sierra Nevada Range (Beesley 1996). With the on-set of the California Gold Rush, harvesting of Sierran forests was associated with providing wood for mines and their associated towns and residences (Beesley 1996); it is noted that photographs and sketches from mining communities depicted "…barren environments around mining settlements" (Beesley 1996). Much of the area around Lake Tahoe was heavily logged in the late 1800's (Taylor 2004), and much of the wood removed from the Lake Tahoe Basin was used in the Comstock silver mining

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region of Nevada (Landauer 2004, Peterson 1996).

With the development of the transcontinental railroad, demand increased for wood to build the Central Pacific and other Sierran Railroads (Beesley 1996). It was estimated that over 300 million board feet alone was needed to construct the wooden snow sheds near the western summit of the railroad (Beesley 1996). Use of narrow gauge railroads for logging increased the efficiency of wood removal (Beesley 1996, Stephens, 2001, Young 2003) and favored the removal of both ponderosa pine (*Pinus ponderosa* Dough. Ex. Laws) and sugar pine (*Pinus lambertiana* Dougl.) over Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco), white fir (*Abies concolor* Gord. & Glend.), and incense cedar (*Calocedrus decurrens* [Torr.] Floren.)(Polkinghorn 1984). The emphasis on the removal of pine was observed by John Leiberg (1902), who was surveying forest conditions of lands in present day El Dorado, Plumas, and Tahoe counties. Leiberg (1902) described his ideas of what future forest conditions might be under these types of harvest practices as follows: *"The Old Forest of the west slope of the Sierra will have been cut away, and the young growth will consist largely of red fir, white fir, and incense cedar."* After World War II ended, demand for lumber increased on Federal Lands (Beesley 1996). This demand led to an increasing emphasis on silvicultural prescriptions which maximized growth and yield (Hirt 1994).

Today, the role of fire in coniferous forests, both in shaping past "stand structure" (definition Helms 1998) and in its ability to shape future structure, is a central force driving both the direction and political debate around forest management on public lands (Stephens and Ruth 2005, USDA 2004). Management of public lands now has an emphasis placed on creating stand structures which have some improved level of fire resiliency. This paper compares and discusses results from two recently published papers (Stephens and Moghaddas 2005a, c). The purpose of this paper is to demonstrate stand structures which contribute to effective fuel treatments, and to provide stand structure data which will help managers design desired conditions for future fuel treatment projects.

Methods

Study Site

Both studies were undertaken in Sierra Nevada mixed conifer forests in the north-central Sierra Nevada at the University of California Blodgett Forest Research Station (Blodgett), approximately 20 km east of Georgetown, California. Blodgett Forest is located at latitude 38° 54′ 45″ N, longitude 120° 39′ 27″ W, between 3,600 and 4,600 ft above sea level, and encompasses an area of 4,400 acres. Species composition, site productivity, and management history (Olson and Helms 1996) of Blodgett forests are representative of 420,000 acres of high site California mixed conifer forestland (Davis and Stoms 1996, Hickman 1993, Mayer and Laudenslayer 1988).

Silvicultural Treatments

Seven fire and fire surrogate and traditional silvicultural treatments are discussed in this paper. Further details on treatment and statistical analysis of vegetation structure, coarse woody debris and fuel characteristics, and fire performance of these stands can be found in three recently published papers (Stephens and Moghaddas 2005a, b, c). The seven silvicultural treatments, which are the four fire and fire surrogate treatments (FFS), and the three traditional silvicultural methods (Traditional), are described below.

No Treatment (FFS)

This treatment assesses the effectiveness of no treatment in managed, second growth mixed conifer forests. The no treatment units had been previously thinned from below (in harvests prior to initiation of the study) using a lop and scatter treatment of activity slash (CDF 2003b) with retention of sub-merchantable material (less than 10 inches dbh).

Fire Only (FFS)

The fire only units had been previously thinned from below (in harvests prior to initiation of the study) using a lop and scatter treatment of activity slash (CDF 2003b) with retention of sub-merchantable material (less than 10 inches dbh). Fire only units were burned with no pre-treatment of fuels except felling of snags and removal of ladder fuels adjacent to fire lines for firefighter safety. Ignition was completed using strip head-fires (Martin and Dell 1978), one of the most common ignition patterns used to burn forests in the western US. All prescribed burning (fire only and mechanical plus fire treatments) was conducted during a short period (10/23/2002 to 11/6/2002) with the majority of burning being done at night (Knapp et al. 2004). Night burning was preferred because relative humidity, air temperature, wind speed, and fuel moistures were within pre-determined levels to produce the desired fire effects.

Mechanical Only (FFS)

Mechanical only treatment units had a two-stage prescription. In 2001, stands were commercially thinned from below to maximize crown spacing while retaining 125 to 150 ft^2/ac of basal with the silvicultural goal to produce an even species mix of residual conifers. Slash treatment was a lop and scatter of limb wood and tree tops from the harvested trees to an average depth of less than 30 inches (CDF 2003b). Following the commercial harvest, approximately 90 percent of understory conifers and hardwoods between one and 10 inches diameter at breast height (dbh) were masticated in place using an excavator mounted rotary masticator.

Mechanical Plus Fire (FFS)

Mechanical plus fire experimental units underwent the same treatment as mechanical only units, but in addition, they were prescribed burned using a backing fire (Martin and Dell 1978).

Individual tree selection (Traditional)

Thinning of trees across all diameter classes favoring removal of damaged, diseased, and suppressed conifers. The silvicultural goal is to recruit new cohorts of conifers and hardwoods, and to develop and maintain an uneven sized forest structure with multiple canopy layers. Minimum size of trees harvested are typically at least 10 inches dbh. Sub-merchantable material (less than 10 inches dbh) is retained. Post harvest fuel treatment includes lop and scatter of limb wood and tree tops from the harvested trees to an average depth of less than 30 inches (CDF 2003b).

Thin from below (Traditional)

Low thinning favored residual forest composed of largest diameter trees in stand. Minimum size of trees harvested are typically at least 10 inches dbh; sub-merchantable material (less than 10 inches dbh) is retained. Post harvest fuel treatment includes lop and scatter of limb wood and tree tops from the harvested trees to an average depth of less than 30 inches (CDF 2003b). The silvicultural goal is to produce an open understory structure with many large overstory trees.

Overstory Removal (Traditional)

This method called for the removal of all trees greater than 18 inches dbh while meeting minimum stocking standard (125 ft^2/acre) (CDF 2003b). Sub-merchantable material (less than 10 inches dbh) is retained. Post harvest fuel treatment includes lop and scatter of limb wood and tree tops from the harvested trees to an average depth of less than 30 inches (CDF 2003b). The silvicultural goal is to release intermediate and suppressed trees and maximize harvest volume.

Statistical Assessment of Vegetation Structure, Fuels Characteristics, and Fire Performance

Vegetation was measured using 1/10th acre circular plots installed in each treatment unit on a systematic grid. Tree species, dbh, total height, height to live crown base, and crown position (dominant, co-dominant, intermediate, suppressed) were recorded for all trees greater than 4.5 inches dbh. Similar information was also recorded for all trees greater than 4.5 feet tall on a 1/100th acre nested subplot in each 1/10th acre plot. Surface and ground fuels were sampled with transects at each of the plots using the line-intercept method (van Wagner 1968, Brown 1974).

Fire behavior was modeled under the upper 90th percentile fire weather conditions. Percentile weather was computed using Fire Family Plus (Main et al. 1990). Forty-one years (1961 to 2002) of weather data from the Bald Mountain Remote Access Weather Station (NFAM, 2004), 2.5 miles west of Blodgett Forest, were analyzed with Fire Family Plus Software to determine percentile weather conditions. Fuels Management Analyst was used to model fire behavior, crowning index, torching index, scorch height, and tree mortality (Carlton, 2004). Torching and crowning indices are the 20-foot wind speed required to initiate torching (passive crown fire) or sustain a crown fire (active crown fire) within a stand (Scott and Reinhardt 2001).

Analysis is based on three replicates of each silvicultural system described. Analysis of variance (ANOVA) was used to determine if significant differences ($p < 0.05$) existed in vegetation structure (trees ac^{-1}, basal area ac^{-1}, height to live crown base, tree height, crown bulk density, and quadratic mean diameter), stand density index, 1-100 combined fuel load, crowning, and torching index for each silvicultural system. If significant differences were detected, a Tukey-Kramer HSD test was performed to determine which specific silvicultural system or reserve was different from another (Zar 1999). The Jump Statistical Software package (Sall et al. 2001) was used in all analyses.

Results

Vegetation, fuel, and fire performance characteristics are summarized in *table 1*. Stand structure of traditional silvicultural and Fire Surrogate treatments were statistically similar in terms of stand density and tree height to crown base. The quadratic mean diameter of the Fire Surrogate mechanical plus fire was significantly higher than all other treatments. While tree height to crown base were statistically similar between treatments which did not incorporate fire, the relatively lower height to crown base and relatively higher surface fuel loads in traditional silvicultural systems affected the modeled potential for torching in these stands. Traditional silvicultural systems, which did not include removal of sub merchantable material and used only a lop and scatter of activity fuels, typically had a relatively higher likelihood of torching than fire surrogate treatments, which incorporated burning as a surface fuel treatment (*table 1*). The Fire Surrogate mechanical plus fire treatment had the lowest potential for crown fire when compared with the traditional thin from below treatment and the Fire Surrogate Study fire only treatment. Crown fire potential in the individual tree selection, overstory removal, and Fire Surrogate Study mechanical only treatments were similar.

Discussion

An understanding of stand conditions which meet fire performance goals is critical to effective fuel treatment planning. There is consensus that reducing surface, ladder, and some degree of canopy fuels, in conjunction with each other, can mitigate fire behavior at a stand level (Peterson et al, 2005). There is also general consensus that “no treatment”, particularly in second growth stands that have been subjected to past harvest and fire suppression, will not improve fire performance in a given stand (Agee 2002, Stephens and Moghaddas 2005a, c). Data from the studies discussed support these fuel reduction concepts. Agee and Skinner (2005) have outlined four principles of fuel treatments which should be integrated when implementing treatments with a goal of enhancing fire resiliency (these principles were discussed by Dr. Agee in his presentation at this conference). Implementation of these treatments should emphasize them in order from (1) to (4) at the stand level: (1) reducing surface fuels, (2) increasing height to live crown base, (3) decreasing crown density, and (4) retaining the largest trees in the stand through thinning. Implementation of these treatments should emphasize these treatments in order (1-4) at the stand level. The importance of treating surface fuels as part of a larger fuel treatment strategy has been identified in previous research (Stephens 1998).

In this comparison, stands which performed the best with respect to potential fire behavior incorporated most or all of the four principles of fuel reduction (Agee and Skinner 2005, Peterson et al. 2005). Fire Surrogate treatments (fire only and mechanical plus fire), which implemented surface fuel, ladder fuel, and crown fuel reduction while retaining the largest trees in the stand, performed best. Performance of the Fire Surrogate mechanical treatments was followed next by the traditional thinning from below. Within traditional silvicultural treatments, individual tree selection and overstory removal performed most poorly with respect to torching, though they did have a higher crowning index than the FFS fire only treatment and the traditional thinning from below. The FFS no treatment performed most poorly in all stands, indicating that previously managed second growth stands with residual activity fuels may need additional treatment to improve stand level fire performance. On the other hand, use of a whole tree harvest system, which include removal of sub-merchantable materials (<10-inch dbh) and/or prescribed burning of surface fuels, may limit deposition of harvest related activity fuels (Agee and Skinner 2005) and somewhat improve performance of traditional silvicultural systems.

When implementing fuel treatments, it is important to consider the short- and long-term tradeoffs that come with modifications of stand structure. Traditional silvicultural systems typically maintain higher stocking levels than Fire Surrogate

Table 1. *Average vegetation, fuel, and fire performance (torching and crowning index) characteristics and standard errors of seven silvicultural treatments that were the four fire and fire surrogate treatments (FFS) and three traditional silvicultural methods (Traditional). Mean values in a column followed by the same letter are not significantly different (P > 0.05).*

	Tree height (feet)		Height to live crown base (feet)		Trees per acre		Basal area (ft^2 per acre)		Quadratic mean diameter (inches)		Stand density index (Reineke, 1933)		Crown bulk density (pounds per ft^3)		Total 1, 10, 100 hour fuels (tons per acre)		Torching index (miles per hour)		Crowning index (miles per hour)	
	Ave	*SE*	Ave	*SE*	Ave	*SE*	Ave	*SE*	Ave	*SE*	Ave	*SE*	Ave	*SE*	Ave	*SE*	Ave	*SE*	Ave	*SE*
No Treatment (FFS)	51.1^{d}	*2.5*	24.6	*1.8*	49.2^{a}	*34.1*	246.2^{a}	*13.0*	10.1^{bc}	*0.1*	452^{a}	*26*	0.005^{ab}	*0.0004*	6.4	*0.6*	18^{c}	*0*	21^{d}	*1*
Fire Only (FFS)	58.5^{cd}	*1.5*	24.2	*1.0*	178.7^{c}	*13.0*	208.3^{a}	*10.6*	14.7^{b}	*0.2*	329^{ab}	*18*	0.0045^{abcd}	*0.0002*	2.0	*0.5*	308^{b}	*24*	22^{c}	*1*
Mechanical plus Fire (FFS)	74.6^{ab}	*3.1*	31.2	2.5	96.7^{c}	*8.5*	171.3^{ab}	*11.0*	18.2^{a}	*1.4*	248^{b}	*10*	0.0026^{d}	*0.0001*	2.1	*0.2*	415^{a}	*10*	33^{a}	*1*
Mechanical Only (FFS)	67.0^{bc}	*1.9*	31.1	*1.8*	173.6^{c}	*56.5*	178.5^{ab}	3.4	14.8^{b}	2.2	285^{ab}	22	0.0028^{c}	*0.0001*	7.6	*0.3*	46^{c}	*4*	31^{abc}	*1*
Thin From Below (Traditional)	52.8^{de}	*1.5*	33.1	*1.2*	229.6^{ac}	*18.9*	209.7^{a}	*27.7*	12.9^{bc}	*0.3*	346^{ab}	*42*	0.0047^{abcd}	*0.0005*	7.5	*2.4*	46^{c}	*4*	23^{c}	2
Individual Tree Selection (Traditional)	51.6^{de}	*2.2*	20.6	*3.8*	232.4^{ac}	*53.5*	172.2^{ab}	*18.2*	11.9^{bc}	*0.7*	296^{ab}	*38*	0.0038^{abcd}	*0.0003*	11.4	*3.3*	34^{c}	*9*	27^{abcd}	*1*
Overstory Removal (Traditional)	44.5^{e}	*5.3*	21.4	*6.2*	258.8^{ac}	*85.0*	110.3^{b}	*35.9*	9.0^{c}	*0.8*	211^{b}	*65*	0.0041^{abcd}	*0.0007*	10.0	*3.4*	36^{c}	*10*	26^{abcd}	*4*

treatments. Excessive reduction of canopy fuels can reduce annual volume growth, thereby possibly hindering long-term volume growth strategies. Maintaining higher stand densities may leave stands prone to other risks, including insect attack (Powell 1999). It is also important to understand the role of regeneration in any silvicultural system in order to sustain desired species and age class composition over the long term. The range of light conditions created by different stand structures may favor the germination and growth shade of intolerant species, depending on light availability at the forest floor (Ansley and Battles 1998). Finally, where feasible, treatments should be placed on the landscape to optimize their effectiveness in conjunction with past treatments or fires, topography, local weather patterns, adjacent vegetation types, and protection of resources at risk (Stratton 2004, Finney 2001).

Conclusion

Current stand structure across much of the coniferous forests of the Sierra Nevada has been heavily influenced by past management practices. Many past silvicultural systems did not emphasize fire performance as a silvicultural objective. Primary fuel surface treatments consisted of lop and scatter treatments, which tended to increase surface fire intensity until slash had adequately broken down. In dry forest systems, slash in the 100- and 1,000-hour size classes can remain in Sierra Nevada forests for 20-30 years (Stephens and Moghaddas 2005c). It is important for managers to understand the history of the stand prior to writing prescriptions. Even-aged, relatively young (less than 150 year old) stands that have been exposed to decades of past management practices will need to be treated differently than older stands that have had little impact of past harvest but have been exposed to intensive fire suppression. Integrating surface and ladder fuel treatments into silvicultural prescriptions early in the planning process is crucial. Selecting harvest systems (whole tree vs. traditional lop and scatter) can affect the amount of fuels left on site (Agee and Skinner 2005).

