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Jerry F. Franklin and K. Norman Johnson

Moist Forest Principles and Prescriptions (emphasizing the creation of diverse early successional ecosystems through variable retention regeneration harvest)

Background

Moist Forests are characterized by disturbances that typically include a significant stand-replacement component at intervals of one to several hundred years; subsequently forests require centuries to redevelop the characteristic massiveness and structural complexity of old-growth forests. Key elements of a Moist Forest restoration strategy include:

- *Reserving existing older forest stands not already incorporated into reserves,* as well as individual older trees found within younger stands proposed for management, using a threshold age;

- *Accelerating development of complexity in existing young forest stands*, using a variety of silvicultural approaches, including variable density thinning in plantations;

- *Implementing regeneration harvests in younger forests located within Matrix (areas allocated to sustained timber production as one of their objectives), applying a variable retention harvest system.* Significant structural and compositional elements (e.g., individual trees, snags, and logs and small intact forest patches) would be retained to sustain biota and enrich the post-harvest stand;

- *Encouraging development of early successional ecosystems immediately following harvest*, in part by using less aggressive approaches to site preparation and tree regeneration. Early successional ecosystems are pre-forest ecosystems that occupy forested sites between a stand-replacement disturbance and re-establishment of a closed forest canopy. These ecosystems are typically rich in biodiversity, provide conditions for many species that are habitat specialists, and have unique functional roles. Well-developed examples of such ecosystems are currently rare in Moist Forest landscapes because of intensive approaches to establishing new forests, such as dense tree planting and efforts to control competing vegetation (Swanson et al. 2011); and

- *Incorporating all of the above into a comprehensive silvicultural system that also includes longer rotations* (e.g., 100-160 years) and multi-aged and mixed-species stands, thereby increasing both the diversity and resiliency of managed federal forest lands and increasing future social options.

Variable Retention Harvesting

This document provides a guide to the concept and application of variable retention regeneration harvesting (VRH) for Moist Forests. It also includes a description of a silvicultural system that uses the principles of ecological forestry and that emphasizes creation of diverse early successional ecosystems as a stage in forest development.

General goals in applying VRH include:

- Providing for continuity of forest structure, function, and biotic composition between forest generations;
- Regenerating a new cohort of trees;
- Life-boating plant and animal species by providing critical habitat, food sources, and micro-environmental conditions;
- Structurally enriching the post-harvest ecosystems, including the early successional (pre-forest-closure);
- Providing conditions that allow development of the early successional (pre-forest) ecosystem; and
- Altering visual conditions from within and outside of the harvest unit.

VRH involves the retention of structures, organisms, and conditions from a pre-harvest forest stand for incorporation into the post-harvest ecosystem and, ultimately, forest stand. These can include individual structures, such as old trees and snags, small (e.g., 1/4 to 4+ acre) intact areas of the pre-harvest stand, or patches of ecologically important conditions found in the pre-harvest stand (e.g., seeps and rock outcrops).

VRH is intended to be a part of a comprehensive silvicultural system. Elements of that silvicultural system can include provisions to allow significant expression of the early successional (pre-forest) stage, regeneration of a new tree cohort, development and intermediate management of mixed-age stands, and long rotations (e.g., >120 years). Goals of this silvicultural system include creation of more resilient and diverse ecosystems and landscapes, maintenance of well-regulated high-quality hydrologic systems, sustained availability of culturally important resources, and a continuous flow of diverse wood and other forest products.

Prescriptions for Variable Retention Harvesting

Prescriptions for VRH can and will be highly variable depending upon the management goals and stand conditions. A general prescription that has been used in some pilot projects in southwestern Oregon involves retention of 20 to 30% of the pre-harvest forest. The majority of the retention is in the form of small (e.g., ½- to 3-acre) intact patches (*"aggregates"*), which are not entered during the harvesting operation. Additional retention occurs as individual trees and snags and small clusters of trees. Retention of some of the individual trees will be to provide candidates for snag creation either using fire (broadcast slash burning) or mechanical means.

Portions of riparian buffers that extend into harvest units can sometimes be appropriately counted as contributing to some portion of the retention target. Limitations on credit for riparian buffers are necessary, though, because such buffers typically are spatially concentrated at the margins of harvest units, rather than well distributed throughout the unit. To achieve the stated goals for retention, retained structures need to be reasonably well distributed throughout the harvest unit; conversely, creation of large areas totally lacking in retention should be avoided (see below).

Selection of the Aggregates for Retention

Several types of areas will be candidates for location of retained pre-harvest patches or aggregates. These include:

- Representative patches of the pre-harvest forest stand (required);
- Locations of old-growth trees (the F-J restoration strategy requires retention of all old trees within a harvest unit);
- Concentrations of large woody debris
- Locations of large snags selected for retention;
- Special habitats within the harvest unit, such as seeps, rocky outcrops, and other areas of high species diversity;
- Patches dominated by hardwoods (if not adequately covered above); and
- Areas that can facilitate Northern Spotted Owl foraging (applicable where wood rats are a significant part of the NSO diet).

Distribution of the Aggregates

The goal in locating aggregates is to have them well distributed throughout the harvest unit. Logging methods will constrain potential locations, particularly on areas logged by cable systems. *The intent is to avoid creating large open areas lacking retention but, also, provide conditions suitable for development of early successional ecosystems and regeneration of shade-intolerant species.* Obviously, for a given level of retention (e.g., 30%) there will be tradeoffs between aggregate size and distributional objectives. E.g., focusing on distribution may require creating more small aggregates rather than a few large aggregates. On the other hand, some objectives, such as facilitation of Northern Spotted Owl foraging, may require larger aggregates.

Although we do not advocate a specific formula for distribution of aggregates some practitioners have done so. For example, foresters adopted a distributional requirement in applying VRH in British Columbia. In the Clayoquot Sound area of Vancouver Island, all portions of a harvest unit had to be within 2 tree heights of an aggregate or the edge of unit. The BC legislature in their legal definition of VRH require that 50% of the harvest area be within 1 tree height of an aggregate or edge (riparian buffers count in meeting this requirement).

Selection of Individual Structures for Retention

Some distributed tree structures are typically desirable, generally in the form of individual trees or small clusters of trees. These should be dominant or co-dominant trees of species more capable of tolerating an open condition (e.g., Douglas-fir). Some old-growth trees might also be retained as individual trees rather than as part of aggregates. Some of these trees are expected to provide a source of distributed snags and down logs within the harvest unit. Logging considerations will largely dictate available locations for these structures.

Post-Harvest Reforestation Strategy for VRH Units

Goals in subsequent treatment of the harvested sites will be to provide for significant development or expression of the early successional ecosystem that dominates between a stand-replacement disturbance and re-establishment of a closed forest canopy. Tree planting will be conducted at low levels (e.g., 200 +/- trees/acre to assure basic levels of restocking; natural regeneration will be expected to significantly augment the planting over time. Aggressive artificial regeneration and other treatments to hasten re-establishment of a forest canopy will not be undertaken, so as to allow for development of the early successional ecosystem.

Slash treatment can be accomplished either mechanically or by fire, but aggregates should be protected during such treatments.

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June 1, 2011

A Guide to Creating Diverse Early Successional Ecosystems through Variable Retention Regeneration Harvest on the Coos Bay District of the BLM

Jerry Franklin

Norm Johnson

Background

Moist Forests, such as occur on the Coos Bay District, are characterized by disturbances that typically include a significant stand-replacement component at intervals of one to several hundred years; subsequently forests require centuries to redevelop the characteristic massiveness and structural complexity of old-growth forests. Key elements of a Moist Forest restoration strategy include:

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developed examples of such ecosystems are currently rare in Moist Forest landscapes because of intensive approaches to establishing new forests, such as dense tree planting and efforts to control competing vegetation (Swanson et al. 2011); and

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maintenance of well-regulated high-quality hydrologic systems, sustained availability of culturally important resources, and a continuous flow of diverse forest products.

Initial Application on the Coos Bay District

The general prescription proposed in the Coos Bay Pilot Project is retention of 20 to 30% of the pre-harvest forest. The majority of the retention will be in the form of small (e.g., ½- to 3-acre) intact patches ("*aggregates"*) that are not entered during the harvesting operation. Additional retention will occur as individual trees and snags and small clusters of trees. Retention of some of the individual trees will be to provide candidates for snag creation either using fire (broadcast slash burning) or mechanical means.

Riparian buffers that extend into harvest units can often be counted as contributing to some portion of the retention target. Limitations on credit for riparian buffers are necessary, though, because such buffers typically are spatially concentrated in portions of harvest units, rather than well distributed throughout the unit. To achieve the stated goals for retention, retained structures need to be reasonably well distributed through the harvest unit; conversely, creation of large areas lacking in retention should be avoided (see below).

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Foresters in British Columbia mandated a distributional requirement in applying VRH. In the Clayoquot Sound area of Vancouver Island, all portions of a harvest unit had to be within 2 tree heights of an aggregate or the edge of unit. The BC legislature in their legal definition of VRH require that 50% of the harvest area be within 1 tree height of an aggregate or edge (riparian buffers count in meeting this requirement).

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General Technical Report
PSW-GTR-220
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An Ecosystem Management Strategy for Sierran Mixed-Conifer Forests

Malcolm North, Peter Stine, Kevin O'Hara, William Zielinski, and Scott Stephens

The Forest Service of the U.S. Department of Agriculture is dedicated to the principle of multiple use management of the Nation's forest resources for sustained yields of wood, water, forage, wildlife, and recreation. Through forestry research, cooperation with the States and private forest owners, and management of the National Forests and National Grasslands, it strives—as directed by Congress—to provide increasingly greater service to a growing Nation.

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Cover photos, clockwise from top: Lodgepole and white pine forest on the Lee Lake Trail, by Malcolm North; fisher, by Bill Zielinski; Aspen Valley mixed conifer, by Malcolm North; prescribed fire, by Malcolm North; and heavy fuel load, by Eric Knapp.

Abstract

North, Malcolm; Stine, Peter; O'Hara, Kevin; Zielinski, William; Stephens, Scott. 2009. An ecosystem management strategy for Sierran mixed-conifer forests. Gen. Tech. Rep. PSW-GTR-220. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 49 p.

Current Sierra Nevada forest management is often focused on strategically reducing fuels without an explicit strategy for ecological restoration across the landscape matrix. Summarizing recent scientific literature, we suggest managers produce different stand structures and densities across the landscape using topographic variables (i.e., slope shape, aspect, and slope position) as a guide for varying treatments. Local cool or moist areas, where historically fire would have burned less frequently or at lower severity, would have higher density and canopy cover, providing habitat for sensitive species. In contrast upper, southern-aspect slopes would have low densities of large fire-resistant trees. For thinning, marking rules would be based on crown strata or age cohorts and species, rather than uniform diameter limits. Collectively, our management recommendations emphasize the ecological role of fire, changing climate conditions, sensitive wildlife habitat, and the importance of forest structure heterogeneity.

Keywords: Climate change, ecosystem restoration, forest heterogeneity, forest resilience, topographic variability, wildfire.

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Introduction

In recent years, there has been substantial debate over Sierra Nevada forest management. All perspectives on this debate inevitably cite "sound science" as a necessary foundation for any management practice. Over the dozen years since publication of the last science summary, the Sierra Nevada Ecosystem Project (SNEP 1996), many relevant research projects have published findings in dozens of scientific journals, yet these have not been synthesized or presented in a form that directly addresses current land management challenges.

Over the dozen years since publication of the last science summary, the Sierra Nevada Ecosystem Project, project findings have not been synthesized or presented in a form that directly addresses current land management challenges.

Current management usually cites a "healthy forest"[1] as a primary objective. It is difficult, however, to define forest "health," and, as a broad concept, "a healthy forest" provides few specifics to guide management or assess forest practices. Various constituencies have different ideas of forest health (i.e., sustainable timber production, fire resilience, biodiversity, etc.) making forest health unclear as an objective (Kolb et al. 1994). A premise of silviculture is that forest prescriptions can be tailored to fit a wide variety of land management objectives, once those objectives are defined. We attempt to define some of the key management objectives on National Forest System lands in the Sierra Nevada and how they might be approached through particular silvicultural prescriptions.

In this paper, we focus on summarizing forest research completed at different scales and integrating those findings into suggestions for managing forest landscapes. Although many experiments and forest treatments still occur at the stand level, ecological research and recent public input have emphasized the need to address cumulative impacts and coordinate management across the forest landscape. We believe our synthesis has some novel and highly applicable management implications. This paper, however, is not intended to produce new research findings for the academic community; rather it is an effort to provide managers of Sierran forests with a summary of "the best available science." Some of the suggestions in this paper are already used in different Forest Service management practices.

There are several aspects of forest management that this paper does not address, but we would like to particularly note two omissions. The USDA Forest Service is charged with multiple-use management, which can include more objectives (e.g., socioeconomic impacts) than our focus on ecological restoration of Sierran forests. Restoration practices need both public and economic support to be socially and financially viable. Also, we do not specifically address the issues of water yield and quality in this paper, although water is one of the Sierra's most important resources.

[1] See definition in http://www.whitehouse.gov/infocus/healthyforests/Healthy_Forests_v2.pdf.

Although our focus is on forest conditions, the suggested management practices may also make forests more resilient to disturbances including climate change. Management practices that help restore the forest headwaters of Sierran watersheds will benefit water production and quality for downstream users.

Recent Scientific Information

Current Sierra Nevada forest management is often focused on landscape strategies intended to achieve immediate fuel reduction (e.g., strategically placed area treatments [SPLATs] [Finney 2001], defensible fuel profile zones [DFPZs], and defense zones) (SNFPA 2004). Fire scientists have developed effective models for the strategic placement of these fuel treatments across forest landscapes accounting for practical limitations of how much area can actually be treated in the coming decades (Finney 2001, Finney et al. 2007). These models have been particularly valuable for optimizing and prioritizing fuel treatment locations, and comparing likely fire behavior between treated and untreated landscapes (Ager et al. 2007, Bahro et al. 2007, Finney et al. 2007, Stratton 2006). Although these models have assisted managers in the strategic placement of fuel treatments, they don't have the capacity to evaluate ecosystem responses to treatments. Treatments often rely upon various diameter limits for mechanical tree removal and treat only a portion of the landscape, roughly 20 to 30 percent, relegating most of the forest matrix to continued degradation from the effects of fire suppression. With a focus on evaluating fire intensity and spread, these fuel strategies do not explicitly address how forests might be ecologically restored or wildlife habitat enhanced. Without addressing these issues, treatments often face legal challenges resulting in fuel-treated acres falling far behind Forest Service goals (e.g., approximately 120,000 ac/yr in the Sierra Nevada [Stephens and Ruth 2005]).

Fuel strategies do not explicitly address how forests might be ecologically restored or wildlife habitat enhanced. Without addressing these issues, treatments often face legal challenges.

We have learned much in recent years that can contribute to how forests are managed within strategically placed fuel treatments and throughout the landscape matrix. The Forest Service is already using many ideas in this paper. In other instances, litigation, limited funding, and regulations have fostered practices, such as thinning to a diameter limit or limited use of prescribed fire, that no one is happy with. We hope this science summary contributes to revising and removing some of these restraints.