Modification of fire behavior and severity will likely continue to be a driving force in forest management, particularly on public lands. In most cases, this goal will have to be integrated with other forest values and constraints including protection of water resources, scenic values, air quality, wildlife habitat, recreational use, and limited budgets. Integration with these other values may decrease the effectiveness of fuel treatments if they require retention of ladder fuels or limit post-treatment prescribed burn activities.

Acknowledgements

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Delayed Conifer Tree Mortality Following Fire in California[1]

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Abstract

Fire injury was characterized and survival monitored for 5,246 trees from five wildfires in California that occurred between 1999 and 2002. Logistic regression models for predicting the probability of mortality were developed for incense-cedar, Jeffrey pine, ponderosa pine, red fir and white fir. Two-year post-fire preliminary models were developed for incense-cedar, Jeffrey pine, ponderosa pine and white fir. Three- and four-year post-fire models are presented for white fir and red fir, respectively. Mortality was predicted using percent crown length kill and cambium kill in all optimal models. Diameter at breast height was also a significant variable in all models except for red fir. A pre-bud break model for pine using crown length scorch was also developed. Additional models are provided for each species without the cambium injury variable to show the predictive capability lost when this variable is not assessed. A comparison between bark char classification and cambium condition status was also performed to determine the validity of using bark char classifications as a surrogate for cambium sampling. Light and deep bark char codes are relatively accurate in predicting live and dead cambium, respectively. However, the moderate bark char rating is not a good predictor of cambium status.

Introduction

The number of forested acres burned by high intensity wildfire in California has increased over the past several years. This increase is generally attributed to high stand densities of smaller diameter trees and the accumulation of dead fuels that have developed in response to management activities such as fire suppression. High intensity forest fires typically result in considerable tree mortality. This mortality can be immediate, due to the complete consumption of living tissue during the fire, or can be delayed, occurring over the course of a few years, as a result of fire injuries to the crown, bole, and roots and subsequent insect activity. The ability to accurately predict the probability of mortality of these fire-injured trees is critical when making most post-fire management decisions. Post-fire management activities, such as salvage logging, fuels treatments, and reforestation, are often based on economics and ecological considerations, both of which need to account for the current and expected levels of tree mortality.

Although there are numerous publications reporting findings with respect to fire injuries and conifer tree survival from many areas across the western United States

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(Lynch 1959, Bevins 1980, Peterson and Arbaugh 1986, Ryan and Reinhardt 1988, Ryan and Frandsen 1991), publications based on similar work completed in California are limited. Existing publications cover a limited number of species from single fires (Mutch and Parsons 1998, Borchert et al. 2002), provide a good description and method of application of the criteria for survival but present only generalizations as marking guidelines (Wagener 1961), or attempt to relate prescribed fire characteristics to mortality of mixed conifer species for use in achieving prescribed fire objectives, not to provide salvage marking guidelines from post-fire measurements (Stephens and Finney 2002).

Most researchers that have examined the effect of fire on conifers have concluded that crown injury is the most important predictive variable for mortality (see Fowler and Sieg 2004 for review). Crown injury is typically quantified by percentage of crown killed, described by either a length or height measurement (Herman 1954, Bevins 1980, Harrington 1993), or by percentage of crown volume killed (Reinhardt and Ryan 1989, Finney 1999, Weatherby et al. 2001, Borchert et al. 2002, McHugh and Kolb 2003).

In addition to crown injury, cambium death caused by lethal heating of the tree bole is another important predictive variable for mortality. Duff smoldering around the base of trees or extensive flame exposure to the bole can cause some level of cambium injury, which is dependent on bark thickness (Reinhardt and Ryan 1989). The fact that fire-killed cambium can contribute to subsequent tree mortality is rarely disputed. However, how much dead cambium causes tree mortality, either alone or in combination with crown kill, and how to accurately assess cambium condition while limiting direct cambium sampling, has not been determined for many species.

Various methods have been used to quantify cambium kill, from direct sampling of the cambial tissue (Bevins 1980, Ryan et al. 1988, Peterson and Arbaugh 1989, Ryan and Frandsen 1991) to indirect measures, such as amount or height of bark scorch or degree of bark char (Herman 1954, Peterson 1984, Wyant et al. 1986, Finney 1999, Borchert et al. 2002, McHugh and Kolb 2003). Limited studies exist that have analyzed bole char characteristics and compared them to actual cambium kill. Ryan (1982a) indicates that the depth of bark charring is not an adequate indicator of cambium kill without correlating cambium condition for each depth of char class. Ryan et al. (1988) found the number of dead cambium samples to be the most important predictor of mortality for Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco). They concluded that the ability to accurately predict mortality may be greatly limited if cambium kill is not considered. Wagener (1961), the only author prior to our study to assess cambium condition from work in California, concluded that, even in a single species, there are wide differences in bark thickness, depth of crevices, and size of bark ridges. For application in marking guidelines, he suggested using degree and location of bark char to determine where to sample for cambium injury.

The objective of our study was to develop mortality models for five conifer species in California that land managers can use to predict post-fire tree mortality. We used percent crown length kill, cambium kill, diameter at breast height (dbh), and insect attacks as potential variables for all models. We also compared the accuracy of established bole char classifications in predicting the degree of cambium injury.

Methods

Study Sites

A total of 5,246 fire-injured trees were characterized and monitored for survival in five wildland fires that occurred between 1999 and 2002 in California (*table 1*). The data set included a range of tree sizes and fire injuries for incense-cedar (*Calocedrus decurrens* [Torr.] Florin), Jeffrey pine (*Pinus jeffreyi* Grev. & Balf.), ponderosa pine (*Pinus ponderosa* Laws), red fir (*Abies magnifica* A. Murr.) and white fir (*Abies concolor* [Gord. & Glend.] Lindl.). For the purposes of the analyses, Jeffrey pine and ponderosa pine were combined into one yellow pine group because of similar physical characteristics. Study sites comprised a wide geographical area in California extending from the southern end of the Cascade range to the southern end of the Sierra Nevada range (*fig. 1*). All fires were in the Sierra Nevada mixed conifer forest type (SAF Type 243), with the exception of the Cone fire, which occurred in the Interior ponderosa pine type (SAF Type 237) (Eyre 1980) (*table 2)*.

Table 1—*Number of trees by fire and by species. Yellow pine includes ponderosa and Jeffrey pine.*

Fire	Red Fir	White Fir	Incense-Cedar	Yellow Pine
Bucks	112	124	-	-
Cone	-	-	-	923
McNally	-	1866	781	1046
Star	-	199	-	-
Storrie	94	101	-	-
Total	206	2290	781	1969

Table 2—*Date of wildfire occurrence and general site characteristics.*

Fire	Month/year burned	Elevation (m)	Forest Type	MAP (cm)
Bucks	August, 1999	1400-1500	Sierra Nevada mixed conifer	152-178
Cone	September, 2002	1750-1800	Interior ponderosa pine	50-75
McNally	July, 2002	1700-2750	Sierra Nevada mixed conifer	50-75
Star	August, 2001	1550-1950	Sierra Nevada mixed conifer	152-178
Storrie	August, 2000	1650-1950	Sierra Nevada mixed conifer	152-178

Sampling

Personnel from local forest districts identified sites of mixed fire severity for inclusion in this project. For the Cone and McNally fire, individual fire-injured trees were selected from these areas in an attempt to fill a matrix of different crown and cambium injury levels, size classes, and species. Crews were instructed to fill each category with 30 trees from all the available trees in the area. In any given area, some trees may not have been sampled because they fit into a category that was already filled, or they may have been inadvertently missed as the crew worked the stand.

Although the target of 30 trees for every category was not met, this sampling gave us a broad range of fire injuries and size classes needed to test our objectives. For all other fires, crews selected trees with higher levels of crown kill, but were given no size or cambium injury selection criteria.

Initial assessments of tree condition were completed during the summer of the year following the fire for all fires with the exception of the Bucks Fire. Initial

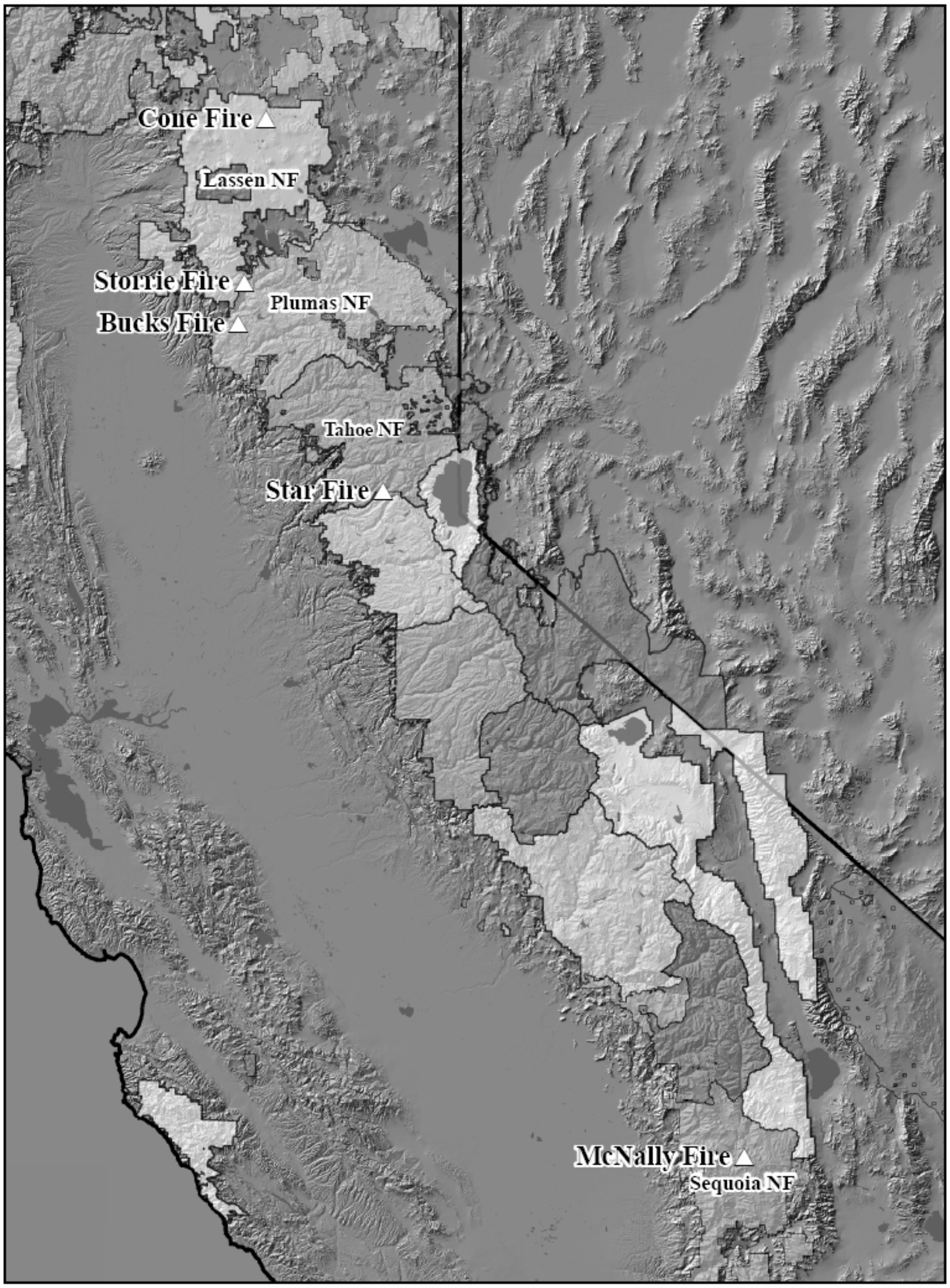

Figure 1--Map of fires included in analysis.

assessment for trees in the Bucks Fire occurred two years post-fire. For all Jeffrey and ponderosa pines, initial assessment occurred after bud break, as recommended by Wagener (1961). Tree status was reassessed annually for all fires. The last measurement year was 2004. A one-year post-fire status assessment was completed in the fall of the same year as the initial assessment for trees in the Cone, McNally, and Star fires. Trees were recorded as dead if no green foliage was visible, or if trees had either fallen or snapped off. Data collected included species, dbh, tree height, crown length killed, crown volume killed (McNally and Cone only), cambium kill rating (CKR), and post-fire insect activity. For pine trees in the McNally and Cone fires, length and volume estimates were also obtained for crown scorch. A bark char code (Ryan (1982a) was also determined by quadrant for each tree in the McNally and Cone fires to compare with cambium kill.

Crown kill refers to the portion of the crown that no longer has living tissue. It includes both tissue consumed by flames and tissue killed by convective heating during the fire. The linear measurement of crown kill was obtained by measuring the pre-fire crown length and the length of the remaining live crown to the nearest foot after estimating pre-fire crown base and height of crown kill. Pre-fire crown base was estimated by the presence of scorched needles or partially consumed needles and fine branches. Variations in crown kill pattern were averaged to obtain one crown kill length value. Lengths were measured using a clinometer or laser range finder. We calculated the percentage of pre-fire crown length killed (PCLK) by dividing the crown length killed by the original live crown length.

Percent crown volume killed (PCVK) was also determined for trees in the McNally and Cone fires. We visually estimated the volumetric proportion of crown killed compared to the space occupied by the pre-fire crown volume to the nearest five percent (Ryan 1982b). Obtaining both a length measurement and a volume estimate for crown kill enabled us to compare the predictive ability of the two variables. Also, collecting percent crown volume killed enabled us to compare our models with previously published probability models that use a volume estimate to predict mortality of fire-injured trees in California.

For ponderosa and Jeffrey pines on the McNally and Cone fire, we also assessed percent crown length scorched (PCLS) and percent crown volume scorched (PCVS). Crown scorch refers to the portion of the crown where needles are heat killed, but buds remain alive. The large buds of these species are more protected than those on true firs and incense-cedar and, consequently, often survive fire even when the surrounding needles are killed. A pine tree could feasibly have 100 percent crown scorch with little crown kill. For all other species in the study, crown kill is the equivalent of crown scorch.

Bole injury was assessed by first visually dividing the tree bole into four quadrants based on cardinal directions. The cambium was sampled in the center of each quadrant to obtain a cambium kill rating (CKR) for each tree. This was accomplished by drilling through the bark to the sapwood, within 7.5 cm of ground-line, using a power drill equipped with a 2.5 cm hole saw bit. Each sample was visually inspected in the field for color and condition of the tissue. Dead cambium is darker in color, often resin soaked and hard or gummy in texture. Live cambium is lighter in color, moist and rather pliable. Dead cells in the cambium zone also lose their plasticity which may allow the bark and wood to separate more easily (Ryan 1982a). A rating between zero and four was recorded for each tree by totaling the number of quadrants with dead cambium. In the McNally and Cone fires, a bark char code based on the charring that occurred in the majority of each quadrant within 30 cm of the ground line was also recorded following the methods in Ryan (1982a). Bark char was classified as unburned, light, moderate, or deep. Light charring has some blackened areas on the bark, but unburned portions also remain. With moderate charring, all bark is blackened, but the bark characteristics remain. When deep charring occurs, the bark characteristics are no longer discernable.

Insect activity was recorded for each tree. For white and red fir, the circumference of the bole with ambrosia beetle (*Trypodendron* and *Gnathotrichus* spp.) boring dust was recorded to the nearest ten percent. For ponderosa and Jeffrey pine, the number of red turpentine beetle (*Dendroctonus valens* LeConte) pitch tubes on the bole was recorded. Trees that showed signs of attack by primary bark beetles (*Dendroctonus jeffreyi*, *D. brevicomis*, *D. ponderosae*, and *Scolytus ventralis*) were not selected. Assessments of insect activity were limited to visual signs on the bole. No bark was removed to determine the success of the beetle attacks.

Statistical Analysis

Logistic regression was used to develop separate mortality models for red fir, white fir, incense-cedar, and yellow (ponderosa and Jeffrey) pine (SAS Institute v. 9.1). The predicted probability of mortality was estimated based on the dependent variable post-fire tree status, where dead trees were coded as one, and live trees as zero. An optimal model was developed for each species group from the following independent variables: percent crown length killed (PCLK), cambium kill rating (CKR), dbh (cm), and beetle presence (1) or absence (0). All models use the form:

$$P_m = 1/[1 + \exp(-(b_0 + b_1x_1 + b_2x_2 + \text{etc.}))]$$

Where:

P_m = predicted probability of mortality

b_0 = intercept

b_1, b_2, etc. = regression coefficient for x_1, x_2, etc.

x_1, x_2, etc. = value of fire injury variable (PCLK, CKR, dbh, AB, or RTB)

For ambrosia beetle (AB) and red turpentine beetle (RTB), only presence or absence of boring dust was used after determining that it performed equally well in the model compared to using the percent of the bole circumference with boring dust or number of pitch tubes, as initially assessed. The plot of the logit against CKR showed a linear increase. We therefore treated CKR as an ordinal variable and modeled it as a continuous variable (Hosmer and Lemeshow 2000).

PCLK was the common crown injury variable for all trees in the data set and is used in all models. It was a significant variable for determining the predicted probability of mortality for all species. We also tested the performance of the models when PCLK was squared or cubed and only report the model which contains the version of PCLK that performed the best. An optimal yellow pine model using percent crown length scorched (PCLS) is also presented for use when assessing crown injury prior to bud break. Models were also developed using percent crown volume killed (PCVK), or percent crown volume scorched (PCVS), to compare to those published by Mutch and Parsons (1998) and Stephens and Finney (2002). These comparisons are discussed, but the models are not reported in the paper.

The models predict probability of mortality for two, three or four years post-fire depending on the species. The majority of our data set includes only two-year post-fire mortality. Our two-year models should be considered preliminary, as additional mortality is expected. We chose the final models based on the combination of the lowest -2 log likelihood value (-2LogL) and the highest receiver operator characteristic curve value (ROC). The ROC reflects the accuracy of the model in classifying live and dead trees, with a value of 0.5 being no better than chance, and 1.0 indicating a perfect fit (Saveland and Neuenschwander 1990). After each optimal mortality model was developed from the potential variables, the CKR variable was dropped to create a second model. If the optimal model included a significant beetle attack variable, a third model was developed without the CKR or beetle variables. We created these second and third models to assess the importance of the CKR and beetle attack variables to predict tree mortality, and to give land managers the ability to compare probabilities of mortality when these specific fire injury criteria are not assessed.

Results and Discussion

Bark Char Classification versus Cambium Kill Rating

The bark char classification was compared to the cambium status for each quadrant per tree by species for all trees in the Cone and McNally fires. Bark charring is often used as a surrogate for direct cambium sampling because it can be obtained quickly. It is unknown how well the bark char classification system predicts cambium status. The following comparisons between bark char classification and cambium condition were similar across all dbh classes. We assessed very few unburned and light bark char ratings in our fires, and, with the exception of light bark char for yellow pine, few had dead cambium (*fig. 2*). While light charring on yellow pine equated to a dead cambium sample 45 percent of the time, only three percent of the pine samples were classified as light (*fig. 2,* pie chart inset). Because our sample size is so large (n=18,464 quadrants), we believe that light bark charring on white fir, incense-cedar, and yellow pine is relatively uncommon in wildfires and, therefore, would have little impact on the post-fire management decisions, if all light charring were assumed to have live cambium.

For our trees, the moderate bark char class described by Ryan (1982a) did not accurately predict cambium status (*fig. 2*). Fifty percent of the quadrants classified as moderate bark char had dead cambium samples. The majority of total recordings for bark char by quadrant were classified as moderate across all species, with 59 percent for incense-cedar, 60 percent for white fir, and 78 percent for yellow pine. Based on these results, reaching the conclusion that quadrants with moderate bark char equate to those with dead cambium would result in an incorrect determination 50 percent of time. Dead cambium was associated with deep bark charring for approximately 80 percent of the samples across all species.

When salvage marking guidelines include a measure of cambium condition, additional time is required to assess each tree. Based on our results, direct cambium sampling could be reduced by 20-40 percent by using unburned, light and deep bark char classes as a substitute. Moderately charred quadrants would still require direct sampling. Based on the inconsistency of the moderate bark char classification and the fact that most of the quadrants in our data set were classified as moderate, we did not use bark char as a variable in the logistic models.

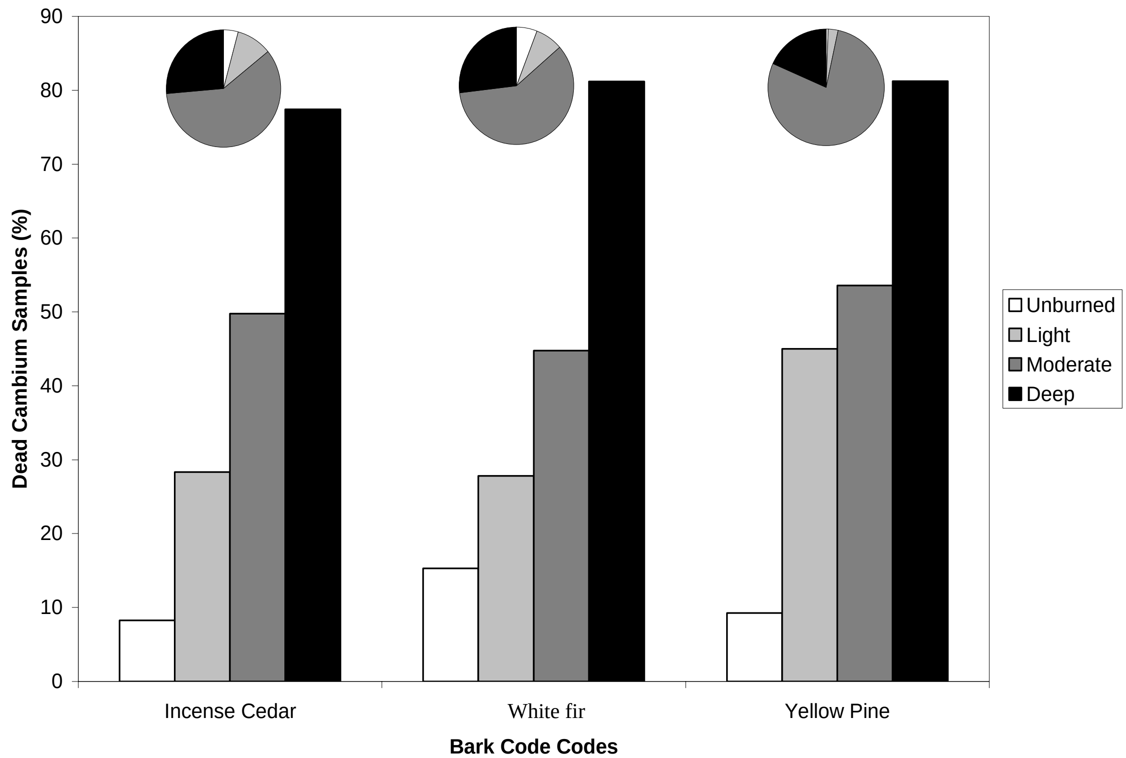

Figure 2--Percent of dead samples by bark char code for incense-cedar, white fir and yellow (ponderosa and Jeffrey) pine. Pie charts show distributions of bark char codes by species. Cambium status was determined by taking a 2.5 cm diameter sample within 7.5 cm of ground line in the middle of each quadrant. Bark char was assessed for each quadrant within 30 cm of ground line based on the classification that best fit the majority of the charring.

Mortality Models

Red Fir

The data set for red fir includes trees in the Bucks Fire and the Storrie Fire. Tree status four years post-fire was used for model development. All variables were significantly different between live and dead trees except for dbh (*table 3*). The total mortality over the four-year period following the fires was 22 percent.

The optimal model for predicting red fir mortality includes percent crown length kill (PCLK), cambium kill rating (CKR), and ambrosia beetle attacks (AB) (*table 4*). Modeled probability of mortality increased as PCLK increased for all ratings of cambium kill (*figs. 3, 4*). When the CKR is excluded from the optimal model, the ROC value drops from 0.83 to 0.72 (grey line in *figs. 3, 4*, model 2). Contrary to the CKR curves displayed for all models, we would expect every tree with a crown length kill equal to 100 percent to die, regardless of the CKR. The maximum PCLK for our study trees was 89 percent.

Table 3--*Mean characteristics of variables for red fir. P-values test differences between live and dead values.*

Variable[1]	Mean (n=206)	SE	Range	Live Mean (n=160)	Dead Mean (n=46)	p-value
Dbh	42.2	1.2	15-105	42.4	41.7	0.8221
PCLK	42	1.8	0-89	38	52	0.0009
CKR	1.6	0.1	0-4	1.3	2.3	<0.0001
AB	13	1.5	0-100	10	25	0.0017

[1] Dbh – diameter at breast height (cm); PCLK – percent crown length killed; CKR – cambium kill rating; AB – percent of bole circumference with boring dust.

Table 4--*Red fir mortality models four years post-fire. Model 1 is the statistically best, optimal model. The class effect ambrosia beetle (AB) is modeled as 1 when present and -1 when absent.*

Model	Intercept	PCLK	CKR	AB	-2LogL	ROC
1	-4.2066	0.0330	0.8702	0.4619	165.18	0.83
2	-2.1342	0.0221	-	0.6218	195.06	0.72
3	-2.3431	0.0240	-	-	207.35	0.67

Ambrosia beetles attack weakened, recently dead, freshly felled, or other unseasoned or moist wood (Furniss and Carolin 1977). They penetrate the bark, sapwood, and sometimes heartwood, thus providing an entry court for fungi and other organisms to begin the decomposition process. Their piles of white boring dust on tree boles often provide a good external indicator that a tree is not healthy. A tree with AB present has a higher predicted probability of mortality than a tree with similar PCLK and CKR but no AB (*figs. 3, 4*, model 1).

We found CKR to be the most important predictor of red fir mortality (ROC =0.74 for CKR alone). Wagener (1961) found that cambium killing involving more than 25 percent of the bole circumference would greatly affect survival for several conifer species, including red fir. Ryan et al. (1988) reported the number of dead samples to be the most important predictor of mortality for fire-injured Douglas-fir. When CKR is not used in the models, mortality predictions would be slightly overestimated at low levels of cambium kill and greatly underestimated at higher levels of cambium kill (*figs. 3, 4*, models 2 and 3).

Dbh was not a statistically significant variable in our red fir model. This differs from most other tree mortality equations, although there is no other published model for red fir mortality to which to directly compare. Tree size is widely recognized as an important factor in resistance to fire injury due to an increase in basal crown height and bark thickness as tree height and diameter increase (Ryan et al. 1988). The difference in mean dbh for our red fir trees was not statistically significant between live and dead trees (*table 3*). It is unclear why dbh was not correlated with mortality for red fir, as it is for all other species. A significant relationship might develop with an increase in the sample size.

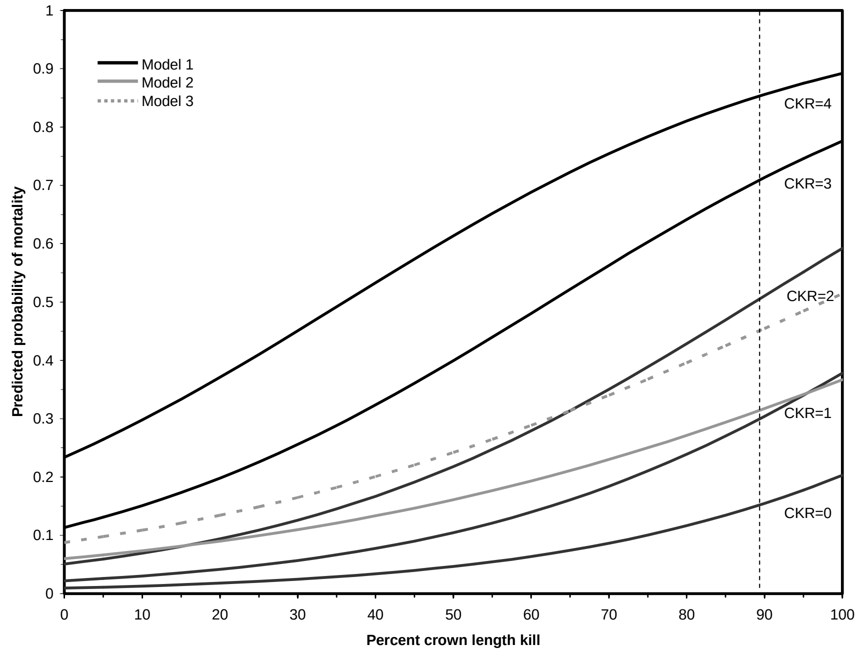

Figure 3–Year 4 red fir mortality curves for trees with no signs of ambrosia beetle attack. Solid black lines indicate optimal mortality model (model 1). Grey line is model 2 mortality curve. Dashed line is model 3 mortality curve. The vertical dashed line shows the upper limit of the data set (maximum PCLK = 89%).

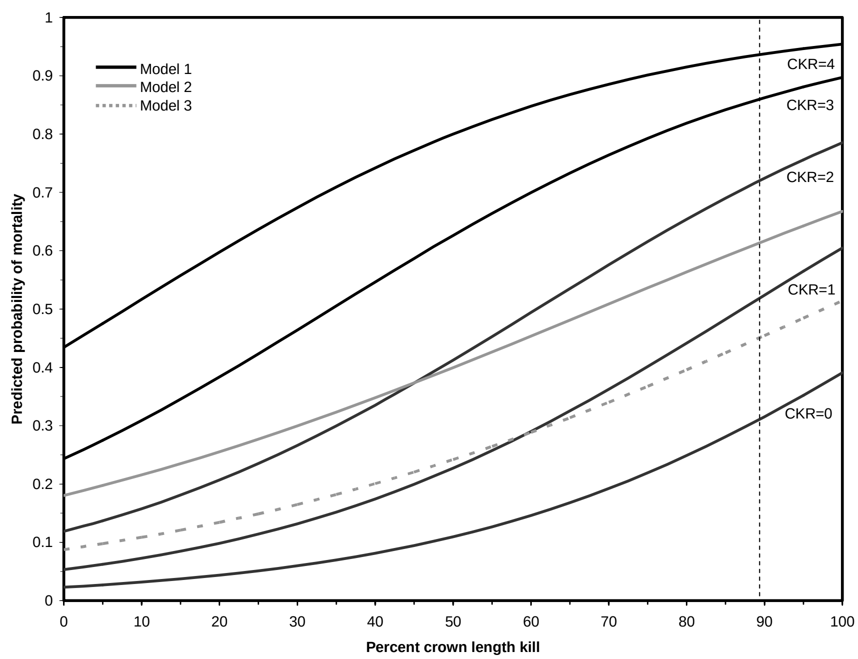

Figure 4–Year 4 red fir mortality curves for trees with signs of ambrosia beetle attack. Solid black lines indicate optimal mortality model (model 1). Grey line is model 2 mortality curve. Dashed line is model 3 mortality curve. The vertical dashed line shows the upper limit of the data set (maximum PCLK = 89%).

When developing post-fire salvage marking guidelines for red fir based on these models, both the desired level of predicted probability of mortality and the intensity of individual tree sampling need to be determined. Models 2 and 3 are displayed to facilitate a comparison in the predicted probabilities of mortality when statistically significant variables are excluded. Based on comparing our models, if only PCLK and AB are used and AB is present, the highest predicted probability of mortality possible is 0.66 (*fig. 4*). In model 3, where only PCLK is used, the highest predicted probability of mortality possible is 0.52. Of the variables included in the optimal model, percent crown length kill and the presence or absence of AB are both easily obtained by a quick observation of the crown and bole. Obtaining a cambium kill rating involves direct cambium sampling in each quadrant, which requires additional time to assess each tree. Land managers should be aware that not assessing for cambium injury can greatly decrease their ability to accurately predict the mortality of fire-injured red fir. We do not have data on bark char classifications compared to cambium condition for red fir, however, based on our results with other species, we can presume that the classifications of unburned, light, and deep bark char would be appropriate to use in place of direct cambium sampling. Moderately charred quadrants would still need a direct assessment.

Incense-cedar

All incense-cedar data was collected on the McNally fire and the status of trees two years post-fire was used in model development. All variables were significantly different between live and dead trees (*table 5*). Through 2004, the second year post-fire, twelve percent of our sample trees had died.

Table 5--*Mean characteristics of variables for incense-cedar. P-values test differences between live and dead values.*

Variable[1]	Mean (n=781)	SE	Range	Live Mean (n=688)	Dead Mean (n=93)	p-value
Dbh	51.5	0.9	25.4-166.4	52.2	46.3	0.0137
PCLK	40	1.1	0-98	34	79	<0.0001
PCVK	44	1.2	0-95	38	85	<0.0001
CKR	2.1	0.0	0-4	2.0	2.8	<0.0001

[1] Dbh – diameter at breast height (cm); PCLK – percent crown length killed; PCVK – percent crown volume killed; CKR – cambium kill rating.