In this paper, we first summarize recent science findings on fuel dynamics that might improve current fuel treatment practices. Even with these changes, however, Sierra Nevada forest management still lacks an explicit strategy for enhancing forest resilience and wildlife habitat, or managing the majority of the forested landscape outside fuel-treated areas. To incorporate these goals into

current management, we then examine recent research on the ecological role of fire, forest resilience under changing climate conditions, and habitat requirements of sensitive wildlife. Research in all of these areas stresses the ecological importance of forest heterogeneity. Knowing the restoration importance of fire, we determined the pattern and stand structures for implementing this heterogeneity based on how fuel and fire dynamics varied topographically. We discuss how these variable forest conditions could be implemented with revised silvicultural practices. Finally we summarize the paper's content in short bullet points, distilling the applied management implications and listing research needed to improve and modify implementation.

Fuel Dynamics and Current Management Practices

Forest fuels are usually assessed in three general categories: surface, ladder, and canopy bulk density (Agee et al. 2000). Fuel treatments often focus on ladder fuels (generally defined to be variably sized understory trees that provide vertical continuity of fuels from the forest floor to the crowns of overstory trees [Keyes and O'Hara 2002, Menning and Stephens 2007]). Some studies and models, however, suggest a crown fire entering a stand is rarely sustained (i.e., sustained only under extreme weather conditions) if understory fuels are too sparse to generate sufficient radiant and convective heat (Agee and Skinner 2005, Stephens and Moghaddas 2005). Surface fuels merit as much attention as ladder fuels when stands are treated. Prescribed fire is generally the most effective tool for reducing surface fuels.

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One approach to developing fuel prescriptions, similar to current Forest Service procedures, is using modeling software to understand how the load of different fuel sizes and weather conditions affect predicted fire intensity. For example, Stephens and Moghaddas (2005) have modeled fire behavior and weather using Fuels Management Analysis (FMA) (Carlton 2004) and Fire Family Plus software (Main et al. 1990), respectively. The FMA uses two modules, Dead and Down Woody Inventory (data supplied by the Brown 1974 fuel inventory) and Crown Mass (data supplied by inventories of trees by species, size, height, and crown ratio), to model a stand's crowning and torching indices (the windspeed needed to produce an active and passive crown fire, respectively), scorch height, and tree mortality. All four outputs can be controlled by changing surface and ladder fuels, giving managers an opportunity to interactively develop target fuel conditions for a desired fire behavior. Fuels can be reduced until the crowning and torching indices are higher than conditions that are likely to occur even under extreme weather events (e.g., Stephens and Moghaddas 2005).

In addition to ladder and surface fuels, managers have been concerned with reducing canopy bulk density in DFPZs and the defense zone of wildland urban interaces (WUI). Overstory trees are commonly removed, and residual trees are evenly spaced to increase crown separation. The efficacy of canopy bulk density reduction in modifying fire behavior is largely a function of weather conditions. Research has suggested there is often limited reduction in crown fire potential through overstory thinning alone, without also treating surface fuels (Agee 2007, Agee and Skinner 2005, Agee et al. 2000, Stephens and Moghaddas 2005). However, some field observations (JoAnn Fites Kaufmann, Forest Service Enterprise Team, Steve Eubanks, Tahoe National Forest) suggest that under severe weather conditions (e.g., sustained high winds) or on steep slopes, crown separation may reduce the risk of crown fire spread. Fire behavior under extreme conditions is still difficult to model, and, furthermore, what constitutes "extreme" (because many wildfires occur under hot, windy conditions) has not been defined (for the Southwest see Crimmins [2006]). In forests adjacent to homes or key strategic points, managers may want to reduce canopy bulk density to reduce potential fire severity under all possible weather scenarios. Outside of those cases, the value of crown separation in preventing crown fire spread may be limited (Agee et al. 2000, Stephens and Moghaddas 2005).

Mixed-conifer forests were highly clustered with groups of trees separated by sparsely treed or open gap conditions.

A concern with the widespread use of canopy bulk density thinning in defensible fuel profile and defense zones is the ecological effects of the regular tree spacing (fig. 1). In the Sierra Nevada, historical data (Bouldin 1999, Lieberg 1902), narratives (Muir 1911), and reconstruction studies (Barbour et al. 2002, Bonnicksen and Stone 1982, Minnich et al. 1995, North et al. 2007, Taylor 2004) indicate mixed-conifer forests were highly clustered with groups of trees separated by sparsely treed or open gap conditions. This clustering can be important for regenerating shade-intolerant pine (Gray et al. 2005, North et al. 2004, York and Battles 2008, York et al. 2003), increasing plant diversity and shrub cover (North et al. 2005b), moderating surface and canopy microclimate conditions within the tree cluster (North et al. 2002, Rambo and North 2009), and providing a variety of microhabitat conditions for birds (Purcell and Stephens 2006) and small mammals (Innes et al. 2007, Meyer et al. 2007a). Studies in Baja's Sierra San Pedro del Martir (SSPM) forests also indicate forest structures (live trees, snags, logs, and regeneration) are highly clustered (Stephens 2004, Stephens and Fry 2005, Stephens and Gill 2005, Stephens et al. 2007a). This forest in Mexico shares many characteristics of mixed-conifer forests found in the Sierra Nevada but has had little fire suppression and has not been harvested. Although these Baja forests have a different weather pattern than California's Sierra Nevada (Evett et al. 2007), they can provide some insight

Eric Knapp

Figure 1—Regular spacing of "leave" trees in a defensible fuel profile zone.

into the structure and ecological dynamics of a mixed-conifer forest with an active fire regime. A recent study of stressed SSPM Jeffrey pine/mixed-conifer forests where a 2003 wildfire was preceded by a 4-year drought, found spatial heterogeneity was a key feature in forest resiliency (Stephens et al. 2008). A clumped tree distribution, where groups are separated by gaps, might also slow crown fire spread (fig. 2), but we do not know of any studies that have examined this idea. Studies in other mixed-conifer forests (e.g., Klamath Mountains and eastern Washington) imply this heterogeneity may be an important characteristic of frequent fire's effect on mixed-conifer forests (Hessburg et al. 2005, 2007; Taylor and Skinner 2004). Fuel treatments that produce uniform leave tree spacing reduce this ecologically important spatial heterogeneity.

Managing surface fuels and reducing the use of regular leave-tree spacing can improve current fuel treatments. These changes, however, have not addressed a fundamental public concern that current forest management lacks explicit strategies for ecological restoration and provision of wildlife habitat.

Ecological Restoration Using Fire

Fire plays a pivotal role in reshaping and maintaining mixed-conifer ecosystems. Fire was once very common in most of the Sierra Nevada and has been a primary force shaping the structure, composition, and function of mixed-conifer forests (Fites-Kaufman et al. 2007, Franklin and Fites-Kaufman 1996, McKelvey et al.

Carl Skinner

Figure 2—An example of the clumped tree distribution and canopy gaps produced by an active fire regime. The photograph is an aerial view of the Beaver Creek Pinery, which has experienced very little fire suppression.

Fire is both a viable fuel-treatment tool and an important jumpstart for many ecosystem processes stalled by accumulating surface fuels and the absence of frequent burning

1996, Stephens et al. 2007b). Management strategies need to recognize that, in many situations, fire is both a viable fuel-treatment tool (Agee and Skinner 2005; Stephens et al. 2009) and an important jumpstart for many ecosystem processes stalled by accumulating surface fuels and the absence of frequent burning (North 2006). The main effect of low-intensity fire is its reduction of natural and activity (i.e., resulting from management activities) fuels, litter, shrub cover, and small trees. These reductions open growing space, provide a flush of soil nutrients, and increase the diversity of plants and invertebrates (Apigian et al. 2006, Knapp et al. 2007, Moghaddas and Stephens 2007, Murphy et al. 2006, Wayman and North 2007). By reducing canopy cover, fire also increases habitat and microclimate heterogeneity at site, stand, and landscape levels (Chen et al. 1999, Collins et al. 2007, Concilio et al. 2006, Falk et al. 2007, Hessburg et al. 2007, Miller and Urban 1999). Fire is an indispensable management tool, capable of doing much of the work to restore ecological processes (Bond and van Wilgen 1996, Covington et al. 1997, North 2006, Stephenson 1999, Sugihara et al. 2006).

By itself, prescribed fire will be difficult to apply in some forests owing to fuel accumulations, changes in stand structure, and operational limitations on its use.

Mechanical treatments can be effective tools to modify stand structure and influence subsequent fire severity and extent (Agee et al. 2000, Agee and Skinner 2005) and are often a required first treatment in forests containing excessive fuel loads. Prescribed fire is generally implemented very carefully, killing only the smaller size class trees (Kobziar et al. 2006). In some cases, it is ineffective for restoring resilience, at least in the first pass (Ritchie and Skinner 2007). For example, prescribed fire may not kill many of the larger ladder-fuel or co-dominant true fir trees that have grown in with fire suppression (Knapp and Keeley 2006, North et al. 2007). In many stands, mechanical thinning followed by prescribed fire may be necessary to achieve forest resilience much faster than with prescribed fire alone (Schwilk et al. 2009, Stephens et al. 2009).

Some forests cannot be prescription burned, at least as an initial treatment, because of air quality regulations, increasing wildland home construction, and limited budgets. Yet restoration of these forests still depends on modifying fuels because it reduces wildfire intensity when a fire does occur (Agee and Skinner 2005) and can produce stand conditions that simulate **some** of fire's ecological effects (Innes et al. 2006, Stephens and Moghaddas 2005, Wayman and North 2007). Mechanical control of fuels allows fire, both wildland fire and prescribed fire, to be more frequently used as a management tool.

Climate Change

Forest restoration has often examined past conditions, such as the pre-European period, as a basis for developing management targets. With climate change, however, is restoring forests to these conditions even an appropriate goal? Returning to a pre-European condition, is unlikely to be feasible, because in addition to climate, livestock grazing and Native American ignitions have changed (Millar and Woolfenden 1999, Millar et al. 2007). Rather than strive for restoration of a fixed presettlement condition, managers could increase tree, stand, and landscape resiliency.

Forest restoration has often examined past conditions, such as the pre-European period, as a basis for developing management targets. With climate change, however, is restoring forests to these conditions even an appropriate goal?

Research suggests global mean minimum temperatures may have already begun to rise (Easterling et al. 1997). One effect of this change for western forests would be earlier spring melt of mountain snowpacks. An analysis of Western U.S. fire season length over the last 50 years suggests that during the last two decades, fires begin earlier in the spring and occur later in the fall possibly owing to this trend in elevated nighttime minimum temperatures (Westerling et al. 2006). An analysis of fire severity and size in California has found an increase in both, along with a regional rise in temperature (Miller et al. 2009). Climate change effects on precipitation have been more difficult to predict with models suggesting regional

differences. For example, some models predict an increase in precipitation for northern California, some predict a decrease, and others suggest little change (Hayhoe et al. 2004, Lenihan et al. 2003). Most models predict the southern Sierra will receive less precipitation, with a higher percentage of it occurring as rain rather than snow (Miller et al. 2003). Climate models suggest there will be more frequent and stronger shifts between El Niño and La Niña events making changes in average precipitation difficult to predict. Perhaps one point of consensus is that most modelers agree the climate will become more extreme, suggesting oscillations between wet and drought conditions will be more common.

Species will not simply shift up in elevation or latitude in response to warming conditions.

The potential effects of these changes on vegetation, fire, and wildlife are largely speculative (Field et al. 1999, Skinner 2007). Studies of past vegetation communities under a range of climates show unique plant assemblages without modern analogs (Millar et al. 2007, Williams and Jackson 2007). This suggests species will not simply shift up in elevation or latitude in response to warming conditions. Some general predictions that grouped species by functional categories have predicted an increase in broad-leaved over needle-leaved species, a general increase in ecosystem productivity (i.e., total biomass), and a decrease in forest and an increase in shrub and grasslands (Lenihan et al. 2003). Changes in forest understory may vary depending on existing vegetation and the synergistic effects of increasing nitrogen enrichment from pollution and increased herbaceous fuels affecting burn intensity and frequency (Hurteau and North 2008). If Sierra precipitation decreases or experiences more frequent, intense La Niña events, forests are likely to become more drought stressed. One study examining several decades of mixed-conifer demography trends (van Mantgem and Stephenson 2007) suggests a recent increase in mortality may be related to increased drought stress from a warming climate. This drought stress would make current, high-density, Sierra forests more susceptible to pest and pathogen mortality, particularly from bark beetles (Ferrell 1996, Fettig et al. 2007, Maloney et al. 2008, Smith et al. 2005).

Managing forests under these conditions will be challenging. In the face of uncertainty, Millar et al. (2007) have suggested managers consider adaptive strategies focused on three responses: resistance (forestall impacts and protect highly valued resources), resilience (improve the capacity of ecosystems to return to desired conditions after disturbance), and response (facilitate transition of ecosystems from current to new conditions). All of these strategies acknowledge the influence of climate change and suggest management may fail if focused on re-creating past stand conditions using strict structural targets.

Although historical forest conditions may not provide numerical guidelines, the past still has lessons for managing Sierran forests. Historical forests can provide a better understanding of the ecological processes that have shaped mixed-conifer forest and the habitat conditions to which wildlife have adapted (Falk 1990, Society for Ecological Restoration 1993). All reconstruction studies, old forest survey data sets, and 19th-century photographs (Gruell 2001, McKelvey and Johnson 1992) suggest that frequently burned forests had very low tree densities. For example, in the early 20th century, Lieberg (1902) estimated that stem density in the northern California forests he surveyed was only 35 percent of its potential because of mortality from frequent fire. Studies reconstructing pre-European conditions all indicate that forests had a greater percentage of pine, a clustered pattern with highly variable canopy cover, and a high percentage of the growing stock in more fire-resistant, large-diameter classes. These past conditions give general guidance but should not be taken as strict numerical targets for density or diameter distribution in silvicultural prescriptions. What these reconstructions do provide is inference about the cumulative process effects of fire, insects, pathogens, wind, and forest dynamics on stand structure and composition, producing forests resilient to most disturbances, including wildfire. A modeling comparison of different stand structures grown over 100 years, including those produced by fuel treatments (Hurteau and North 2009), found a low-density forest dominated by large pines was most resilient to wildfire, sequestered the most carbon, and had the lowest carbon dioxide (CO_2) emissions and thus contributed less to global warming. An analysis of carbon emissions and storage from different fuel treatments, found understory thinning followed by prescribed fire produced the greatest reduction in potential wildfire severity without severely reducing carbon stocks (North et al., in press). As climate changes, managing the process or behavior of fire (i.e., manipulating fuels to influence burn intensity) may produce more resistant and resilient forests than managing for a desired number and size of trees.

As climate changes, managing the process or behavior of fire may produce more resistant and resilient forests than managing for a desired number and size of trees.

An important benefit of forest management focused on affecting fire behavior is that in areas of wildland fire and prescribed burning, forest structure and composition are allowed to reestablish to modern dynamic equilibrium by adapting to fire that occurs under current climate and ignition conditions (Falk 2006, Stephenson 1999). A recent analysis of fire severity data by 10-yr periods in Yosemite's mixed-conifer forest (Collins et al. 2009) revealed a fair degree of stability in the proportion of area burned among fire severity classes (i.e., unchanged, low, moderate, high). This suggests that free-burning fires, over time, can regulate fire-induced effects across the landscape.