The preliminary optimal model for predicting incense-cedar mortality within two years post-fire includes percent crown length kill cubed ($PCLK^3$), cambium kill rating (CKR) and dbh (*table 6*). PCLK is the most important predictor of mortality. Mortality equations developed from our incense-cedar data predict a low probability of mortality (less than 25 percent) until PCLK reaches approximately 70 percent when CKR equals four and 90 percent when CKR equals zero. As PCLK increases above these levels, the predicted mortality increases considerably. In this study, 85 percent of the observed incense-cedar mortality occurred in trees with greater than 65 percent PCLK.

CKR is also a significant variable accounting for slight increases in the predicted mortality with increasing cambium kill ratings (*fig. 5*). When CKR is dropped from the model, for PCLK greater than 50 percent, mortality of trees with a CKR of four would be slightly under predicted while mortality of trees with a CKR less than

Table 6–*Incense-cedar mortality models two years post-fire. Model 1 is the statistically best, optimal model.*

Model	Intercept	PCLK^3 (%)	CKR	Dbh	-2LogL	ROC
1	-4.9639	0.0000068	0.5398	-0.0143	325.20	0.92
2	-4.2505	0.0000068	-	N.S.	348.68	0.90

three would be slightly over predicted (*fig. 5*, model 2). Contrary to the CKR curves displayed in model 1 (*fig. 5*), where the predicted mortality never reaches 1.0, we would not expect any tree to survive with 100 percent crown kill. Also, there were no trees in our data set with PCLK > 95 percent and CKR < 2, which helps explain why the curves do not approach 1 when PCLK equals 100 percent for low CKR's. The maximum crown length kill for our study trees was 98 percent. Dbh is significant in the model, however, it does not greatly affect the predicted mortality for trees with similar percent crown length kill and cambium kill rating.

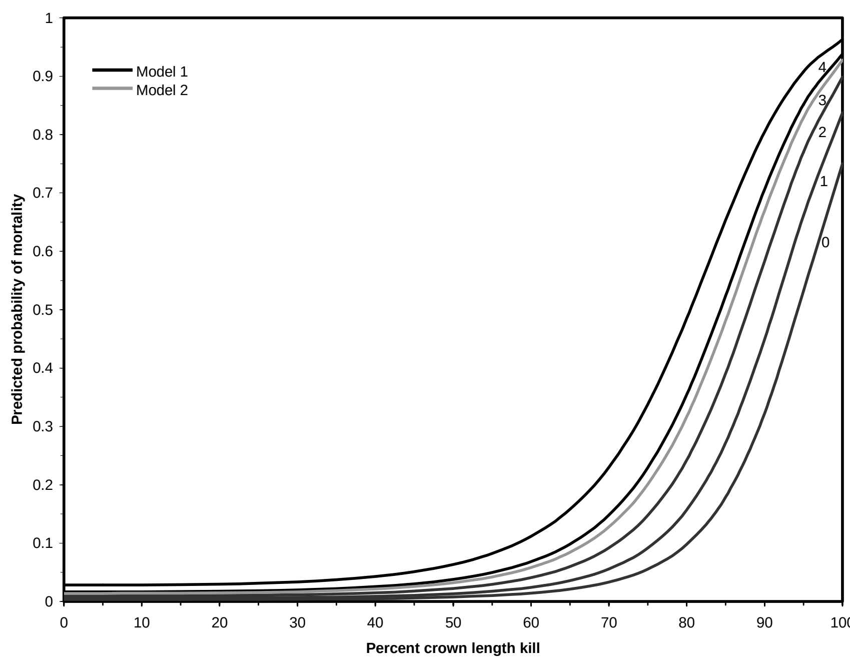

Figure 5–Year 2 incense-cedar mortality curves by cambium kill rating and dbh of 51.5 cm. Solid black lines indicate optimal mortality model (model 1). Grey line following CKR=3 line is model 2 mortality curve.

The mean dbh for incense-cedar was significantly lower for dead versus live trees (*table 5*). This is similar to the results reported in Stephens and Finney (2002), where their incense-cedar model also showed a lower predicted probability of mortality with increasing dbh. These similar results are likely due to both incense-cedar data sets being heavily weighted towards smaller diameter trees.

Model 2 (gray dashed line in *fig. 5*) was developed to compare the probabilities of mortality when CKR is excluded. Note that dbh becomes insignificant in the model when CKR is dropped (*table 6*). When CKR is dropped, there is little effect on the performance of the model (ROC = 0.92 vs. 0.90) revealing the relative importance of PCLK over CKR for incense-cedar. Since CKR only accounts for minimal differences in the predicted mortality and requires additional time for sampling, land managers may choose to disregard cambium sampling without losing much predictive accuracy.

White Fir

The data set for white fir trees from the Bucks, Storrie, and Star fires were combined and the status of trees three years post-fire was used for model development. The model for white fir on the McNally fire is presented separately as a preliminary two-year status model. There was a significant difference between live and dead trees for all variables collected for both the three-year and two-year data sets (*table 7*). Mean dbh was higher for dead trees than for live trees in both data sets. Total mortality observed over the three-year post-fire period was 43 percent compared to 50 percent after only two years on the McNally fire.

Table 7--*Mean characteristics of variables for white fir. The McNally fire was analyzed separate from other fires. P-values test differences between live and dead values.*

Variable[1]	Mean	SE	Range	Live Mean	Dead Mean	p-value
			Bucks, Star, and Storrie Fires (3 years post-fire)			
	(n=424)			(n=242)	(n=182)	
Dbh	54.5	1.0	15.2-134.4	47.3	63.9	<0.0001
PCLK	57	1.4	0-100	45	74	<0.0001
CKR	1.8	0.1	0-4	1.4	2.3	<0.0001
AB	14	1.1	0-100	6	25	<0.0001
			McNally Fire (2 years post-fire)			
	(n=1866)			(n=929)	(n=937)	
Dbh	60.2	0.5	25.4-152.7	56.4	64.0	<0.0001
PCLK	69	0.6	0-100	53	83	<0.0001
PCVK	71	0.6	0-95	55	83	<0.0001
CKR	2.1	0.0	0-4	1.8	2.1	<0.0001

[1]Dbh – diameter at breast height (cm); PCLK – percent crown length killed; PCVK – percent crown volume killed; CKR – cambium kill rating; AB – percent of bole circumference with boring dust.

The optimal three-year model for predicting white fir mortality includes percent crown length kill cubed ($PCLK^3$), cambium kill rating (CKR), dbh, and ambrosia beetle attacks (AB) (*table 8*). *Figure 6* shows the predicted probability of mortality by CKR when AB is assessed but not present. *Figure 7* shows the predicted probability of mortality by CKR when AB is assessed and present. Trees with AB have a higher predicted probability of mortality with the same levels of injury compared to trees without AB (*figs. 6, 7*).

The ROC value is reduced by 0.02 when CKR is not used in the model. Model accuracy further declines when neither CKR nor AB is included. Using the average dbh of 54.5 cm as an example, for a tree with 70 PCLK, if a land manager chose to not evaluate the cambium condition but did assess for AB, and it was not present, the predicted mortality is underestimated by 0.1 to 0.4 when CKR is greater than one (*fig. 6*). For the same tree, predicted mortality would be overestimated by as much as 0.2 when there is no cambium kill. When AB boring dust is present and cambium condition is not assessed, the predicted mortality follows a curve similar to when

CKR equals two (*fig. 7*). If a land manager chose to only evaluate PCLK, mortality of trees with higher amounts of cambium kill would be under predicted.

Table 8--*White fir mortality models three years post-fire. Model 1 is the statistically best, optimal model. The class effect ambrosia beetle (AB) is modeled as 1 when present and -1 when absent.*

Model	Intercept	$PCLK^3$ (%)	CKR	Dbh	AB	-2LogL	ROC
1	-5.3456	0.000006	0.6584	0.0367	0.5308	319.37	0.91
2	-3.5603	0.000005	-	0.0296	0.7338	351.85	0.89
3	-4.2829	0.000006	-	0.0397	-	381.284	0.87

The preliminary optimal model for white fir using two-year data from the McNally fire includes the same variables as the three-year white fir model with the exception of ambrosia beetle (*table 9*). Ambrosia beetle was only observed on a few trees in the McNally fire and there was no significance difference between live and dead trees for this variable. Excluding the CKR from the model does not change the ROC value, which is evidence that crown kill is a more important criterion for predicting mortality within two years.

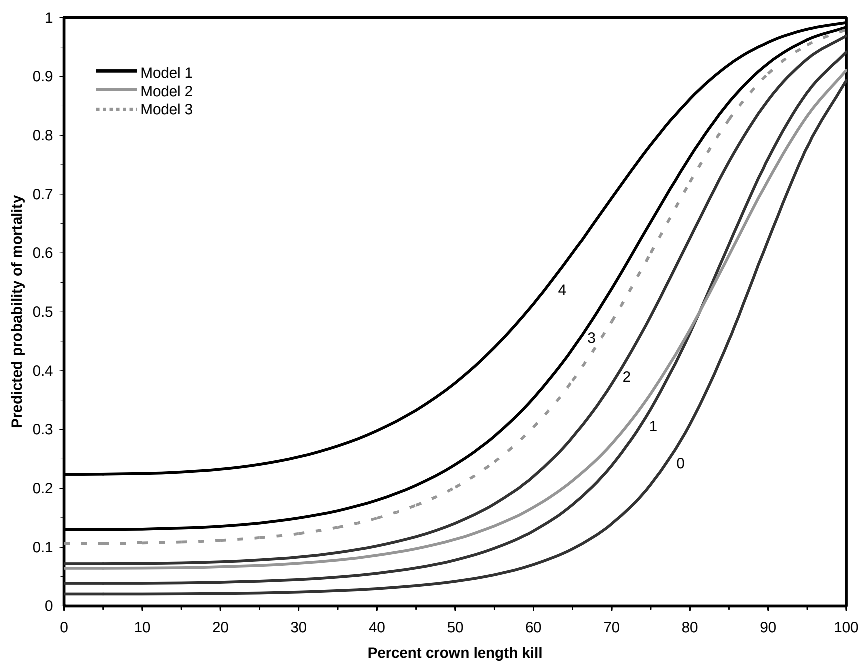

Figure 6--Year 3 white fir mortality curves for trees with no signs of ambrosia beetle attack and a dbh of 54.5 cm. Solid black lines indicate optimal mortality model (model 1). Grey line is model 2 mortality curve. Dashed line is model 3 mortality curve.

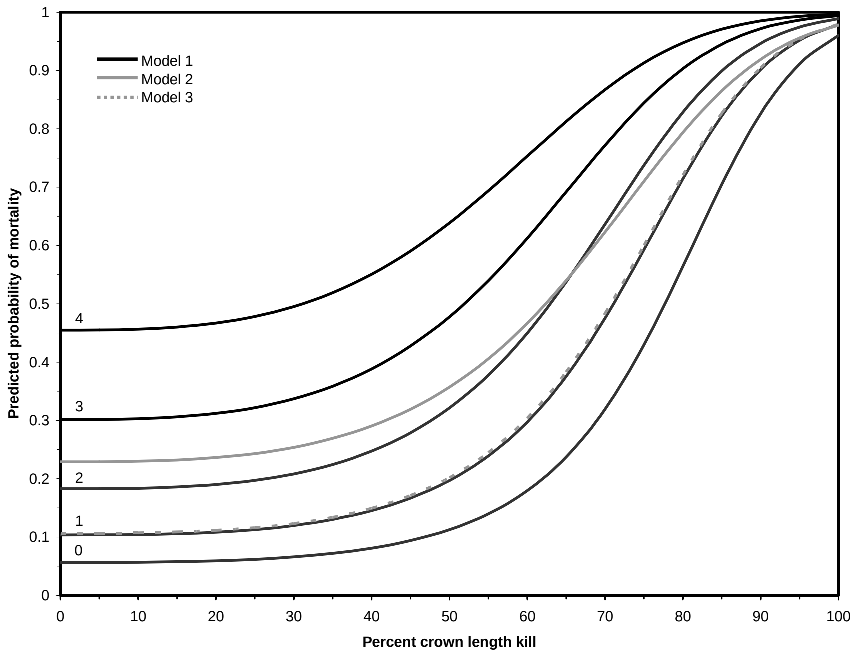

Figure 7–Year 3 white fir mortality curves for trees with signs of ambrosia beetle attack and a dbh of 54.5 cm. Solid black lines indicate optimal mortality model (model 1). Grey line is model 2 mortality curve. Dashed line is model 3 mortality curve.

Table 9--*White fir mortality models two years post-fire. Model 1 is the statistically best, optimal model.*

Model	Intercept	$PCLK^3$ (%)	CKR	Dbh	-2LogL	ROC
1	-4.2913	0.000006	0.2185	0.0174	1669.23	0.87
2	-3.7578	0.000006	-	0.0162	1689.69	0.87

Cambium kill becomes more important in white fir models in the third year post-fire. When comparing the predicted mortality between the two- and three-year white fir models, the probability of mortality is much higher in year three for trees with a CKR equal to three or four (*fig. 8*). The difference in predicted mortality between CKR's of three and four also widens in the year three model (*fig. 8*). As PCLK increases above 70 percent, the differences between the year two and year three three lines decrease. As crown kill approaches 100 percent, trees do not have enough photosynthetic capacity remaining to sustain life, regardless of the amount of cambium kill. If enough photosynthetic capacity remains, trees with terminal levels of cambium injury may take a longer period of time to die. Foliage may remain green for several years due to water being conducted upward through the uninjured xylem. However, the inability of the dead phloem to transport carbohydrates to the roots results in a slow starvation and eventual death.

The mean dbh for dead trees was higher than live trees in both the two- and three-year data sets. Our white fir models have higher predicted probabilities of

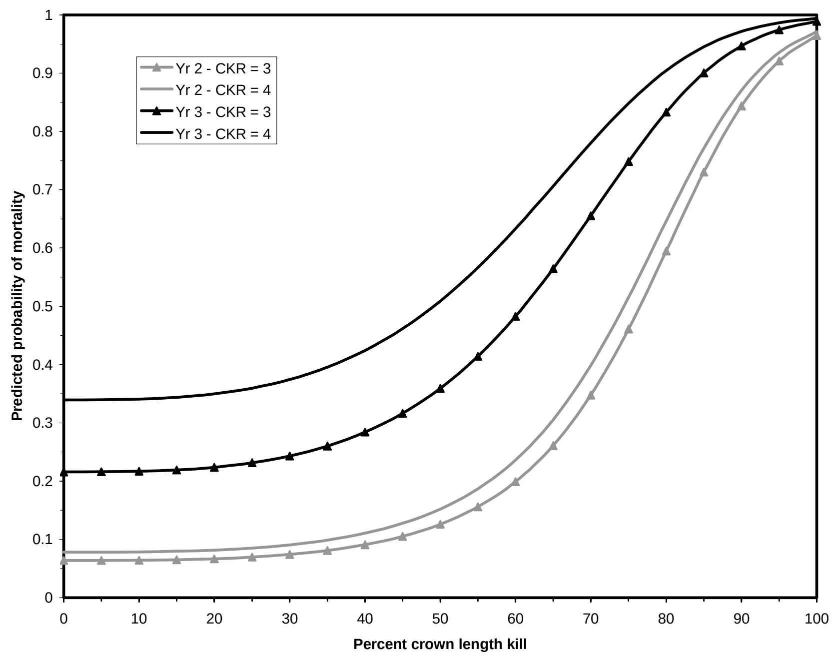

Figure 8--Comparison of years 2 (grey) and 3 (black) white fir mortality curves where dbh equals 54.5 cm. Year 3 curves include both attacked and unattacked trees. Only cambium kill ratings of three and four are shown.

mortality with increasing dbh when comparing trees with similar crown and cambium kill. Our results are contrary to those reported by Stephens and Finney (2002) and Mutch and Parsons (1998) for white fir. Their models show a decrease in predicted mortality as dbh increases, which may be due to the small number of large trees in their data sets. An increase in the predicted mortality as dbh increases has only previously been reported for ponderosa pine (Ryan and Frandsen 1991, McHugh and Kolb 2003). We also report similar findings for Jeffrey and ponderosa pine in this paper.