Sensitive Wildlife

A strategy for mixed-conifer ecological restoration will conserve wildlife and minimize habitat impacts for both the broader animal community as well as the specific needs for a subset of species of concern. For over 15 years, Sierran forest management devoted significant effort to meeting the needs of old-forest-associated species, particularly the California spotted owl (*Strix occidentalis occidentalis*) (Verner et al. 1992) and the Pacific fisher (*Martes pennanti*). Sound wildlife management strategies balance species needs (both sensitive and common) at a variety of spatial (microsite to foraging landscape) and temporal (immediate to long-term population viability) scales (Noss et al. 1997).

Managing for owl and fisher viability needs to account for a few shared characteristics of these top tropic species, including territoriality, large home range size, strong associations with late-seral forest structures, and long-distance travel for foraging contribute to improved owl and fisher viability. Both species are strongly associated with Sierran forest stands characterized by large trees and dense canopy cover (Verner et al. 1992, Zielinski et al. 2004b). These features are consistently selected by spotted owls for nesting (North et al. 2000), and by fishers for denning and resting sites in the Sierra Nevada (Mazzoni 2002; Zielinski et al. 2004a, 2004b) and elsewhere. Fishers use cavities in living and dead conifers and hardwoods (particularly California black oak [*Quercus kellogii* Newb.]) as daily refuges, and tend to select the largest individual trees in dense canopy stands (fig. 3). Individual trees are rarely reused as rest structures, at least consistently from night to night (Zielinski 2004b), so many different large trees are required. This behavior makes provision of resting habitat critical to fisher conservation (Zielinski 2004b). Spotted owls also use many different large trees within their home range for roosting (Verner et al. 1992). Large decadent trees are less common in the Sierra Nevada than they once were (Bouldin 1999), and providing for this structure requires protecting existing large trees, managing for their future development, and reducing major threats (i.e., high-severity fire and pest mortality).

Prey species are associated with a variety of forest conditions suggesting that habitat heterogeneity at different spatial scales across the landscape may be desirable for sustaining adequate food supplies.

Foraging habitat, unlike resting habitat, is much easier to provide for spotted owls and fishers. The fisher's diet is very diverse and includes a variety of small mammals, birds, reptiles, fruits, and insects (Zielinski et al. 1999). Owls have a somewhat more specialized diet. In most locations they tend to prey on woodrats (*Neotoma* spp.), northern flying squirrels (*Glaucomys sabrinus*), and deer mice (*Peromyscus maniculatus*), at least during nesting season (Forsman et al. 2004, Williams et al. 1992). Although our current knowledge of fisher and owl foraging habitats is fairly limited, we do know that their array of prey species are associated with a variety of forest conditions suggesting that habitat heterogeneity at different

Rebecca Green

Figure 3—Pacific fisher resting on a limb of a large black oak. Although this picture is from the Klamath Mountains, it is typical of the stand conditions associated with fisher resting locations.

spatial scales across the landscape may be desirable for sustaining adequate food supplies (Carey 2003, Coppeto et al. 2006, Innes et al. 2007, Meyer et al. 2005). A cautious strategy would be emulating patterns created by natural disturbance to provide a heterogeneous mix of forest habitat across a managed landscape (Lindenmayer and Franklin 2002, North and Keeton 2008).

Management of Large Structures

Much of the public apprehension over forest management practices stems from possible impacts to old-forest-associated species such as the Pacific fisher, California spotted owl, and northern goshawk (*Accipiter gentilis*). All three of these sensitive species depend on a forest structure usually dominated by large trees, snags, and downed logs, which provide suitable substrate for nesting, denning, and resting sites. Retaining these large snags and logs may increase fire hazard in these favorable habitat microsites, particularly in warming climate conditions. In some stands that have been depleted of larger trees, the best available structures may be intermediate-sized trees, generally defined as the 20- to 30-inch size class for conifers. In these stands, retaining conifers of this size is important not only for immediate wildlife needs, but also because they will become the next generation of large trees, (and eventually) snags and logs. Fisher rest structures include live trees

(e.g., cavities, broken tops), snags (e.g., cavities, broken tops, stumps), platforms (nests, mistletoe growths, witches' brooms), logs, and ground cavities (Zielinski et al. 2004b). We do not yet have a good understanding of how best to distribute potential rest sites or how many are needed.

Other Key Structures and Habitats

Other forest features that may be important to sensitive species as well as the broader wildlife community include hardwoods, shrubs, "defect" trees, and riparian corridors. Hardwoods, particularly black oak, are increasingly regarded as an important species for providing food and cavities. Many small and large mammals and birds use acorns as a food source (McShea 2000), particularly in large masting years (Airola and Barrett 1985, Morrison et al. 1987, Tevis 1952). Oaks often have broken tops and large cavities from branch breakage, and are frequently used for resting and nesting sites by small mammals (Innes et al. 2007), forest carnivores (Zielinski et al. 2004b), and raptors (North et al. 2000, Richter 2005). In many areas, hardwoods are in decline because they have become overtopped and shaded by conifers. The larger oaks likely germinated and had much of their early growth in more open forest than exists today (Zald et al. 2008). Provisions are needed to create open areas within stands to facilitate hardwood recruitment. Thinning around large oaks that are prolific seed producers creates open conditions that favor oak regeneration. However, thinning around large, cavitary oaks that are currently shaded is a difficult decision. It is important to balance thinning to prolong the life of the oak against the possibility that reducing the canopy around the oak will decrease the overall habitat value of the rest structure. Managers might consider thinning around some, but not all, cavitary oaks if several are present within a stand.

In fire-suppressed forests, shrubs are often shaded out (Nagel and Taylor 2005, North et al. 2005b) reducing their size, abundance, and fruit and seed production in low-light forest understories. Anecdotal narratives (Lieberg 1902, Muir 1911), a forest reconstruction (Taylor 2004), and a few early plot maps[2] suggest shrub cover in active-fire conditions might have been much higher than in current forests, mostly owing to large shrub patches that occupied some of the gaps between tree clusters (fig. 4). In SSPM's active-fire Jeffrey pine/mixed-conifer forests, Stephens et al. (2008) found shrub cover was highly spatially variable, and often occurred in high-density patches. Some birds (Robinson and Alexander 2002) and small mammals, including spotted owl prey such as the woodrat (Coppeto et al. 2006,

[2] Eric Knapp. 2008. Personal communication. Research ecologist, USDA Forest Service, Silviculture Laboratory, 3644 Avtech Parkway, Redding, CA. 96002.

Robert Dunning

Figure 4—Photograph taken in 1929 of mixed-conifer forest before thinning and approximately 40 years after the last fire, near Pinecrest, California. Note the extensive cover of understory shrubs, particularly under the canopy gap in the foreground.

Innes et al. 2007), are associated with these habitat patches. We also know that species of *Ceanothus* are an important source of available nitrogen (Erickson et al. 2005, Johnson et al. 2005, Oakley et al. 2006) that persists even after the shrubs have been removed by fire (Oakley et al. 2003). In forests where shrubs are currently rare, it is important for managers to consider protecting what shrubs remain and increasing understory light conditions for shrub establishment and patch expansion. Patch size and configuration of such habitat should vary (see discussion on habitat heterogeneity in the next section).

Forest management practices have sometimes removed decadent, broken-topped, or malformed trees that are actually some of the most important features of habitat for many wildlife species (Mazurek and Zielinski 2004, North et al. 2000, Thomas et al. 1976, Zielinski et al. 2004b). These "defect" trees are some of the rarest structures in current forest conditions, often rarer than large trees. Successful management strategies might consider incorporating a means of preserving what remains and adding more of these features across the landscape. The Green Diamond Resource Company developed a guide for identifying and ranking the potential habitat quality of these forest structures in the Klamath Mountains. Developing a similar guide for Sierra forests would be extremely useful.

Connecting habitat within a landscape using corridors has been extensively studied, but results often indicate that suitable forest conditions within the corridor and the optimal distribution of corridors differs by species (Hess and Fischer 2001). In the Sierra Nevada, with its extended summer drought, riparian forests may be particularly important habitat and movement corridors for many species. Owing to greater soil development and moisture retention, these corridors usually provide more vegetative cover, have greater plant and fungal abundance and diversity (Meyer and North 2005), and a moderated microclimate (Rambo and North 2008). Many small mammals are found in greater abundance in riparian areas (Graber 1996, Kattelmann and Embury 1996, Meyer et al. 2007a), and some of these species are selected prey of old-forest-associated species. Initial observations of fisher[3] (Seglund 1995, Zielinski et al. 2004a) suggest that riparian areas may be preferred movement corridors. Riparian corridor width would be better defined by overstory and understory vegetation than the set distances of 150 and 300 feet that are specified in the Sierra Nevada Forest Plan Amendment (SNFPA 2004).

[3] Katherine Purcell. 2008. Personal communication. Research wildlife biologist, USDA Forest Service, Forestry Sciences Laboratory, 2081 E Sierra Ave., Fresno, CA 93710.

Riparian forests are less moisture limited than upland areas, are highly productive, and now have some of the heaviest ladder and surface fuel loads of any Sierran forest communities (Bisson et al. 2003, Stephens et al. 2004). Recent Western U.S. research suggests that although reduced, fire is still a significant influence on riparian forest structure, composition, and function in forests with historically frequent, low-intensity fire regimes (Dwire and Kauffman 2003, Everett et al. 2003, Olson 2000, Pettit and Naiman 2007, Skinner 2003). Although fire in Sierran riparian areas was probably less frequent than in surrounding uplands, we do not yet know what its historical frequency, intensity, and extent was in stream corridors. When inevitable wildfires burn these corridors, they are likely to be high-severity crown fires that can denude riparian areas of vegetation (Benda et al. 2003). Any management activity in riparian areas, including no action, has risks. We suggest that riparian corridors be treated with prescribed fire in spring or late fall (after rains) to help reduce surface fuels (Beche et al. 2005). In moist conditions, some observations[4] suggest that low-intensity prescribed fire can reduce fuels while maintaining high canopy cover and large logs if fuels have high moisture content.

We suggest that riparian corridors be treated with prescribed fire in spring or late fall (after rains) to help reduce surface fuels.

Importance of Heterogeneity

A management strategy that includes methods for increasing forest heterogeneity at multiple scales will improve habitat quality and landscape connectivity. Creating vertical and horizontal heterogeneity in forests with frequent fire, however, has been a challenge. Multilayered canopies, often associated with Pacific Northwest old-growth forests (Spies and Franklin 1988), are not the best model for Sierran mixed-conifer forests because when adjacent trees are multilayered, the continuity of vertical fuels can provide a ladder for surface fire into the overstory canopy. Horizontal heterogeneity, however, used to be relatively common in Sierran mixed-conifer forests (Franklin and van Pelt 2004, Knight 1997, Stephens and Gill 2005). All of the Sierran reconstruction studies (Barbour et al. 2002, Bonnicksen and Stone 1982, Minnich et al. 1995, North et al. 2007, Taylor 2004) suggest mixed-conifer forests, under an active fire regime, had a naturally clumped distribution containing a variety of size and age classes.

[4] Dave McCandliss, 2008. Personal communication. Fire management officer, USDA Forest Service, Sierra National Forest, 1600 Tollhouse Road, Clovis, CA 93611.

Within-Stand Variability

At the stand level, vertical heterogeneity can still be provided by separating groups of trees by their canopy strata (fig. 5). For example, a group of intermediate-size trees that could serve as ladder fuels might be thinned or removed if they are growing under large overstory trees. The same size trees in a discrete group, however, might be lightly thinned to accelerate residual tree growth or left alone if the group does not present a ladder fuel hazard for large, overstory trees. These decisions could be made using the revised silvicultural markings proposed (see "Allocation of Growing Space" section), where growing space is allocated by leaf area index among trees in different height strata. This strategy would produce within-stand vertical heterogeneity, albeit in discrete tree clusters, which will contribute to horizontal heterogeneity.

Drawing courtesy of Robert van Pelt.

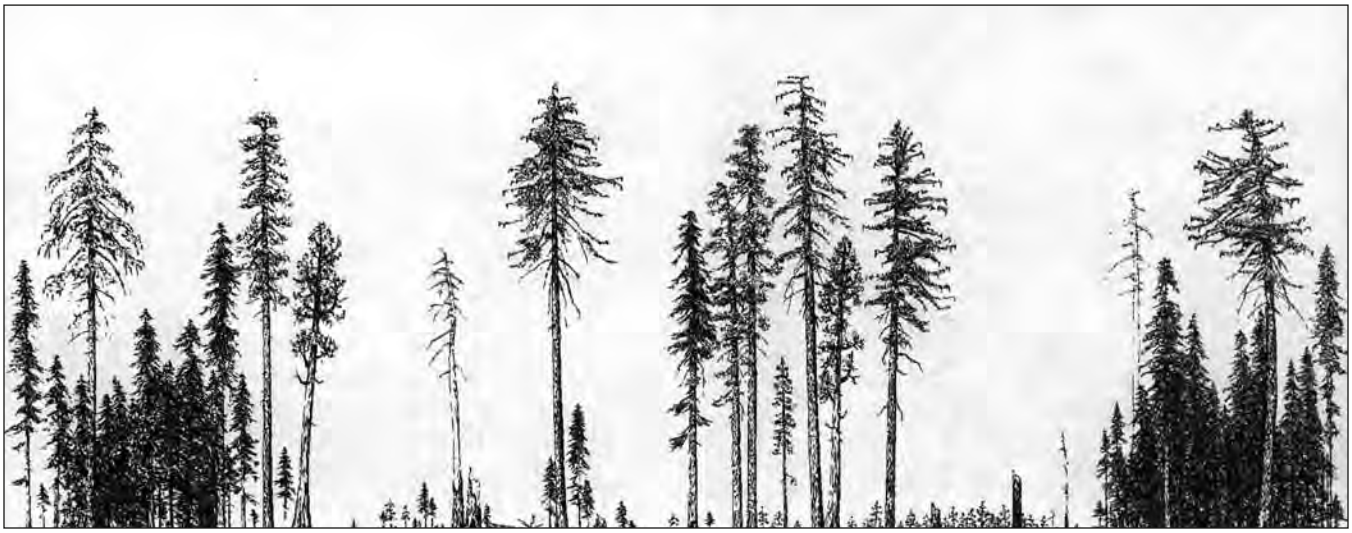
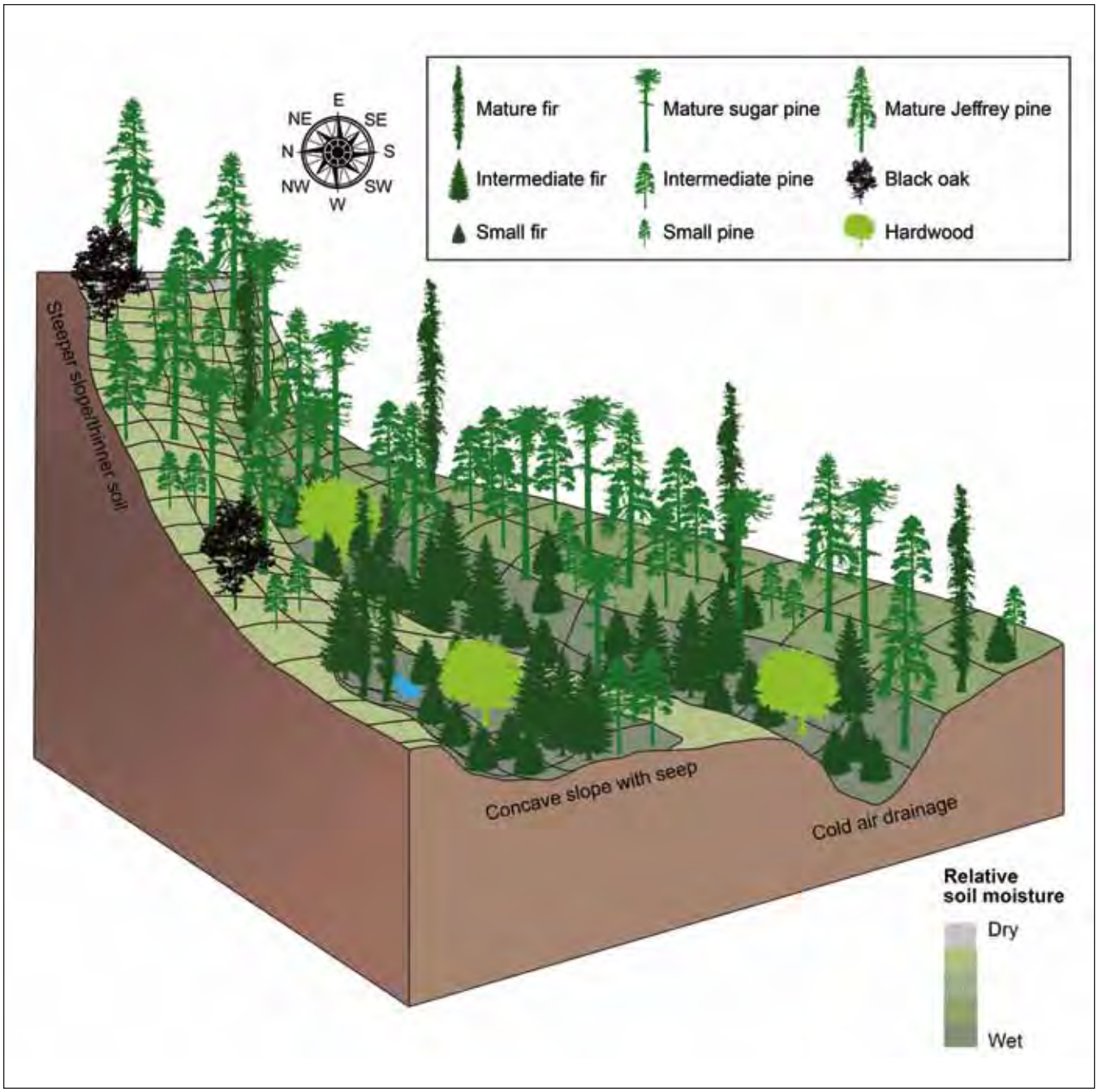


Figure 5—Transect of a mixed-conifer forest in Yosemite National Park's Aspen Valley, which has experienced three understory burns within the last 50 years. Note that the stand has vertical heterogeneity but that trees in different canopy strata tend to be spatially separated.

To increase horizontal heterogeneity, we suggest using microtopography as a template (Sherlock 2007) (fig. 6). Wetter areas, such as seeps, concave pockets, and cold air drainages, may have burned less frequently or at lower intensity (fig. 7). Limiting thinning to ladder fuels in these areas is suggested because with their potentially higher productivity and cooler microclimate, they can support greater stem densities, higher canopy cover, and reduced fire effects. A concern with current uniform fuel reduction is that these microsite habitats associated with sensitive species would be eliminated. Surface fuel loads at these microsites should still be reduced to lower their vulnerability to high-intensity fire.



Mature tree drawiings courtesy of Robert van Pelt

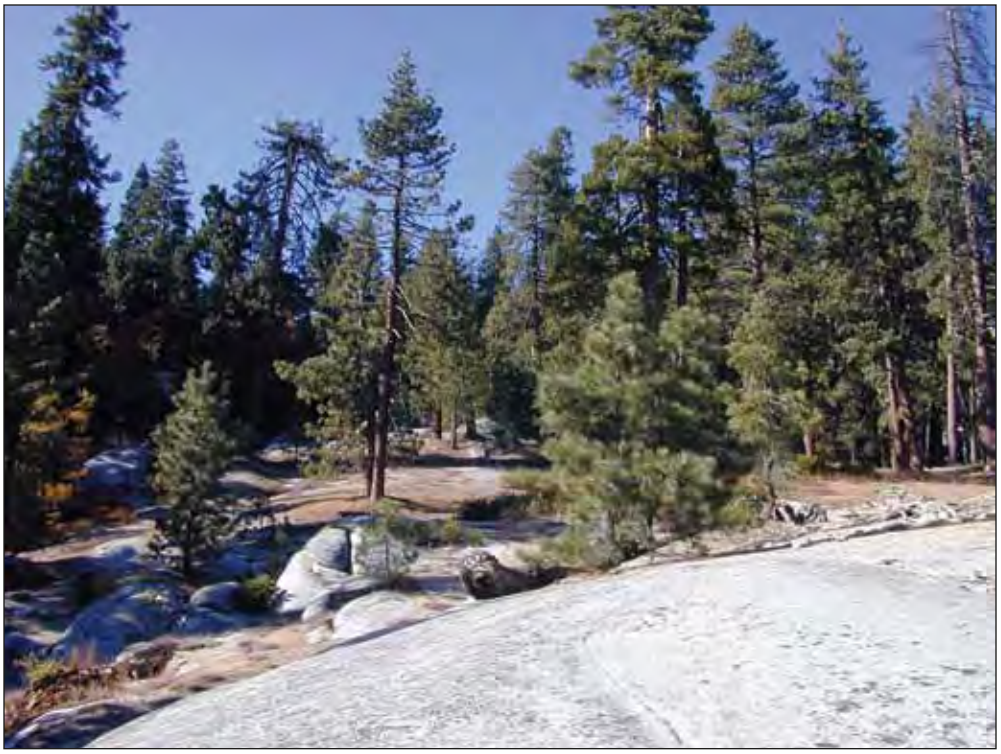
Figure 6—Stand-level schematic of how forest structure and composition would vary by small-scale topography after treatment. Cold air drainages and concave areas would have high stem densities, more fir and hardwoods. With increasing slope, stem density decreases and species composition becomes dominated by pines and black oak.

Malcolm North

Figure 7—Mixed-conifer stand structure at Aspen Valley, Yosemite National Park, produced by frequent, low-intensity fire. Note the higher stem density and hardwoods in the seep drainage in the background.

Within a stand, varying stem density according to potential fire intensity effects on stand structure would create horizontal heterogeneity.

In contrast, upslope areas, where soils may be shallower and drier and where fire can burn with greater intensity, historically had lower stem densities and canopy cover (Agee and Skinner 2005) (fig. 8). On these sites, thinning might reduce the density of small or, where appropriate, intermediate trees and ladder and surface fuels toward a more open condition. In some circumstances this thinning may reduce water stress, accelerating the development of large residual trees (Kolb et al. 2007, Latham and Tappenier 2002, McDowell et al. 2003, Ritchie et al. 2008). Within a stand, varying stem density according to potential fire intensity effects on stand structure would create horizontal heterogeneity.



Malcolm North

Figure 8—Upslope stand conditions where thinner soils and rock outcrops are often associated with drier conditions, and lower density forests, which burned frequently.

Landscape-Level Forest Heterogeneity

Landscapes with an active fire regime are highly heterogeneous. In Baja's active-fire Jeffrey pine/mixed-conifer forests, Stephens et al. (2007a) found that "average" stand characteristics such as snag density, large woody debris, tree density, basal area, and surface fuel loads were rare (approximately 15 to 20 percent of the sampled stands) and varied by an order of magnitude among localized (0.25-ac) plots. Studies in the Sierra Nevada (Fites-Kaufman 1997, Urban et al. 2000) and Klamath Mountains (Beaty and Taylor 2001, Taylor and Skinner 2004) found that mixed-conifer structure and composition varied by fire patterns that were controlled by landscape physiographic features (fig. 9). Fire intensity, and consequently a more open forest condition, increased with higher slope positions and more southwesterly aspects. In eastern Washington mixed-conifer forests, Hessburg et al. (2005, 2007) also found a heterogeneous historical forest landscape shaped by topographic influences on fire behavior. Cumulatively these studies suggest that forest landscapes varied depending on what structural conditions would be produced by topography's influence on fire frequency and intensity.

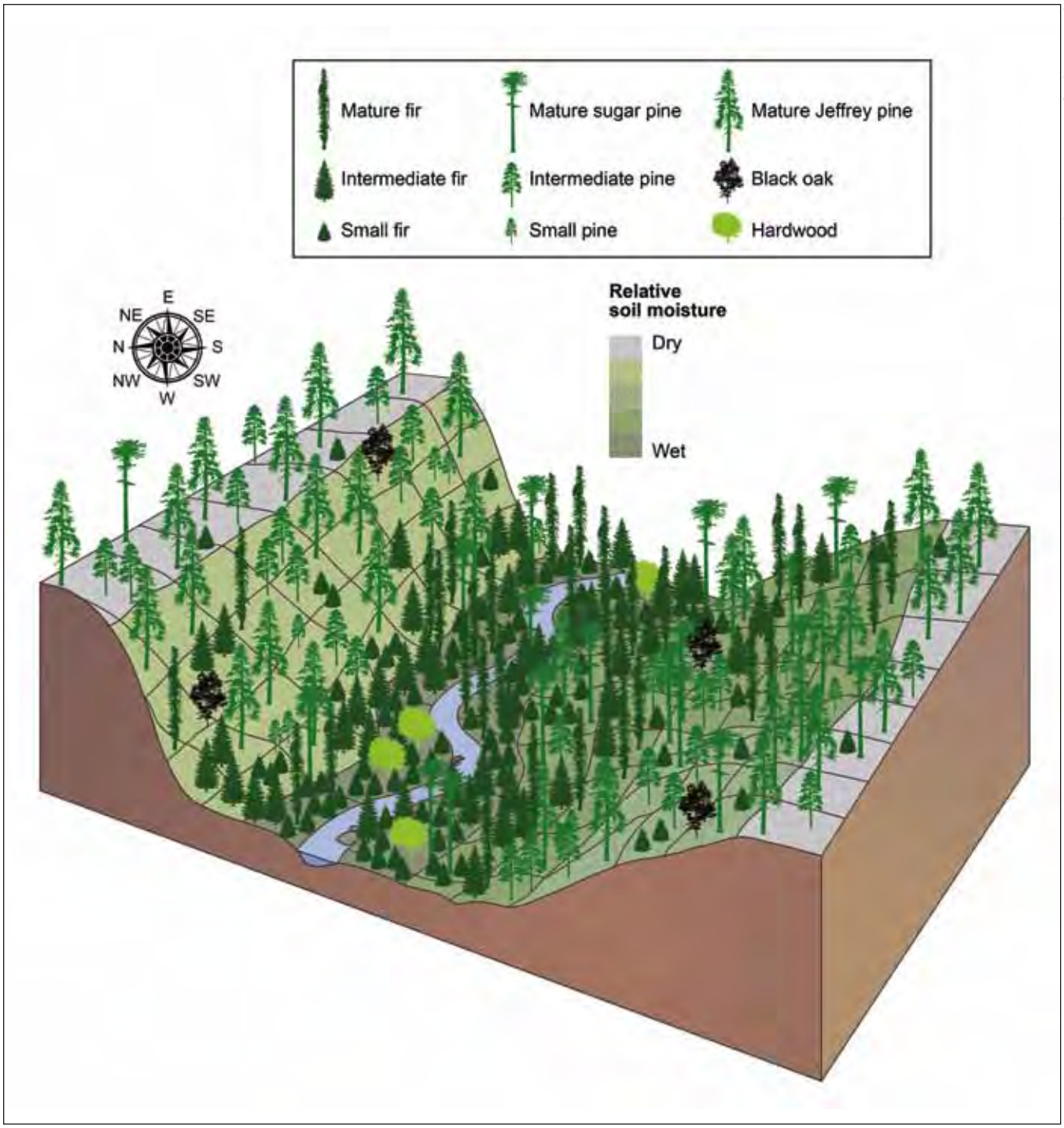
These studies suggest that forest landscapes varied depending on what structural conditions would be produced by topography's influence on fire frequency and intensity.

Eric Knapp

Figure 9—Landscape variation in burn intensity on the Moonlight Fire (2007), Plumas National Forest.

We suggest creating landscape heterogeneity in the Sierra Nevada by mimicking the forest conditions that would be created by the fire behavior and return interval associated with differences in slope position, aspect, and slope steepness.

We suggest creating landscape heterogeneity in the Sierra Nevada by mimicking the forest conditions that would be created by the fire behavior and return interval associated with differences in slope position, aspect, and slope steepness (Sherlock 2007). In general, stem density and canopy cover would be highest in drainages and riparian areas, and then decrease over the midslope and become lowest near and on ridgetops (fig. 10). Stem density and canopy cover in all three areas would be higher on northeast aspects compared to southwest. Stand density would also vary with slope becoming more open as slopes steepen.



Mature tree drawiing courtesy of Robert van Pelt

Figure 10—Landscape schematic of variable forest conditions produced by management treatments that differ by topographic factors such as slope, aspect, and slope position. Ridgetops have the lowest stem density and highest percentage of pine in contrast to riparian areas. Midslope forest density and composition varies with aspect: density and fir composition increase on more northern aspects and flatter slope angles.

Revising Silvicultural Prescriptions

A new silviculture for Sierran mixed-conifer forest that balances ecological restoration and wildlife habitat with fuel reduction can meet multiple forest objectives. By necessity, recent Sierran silviculture has first been focused on reducing fire severity through fuel reduction. For many reasons, including maintaining or restoring resilient forests, public safety, and property loss, fuel reduction remains a priority. We suggest that, with some modification, wildlife and ecological objectives can also be met.

Importance of Tree Species

Diameter-limit prescriptions applied equally to all species will not remedy the significant deficit of hardwoods and pines in current forests (Franklin and Fites-Kaufmann 1996, SNFPA 2004). Prescriptions that differ by species can retain hardwoods, which are important for wildlife, and favor pines that can increase the forest's fire resilience. Given their current scarcity in many locations, there are few instances that warrant cutting either hardwoods or pines in mixed-conifer forests.

Retention of "Defect" Trees

Given the wildlife habitat value of large trees with multiple tops, rot, cavities, etc., managers may want to retain them whenever possible. These growth forms often result from disease or injury (e.g., from lightning, wind breakage, and being struck by adjacent falling tree), and are important structural features for many wildlife species. Disease incidence does not necessarily indicate that a tree is genetically more susceptible and therefore should be "culled" (Tainter and Baker 1996). Modern Sierran forests have a significant shortage of these "decadent" but essential habitat structures (McKelvey and Johnson 1992).