Mutch and Parsons (1998) developed a model for white fir based on the status of trees five years after a prescribed fire. The majority of white fir trees in their study were less than 50 cm dbh. When modeling only percent crown volume killed (PCVK) and dbh, for trees with dbh less than 50 cm, their model predicts higher mortality than our model. The predicted mortality for trees smaller than 50 cm dbh with 70 percent crown kill is 0.28 in our model compared to 0.84 in theirs. This large difference in the probabilities of mortality between models begins to decrease as trees get larger. However, the predictive capability of their model for trees larger than 50 cm dbh is questionable because their data range is limited.

Stephens and Finney (2002) developed a model for white fir based on the status of trees three years post-fire. Their data set included a minimum of five trees per five cm diameter class for white fir between 5-65 cm dbh. Their mean dbh for white fir was 20.3 cm compared to a mean of 60.2 cm for our trees. For trees less than 50 cm dbh, the predicted probabilities of mortality are very similar for all levels of crown injury between models when using only PCVK and dbh. As trees get larger than 50 cm dbh, the Stephens and Finney model dramatically underestimates tree mortality. For example, our predicted mortality for trees greater than 75 cm dbh with 70 percent crown volume kill is 55 percent compared to less than 10 percent for theirs. It should be noted that the Stephens and Finney model is intended for use by forest managers planning prescribed fires. Their data were collected before and after a prescribed fire and the majority of their mortality was in the smaller size classes, as would be expected burning under prescribed fire conditions. Their lack of data for trees greater than 65 cm dbh and the dramatic differences in the predicted probabilities of mortality for larger trees between our model and theirs illustrates the concern of using models beyond the authors' intent and extrapolating beyond the data used for model development.

Yellow Pine

The data set for yellow pine includes Jeffrey and ponderosa pine trees in the Cone and McNally fires. Tree status two years post-fire was used for model development. There was a significant difference between live and dead trees for all variables collected (*table 10*). Average dbh was higher for dead trees than live trees. Sixty-five percent of the trees died in the first two years post-fire.

Table 10--*Mean characteristics of variables for yellow pine. P-values test differences between live and dead values.*

Variable[1]	Mean (n=1974)	SE	Range	Live Mean (n=682)	Dead Mean (n=1292)	p-value
Dbh	62.6	0.6	25.4-160.8	56.7	65.8	<0.0001
PCLK	64	0.5	0-100	42	76	<0.0001
PCLS	85	0.4	0-100	73	91	<0.0001
PCVK	71	0.5	0-95	52	81	<0.0001
PCVS	87	0.4	0-100	76	92	<0.0001
CKR	2.4	0.0	0-4	1.5	2.8	<0.0001
RTB	3	0.1	0-31	1	4	<0.0001

[1] Dbh – diameter at breast height (cm); PCLK – percent crown length killed; PCLS – percent crown length scorched; PCVK – percent crown volume killed; PCVS – percent crown volume scorched; CKR – cambium kill rating; RTB – number of red turpentine beetle pitch tubes on bole.

The preliminary optimal model to predict mortality within two years post-fire includes crown length kill squared ($PCLK^2$), cambium kill rating (CKR), dbh, and red turpentine beetle (RTB) as variables (*table 11*, model 1). While leaving CKR out of the model only reduced the ROC slightly, the graphs of the mortality curves illustrate the reduced accuracy when cambium condition is not assessed (*figs. 9*, *10*, model 2). Excluding RTB and CKR reduce model accuracy even further. The predicted probability of mortality for trees that were attacked by RTB would be underestimated if RTB and CKR were not assessed (*fig. 10*, model 3). Conversely, if RTB and CKR were not assessed for individual trees that were not attacked by RTB, the predicted probability of mortality would be overestimated (*fig. 9*, model 3).

All three models predict increasing probabilities of mortality with increasing dbh. This is similar to results in McHugh and Kolb (2003) for ponderosa pine models developed using wildfire alone and prescribed and wildfire combined data sets, but contrary to the prescribed fir models reported in Stephen and Finney (2002) and McHugh and Kolb (2003). Most often, the objective of a prescribed fire is to limit mortality of the overstory while reducing fuel loadings and ingrowth of smaller trees. Therefore, a data set from a prescribed burn likely does not contain many

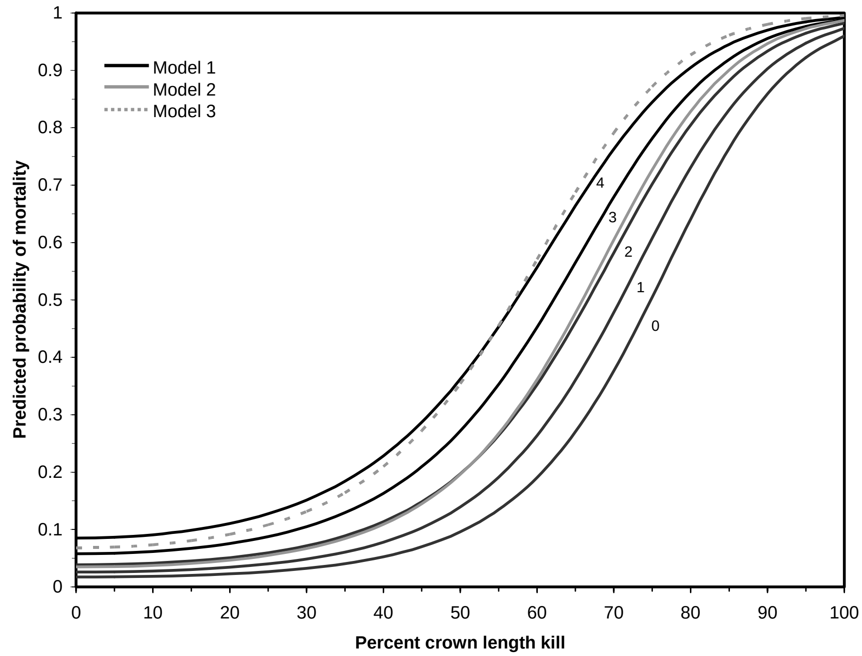

Figure 9--Year 2 yellow pine mortality curves for trees without red turpentine beetle pitch tubes and a dbh of 62.6 cm. Solid black lines indicate optimal mortality model (model 1). Grey line is model 2 mortality curve. Dashed line is model 3 mortality curve.

Table 11–*Yellow pine mortality models two years post-fire for use after bud break. Model 1 is the statistically best, optimal model. The class effect red turpentine beetle (RTB) is modeled as 1 when present and -1 when absent.*

Model	Intercept	$PCLK^2$	CKR	Dbh	RTB	-2LogL	ROC
1	-4.3202	0.000723	0.4185	0.0188	0.9048	1294.79	0.92
2	-3.7431	0.000765	-	0.0219	0.9515	1355.70	0.92
3	-3.1647	0.000805	-	0.0088	-	1530.79	0.89

larger, overstory trees with high levels of crown and cambium kill. The differences in tree size and fire type could account for the different effects of dbh when predicting mortality.

In ponderosa and Jeffrey pines, extensive heat killing of foliage may occur with only light injury to buds and twigs (Wagener 1961). Delaying the evaluation of fire-injured pines until after bud break results in a more accurate determination of the residual amount of live crown. However, the ability to predict mortality of pine trees prior to bud break may be useful for land managers that want to expedite tree removal to limit wood deterioration. Our optimal model for use in Jeffrey and ponderosa pine trees prior to bud break includes percent crown length scorch squared ($PCLS^2C$), cambium kill rating (CRK), and dbh (*table 12,* model 1). The models using PCLS do not predict mortality as accurately as the percent crown length kill (PCLK) models (Model 1 ROC = 0.92 vs. 0.87). We did not include red turpentine beetle in the model, as few beetles would fly prior to bud break the year after the fire.

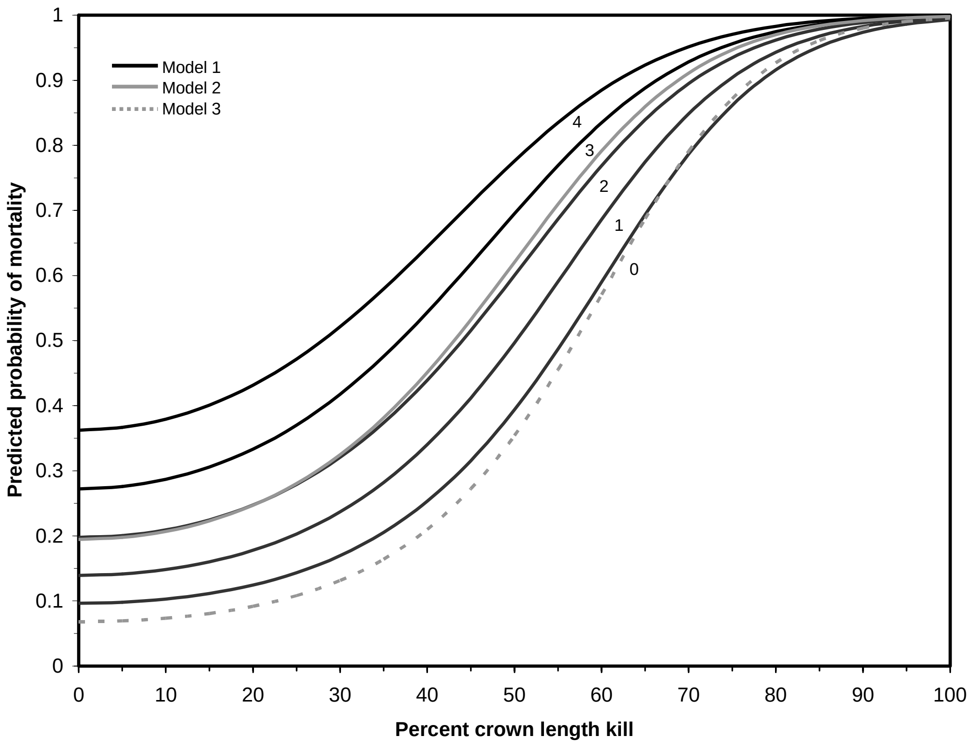

Figure 10–Year 2 yellow pine mortality curves for trees with red turpentine beetle pitch tubes and a dbh of 62.6 cm. Solid black lines indicate optimal mortality model (model 1). Grey line is model 2 mortality curve. Dashed line is model 3 mortality curve.

Table 12–*Yellow pine mortality models two years post-fire for use prior to bud break. Model 1 is the statistically best, optimal model.*

Model	Intercept	$PCLS^2$	CKR	Dbh	-2LogL	ROC
1	-6.8243	0.000568	0.6688	0.0285	1675.73	0.87
2	-5.5637	0.000578	-	0.0308	1903.87	0.81

Once bud break has occurred, the models using percent crown length kill are preferable.

Our model using PCVS and dbh for trees equal to 50 cm dbh was very similar when comparing with the model by Stephens and Finney (2002). Above 75 percent PCVS, our model predicted slightly higher probabilities of mortality. This discrepancy between predicted probabilities of mortality increases greatly as trees get larger. The lower predicted probabilities in their models compared to ours may be attributed to the small overlap between the data sets. Our data set contains much larger trees (average of 62.6 cm dbh versus 26.3 cm dbh). Their lack of data for trees greater than 60 cm dbh and the dramatic differences in the probabilities of mortality for larger trees between our model and theirs again illustrate the concern of using models beyond the authors' intent and extrapolating beyond the data used for model development.

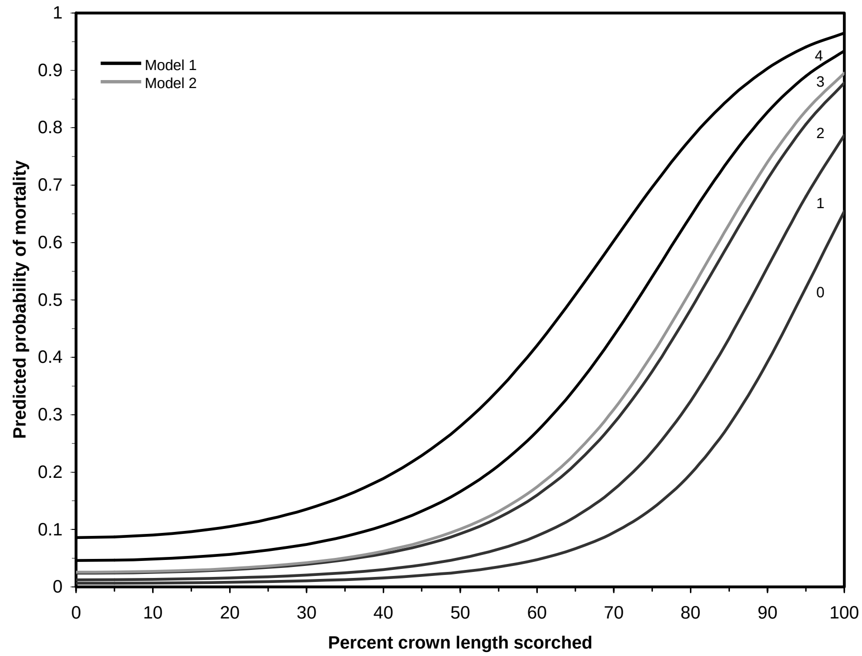

Figure 11–Year 2 yellow pine mortality curves percent crown length scorched (model 1) by cambium rating for a dbh of 62.6 cm. Solid black lines indicate optimal mortality model (model 1). Grey line is model 2 mortality curve. Models are intended for use before bud break occurs within one year following fire.

Conclusion

We found that percent crown length killed and the number of quadrants with dead cambium samples to be the most important variables for predicting post-fire mortality for mixed conifer species in California. The size of the tree, in terms of diameter at breast height, was also important in the models for all species except red fir. For white fir, ponderosa and Jeffrey pines, larger trees were more likely to die than smaller trees given the same level of crown and cambium injury. This could be due to larger trees having greater duff accumulations, leading to increased smoldering times and the potential for more root injury. Larger trees may also be less vigorous than smaller trees, reducing their capability to recover from fire caused injuries (McHugh and Kolb 2003). The opposite was true for incense-cedar, where smaller trees succumbed more often, given the same level of crown and cambium injury. The variables indicating the presence of ambrosia beetle boring dust on red and white fir and red turpentine beetle pitch tubes on Jeffrey and ponderosa pines increased model accuracy in all equations.

Land managers need the ability to predict mortality following wildfires to plan tree removal and regeneration projects. The logistic models presented in this paper were specifically developed for use in predicting post-fire mortality for salvage marking, but may also be useful in determining future stocking levels and planning fuels treatments. These models enable managers to select a desired level of predicted probability of mortality based on land management objectives. The logistic curves for additional models are also provided to demonstrate the decrease in accuracy when significant variables are removed. Each variable in a model requires additional time for assessment in the field. Estimating crown injury takes the least amount of time, followed by assessing for insect activity and measuring dbh. Sampling the cambium in each quadrant is the most time consuming, but based on the significance of CKR in all our models, not sampling for it may result in great inaccuracies in model predictability, with the possible exception of incense-cedar. Compared to using the optimal models for developing marking guidelines, using less accurate models will likely result in leaving more dead trees than desired on the landscape or removing more trees that would have survived. The implementation of a bark char classification system that equates to unburned and light char as no cambium kill and deep char as dead cambium could be used as an reasonably accurate alternative to cambium sampling. Bole quadrants with moderate char, which was the most common char rating in our study, would still require direct sampling due to the poor correlation with cambium status.

Crown injury can be assessed using a variety of methods. We chose to use percent crown length killed in all of our models because it was a common variable between data sets. Unlike conclusions drawn by other authors suggesting that a volume estimate is more accurate (Peterson 1985, Stephens and Finney 2002), we did not find great differences in predictive accuracy between models developed with percent crown length killed versus models developed with percent crown volume killed. The selection of one method of crown injury assessment over another can therefore be based on the assessor’s preferred method. However, volume versus length killed estimates are not interchangeable in the models and only percent crown length killed should be used with the models in this paper.

The logistic models in this paper for incense-cedar, Jeffrey pine, ponderosa pine and white fir are based on tree status two years post-fire. Mortality of study trees is still occurring in all fire areas and incorporating additional annual assessments into model development should improve accuracy. For fires where assessments have been made over a longer period of time, the majority of the mortality occurred within three years post-fire. Delayed mortality, in terms of crown death, may take several years to occur for trees with fatal levels of cambium kill. For white fir, cambium kill was a more important variable in the three-year model compared to the two-year model. We anticipate similar results for our remaining species.