Revising the Desired Diameter Distribution

The proposed silvicultural approach is a multiaged-stand strategy driven by the need for wildlife habitat, fire-resistant stand structures, and restoration of stand and landscape patterns similar to active-fire conditions in mixed-conifer forests. Although we use the term multiage, we are most interested in size and structure, and their associated ecological attributes. Multiaged stands are a flexible means of including variable stand structures with two or more age classes and integrating existing stand structures into silvicultural prescriptions. More traditional forms of uneven-age silviculture were heavily reliant on achieving a reverse-J diameter distribution that reduced large-tree retention (O'Hara 1998). Past silviculture has often changed the slope of this line (i.e., adjusting the q-factor [Smith et al. 1996])

in response to different forest types and stand conditions, but has not fundamentally changed the shape of the curve or its allocation of growing space. The reverse-J diameter distribution prescribes a stand structure with a surplus of small trees and limited space for large trees. Such a distribution is inconsistent with historical Sierran mixed-conifer forests where fire reduced small-tree abundance while retaining fire-resistant, large-diameter trees (North et al. 2005a, 2007) (fig. 11). Research suggests that fire-prone forests rarely had reverse-J diameter distributions (Bouldin 1999; O'Hara 1996, 1998; Parker and Peet 1984).

Research suggests that fire-prone forests rarely had reverse-J diameter distributions.

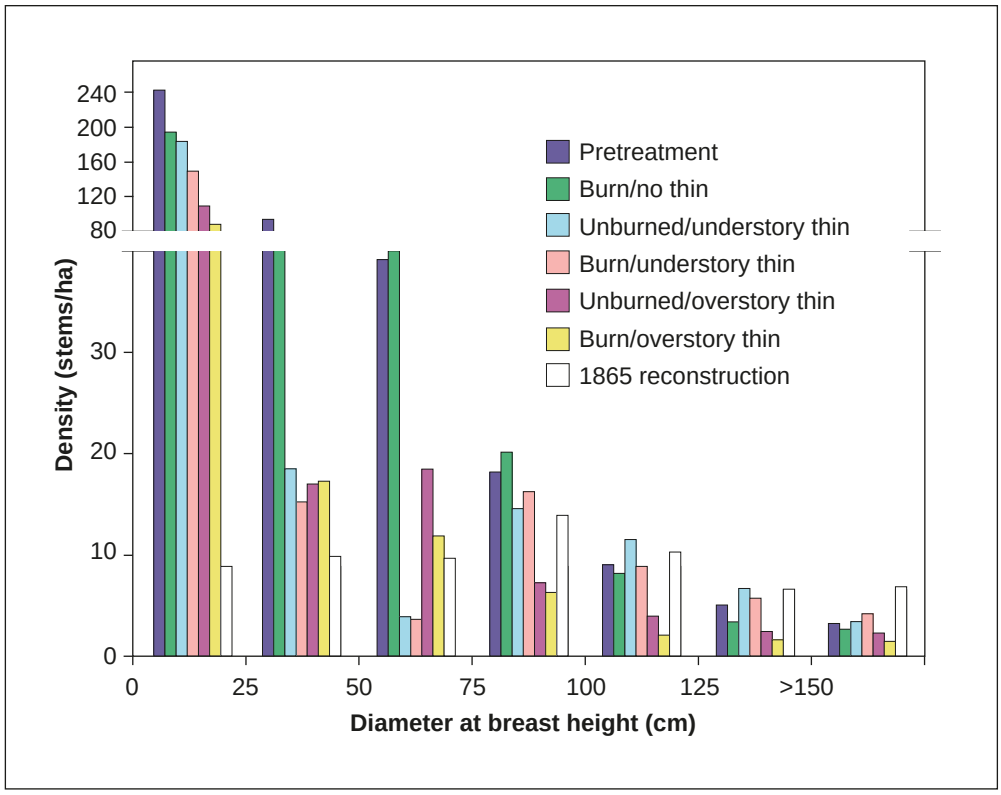


Figure 11—Density of live trees (stems per hectare) in seven size classes for seven conditions in the Teakettle Experiment. The five fuel-reduction treatments (prescribed burn only, understory thin, understory thin and burn, overstory thin, and overstory thin and burn) retain the same reverse-J-shaped diameter distribution as the pretreatment (fire-suppressed old growth) and do not approximate the reconstruction of the diameter distribution in 1865 active-fire conditions. Reconstruction methods probably underestimate the number of small stems in 1865 active-fire conditions, but even a three- to fourfold increase would not produce a reverse-J distribution (reprinted from North et al. 2007).

Groups of Large Trees

Clusters of intermediate to large trees (i.e., >20 inches diameter at breast height [d.b.h.]) are sometimes marked for thinning with the belief that they are overstocked and thinning would reduce moisture stress. Some evidence, however, suggests these groups of large trees may not be moisture stressed by within-group competition

because they have deep roots that can access more reliable water sources including fissures in granitic bedrock (Arkley 1981, Hubbert et al. 2001, Hurteau et al. 2007, Plamboeck et al. 2008). Reconstructions of Sierran forests with active fire regimes (Barbour et al. 2002, Bonnicksen and Stone 1982, Minnich et al. 1995, North et al. 2007, Taylor 2004) have consistently found large trees in groups. These groups, however, can be at risk if intermediate and small trees grow within the large tree groups. Thinning these small and intermediate trees will reduce fire laddering.

Managing the Intermediate Size Class

Many studies have documented the importance of large tree structures in forests for many ecological processes and their value for wildlife habitat (see summaries in Kohm and Franklin 1997, Lindenmayer and Franklin 2002). However, "large" varies with forest type and site productivity, and there is no set size at which a tree takes on these attributes. We only address this question of 20- to 30-in d.b.h. trees because it is so pivotal in the current management strategies for Sierran forests and is driving much of the discussion around fuel treatment thinnings.

If trees larger than 10 to 16 inches in d.b.h. are thinned, it is important to provide reasons other than for ladder-fuel treatment.

What is achieved by thinning intermediate sized (20- to 30-in d.b.h.) trees? Some research suggests that for managing fuels, most of the reduction in fire severity is achieved by reducing surface fuels and thinning smaller ladder-fuel trees (see summaries in Agee et al. 2000, Agee and Skinner 2005, Stephens et al. 2009). What is considered a ladder fuel differs from stand to stand, but typically these are trees in the 10- to 16-in d.b.h. classes. If trees larger than this are thinned, it is important to provide reasons other than for ladder-fuel treatment. These may include additional fuel reduction such as thinning canopy bulk density in strategic locations. Or it could be other ecological objectives such as restoration of an active-fire stand structure, managing for open habitat that includes shrubs, or accelerating the development of large leave trees. Although large trees are often old, studies have found diameter growth increases significantly when high densities of adjacent small stems are removed (Das et al. 2008, Latham and Tappeiner 2002, McDowell et al. 2003, Ritchie et al. 2008, Skov et al. 2004). There may be socioeconomic purposes for harvesting intermediate-sized trees such as generating revenue to help pay for fuel treatment or providing merchantable wood for local sawmills (Hartsough et al. 2008). Clear statement of the objectives for thinning intermediate-sized trees will help clarify management intentions.

Under what conditions could intermediate trees be thinned?

Under what conditions could intermediate trees be thinned? We suggest the following criteria but stress that these criteria are based on working hypotheses. The first selection criteria is species. Thinned intermediate-size trees should only be fire-sensitive, shade-tolerant species such as white fir (*Abies concolor* (Gord.

& Glend.) Lindl. ex Hildbr.), Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) and incense-cedar (*Calocedrus decurrens* (Torr.) Florin). In mixed-conifer forest, attempt to keep intermediate-size pines and hardwoods because of their relative scarcity and importance to wildlife and fire resilience. A second criterion would be tree growth form. Some intermediate-size trees can still function as ladder fuel, particularly those that were initially grown in more open conditions (fig. 12). These trees can have live and dead limbs that extend down close to the forest floor providing a continuous fuel ladder. A third condition is middle to upper slope topographic position. In these slope positions, some thinning of intermediate-size trees may help accelerate the development of large "leave" trees. We suggest, however, that these criteria not be applied to riparian areas, moist microsites often associated with deeper soils, concave topography, or drainage bottoms because these areas may have supported higher tree densities and probably greater numbers of intermediate-size trees (Meyer et al. 2007b).

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Figure 12—Fire suppressed stand in the Teakettle Experimental Forest with white fir (*Abies concolor*) 20- to 30-in d.b.h. with ladder fuel potential.

Allocation of Growing Space

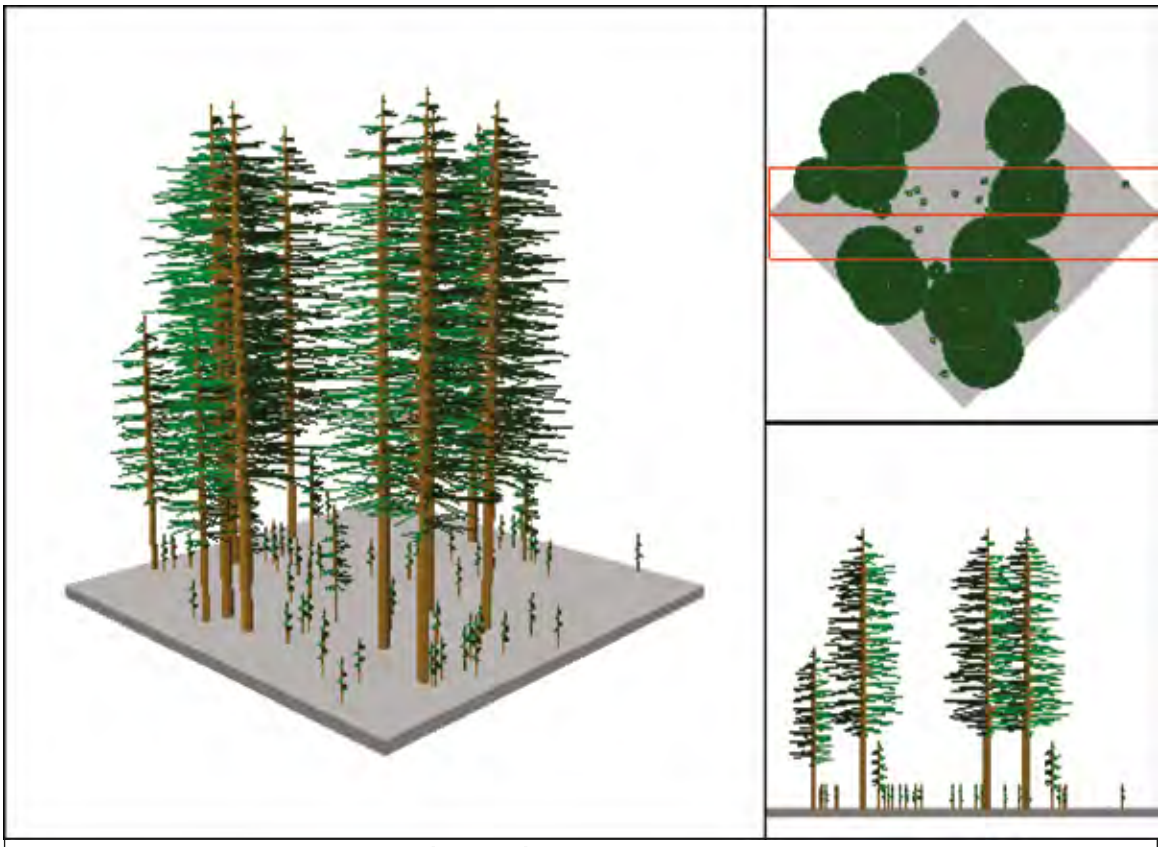
We propose a form of multiaged silviculture for Sierra mixed-conifer forest that is flexible to meet diverse forest objectives, that would retain existing large trees and promote recruitment of more large structures, and that provides for sustainability. The silvicultural system is based on leaf area representing the occupied growing space of trees and stands. By segmenting stand-level leaf area index among canopy strata, we can develop tools to allocate growing space and provide flexibility for creating heterogeneous stand structures and meet ecological objectives (fig. 13) (O'Hara 1996, O'Hara and Valappil 1999). For example, leaf area could be allocated primarily to larger trees in one stand where these large trees are present and important structural components. In other stands, large trees may be absent and leaf area is allocated to developing cohorts to expedite development of large structural features. Trees are harvested and timber is an output, but the silvicultural system's focus is on retained stand structures, not what is removed for harvest. On the ground, this system provides for a diverse stand structure with both vertical (in discrete groups) and horizontal heterogeneity. It is prescribed one stand at a time and creates landscape-level heterogeneity by varying the stocking regime. Treatments are intended to create a mixture of structures sustained throughout the period between active management entries.

The proposed silvicultural system recognizes canopy strata as the primary unit for allocation of growing space. Within these strata, space is allocated to species or species groups. A resulting stocking matrix might consist of three canopy strata and three species groups (e.g., pines, white fir and incense-cedar, and others) providing for a stocking matrix with nine cells. This approach generally simplifies the marking of trees and also can modify species composition (O'Hara et al. 2003). This silvicultural revision will, however, require a new research project to adapt the MultiAge Stocking Assessment Model (MASAM) to Sierra Nevada mixed-conifer forests.

The risks of carefully considered active forest management are lower than the risks of inaction and continued fire suppression in the Sierras' fire-prone forest types.

Conclusion

A central premise of this paper is that the risks of carefully considered active forest management are lower than the risks of inaction and continued fire suppression in the Sierras' fire-prone forest types. We recognize the need to address specific management priorities (e.g., sensitive species) while developing practical and ecologically sound silvicultural guidelines. Many of the ideas contained within this ecosystem management strategy are not new, but their implementation will require some innovations, and they may provide a greater range of management options than do current practices. Our scientific understanding of mixed-conifer



Ponderosa pine MASAM — OREGON					
	User-Specified Variables				
TOTAL Leaf area index (LAI)	6				
	Cohort 1	Cohort 2	Cohort 3	Cohort 4	TOTAL
Number of trees/cohort/acre	25	40	55		120
Percent of LAI/cohort	50	35	15		100
	Diagnostic Information				
	Cohort 1	Cohort 2	Cohort 3	Cohort 4	TOTAL
Leaf area index/cohort ECC	3.0	2.1	0.9	0	6.0
Leaf area index/cohort BCC	1.3	0.7	0	0	2.0
Leaf area/tree (ft^2) ECC	5,227.2	2,286.9	712.8	0	—
BA/cohort (ft^2/ac) ECC	72.5	47.6	19.8	0	139.8
BA/cohort (ft^2/ac) BCC	29.7	14.4	0	0	44.1
Average volume increment/tree (ft^3/yr) ECC	1.02	0.64	0.18	0	—
Average volume increment/CC (ft^3/ac/yr)	20.7	16.3	5.0	0	42.0
Quadratic mean DBH/cohort (in) ECC	23.1	14.8	8.1	0	—
Tree vigor (in^3/ft^2/yr)	0.408	0.460	0.440	0	—
Stand density index ECC	95.2	74.6	39.4	0	209.2
Stand density index BCC	46.6	28.7	0	0	75.3

Figure 13—MultiAged Stocking Assessment model of a three-strata (or three-cohort) Oregon ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.) stand. Growing space can be allocated in a variety of patterns providing flexibility in stand structure design (from O'Hara et al. 2003). (— = not calculated.)

ecosystems is rudimentary, and, therefore, it is important to continue learning from these strategies as they are applied. We have tried to identify information that is supported by many studies, that is suggested by fewer but often recent studies, and that we can only infer from lines of evidence or observation but do not yet know with any degree of certainty. In the "Research Needs" section below, we identify some of the topics raised in this paper that need further investigation, although management will also be improved by trying some of the proposed strategies and learning what works and what fails.