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Effects of Fuel Reduction Treatments on Breeding Birds in a Southern Appalachian Upland Hardwood Forest[1]

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Abstract

In the past, fires set by American Indians and settlers shaped much of the southern Appalachian forest by reducing the shrub layer and maintaining an open understory. Since the 1930's, fire exclusion has allowed the development of a thick shrub layer and accumulation of woody debris. This fuel buildup contributes to the potential for wildfire in many ecosystems. Recently, the need for fuel reduction, using techniques such as prescribed fire or mechanical treatments, has received national attention. However, the impacts of such habitat manipulations on breeding birds are not well understood, especially in southern hardwood ecosystems. As part of the multidisciplinary National Fire and Fire Surrogate Research Project, we compared the effects of three fuel reduction techniques and controls on breeding birds, using 50m point counts in four, 14-ha treatments within each of three replicate blocks at the Green River Game Land, Polk County, North Carolina. Treatments were: (1) prescribed burning (B), (2) mechanical felling of shrubs and small trees (M), (3) mechanical felling + burning (MB), and (4) controls (C). Breeding birds were surveyed using point counts during 2001-2004. Mechanical understory felling treatments were conducted in winter 2001-2002, and prescribed burning in spring 2003. Hence, bird responses to all four treatments were compared only for 2003 and 2004. After prescribed fire (2003), leaf litter depth decreased in B and MB, and snag densities and canopy openness increased in MB. Shrub cover was significantly lower in all fuel reduction treatments than in C. Total breeding bird abundance was similar among treatments each year except 2003, when it was higher in C than M. Species richness was similar among treatments except in 2004 when it was higher in MB. Shrub forager abundance was highest in C in 2003, and higher in C than in B during 2004. The abundance of shrub nesters was also lower in B and M than in MB in 2004. Responses were most evident at the species level. Most species showed no detectable response to treatments. During 2003 Worm-eating Warbler abundance was lower in MB than C or M, and, in 2004, it was lower in both B and MB than C or M. Hooded Warblers were more abundant in C than any fuel reduction treatments during 2003 and 2004. Indigo Buntings, which are associated with open habitats, were most abundant in MB during 2004. Fuel reduction treatments affected individual bird species differently, and responses appeared to be associated with changes in habitat structure. To fully understand how fire and fire surrogates for fuel reduction affect breeding bird communities, post-fire surveys of birds and vegetation structure must continue for several years.

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Introduction

In the past, fires set by American Indians and settlers shaped much of the southern Appalachian forest by reducing the shrub layer and maintaining an open understory (Lorimer 1993). Fire was used first by American Indians to improve conditions for travel and game, and later by settlers to improve hunting conditions and pasturage for livestock (see Lorimer 1993). Ayres and Ashe (1905) reported fire scars on 80% of trees in a systematic survey of 6.5 million acres of the southern Appalachian region. Fire scar evidence in the Great Smoky Mountains indicated that the fire return interval there was $\leq$ 10 years prior to 1940 (Harmon 1982).

Beginning in the 1930's, forest fires began to be viewed as destructive, and were suppressed or excluded where possible (Lorimer 1993). Fire exclusion promoted the buildup of fuels, such as woody debris and ericaceous shrubs, enhancing the likelihood of wildfire. Today, prescribed burning is again employed, now as a forest management tool for ecosystem restoration, oak regeneration, understory control, and fuel reduction, to reduce the likelihood of wildfire. Mechanical methods to reduce the forest understory are also used in lieu of prescribed fire when burning is not feasible or practical. Yet the effects of fire and fire surrogates for fuel reduction on breeding birds is poorly known, especially in the southern Appalachians.

In 2000, the National Fire and Fire Surrogate Study (NFFS) was initiated by the Joint Fire Science Program to research impacts of fuel reduction treatments on multiple components of forested ecosystems across the U.S. (Youngblood et al. 2005). The Green River Game Land in Polk County, NC was selected to represent the southern Appalachian upland, hardwood forest ecosystem in the NFFS study during 2001. This site was added to the original study through funding from the National Fire Plan. As part of the national NFFS, we studied community and species-level responses of breeding birds to three fuel reduction treatments (prescribed burning, mechanical understory removal, and mechanical understory removal + prescribed burning) and controls in the southern Appalachians.

The influence of vertical and horizontal vegetation structure on bird communities is well established (MacArthur and MacArthur 1961, Mauer et al. 1981). However, species differ in their habitat requirements, and might be expected to respond differently to changes in habitat attributes that are created by forest management activities. Several studies report higher bird species richness, diversity, and density in sites that were disturbed by management activities compared to mature undisturbed forest (Annand and Thompson 1997, Baker and Lacki 1997). Several bird species require habitat that has been recently disturbed by fire or by large-scale, high-intensity disturbance (Klaus et al. 2005). However, the abundance and nesting success of some ground and shrub nesting species may decline where the shrub or leaf litter layer is reduced or removed through mechanical means (Rodewald and Smith 1998) or by burning (Aquilani et al. 2000, Artman et al. 2001).

Other than understory reductions, fuel reduction treatments may affect habitat attributes by increasing light levels and primary productivity, which in turn may increase food resources for breeding birds by promoting higher insect densities and fruit production (Blake and Hoppes 1986). Prescribed fire may additionally create snags (Van Lear 2000), which benefit cavity nesters, such as woodpeckers (Lanham and Guynn 1996, Saab et al. 2004, Giese and Cuthbert 2003).

Other studies examine how burning alone or other silvicultural practices affect breeding birds. However, to our knowledge, none compare the effects of three commonly used understory reduction techniques on breeding bird communities and species using a replicated experimental design. Land managers need to know how different fuel reduction methods affect breeding birds to better manage populations and communities in conjunction with forest management. Our objectives were to determine whether and how breeding bird communities and individual species respond to fuel reduction treatments by prescribed burning and/or mechanical understory removal. Specifically, we examine differences in total bird abundance and species richness, foraging and nesting guilds, and individual species among three fuel reduction treatments and controls in the southern Appalachians before--and for two breeding seasons after--all treatments were implemented.

Methods

Our study was conducted on The Green River Game Land in Polk County, North Carolina. The Game Land lies entirely within the mountainous Blue Ridge Physiographic Province of Western North Carolina. Soils are primarily of the Evard series (fine-loamy, oxidic, mesic, Typic Hapludults) which are very deep and well-drained in mountain uplands (USDA Natural Resources Conservation Service 1998). There are also areas of rocky outcrops in steeper terrain. Forest stands are composed mainly of oaks (*Quercus* spp.) and hickories (*Carya* spp.). Shortleaf (*Pinus echinata*) and Virginia (*P. virginiana*) pines are found on ridgetops, and white pine (*P. strobus*) occurs in moist coves. Elevation ranges from approximately 366-793m.

We selected three study areas (blocks) within the Game Land. First and second order streams bordered and/or traverse all three replicate blocks. Study blocks were selected based upon stand size (large enough to fit all four treatments), stand age, cover type, and management history, to insure consistency in baseline conditions among the treatments. Stand ages varied from 80 to 120 years (Waldrop 2001). Minimum treatment size (four within each block) was 14 ha to allow for 10-ha treatment “core” areas, with 20m buffers around each. None of the sites had been thinned during the past ten years and none had been burned in at least five years. Oaks dominated all sites. Other dominant species included pignut hickory (*C. glabra*), mockernut hickory (*C. tomentosa*), and shortleaf pine (*P. echinata*). Generally, thick shrub layers occurred along ridge tops and on upper southwest-facing slopes. Predominant shrubs were mountain laurel (*Kalmia latifolia*) and rhododendron (*R. maximum*).

Three treatments and an untreated control (C) were randomly assigned to each of the three study blocks. Treatments were: (1) fuel reduction by mechanical understory felling (M), (2) fuel reduction by prescribed burning (B), and (3) fuel reduction by mechanical understory felling + prescribed fire (MB). Mechanical understory felling treatments were conducted during winter 2001-2002. The understory thinning was conducted using chainsaws, and included all mountain laurel, rhododendron, and trees >1.8 meters tall and <10.0 cm in diameter at breast height (dbh). Fuels were not removed from the site due to the high cost of operating in steep terrain. Prescribed burns were conducted in B and MB treatments on March 12 or 13, 2003. One block was burned by hand ignition using spot fire and strip-headfire techniques. The other blocks were ignited by helicopter using a spot fire technique. The objectives of prescribed burning were to remove the shrub layer and create a few snags for wildlife habitat.

We censused bird communities using three 50m radius (0.785 ha area) point counts spaced 200m apart in each treatment (Ralph et al. 1993). Each point was surveyed for 10 minutes during three separate visits between 15 May and 30 June during 2001-2004. Point counts were conducted within four hours of sunrise. All birds that were seen or heard were recorded. We rotated the times for point counts among the three visits to each treatment to avoid time-of-day biases. Bird abundance for each treatment was calculated by averaging across the three visits and the three censuses for each year, and extrapolating the average number per point count to number per ha. Species richness was calculated for each treatment by summing the number of species detected during all three visits and point counts each year.

We measured post-treatment habitat features during the first growing season after treatment implementation (2002 for M, 2003 for B, C, and MB). Snag ($\geq$10 cm dbh) density and percent cover of shrub species $\geq$1.4m high were measured within ten, 0.05-ha (10x50m) plots that were spaced systematically within each treatment. Leaf litter depth was measured at three locations along each of three randomly oriented, 15m transects originating at gridpoints that were spaced at 50m intervals throughout treatment areas. Canopy openness was measured at two randomly selected points within each treatment during summer (leaf on) 2003 using a spherical densiometer held at breast height.

We used one-way, ANOVAs (SAS 1990) in a complete randomized block design for each year separately (because treatments were implemented incrementally) to examine differences among the four treatments in total bird abundance, species richness, abundance of select, common species (if n > 30 detections during the four-year study period), and the abundance of birds in nesting and foraging guilds. Each species was assigned a single nesting (ground, shrub, tree, and cavity) and foraging (ground, shrub, and tree) guild (adapted from Hamel 1992). For vegetation analyses, we used ANOVAs to determine post-treatment differences in snag ($\geq$10 cm DBH) density, percent tall ($\geq$1.4m ht) shrub cover, leaf litter depth (Waldrop 2001), and canopy openness. Percentage data (shrub cover and canopy openness) was square-root arcsine transformed for analyses. We used least squares means tests to determine significant differences among treatments. We considered $P < 0.10$ as statistically significant due to high among-site variability in bird detections.

Results

Fire intensities varied within and among sites, but were generally moderate to high. Flame lengths of one to two meters occurred throughout all burn units but reached up to five meters in localized spots where topography or intersecting flame fronts contributed to erratic fire behavior. Loading of fine woody fuels in MB sites was essentially double that of C and M sites due to felling of the shrub layer. Measured temperatures were generally below 120°C in B sites but often exceeded 800°C in MB sites.

Both snag density and canopy openness were higher in MB than the other treatments in 2003 after all treatments had been implemented (*table 1*). Tall ($\geq$1.4m ht) shrub cover was

Table 1—*Mean ($\pm$SE) post-treatment number of snags (per ha), percent cover of tall (>1.4m ht) shrubs, percent canopy openness, and leaf litter depth (cm) in three treatments: burned (B), mechanical understory felling (M), mechanical understory felling followed by burning (MB), and controls (C) (n = 3 each), Green River Game Land, Polk County, NC. P-values represent block effects (P_{Block}) and treatment effects ($P_{Treatment}$). Differences among treatments were determined by least squares means tests, and are denoted by different letters within rows.*

Feature	*Treatment*				P_{block}	$P_{treatment}$
	B	C	M	MB		
Snags/ha	72.7 ± 19.0^{A}	68.0 ± 9.0^{A}	52.7 ± 4.4^{A}	152.0 ± 25.3^{B}	0.74396	0.0309
Canopy openness (%)	2.6 ± 1.1^{A}	1.6 ± 0.4^{A}	3.0 ± 0.8^{A}	12.8 ± 5.0^{B}	0.2047	0.0280
Shrub cover (%)	4.7 ± 2.8^{A}	20.0 ± 3.9^{B}	1.4 ± 0.2^{A}	0.2 ± 0.2^{A}	0.5763	0.0078
Litter Depth (cm)	0.9 ± 0.1^{A}	4.2 ± 0.5^{B}	5.5 ± 0.2^{C}	0.5 ± 0.1^{A}	0.1389	<0.0001

significantly lower in all fuel reduction treatments (B, M, and MB) than in C *(table 1).* Leaf litter depth was significantly lower in both burned treatments (B and MB) than in either unburned treatment, and was slightly greater in M than C *(table 1).* No block effects were detected for habitat measurements.

We detected 45 breeding bird species within our point counts during the four year study period. Total breeding bird abundance was similar among treatments each year except 2003 when it was higher in C than in M *(fig. 1).* Species richness was similar among treatments except in 2004, when it was higher in MB than in other treatments

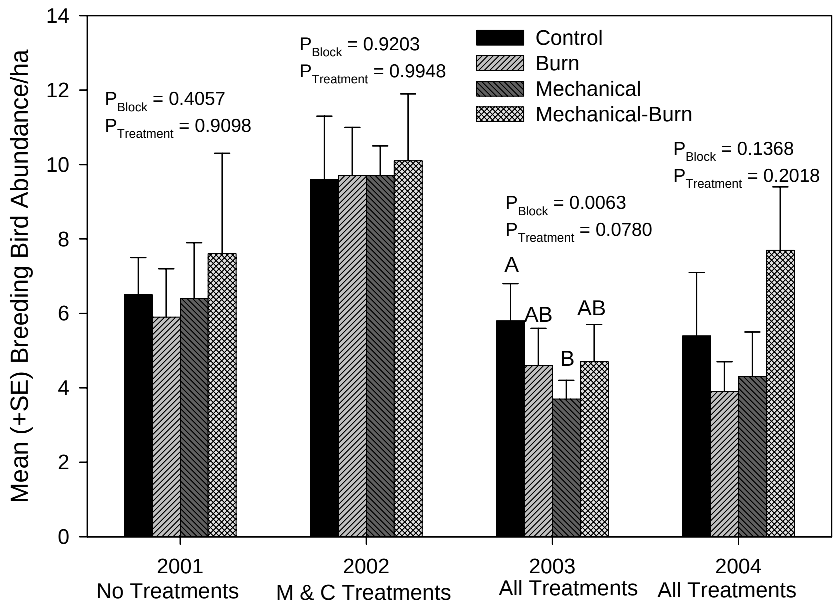

Figure 1—Mean (+SE) total abundance of breeding bird species in 3 treatments: burned (B), mechanical understory felling (M), mechanical understory feeling followed by burning (MB), and controls (C) (n=3 each), Green River Game Land, Polk County, NC. Treatments were not yet implemented in 2001 (pretreatment); in 2002 only (C) (including C and B) and M (including M and MB) were implemented; in 2003 and 2004 all four treatments were implemented. P-values represent block effects (P_{Block}) and treatment effects ($P_{Treatment}$). Significant differences among treatments within years were determined by least square means tests and are denoted by different letters.

(*fig. 2).* The abundance of ground and tree foragers did not differ among treatments in any year *(fig. 3).* However, shrub forager abundance was lower in all fuel reduction treatments immediately after understory reductions (in 2003), and lower in B than C in 2004 *(fig. 3).* Shrub nester abundance was similar among treatments in all years except 2004, when it was higher in MB than in B or M *(fig. 3).* The abundance of tree, ground, and cavity nesters did not differ among treatments in all years.

Responses to fuel reduction techniques were most evident at the species level. Most species, including those associated with closed canopy forest, showed no response to treatments. For example, Tufted Titmice (*Baeolophus bicolor*) and Blue-headed Vireos (*Vireo solitarius*) were common and showed no response to fuel reduction treatments during the study period *(table 2).* Red-eyed Vireos (*Vireo olivaceous*) showed a short-term

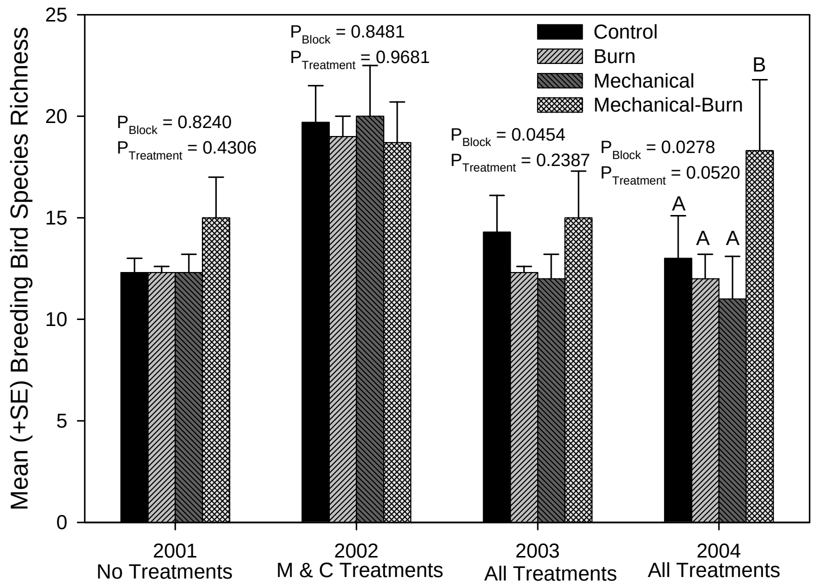

Figure 2—Mean (+SE) breeding bird species richness in 3 treatments: burned (B), mechanical understory felling (M), mechanical understory feeling followed by burning (MB), and controls (C) (n=3 each), Green River Game Land, Polk County, NC. Treatments were not yet implemented in 2001 (pretreatment); in 2002 only (C) (including C and B) and M (including M and MB) were implemented; in 2003 and 2004 all 4 treatments were implemented. P-values represent block effects (P_{Block}) and treatment effects ($P_{Treatment}$). Significant differences among treatments within years were determined by least square means tests and are denoted by different letters.

(in 2003) reduction in numbers in response to all fuel reduction treatments, but differences were not detected by 2004. During 2003 (after treatments had been implemented), the ground-nesting Worm-eating Warbler (*Helmitheros vermivorous*) was less abundant in MB than in C or M, and, in 2004, it was less abundant in both B and MB compared to C or M *(table 2).* However, in 2001 (before any treatments were implemented), they were less abundant in B than in C or M *(table 2).* Hooded Warblers (*Wilsonia citrina*) also declined in numbers in all fuel reduction treatments, compared to C, following fuel reduction treatments during 2003 and 2004 *(table 2).* Indigo Buntings (*Passerina cyanea*), which are associated with young, open habitat, were more abundant in MB than other treatments during 2004 *(table 2).* Block effects were detected for some species and community-level measures during some years, indicating that there was variability among replicate blocks, at least during some years.

Discussion

The fuel reduction treatments resulted in dramatic changes in habitat structure. Tall shrub cover averaged $\geq$ 4.3 times lower in all fuel reduction treatments compared to C. Leaf litter depth in both burned treatments (B and MB) averaged $\geq$ 4.6 times less compared to unburned treatments (M or C). However, snag densities and canopy openness were much higher in MB compared to other treatments, likely contributing to our observed changes in the abundance of some breeding bird species. In MB, snag density doubled, with a corresponding eightfold increase in canopy openness compared to C. High overstory mortality in

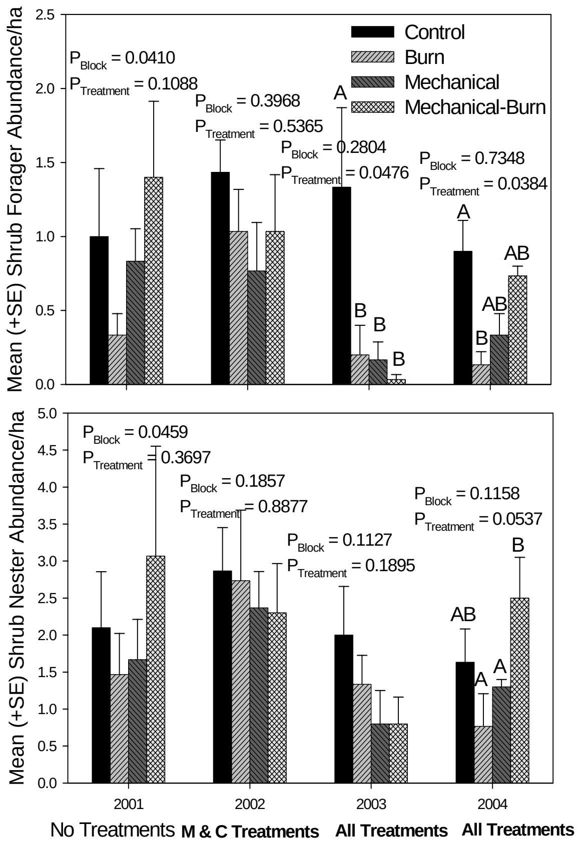

Figure 3—Mean (+SE) abundance of a) shrub foragers, and b) shrub nesters in 3 treatments: burned (B), mechanical understory felling (M), mechanical understory feeling followed by burning (MB), and controls (C) (n=3 each), Green River Game Land, Polk County, NC. Treatments were not yet implemented in 2001 (pretreatment); in 2002 only (C) (including C and B) and M (including M and MB) were implemented; in 2003 and 2004 all 4 treatments were implemented. P-values represent block effects (P_{Block}) and treatment effects ($P_{Treatment}$). Significant differences among treatments within years were determined by least square means tests and are denoted by different letters.

MB sites was likely due to the stress of hotter fires resulting from higher fuel loadings. Xeric site conditions, prolonged drought, and root-borne pathogens may also have contributed to tree mortality in response to burning.

In our community-level analyses, we detected little response by breeding birds to the fuel reduction treatments. We found no difference in total bird abundance among the four treatments, except during 2003 when it was lower in M than in C. Artman et al. (2001) also found no difference in total bird abundance in burned and unburned mixed-oak forest in Ohio. This finding is likely because many common species do not show a detectable response; other species may decline or increase, with little net change in total bird abundance.

During 2004, species richness was higher in MB than in the other treatments. This was due to the addition of a few species that are typically associated with open habitat, created in this case by high tree mortality, such as Chipping Sparrows (*Spizella passerina*), Eastern Bluebirds (*Sialia sialis*), and Indigo Buntings. Higher snag densities may also have contributed to more Eastern Bluebirds (which are cavity nesters) in MB, but we did not document nesting. Weakland et al. (2002) also reported that the mosaic of shrub cover and tree mortality created by patchy burns likely promoted higher bird species richness.

Table 2—*Mean (±SE) abundance (per ha) of select breeding bird species in three treatments: burned (B), mechanical understory felling (M), mechanical understory felling followed by burning (MB), and controls (C) (n = 3 each), Green River Game Land, Polk County, NC. P-values represent block effects (P_{Block}) and treatment effects ($P_{Treatment}$). Differences among treatments within years were determined by least square means tests and are denoted by different letters within rows. Treatments were not yet implemented in 2001 (pretreatment); in 2002 mechanical understory felling had been implemented (in M and MB); in 2003 and 2004 all fuel reduction treatments had been implemented.*

Species Common Name *(Scientific Name)*	Year	Treatment B	 C	 M	 MB	P_{bloc}	$P_{treatment}$
Blue-headed Vireo	2001	0.8±0.5	0.6±0.4	0.4±0.2	0.6±0.4	0.0023	0.4471
(Vireo solitarius)	2002	0.9±0.6	0.8±0.3	1.2±0.5	0.7±0.3	0.0857	0.7818
	2003	0.3±0.2	0.3±0.1	0.4±0.2	0.2±0.1	0.3440	0.7620
	2004	0.3±0.3	0.4±0.2	0.6±0.2	0.3±0.2	0.0644	0.6312
Tufted Titmouse	2001	0.3±0.1	0.4±0.2	0.4±0.3	0.2±0.1	0.5914	0.9273
(Baeolophus bicolor)	2002	0.6±0.2	0.5±0.1	0.7±0.3	0.8±0.3	0.4736	0.8487
	2003	0.4±0.2	0.4±0.2	0.1±0.1	0.4±0.1	0.3500	0.5979
	2004	0.4±0.2	0.4±0.2	0.5±0.2	0.7±0.1	0.1203	0.5178
Hooded Warbler	2001	0.3±0.1	1.0±0.5	0.8±0.2	1.0±0.3	0.0408	0.1715
(Wilsonia citrine)	2002	1.0±0.2	1.2±0.2	0.5±0.1	0.7±0.3	0.4011	0.1822
	2003	0.2 ± 0.2^{A}	1.2 ± 0.5^{B}	0.2 ± 0.1^{A}	0.0 ± 0.0^{A}	0.2850	0.0497
	2004	0.1 ± 0.1^{A}	0.8 ± 0.3^{B}	0.3 ± 0.1^{A}	$<0.1\pm0.1^{A}$	0.6631	0.0680
Indigo Bunting	2001	0.0±0.0	0.0±0.0	0.0±0.0	0.3±0.2	0.4219	0.1386
(Passerina cyanea)	2002	<0.1±0.1	0.1±0.1	0.2±0.2	0.2±0.2	0.6016	0.7288
	2003	0.0±0.0	0.0±0.0	0.0±0.0	<0.1±01	0.4219	0.4547
	2004	0.1 ± 0.1^{A}	0.0 ± 0.0^{A}	0.0 ± 0.0^{A}	0.7 ± 0.1^{B}	0.6141	<0.0001
Red-eyed Bunting	2001	1.0±0.5	1.0±0.4	0.9±0.3	1.1±0.2	0.8960	0.9896
(Vireo Olivaceus)	2002	0.8±0.1	0.7±0.2	0.7±0.3	1.1±0.4	0.0556	0.4958
	2003	0.4 ± 0.2^{A}	0.8 ± 0.1^{B}	0.3 ± 0.1^{A}	0.4 ± 0.1^{A}	0.1447	0.0847
	2004	0.3±0.2	0.7±0.2	0.4±0.2	0.4±0.1	0.0339	0.4645
Worm-eating Warbler	2001	0.0 ± 0.0^{A}	0.5 ± 0.2^{B}	0.3 ± 0.2^{B}	0.2 ± 0.2^{AB}	0.0307	0.0741
(Helmitheros vermivorus)	2002	0.4±0.2	0.4±0.1	0.3±0.1	0.3±0.1	0.2553	0.8011
	2003	0.2 ± 0.1^{AB}	0.3 ± 0.1^{A}	0.3 ± 0.1^{A}	0.0 ± 0.0^{B}	0.1053	0.0967
	2004	0.1 ± 0.1^{A}	0.3 ± 0.1^{B}	0.3 ± 0.0^{B}	0.0 ± 0.0^{A}	0.3302	0.0366

At a guild-level, only shrub-associated breeding birds showed a detectable response to the fuel reduction treatments. Interestingly, no response was evident after mechanical understory reductions had been implemented (2002) but before the prescribed burns (which were conducted in 2003). In 2003, shrub foragers showed reduced densities in all fuel reduction treatments; some recovery was evident by 2004 when their numbers were lower only in B compared to C. Artman et al. (2002) also reported declines in ground and shrub nesting and foraging guilds in response to burning in Ohio, but not for other guilds.

Rodewald and Smith (1998) found fewer shrub nesters after an understory removal in an Arkansas oak-hickory forest. In contrast, we found similar numbers of shrub nesters among all treatments during all years except 2004, when they were more abundant in MB than in B or M. Aquilani (2000) reported lower nest success of ground and shrub nesters in an Indiana forest after burning. In our study, canopy foraging and nesting guilds were not affected by the fuel reduction treatments, despite substantially lower canopy cover between MB and the other treatments. Similarly, cavity nesters did not differ in abundance among treatments, despite much higher snag densities in MB. We hypothesize that cavity nesters will increase in MB as snags decay and become suitable for excavation by woodpeckers.

Community-level parameters, such as species richness, total density, or bird abundance within guilds, are useful as indices to gauge the effects of forest management practices. However, they are often too crude to detect changes that may be occurring at the species level. For example, the species richness may be similar among treatments, but the species that are present, or the relative abundance of each, might differ. Similarly, overall bird density does not address what species are more or less abundant in the various treatments. Assignment of birds to guilds may impose rigid categories on species that in fact use many habitats, or components of habitats, for various activities. In our study, however, understory reductions clearly affected birds that use shrubs for foraging and/or nesting at the guild level.

Most bird species showed no significant difference in relative abundance among the fuel reduction treatments, even two years post-burn. For example, Tufted Titmice and Blue-headed Vireos did not differ in abundance among treatments in any year. However, presence, or habitat use by a species, is not necessarily indicative of habitat quality (Van Horne 1983), and must be viewed with caution.

A few species showed a significant response to some or all of the fuel reduction treatments. Red-eyed Vireos nest and forage primarily in tree canopies, and were more abundant in C than in other treatments in 2003, but not during other years. Possibly understory felling in the fuel reduction treatments influenced their abundance in the short-term. Duguay et al. (2001) also found that Red-eyed Vireos were abundant in stands with different levels of basal area, ranging from clearcuts to unharvested stands. Hooded Warblers, which nest and forage primarily in shrubs, occurred in much lower numbers in all fuel reduction treatments compared to C, after all treatments had been implemented (2003 and 2004). This was likely due to reductions in shrub cover in the fuel reduction treatments. However, we did not detect differences in Hooded Warbler abundance among treatments in 2002, after the mechanical understory felling treatments (only) had been done in the M and MB treatments. Worm-eating Warblers are ground nesters, and forage on the ground and in shrubs. They also showed reduced numbers in both burned treatments (B and MB) after the burns had been implemented (but also showed lower numbers in B during 2001 before the burns had been conducted). This is likely due to reductions in shrub cover, and lower leaf litter depth (and probably litter cover) in the burned treatments. In an Ohio oak forest, Artman et al. (2001) also found that some species, including Ovenbirds, Worm-eating Warblers, Hooded Warblers, and Northern Cardinals (*Cardinalis cardinalis*), were negatively affected by 1-4 years of annual burning. Rodewald and Smith (1998) also found fewer Worm-eating Warblers where the understory and (or) tree basal area was reduced.

In contrast, Indigo Buntings showed a significant increase in numbers in the MB treatment during 2004 (15 months after burning). This species is associated with open habitat, but they also nest and forage in shrubs. However, Indigo Buntings did not increase in abundance in B where light and snag numbers were similar to C and M. Their positive response to the MB treatment was likely due to the more open conditions that resulted from tree death after the hotter prescribed fire. Rodewald and Smith (1998) reported a higher density of Indigo Buntings in oak-hickory forest stands that had received understory removal and basal area reduction treatments. Hejl (1994) suggested that bird response to fire varies according to fire severity, and the corresponding post-burn conditions.

After initial changes in vegetation structure from mechanical understory felling or prescribed fire, post-disturbance vegetation structure continues to change rapidly for several years. Snag densities will likely decrease in MB as they decay and fall, and canopy openness will correspondingly increase. The shrub layer will likely recover, and leaf litter depth may increase as dead leaves fall from trees and shrubs. The abundance of food resources for birds, such as insects (Whitehead 2003) and fruit (Greenberg and Levey 2004), also can change over time after disturbance (Blake and Hoppes 1986). Bird species respond differently to habitat structure, so changes in the relative abundance of species might be expected for several years after the initial disturbance. Specifically, we hypothesize that (1) differences in the relative abundance of shrub-associated species will disappear as the shrub layer recovers in fuel reduction treatments, (2) cavity nesters will increase in MB as snags decay, and (3) bird species associated with open habitat will increase in MB as snags fall, at least in the short-term. To fully understand how fire affects breeding birds at the community- and species-level, post-fire surveys of birds and vegetation structure must continue for several years (Raphael et al. 1987).

Summary

Our results indicate that the habitat alterations resulting from mechanical understory felling and (or) burning affect breeding bird communities and species differently. At the guild level, shrub foragers showed the greatest response, with lower abundance in all fuel reduction treatments. In our study, bird response was better detected at the species-level than at the community-level. Most species showed no change in density after the fuel reduction treatments. Some species, such as Hooded Warblers and Worm-eating Warblers, decreased in density after some or all treatments, whereas others, such as Indigo Buntings, increased in abundance. Responses likely differed among bird species depending upon habitat suitability for nesting, foraging, and cover. Heterogeneous habitat conditions, especially within the B and MB treatments, likely reduced our ability to detect some breeding bird response to the fuel reduction treatments. To fully understand how fire and fire surrogates for fuel reduction affect breeding bird communities, post-fire surveys of birds and vegetation structure must continue for several years.