In bringing together authors representing different key disciplines affecting Sierran forests, we did not know whether they would provide contrasting or complementary management concepts; however, each discipline's research findings coalesced around the importance of variable forest structure and fuels conditions.

This project began at the invitation of Forest Service Pacific Southwest Region managers who asked if we could develop a summary of current research to inform mixed-conifer management. In bringing together authors representing different key disciplines affecting Sierran forests, we did not know whether recent fire science, forest ecology, and wildlife biology research would provide contrasting or complementary management concepts or whether the concepts could be translated into silviculture practices. It was soon clear that each discipline's research findings coalesced around the importance of variable forest structure and fuels conditions for ecological restoration, forest resilience, and resulting diversity of wildlife habitat. We know that fire was the most important process influencing these ecosystems and that fire behavior was influenced by topography. This suggests managers could use localized site conditions and landscape position as a guide for varying forest treatments. The various treatments can be based on flexible thinning guidelines using tree species and canopy position to vary retention by site conditions. In sum, our management strategy is based on emulating forest conditions that would have been created by low-intensity, frequent fire throughout the forest matrix.

Summary Findings

Sierra Nevada mixed-conifer forests could benefit from a new management strategy that goes beyond short-term fuel treatment objectives and incorporates long-term ecological restoration and habitat improvement into forestry practices. This strategy is compatible with current landscape fuel treatments (i.e., SPLATs, DFPZs, and WUI defense zones), but strives to incorporate ecological restoration and wildlife habitat needs that have not been explicitly addressed. This strategy can be implemented using a multiage silvicultural system to meet fuel reduction, ecosystem restoration, and wildlife habitat objectives. Important facets of the strategy include:

- **Mechanical fuels management:** When stands cannot be burned, reducing fuels to moderate fire behavior is still a key priority because wildfire is likely to burn the area eventually. A few of the ecological benefits of fire are achieved with mechanical fuel reduction, but thinning is not an effective

substitute for fire in affecting ecosystem processes. Reducing surface fuels is as important as reducing ladder fuels.

- **Limit use of crown separation in fuel treatments:** Sparingly apply canopy bulk density reduction and increased tree crown separation only in key strategic zones. More research is needed, but current models suggest its effects on reducing crown fire spread are limited, and the regular leave-tree spacing does not mimic tree patterns in active-fire-regime forests.
- **The ecological importance of fire:** Prescribed fire can help reduce surface fuels and restore some of the ecological processes with which mixed-conifer forests have evolved.
- **Treatments focused on affecting fire behavior:** Efforts to restore pre-European forest conditions are likely to fail in the face of climate change and also do not provide flexible prescriptions that adapt to different site conditions. Focus treatments on affecting potential fire behavior by manipulating fuel conditions, thereby allowing forests to equilibrate to fire under modern conditions and increasing forest heterogeneity.
- **Retention of suitable structures for wildlife nest, den, and rest sites:** Trees providing suitable structure for wildlife include large trees and trees with broken tops, cavities, platforms, and other formations that create structure for nests and dens. These structures typically occur in the oldest trees. Develop and adopt a process for identifying, and thus protecting, such trees for use by inventory and prescription-marking crews.
- **Stand-level treatments for sensitive wildlife:** Areas of dense forest and relatively high canopy cover are required by California spotted owls, fishers, and other species. Identify and manage areas where, historically, fire would have burned less frequently or at lower severity owing to cooler microclimate and moister soil and fuel conditions for the higher stem and canopy densities that they can support.
- **Large trees and snags:** Given their current deficit in mixed-conifer forest and the time necessary for their renewal, protect most large trees and snags from harvest and inadvertent loss owing to prescribed fire.
- **Landscape-level treatments for prey of sensitive wildlife:** In the absence of better information, habitat for the prey of owls and fishers may best be met by mimicking the variable forest conditions that would be produced by frequent fire. Reductions in stem density and canopy cover would emulate the stand structure produced by local potential fire behavior, varying by a site's slope, aspect, and slope position.

- **Retain hardwoods and defect trees and promote shrub patches:** Hardwoods (particularly black oak) and defect trees (i.e., those with cavities, broken tops, etc.) are valued wildlife habitat and should be protected whenever possible. Increasing understory light for shrub patch development, can increase habitat for some small mammals and birds.
- **Riparian forest fuel reduction:** Prescribed burning of riparian forest will help reduce fuels in these corridors that are also important wildlife habitat.
- **Spatial dispersion of treatments:** Trees within a stratum (i.e., canopy layers or age cohorts) would often be clumped, but different strata would usually be spatially separated for fuel reasons. Give particular attention to providing horizontal heterogeneity to promote diverse habitat conditions.
- **Spatial variation in forest structure:** "Average" stand conditions were rare in active-fire forests because the interaction of fuels and stochastic fire behavior produced highly heterogeneous forest conditions. Creating "average" stand characteristics replicated hundreds of times over a watershed will not produce a resilient forest, nor one that provides for biodiversity. Managers could strive to produce different forest conditions and use topography as a guide for varying treatments. Within stands, important stand topographic features include concave sinks, cold air drainages, and moist microsites. Landscape topographic features include slope, aspect, and slope position.
- **Stand density and habitat conditions vary by topographic features:** Basic topographic features (i.e., slope, aspect, and slope position) result in fundamental differences in vegetation composition and density producing variable forest conditions across the Sierra landscape. Drainage bottoms, flat slopes, and northeast-facing slopes generally have higher site capacity, and thus treatments retain greater tree densities and basal areas.
- **Tree-species-specific prescriptions:** Hardwoods and pines, with much lower densities in current forests compared with historical conditions, would rarely be thinned. Thinning would be focused on firs and incense-cedar. Address pine plantations separately.
- **Silvicultural model and strategy:** Tree diameter distributions in active-fire forests vary but often have nearly equal numbers in all diameter size classes because of periodic episodes of fire-induced mortality and subsequent recruitment. Stand treatments that significantly reduce the proportion of small trees and increase the proportion of large trees compared to current stand conditions will improve forest resilience.

- **Treatment of intermediate-size trees:** In most cases, thinning 20- to 30-in d.b.h. trees will not affect fire severity, and, therefore, other objectives for their removal should be provided. Where those objectives are identified, silvicultural prescriptions would only remove intermediate-size trees when they are shade-tolerants on mid or upper slope sites.
- **Field implementation of silvicultural strategy:** Modify marking rules to ones based on species and crown strata or size and structure cohorts (a proxy for age cohorts) rather than uniform diameter limits applied to all species.
- **Allocation of growing space:** A large proportion of the growing space would be allocated to the largest tree stratum.
- **Assessment of treatment effects:** Emphasis is on what is left in a treated stand rather than what is removed.

Research Needs

Some of our management recommendations are currently based on inferences from studies in other forest types. There are many aspects of Sierra Nevada ecosystems that are still poorly understood. The list below is focused on research needed to investigate and refine some of the suggested management practices. These studies and implementing the suggested strategy will undoubtedly raise new questions. Working together, forest managers and researchers can exchange information and identify unknowns as they develop.

1. Quantify the leaf area and growth relationships needed to develop stocking control relationships for Sierra Nevada mixed-conifer forest. This will allow completion of a Sierra Nevada MASAM for the Kings River Project (KRP) area or any other area in the Sierra where this approach could be implemented. This tool will allow the design and assessment of a variety of multiaged-stand structures that include, among others, older residual trees, development of resilient structures, and accommodation of prescribed burning regimes.

2. Develop and implement an adaptive monitoring strategy to assess the efficacy of a multiaged strategy at both the stand and landscape scales. This information will include both on-the-ground monitoring of treated stands and simulations using Sierra Nevada MASAM. This input will be used to refine the strategy over time and make large-scale assessments of landscape patterns for wildlife habitat, potential fire behavior, and general diversity of

vegetation patterns. A multiaged strategy would be adjusted pending results of monitoring efforts to accommodate other resource objectives such as wildlife, fire, or forest restoration.

3. Assess the potential outcomes of this proposed silvicultural approach on vegetation response and wildlife habitat features of interest. This could be combined with a comparison to other possible silvicultural strategies to evaluate the similarities and differences of approaches. Research would also assess the effects of any treatment on predicted fisher resting habitat using either a predictive microhabitat model (Zielinski et al. 2004b) or a habitat model based on Forest Inventory and Analysis (FIA) protocols (Zielinski et al. 2006).

4. More closely examine the distribution of tree size and canopy density characteristics within female fisher home ranges to establish the means and variances of tree number/density by size class, for both conifers and hardwoods. This would require overlaying the boundaries of female fisher home ranges, which have been estimated on the Sierra and Sequoia National Forests (Mazzoni 2002, Zielinski et al. 2004a), and then using both remotely sensed and ground-based methods to described the vegetation within these areas. Once we have estimates of the average number of, say, white fir between 20 and 30 in d.b.h. per acre within the average female home range, we will be able to compare this and other characteristics with the average number of this species and size class predicted to occur as **residuals** after proposed treatments. If the selected tree size or density characteristic, when measured after treatment, is significantly **lower** than what occurs in female home ranges, then the proposed management activity would not be consistent with fisher conservation.

5. Determine fire histories of riparian areas to identify fire frequency, intensity, and extent. How far does the riparian influence for dampening fire extend away from the stream? What stream characteristics (i.e., bank slope, stream size, etc.) affect the size of the riparian influence zone? What were historical fuel loads in these forests? How can riparian systems be managed to reduce adverse fire effects while maintaining wildlife habitat? In current wildfires, are riparian forests typically experiencing high-intensity crown fires, or are moister fuels and microclimate still dampening fire behavior?

6. Determine how forest structure and composition varied by topographic feature under an active-fire regime in the Sierra Nevada. There have been studies in the Klamath Mountains and eastern Washington, but no information is available for California forests. The research would identify which topographic features matter, and stand structure and fuels loads associated with different physiographic areas.

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Metric Equivalents

When you know:	Multiply by:	To find:
Acres (ac)	0.405	Hectares
Inches (in)	2.54	Centimeters
Feet (ft)	.305	Meters

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Review

The ecology of mixed severity fire regimes in Washington, Oregon, and Northern California

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ABSTRACT

Forests characterized by mixed-severity fires occupy a broad moisture gradient between lower elevation forests typified by low-severity fires and higher elevation forests in which high-severity, stand replacing fires are the norm. Mixed-severity forest types are poorly documented and little understood but likely occupy significant areas in the western United States. By definition, mixed-severity types have high beta diversity at meso-scales, encompassing patches of both high and low severity and gradients in between. Studies of mixed-severity types reveal complex landscapes in which patch sizes follow a power law distribution with many small and few large patches. Forest types characterized by mixed severity can be classified according to the modal proportion of high to low severity patches, which increases from relatively dry to relatively mesic site conditions. Mixed-severity regimes are produced by interactions between top-down forcing by climate and bottom-up shaping by topography and the flammability of vegetation, although specific effects may vary widely across the region, especially the relation between aspect and fire severity. History is important in shaping fire behavior in mixed-severity landscapes, as patterns laid down by previous fires can play a significant role in shaping future fires. Like low-severity forests in the western United States, many dry mixed-severity types experienced significant increases in stand density during the 20th century, threatening forest health and biodiversity, however not all understory development in mixed-severity forests increases the threat of severe wild fires. In general, current landscapes have been homogenized, reducing beta diversity and increasing the probability of large fires and insect outbreaks. Further loss of old, fire tolerant trees is of particular concern, but understory diversity has been reduced as well. High stand densities on relatively dry sites increase water use and therefore susceptibility to drought and insect outbreaks, exacerbating a trend of increasing regional drying. The need to restore beta diversity while protecting habitat for closed-forest specialists such as the northern spotted owl call for landscape-level approaches to ecological restoration.

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1. Introduction

In fire ecology, definitions of mixed severity fire arose from observations that many fires and fire regimes could not be neatly classified as either surface fire or stand replacement dominated disturbances. These fires occupied a middle zone in terms of first order effects leaving highly variable and mixed patterns of lethal and non-lethal outcomes. Definitions of mixed severity also arose from subtraction of more readily defined terms. Ecosystems with low severity fire were easily described as those where surface fire effects tended to dominate, and they were subsequently defined as those where less than 20% of the overstory trees or basal area is killed by the sum of all fire effects (Agee, 1990, 1993). In concept, low severity fires principally reduce the volume and distribution of the most flammable fuels via surface fire activity, and mortality effects are typically minimal (Stephens et al., 2008). At the opposite pole, high severity fires were also readily described as those where crown fire effects tended to dominate, defined by Agee (1990, 1993) as more than 70% of the overstory trees or basal area killed by the sum of all fire effects. High severity fires principally kill trees via torching and running crown fire and often significantly change the volume and distribution of surface and canopy fuels. Mixed severity fires formed the catch-all bin for what remained, by Agee's (1990, 1993) definition those where 20–70% of the overstory trees or basal area are killed by the sum of all fire effects. The broad bin of 20–70% masks a great deal of variability and would benefit from additional subdivisions. Progress toward a better scientific foundation for mixed severity fire will come by stratifying mixed severity regimes by ecological regions and proportion of high severity fire. Brown et al. (2008): stated the case for the latter "...simply to describe a historical fire regime as variable severity is by itself not useful either for characterizing fire as an ecological process or for fire management or ecological restoration purposes. For example, without reference to scale it is possible to conclude that recent variable-severity fires in ponderosa pine forests (i.e., that have included both surface burning as well as large areas of crown mortality) are within a historical range of variability even though areas of crown mortality are orders of magnitude larger than any area that occurred historically (e.g., Romme et al., 2003).We propose that future definitions of variable-severity fire regimes in ponderosa pine and related forests must be accompanied by descriptions of the maximum spatial extent and how often crown fire occurred over a defined period of time".

> It is important to note that canopy damage is not necessarily the same as soil damage and the two measures of severity can be independent of each other (Jain and Graham, 2007; Safford et al., 2009). In general, the severity of impacts cannot be generalized across different components of an ecosystem (e.g. soils, trees, understory vegetation, streams).

Mixed severity fire regimes are poorly understood and poorly documented but in all likelihood were widespread both in the western and eastern US. For example, Schoennagel et al. (2004) estimate that mixed severity regimes account for 17–50% of the major forest types of the Rocky Mountains.

Key ecological and management questions associated with historic[1] mixed-severity regimes center on implications of structurally diverse and temporally variable landscapes for habitats, animal movements, and propagation of disturbances. Consistent with the *intermediate disturbance hypothesis* (Connell, 1978; Petraitis et al., 1989), mixed severity regimes (by definition) produced rich intermediate scale beta diversity, providing a wide variety of habitats across landscapes. Forests in which mixed severity regimes were the norm were likely to support plant and animal species that prefer closed or nearly closed conditions for at least a part of their life history (Spies et al., 2006), as well as early successional and mid-successional specialists, and species that used both early and late-successional conditions [e.g. the California spotted owl (Bond et al., 2009) and the northern spotted owl in the Klamath Mountains (Franklin et al., 2000)].

In this paper we discuss: (i) the likely extent and location of historical forests of the mixed severity fire regime in Oregon, Washington and California, and variation in fire ecology within this large class (ii) the environmental factors that produce mixed-severity fires; (iii) changes to mixed severity landscapes during the 20th century and threats to biodiversity resulting with those changes; and (iv) uncertainties in the knowledge base and research needed to address those uncertainties. In a companion paper we discuss management approaches to reducing losses to remaining old trees and the habitat they represent; and to maintaining an appropriate mix of early, mid, and late successional habitats across landscapes.

2. Ecology and spatial geography of mixed severity disturbance

What exactly is a mixed severity disturbance? At a broad regional scale all wildfire is mixed severity, a fact that limits the usefulness of such scales for ecological interpretation. Moreover, all disturbance processes exhibit heterogeneity at one spatial scale or another, which may manifest within stands, across landscapes, or in some combination of the two. Within the spectrum of possible patterns mixed severity regimes grade into low and high severity regimes without distinct thresholds or patterns. To better understand the nature of mixed-severity regimes, we must look to the ecology, the spatial geography, and the variability of fires and their effects.

Mixed-severity fires create a patchiness of forest structure, composition, and seral status that can be observed and quantified at an intermediate or meso-scale, with patch sizes ranging from a few hundredths up to tens or hundreds of ha, depending on locale and climatic drivers (Fig. 1a). In forest types that were historically dominated by mixed severity regimes, surface and canopy fuels, topography, climatic conditions, and ignitions worked in concert to influence variation in fire frequency, severity, spatial extent, and seasonality. The result was a complex spatio-temporal mix of low, moderate, and high severity patches.

As we discuss in more detail later, the scale of patch sizes and the envelope of burn severity vary with forest type and across the region, however there are also widespread similarities. Studies in both Washington and California have found that patch sizes in mixed severity regimes followed a negative power law

[1] In the context of this paper, "historic" refers to the period prior to settlement by EuroAmericans.

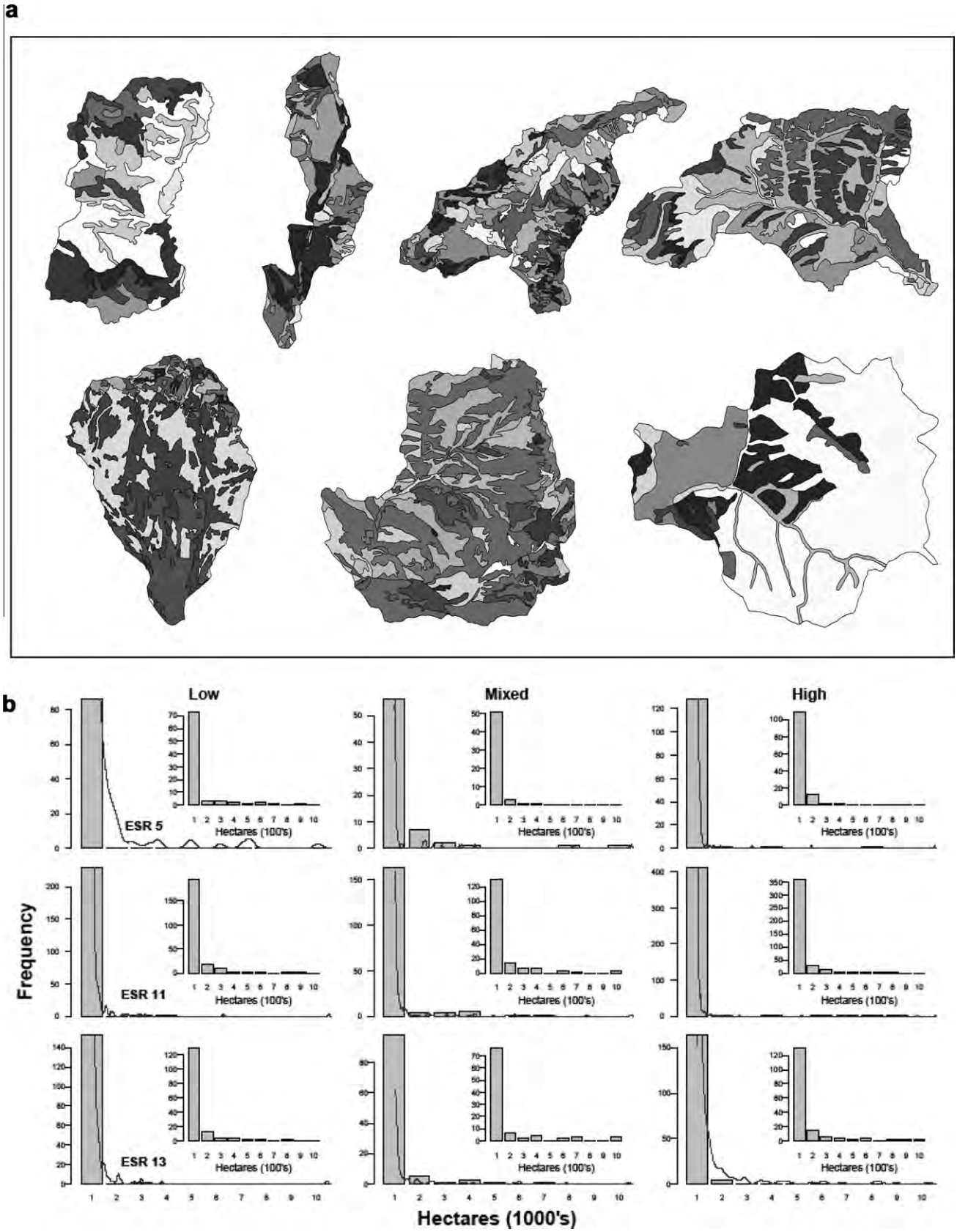

Fig. 1. (a) A sample of 7 historical (ca. 1900) maps of combined cover type and structural class conditions from subwatersheds of the eastern Washington Cascades. Gray tones indicate unique cover type-structural class combinations. Note the highly variable patch sizes. (b) Frequency-size distributions of reconstructed historical (ca. 1900) fire severity patches in three ecoregions of the eastern Washington Cascades. *Low*, *mixed*, and *high* denote severity corresponding with <20%, 20–70%, and >70% of the overstory crown cover or basal area killed by fires, respectively. Data are from Hessburg et al. (2007). (See Hessburg et al., 2000 for complete details).

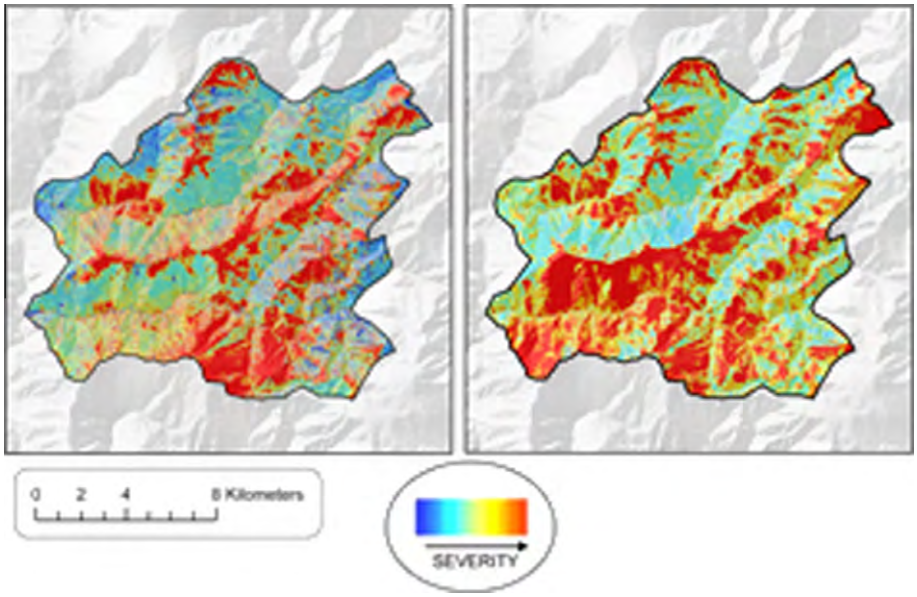

Fig. 2. Comparison of fire severity between the 1987 Silver fire (left) and the 2002 Biscuit fire, which burned through the same area (Thompson et al., 2007).

approximating a Pareto distribution (i.e. many small patches and a few large forming a long tail to the right in the frequency-size histogram). That was the case in two recent fires in Yosemite National Park (central California) where stand replacement patch sizes ranged from .05 (the lower limit of determination) to 90 ha (Collins and Stephens, 2010). In another California study, landscapes exhibited multi-scale patterns of fire sizes that followed a power law distribution for both the meso-scale (50–5000 ha) and for smaller patches embedded within the larger (Moritz et al., 2010). Smaller patch sizes were thought to be driven by endogenous processes, larger by rare or extreme events. Similar results were found in eastern Washington, where patch sizes of low, mixed, and high severity fires ranged from ~1 (the lower limit of determination) to 10,000 ha (Hessburg et al., 2007, Fig. 1b).

The large amount of edge and clumpiness in forest structure, composition, and seral status within and among patches provides a rich intermingling of habitats for early, mid-, and late-successional specialists as well as variety for individual species. As an example of the latter, California spotted owls prefer unburned or lightly burned mixed conifer forests with large trees for roosting and, probably because of prey abundance, moderately or severely burned forest for foraging (Bond et al., 2009). Similarly, Franklin et al. (2000) found that a mosaic including both old forests and early successional patches provided optimal habitat for northern spotted owls in California.

Mixed-severity systems exhibit temporal as well as spatial variability. Depending on climate-vegetation interactions, the proportion of low to high severity patches might vary between fires in a particular locale (e.g. Heyerdahl et al., 2002; Gedalof et al., 2005; Marlon et al., 2009), and the characteristic power law pattern of patch sizes may be altered by extreme fire weather. A recent example is the contrast between the 1987 Silver Fire in SW Oregon, which created a mosaic of low to moderate severity patches, and the 2002 Biscuit Fire, which burned the same area with a preponderance of high severity patches (Fig. 2). The median crown damage (i.e. scorch and consumption) in the Silver fire, which burned under relatively mild weather conditions, was about 16% while the median crown damage for the Biscuit fire, which burned under much hotter and windier conditions was 63% (Thompson and Spies, 2010). Another factor contributing to the higher severity of the Biscuit fire was the high amount of early seral patches produced by the Silver Fire. As we discuss in more detail later, some early seral community types have a high probability of burning severely (Odion et al., 2004; Stephens and Moghaddas, 2005; Thompson et al., 2007). In general, legacies from past fires, as well as from other natural disturbances and land uses, influence fire behavior in a given area, a point we return to later in the paper.

2.1. Where are the forests of the historical mixed severity fire regime?

Fig. 3a shows the geographic distribution of mixed-severity forests in the Pacific Northwest, while Fig. 3b illustrates variation among the forest types in the modal characteristics of fires (because of temporal variation as discussed above, the ratios shown in Fig. 3b should be understood as approximations of central tendencies). In the Interior West (east of the Cascades crest), the Klamath Mountains, low to mid elevations on the western slopes of the Cascades (depending on latitude), portions of the eastern slopes of the Coast Range, and in the Northern and Central Sierra Nevada, forests characterized historically by mixed-severity fire regimes occupied a broad range of environments between forests with predominantly surface fire regimes (dry ponderosa pine, pine-oak, and oak) and subalpine forests dominated by stand replacing regimes.

Spies et al. (2006) grouped forest types that were historically influenced by mixed-severity fire regimes into ponderosa pine (dominated by low severity but experiencing occasional mixed severity in some locales), mixed-conifer/evergreen on dry sites (includes the Douglas-fir zone and driest parts of the other zones), and mixed-conifer/evergreen on mesic sites. Using forest types as defined by Cowlin et al. (1942) for eastern Oregon and Washington, mixed severity fire regimes were probably common in the "pine mixture", "upper slope mixture", "Douglas-fir", and "white fir" types, all four being mixed conifer forest types occupying relatively cool and mesic environments above the forest-shrub land ecotone. The pine mixture and upper slope mixture are distinguished by a relatively high proportion (20–50%) of ponderosa pine, while the other two types have a greater proportion of either Douglas-fir or

Fig. 3a. Bailey Sections and Subsections in Oregon, Washington, and California with forest vegetation types that display mixed severity fire regimes(Bailey, 1995, 2009, http://svinetfc4.fs.fed.us/clearinghouse/other_resources/ecosubregions.html). Section M261A has been modified along the Southwest Oregon coast according to Frenkel (1993) to exclude an area of mostly high severity regimes. Most Sections also contain areas of either low-severity or high-severity regimes (or both). Section alphanumeric codes are: M242B (Western Cascades), M242C (Eastern Cascades), M261A (Klamath Mountains), M261B (Northern California Coast Ranges), M261C (Northern California Interior Coast Ranges), M261D (Southern Cascades Section), M261E (Sierra Nevada), M261F (Sierra Nevada Foothills), M261G (Modoc Plateau), M332G (Blue Mountains), and M333A (Okanogan Highlands).

white fir (Cowlin et al., 1942). Western larch is a common component, particularly in the latter two types.

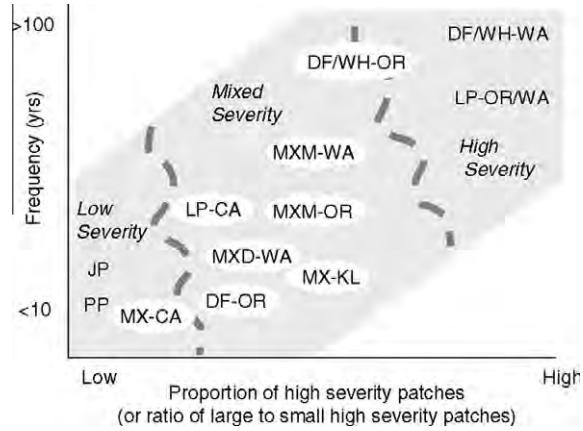

Fig. 3b. General distribution of low, mixed, and high severity disturbance regimes in relation to proportion of high severity patches and frequency of fire for the Pacific Northwest and northern California. The centers of the distributions of a few of the major forest types are shown to illustrate variation. Geographic modifiers illustrate that disturbance regimes of a forest type can vary geographically. PP = ponderosa pine, JP = Jeffrey Pine, DF = Douglas-fir, MX = Mixed conifer, WH = western hemlock; LP = lodgepole pine. WA = Washington, OR = Oregon, KL = Klamath region of California and Oregon, CA = California Cascades and Sierra Nevada.