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Riparian and Upland Vegetation on the Kings River Experimental Watershed, Sierra Nevada, California[1]

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Abstract

The Kings River Experimental Watershed (KREW) is a watershed-level study on headwater streams in the Sierra Nevada, California. Eight perennial streams, from 1500 m (4920 ft) to 2490 m (8170 ft) elevation, have been instrumented and collecting data since 2002. Component research areas of the study include stream flow, water chemistry, sediment, soil chemistry, stream invertebrates, stream algae, riparian and upland vegetation and meteorology. The KREW is part of a larger project called the Kings River Project (KRP). The KRP is a collaboration between the Pacific Southwest Research Station and the Sierra National Forest, with the objective of restoring pre-European-settlement forest conditions in the Forest. To achieve this objective, KRP will implement prescribed fire and uneven-aged management across the landscape, depending on the needs of component studies, and the condition of the land. Treatments for KREW will comprise prescribed fire, mechanical thinning, and a thin/burn combination, and will begin in the fall of 2006. Data collection will therefore include pre-, interim- and post-treatment periods. The vegetation component of KREW is designed to address the effects of these treatments on the herbaceous, shrub and canopy communities of the watersheds. In particular, effects on riparian versus upland communities will be quantified and compared, to assess the effects of fire on riparian vegetation. The arid environment of the southern Sierra creates comparatively narrow riparian bands along streams, usually less than 2 m wide. The role riparian zones play in protecting the stream from disturbance such as fire and logging as been well-studied. However, those studies come primarily from regions with wide, dense riparian zones, such as the Pacific Northwest, and information from the more narrow riparian bands of the Sierra is lacking. Currently, 114 upland and 56 riparian, permanent transects have been established and are censused every summer. Pre-treatment data will be analyzed to determine plant community importance values by transect, to correlate with variables such as stream proximity, as well as provide a background characterization for the vegetation on each watershed. As of August, 2005, 308 taxa have been collected from the eight watersheds and identified to species. Data will be analyzed to determine what proportions of these species are in riparian versus upland communities. Ultimately, post-treatment data will be compared with pre-treatment data to describe changes in the vegetation according to different treatments. This information will help land managers determine how to best approach the endeavor of restoration of the plant community to a pre-settlement state.

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Prescribed Burning Ineffective for Improving Turkey Habitat on a Recently Regenerated Mesic Site in the Southern Appalachian Mountains[1]

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Abstract

Recently regenerated mesic sites in the southern Appalachian Mountains often provide poor brooding areas for wild turkey (*Meleagris gallopavo*) because shade from thick stands of hardwood saplings reduces cover of herbaceous vegetation and the accompanying insects that provide the essential protein needed by young poults. An operational prescribed fire was used to reduce density of 5-yr-old hardwood regeneration on an east-facing cove site that was dominated by saplings of yellow-poplar (YP) (*Liriodendron tulipifera* L.), a mesophytic, shade-intolerant species that regenerates readily from sprouts and stored seeds, and grows rapidly to form dense, "dog-hair" thickets of tall, thin saplings. Sixteen tree species were present on the 21- ac site that was regenerated in 1995 by the shelterwood method, with a residual basal area of 80 ft^2/ac. Before burning, YP accounted for about half of the 11,790±2,673 (±SE) stems/ac, which ranged in height from 2 to 12 ft. The late-spring 2001 flanking and heading fire burned with high intensity and consumed about 78 percent of the 6 tons/ac of fuels present, half of which was logging slash from the largely YP stand that averaged about 14,300 board ft/ac of sawtimber. Postburn sapling top-kill was over 95 percent and herbaceous response was immediate, resulting in a dense cover of fireweed (*Erechtites hieracifolia* L.) and pokeberry (*Phytolacca americana* L.), with lesser amounts of blackberry (*Rubus* spp. L.) and grasses (*Poaceae*).

The fire improved overall wildlife habitat, likely for several years, by increasing browse and soft mast production, but benefits to turkey brooding habitat were short-lived. Over 50 percent of the top killed YP saplings developed basal sprouts, which grew rapidly and reclaimed much of their preburn dominance -- averaging 3 ft in height by fall. In addition, many new YP seedlings originated from germinating seeds stored in the unburned, lower layers of the forest floor and from plentiful wind-blown seeds from nearby stands around the burned site. A second prescribed burn was attempted the following spring to kill new sprouts and seedlings, but failed largely from lack of fuels. Results from this case study suggest that prescribed fire alone may not be a viable method of controlling hardwood saplings on mesic sites to obtain and maintain herbaceous vegetation desirable for turkey brooding habitat.

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Putting Out Fire With Gasoline: Pitfalls in the Silvicultural Treatment of Canopy Fuels[1]

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Abstract

There is little question that forest stand structure is directly related to fire behavior, and that canopy fuel structure may be altered using silvicultural methods to successfully modify forest fire behavior and reduce susceptibility to crown fire initiation and spread. Silvicultural treatments can remediate hazardous stand structures that have developed as a result of the exclusion of low-intensity surface fires: abundant case studies offer evidence of crown fires subsiding upon encountering recently-thinned stands, and modeling studies corroborate. Yet treatments applied to abate one component of crown fire potential may inadvertently promote conditions that exacerbate fire behavior. Canopy fuel treatments typically target one or two parameters of fuel load and contiguity, but they directly or indirectly influence many more related components. In addition, canopy fuel treatments directly affect stand development patterns, and hence future fuel structures and fire behaviors. A review of stand processes associated with thinning suggests nine situations by which silvicultural treatment of canopy fuels can inadvertently exacerbate crown fire hazard or fire severity:

1) translocation of live aerial fuels to dead surface fuel complex
2) inflation of fuelbed depth associated with treatment residues
3) increase in fuel availability due to modified forest floor microclimate
4) enhancement of subcanopy wind penetration and turbulence
5) reduction of duff moisture content associated with reduced canopy shade
6) proliferation of stump sprouts in the lives surface fuel load
7) proliferation of seedling regeneration due to forest floor scarification
8) release of advance regeneration and development of midcanopy fuel layer
9) cessation of overstory crown recession and vertical integration of fuel complexes

Silvicultural manipulations to degraded fire-adapted forest ecosystems offer great promise for restoration, but prescriptions must be examined carefully for their dynamic effect on fuel structures over time. Rather than restoring historic fire regimes, today's fuels management interventions establish new fuel structures and put stands on new trajectories of structural development that have direct implications for future fuel structures and fire behavior. Understanding the many ways that stand structure relates to fire behavior and crown fire hazard helps avoid negative consequences. Understanding forest fuels dynamics, or changes in forest fuel structures over time, helps to forecast the persistence of canopy fuel treatment effectiveness and the extended influence of those treatments on future fire behaviors.

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Thinning and Underburning Effects on Ground Fuels in Jeffrey Pine[1]

R.F. Walker,[2] R.M. Fecko,[2] W.B. Frederick,[2] J.D. Murphy,[2] D.W. Johnson,[2] and W.W. Miller[2]

Abstract

Thinning with cut-to-length and whole-tree harvesting systems followed by underburning were evaluated for their impacts on downed and dead fuel loading by timelag category in eastern Sierra Nevada Jeffrey pine (*Pinus jeffreyi* Grev. & Balf.). Cut-to-length harvesting resulted in an approximate doubling of total fuel loading to 113829 kg ha^{-1} in comparison with the unthinned control treatment, which totaled 55865 kg ha^{-1}. The greatest increases occurred in the 100-hr and 1000-hr categories, amounting to 466% and 354%, respectively, while 1+10-hr fuels increased by only 61%. Ground fuel changes associated with whole-tree harvesting were marginal and ultimately the loading in this treatment did not differ significantly from that in the unthinned treatment regardless of timelag category. The 1+10-hr and total fuel accumulations in the cut-to-length treatment and that of 1000-hr fuels in the whole-tree treatment were positively correlated with harvested basal area and harvested foliage, branch, bole, and total tree biomass. Subsequent consumption during underburning eliminated 1+10-hr and 100-hr fuel additions from cut-to-length harvesting, along with a portion of the natural loading in these categories, but fire was much less effective in reducing the 1000-hr fuels generated by this thinning approach as only 14% of the latter was consumed. Consumption of 1+10-hr, 100-hr, and total fuels in both the cut-to-length and whole-tree treatments was positively correlated with the amounts present within each category before underburning, relationships that extended to the unthinned treatment as well. These results, based on a study conducted on the Tahoe National Forest, provide insight into fuel load modifications resulting from field practices that are being increasingly integrated into comprehensive management efforts to improve forest health in the Sierra Nevada.

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Thinning and Underburning Effects on Productivity and Mensurational Characteristics of Jeffrey Pine[1]

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Abstract

Thinning utilizing cut-to-length and whole-tree harvesting systems with subsequent underburning were assessed for their influence on stand productivity and mensurational variables in uneven-aged Jeffrey pine (*Pinus jeffreyi* Grev. & Balf.) on the Tahoe National Forest. Both intermediate and a combination of dominant and codominant crown class trees were selected within each treatment to evaluate stand productivity based on measurements of pre- and posttreatment ring widths from sample cores, while stand mensurational attributes, derived from trees ≥ 10.2 cm DBH, were collected from permanent measurement plots. These trees were further subdivided into two size categories: (1) ≥ 17.8 cm DBH, ≤ 19.8 m tall and (2) ≥ 25.4 cm DBH, based on their likelihood of either becoming or retaining their status as long-term stand constituents, respectively. Radial growth responses to treatment in both intermediate and dominant/codominant crown class trees clearly demonstrated a thinning effect, with cut-to-length and whole-tree subunits of the stand exhibiting responses ranging from negligible change to substantial increases in posttreatment increment. In contrast, trees in unthinned stand portions exhibited considerable declines in this regard compared to pretreatment values. Neither the fire treatment nor the interaction between thinning and fire treatments exerted a significant influence on radial growth. Comparisons of post- to preburn mortality revealed significant thinning and fire main treatment effects as well as significant interaction between these two treatments in both tree size categories. However, mortality increased most in the small size class within the burned portion of the whole-tree subunit, whereas the values of this variable among the larger trees rose most sharply in that of the cut-to-length subunit. Post- to preburn shifts in live crown, expressed as a percentage of total tree height, were significantly affected by both thinning and fire main treatments in the small and large tree categories, while the interaction of these treatments was also significant in the latter. Within both size classes, decreases in live crown percentage were greatest in the burned portion of the unthinned subunit, with the second highest losses occurring within that of the cut-to-length subunit. These results present land managers with plausible outcomes of differing forest management field practices presently being employed to enhance forest health and reduce wildfire risk in the Sierra Nevada.

[1] Poster presented at the National Silviculture Workshop, June 6-10, 2005, Tahoe City, California.

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Effect of Burn Residue Proximity on Growth of 5 Planted Mixed-Conifer Species After 6 Years[1]

Robert A. York[1] and Robert C. Heald[2]

Abstract

Burned areas represent a significant amount of the forest landscape that can potentially be planted following site preparation techniques that use burning of woody residue. However, managers implementing post-harvest or post-wildfire regeneration efforts face uncertainty in expected performance of seedlings planted in or around post-burn residues (i.e. ash substrates). To address this uncertainty, five species were planted following site preparation beneath a shelterwood overstory in a Sierra Nevada mixed conifer forest. We planted seedlings within, on the edge, and outside of ash substrates following experimental burning of uniform debris piles. After six years, height and radial growth were evaluated with respect to burn pile proximity. For Douglas-fir, sugar pine, ponderosa pine, and giant sequoia, relative and absolute height and radial growth were influenced by burn pile proximity. In general, seedlings planted within burn piles grew better than seedlings planted on the edges and outside of burn piles. Incense cedar growth was not influenced by burn pile proximity. Shrub competition also varied by burn pile proximity, but was only important in explaining Douglas-fir height growth. Mortality for all species was low regardless of burn pile proximity. Further opportunities exist for this study exploring the effects of fire-caused soil nutrient changes on seedling growth over time.

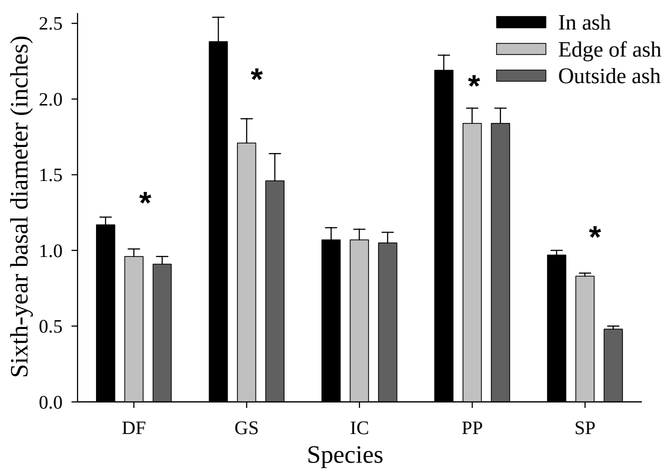

Figure 1--Mean basal diameters (and standard errors) of seedlings six years after planting by ash bed position. ***** denotes a significant (p<0.05) influence of ash bed proximity on mean height using a general linear model with shrub competition as an explanatory variable (ANCOVA).

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Soil Responses to the Fire and Fire Surrogate Study in the Sierra Nevada[1]

Emily E.Y. Moghaddas[2] and Scott L. Stephens[3]

Abstract

The Fire and Fire Surrogate Study utilizes forest thinning and prescribed burning in attempt to create forest stand structures that reduce the risk of catastrophic wildfire. Replicated treatments consisting of mechanical tree harvest (commercial harvest plus mastication of sub-merchantable material), mechanical harvest followed by prescribed fire, prescribed fire alone, and no-treatment controls, were completed at the Blodgett Forest Research Station in fall 2002. We conducted pre-treatment and post-treatment assessments of soil physical, chemical, and biological characteristics. Soil bulk density measures were used to assess soil compaction. At the treatment unit level, there were no differences among treatments in soil bulk density. However, soil bulk density was significantly greater in skid trails of harvested stands compared to undisturbed ground. The presence of skid trails in all treatment units (due to current and past harvest activities) increased the heterogeneity of the soil environment, and may influence treatment effects. Skid trails generally moderated fire effects. Effects such as increased soil pH, increased base saturation, and increased exchangeable calcium were significantly greater in burned undisturbed areas than in skid trails within burned areas. Due to reduced fuels in skid trails, the amount of direct heating and combustion was greatly reduced. Following fire, skid trails had greater total soil carbon and soil carbon:nitrogen ratios than undisturbed areas. In harvested stands, skid trails may occupy ten percent or more of the stand area. Localized treatment effects, such as those within skid trails, must be considered when interpreting overall stand treatment effects.

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The Effect of Mechanical Fuel Reduction Treatments in the Wildland-urban Interface on the Amount and Distribution of Bark Beetle-Caused Tree Mortality[1]

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Abstract

Selective logging, fire suppression, forest succession, and climatic changes have resulted in high fire hazards over large areas of the western USA. Federal and state hazardous fuel reduction programs have increased accordingly to reduce the risk, extent and severity of these events, particularly in the wildland urban interface. In this study, we examined the effect of mechanical fuel reduction treatments on the activity of bark beetles in ponderosa pine, *Pinus ponderosa* Dougl ex. Laws., stands located in Arizona and California, USA. Treatments were applied in both late spring (April-May) and late summer (August-September) and included: (1) thinned biomass chipped and randomly dispersed within each plot, (2) thinned biomass chipped, randomly dispersed within each plot, and raked 2 m from the base of residual trees, (3) thinned biomass lop-and-scattered (thinned trees cut into 1-2 m lengths) within each plot, and (4) an untreated control. The mean percentage of trees attacked by bark beetles ranged from 2.0 percent (untreated control) to 30.2 percent (plots thinned in spring with all biomass chipped). A three-fold increase in the proportion of trees attacked by bark beetles was observed in chipped versus lop-and-scattered plots. Higher levels of bark beetle colonization were associated with spring treatments. Raking chips away from the base of residual trees did not significantly affect attack rates. Several bark beetle species were present including the roundheaded pine beetle, *Dendroctonus adjunctus* Blandford, western pine beetle, *D. brevicomis* LeConte, mountain pine beetle, *D. ponderosae* Hopkins, red turpentine beetle, *D. valens* LeConte, Arizona fivespined ips, *Ips lecontei* Swaine, California fivespined ips, *I. paraconfusus* Lanier, and pine engraver, *I. pini* (Say). *Dendroctonus valens* was the most common bark beetle infesting residual trees.

Based on these results, managers should consider chipping during periods of bark beetle inactivity (e.g., late summer through winter) when possible. Reasonable effort should be made to limit large quantities of chips from directly contacting residual trees. Treatments that promote the desiccation of slash and slow release of monoterpenes prior to chipping should be considered. The implications of these results to sustainable forest management were discussed in detail.

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