In the eastern Washington Cascade Mountains and across the Okanogan Highlands, forest types we classify as having predominantly mixed severity regimes (Fig. 3) represent about 30% of total forest area, while in eastern Oregon they represent about 13% (Cowlin et al., 1942). The total area of mixed regime types is similar between the two states (except for the Douglas-fir type, which is concentrated largely in Washington), however, eastern Oregon has more total forest area and a four-fold greater area of dry ponderosa pine types.

In the southern Cascade Range northern Sierra Nevada, and Klamath Mountains, mixed severity fire regimes are associated with mesic mixed conifer/hardwood forests, Douglas-fir, and red fir forest types (Skinner et al., 2006; Skinner and Taylor, 2006; Collins and Stephens, 2010). Relatively mesic Douglas-fir/hemlock forests in low to mid elevations of the western central Cascades may also have experienced mixed severity fire regimes, occasionally with more surface fire than stand replacing effects, but generally the converse was true (Morrison and Swanson, 1990).

Within a given forest type the characteristics of mixed severity fires vary spatially over environmental gradients that often follow large and small topographic features. In both the eastern Cascade Mountains and the Okanogan Highlands of eastern Washington, Hessburg et al. (2007) found that mixed severity fires occurred in both the mesic and the dry forests of the Douglas-fir and grand fir zones during the pre-suppression era, representing about 58% of that region. In moist mixed conifer, they found that stand replacement fire effects were slightly more widespread in patches than surface fire effects, while in dry mixed conifer, surface fire effects were more widespread by nearly 2:1. Similarly, whereas all of the Klamath Mountains can be classed as having a mixed severity regime, surface fires became more dominant as one moved from the mesic western portions to the dry eastern portions of the range. (The Biscuit fire burned in the relatively mesic northwestern portion). This same pattern likely holds over moisture gradients in any subregion, and across the north-south moisture gradient within the region, with the result that forest type alone is not a good predictor of the proportion of low and high severity patches that

may occur within mixed severity regimes of the region. Some forest types that are typified by mixed severity regimes in Washington fall into a low severity regime in California. That is the case with dry Douglas-fir and mixed conifer types, which in California are typified by low-intensity surface fire rather than mixed severity (Skinner et al., 2006; Skinner and Taylor, 2006).

2.2. What influences the relative proportions of high and low severity?

Any given fire regime is influenced to one degree or another by both top-down and bottom-up forces. From the top-down, spatio-temporal patterns of regional climate influence fire frequency and severity through patterns of seasonality, temperature, and precipitation (Littell et al., 2009). To a certain extent throughout the region, but especially in California and southwestern Oregon, top-down climatic control is exerted largely through the Mediterranean climate of long, dry summers that provide for conditions where fires can readily burn in mixed conifer forests regardless of variation in total annual precipitation (Minnich, 2006). However, fire activity varies year to year, which Littell et al. (2009) correlate with variations in either summer precipitation or summer Palmer Drought Severity Index and other researchers associate with large scale atmospheric phenomena that alter winter precipitation (Norman and Taylor, 2003; Gedalof et al., 2005; Taylor et al., 2008, 2006; Trouet and Taylor 2009, 2010; Trouet et al. 2010). Years with low winter precipitation and high fire risk are associated with a strong atmospheric ridge that blocks moisture from moving onshore (the Pacific North American teleconnection pattern, or PNA), however the strength of this effect varies with phases of both the Pacific Decadal Oscillation (PDO) and the El Nino Southern Oscillation (ENSO). Wet-dry patterns are not necessarily synchronous across the region. In particular, ENSO and the PDO exhibit a dipole between the northwestern and southwestern US with the fulcrum shifting north or south in the vicinity of the Klamaths on a decadal time scale (Westerling and Swetnam, 2003). In consequence, dry years in the Sierras and Klamathsmay or may not correspond to dry years further north (Skinner, 2006; Trouet et al., 2006). Over longer time scales, a 2000 year record of sediment cores from the Oregon Siskiyous shows that large sediment pulses occurred frequently during the Medieval Warm Period, with long periods of low sediment input immediately before and after (the latter corresponding to the Little Ice Age) (Colombaroli and Gavin, 2010). Questions remain about the degree to which that work can be generalized and the relation between sediment yields and fire characteristics.

From the bottom up, local factors (e.g. stand and landscape structure, topography) exert a strong enough influence that locales may burn quite differently even under the same top-down conditions (Skinner et al., 2006; Skinner and Taylor, 2006; Colombaroli and Gavin, 2010 Regional climate is filtered and shaped by broad-scale geologic, vegetative, and geomorphic conditions. For example, Hessburg et al. (2000b, 2004) found that patterns of fire severity eastern Washington were best explained by grouping into eco regions with similar biogeoclimatic influences. The moisture regime of each biophysical setting (precipitation + soil depth + soil organic deposits + evapo-transpiration) interacts with local biotic factors (e.g. fuel bed characteristics, stand composition and structure) and biotic patterns at the landscape scale to determine fire severity regimes (Miller, 2003). In eastern Oregon and Washington, white fir types, so named because land cover was dominated by white or grand fir (50% or more by volume), commonly occurred on relatively cool and moist topographic positions (e.g. north slopes) within the elevation range of ponderosa pine (Cowlin et al., 1942). These provided potential habitat for species that prefer relatively closed forests, and may have been important seed source areas for the invasion of white and grand fir into ponderosa pine dominated patches following the modern era of fire exclusion (e.g. Camp et al., 1997). Because of their relatively high fuel moisture, riparian zones tend to buffer the spread of fires, but with their relatively high biomass may change from fire suppressors to fire corridors if fuel moisture drops sufficiently low (Pettit and Naiman, 2007).

> Humans significantly altered fire regimes even before the EuroAmerican era. In some areas of the Northwest, burning by Native Americans altered the dominant effects of climate and may have converted potentially mixed-severity regimes to frequent, low-intensity ones. For example, in the Little River watershed of the Umpqua National Forest (Oregon), the relationship between fire occurrence (as measured by bole scars) and precipitation (as measured by tree ring widths) changed in the mid-19th century, coincident with declining populations of Native Americans (Carloni, 2005). Fire scars from 1590 to 1820 were not correlated to precipitation, whereas between 1850 and 1950 they were. Similarly, tree regeneration was uncorrelated with fire events between 1590 and 1820, but between 1850 and 1950 there was a highly significant positive correlation. Similar patterns were probably found in areas with high densities of Native Americans in California (Anderson 2005, Stephens et al., 2008; Skinner et al., 2009).

In the more mesic northerly parts of the region, cool, moist northerly aspects may burn with mixed severity while adjoining southerly aspects burn with low severity, or both may burn with mixed severity with stand replacement fire dominating on the northerly aspects and surface fire dominating on southerly. In the more arid Klamath Mountains the opposite is seen, patchy fires dominating on south and west facing aspects and low severity fires dominating on north and east facing aspects (Taylor and Skinner, 1998). Severe weather conditions can override topographic effects to at least some degree. For example, aspect effects were weak in the Biscuit fire, perhaps because the hot, dry winds that drove the fire during its blow-up period came from the NE and drove the fire against aspects that are often considered refugia (Thompson and Spies, 2010). However, in the Megram fire (Northern California, 1999) under similar severe conditions, topography was significantly associated with fire severity patterns (Jimerson and Jones, 2003).

2.3. Vegetation type and structure

Within a given climatic regime, and even within a given fire, densely stocked, uniform forests have a relatively high probability of burning with lethal effects dominating. However, several caveats go with this. In the case of plantations, flammability has been found to depend on the degree and kind of slash treatment during site preparation (Huff et al., 1995; Weatherspoon and Skinner, 1995). Weatherspoon and Skinner (1995) found that techniques which encouraged the growth of grasses (machine piling) resulted in greater fire damage, while those that produced a forb cover (broadcast burning) resulted in less and plantations in which logging slash had not been treated suffered heavily. As fully-stocked stands of shade intolerant species mature, self-pruning raises crown base height and shading discourages the development of other fuel ladders, lessening the chances of fire propagating from ground to crowns (but vulnerability to crown fire via fuel ladders in adjacent stands remains). Such stands are on their way to becoming the mature closed conifer stands that were the most resistant vegetation type in the Biscuit fire (Thompson and Spies, 2009, 2010). There are significant variations on that theme; if stands are too dense individual tree growth is retarded and movement toward a resistant mature stand may be impeded, while the

vegetation that develops beneath an open stand may act as fire suppressants rather than fuel ladders (Agee et al., 2002).

With the exception of extreme fire weather conditions (⩾95th percentile), forests composed of large and very large fire tolerant species tend to burn at low or mixed severity with surface fire effects dominating even when overstory canopies are dense, in part at least because trees are tall with live crowns relatively far above the surface (Stephens and Moghaddas, 2005; Thompson and Spies, 2009; Thompson and Spies, 2010). However, that depends on the dominant tree species, which in turn reflects site characteristics and probably fire history. In the 1987 fire complex in Northern California, stands dominated by ponderosa pine experienced higher severity than those dominated by Douglas-fir or hardwoods, which Weatherspoon and Skinner (1995) attribute to "the fuelbed, usually warmer and drier sites, and generally more open stand structure of ponderosa pine-dominated stands". Weatherspoon and Skinner also suggest that ponderosa stands, with their past history of frequent surface fires, were more impacted by fire exclusion during the 20th century than Douglas-fir stands. In central Oregon's B & B fire, however, ponderosa pine stands burned with lower severity than mixed-conifer, perhaps because the latter had increased fuel loading as a result of recent insect and disease mortality.

Stands dominated by hardwoods tend to burn with lower intensity than conifer-dominated stands (Skinner and Chang, 1996; Skinner et al., 2006), and there are anecdotal examples of mid story hardwoods protecting conifers in the Siskiyous (Perry, 1988; Raymond and Peterson, 2005). Hardwood-dominated stands suffered higher levels of damage during the Biscuit fire than closed conifer stands (Thompson and Spies, 2010), however, crown damage does not necessarily correlate with propagating flames to adjacent crowns (e.g. when damage is due to scorch rather than consumption). In the Biscuit fire, when hardwoods were intermixed with conifers "the hardwood subcanopy affected fire behavior in ways other than serving as a ladder fuel" (Raymond and Peterson, 2005).Raymond and Peterson speculated that mature hardwoods shaded dead fuels and slowed their desiccation, reduced wind speed within stands, and blocked the propagation of heat upwards into conifer canopies. A conifer subcanopy would also produce the first two effects, however the third depends on flammability, a function of chemical content (esp. monoterpenes), hydration, and leaf structure, factors in which conifers and hardwoods differ (Agee et al., 2002). In their study of foliar moisture content in Pacific Northwest species, Agee et al. (2002) concluded that understory grasses would have a dampening effect on flame lengths into September and understory shrubs would have a dampening effect into October. On the other hand, conifers suffered the worst crown damage in the Biscuit fire when in open forests with a shrub understory (Thompson and Spies, 2009). Species composition matters; shrubs in the Thompson and Spies study were predominantly sclerophyllous species that, as we discuss below, may burn readily.

Depending on species composition and age, early successional stands can be quite flammable. In the 2002 Biscuit fire, stands that originated from the 1987 Silver fire (predominantly *Ceanothus* and *Arctostaphylos* shrubs, with intermixed sprouting hardwoods and young conifers) experienced significantly more canopy damage than older, closed conifer forests (Thompson and Spies, 2010). Similarly, in their study of recent fires in Yosemite National Park, Collins and Stephens (2010) found that stands were most susceptible to high severity reburn when they were between 17 and 30 years old. In contrast, in their youngest stages early successional stands have been found to be less flammable and constrain the extent of fires. For example, recently burned forest patches in mesic Sierran mixed conifer forests constrained the extent of subsequent fires when the time since previous fire was 9 years or less (Collins et al., 2009). The period of relatively low flammability likely varies with a number of factors such as the rate at which flammable biomass accretes and interactions between weather and topography. Post-fire management may also have an effect. As mentioned earlier, stands that were salvage logged and planted to conifers following the 1987 Silver fire reburned more severely in the 2002 Biscuit fire than stands that had not been salvaged and planted, although the difference was not large (Thompson et al., 2007; Thompson and Spies, 2010).

2.4. Landscapes: context, fences and corridors, self-organization

The complex mosaic resulting from variety in successional and structural conditions across a broad spectrum of patch sizes potentially affects patterns of burning in at least two ways. Intuitively, fire severity within patches can depend on the larger landscape context, and research supports this idea (Weatherspoon and Skinner, 1995; Fites-Kaufmann, 1997). For example, patches that for environmental and structural reasons are conditioned to burn with mostly stand replacing effects may burn with moderate severity if embedded within a landscape dominated by low-moderate intensity fire (Fig. 4). Likewise, stands conditioned to burn with mostly surface fire effects dominating may burn with mixed or even high severity if embedded within a landscape prone to high-intensity fire (Hessburg et al., 1999a, 1999b). Fire frequency may also depend on context, with mixtures of short- and long-interval types resulting in the frequency of each shifting toward the other (Agee et al., 1990). At a more dispersed level, the variable grain and pattern of the mixed-severity mosaic act as a patchwork quilt of "fences and corridors" that interact with topographic complexity and top-down climatic drivers to either facilitate or resist the movement of fires (or insect outbreaks or species migrations) (Moritz et al., 2010).

What is the dynamic of this pattern? Is it a shifting mosaic, and if so is there a landscape-level self-organizing[2] aspect that constrains overall patterns within a certain envelope (Moritz et al., 2010)? Or are there stand-level self-organizing aspects in which certain structures tend to perpetuate themselves and maintain a semi-static mosaic (Perry, 1995; Skinner and Taylor, 2006; Odion et al., 2009)? The answer to all three questions is probably yes; the mixed severity dynamic is too complex to be neatly pigeon-holed, and different mechanisms may operate at different temporal and spatial scales (Holling, 1992; Perry, 1995; Moritz et al., 2010).

As we have discussed, even within shifting mosaics the patterns of natural wildfires are constrained within certain envelopes by climate and topography. Current studies also support the existence of self-reinforcing community structures, or to use Peterson's (2002) terminology, memories of past disturbances influence responses to future disturbances. In the Biscuit fire, for example, the strongest predictor of relative crown damage was crown damage in the Silver fire, which burned 15 years previously within the same area (Thompson and Spies, 2010). Followed to its logical conclusion, and depending on fire return intervals, system memory would tend to push the landscape toward a binary condition of early successional shrub fields and older closed conifer forests. Because fires that are sufficiently severe have some impact even on relatively resistant forests (Thompson and Spies, 2010), repeated fires at sufficiently short intervals could erode the binary landscape structure over time, however Moritz et al. (2010) argue that lagged effects of past fires and recovery rates (sensu Peterson, 2002) would prevent that from happening and maintain structural diversity on the landscape. Moreover, variability in fire return intervals

[2] Self-organization refers to the tendency for system dynamics to emerge from internal interactions and structure rather than outside forces. "Self-reinforcing" is a similar concept in which system structure leads to processes that tend to maintain the structure. The dynamics of any system are likely to be driven by both external and internal forces.

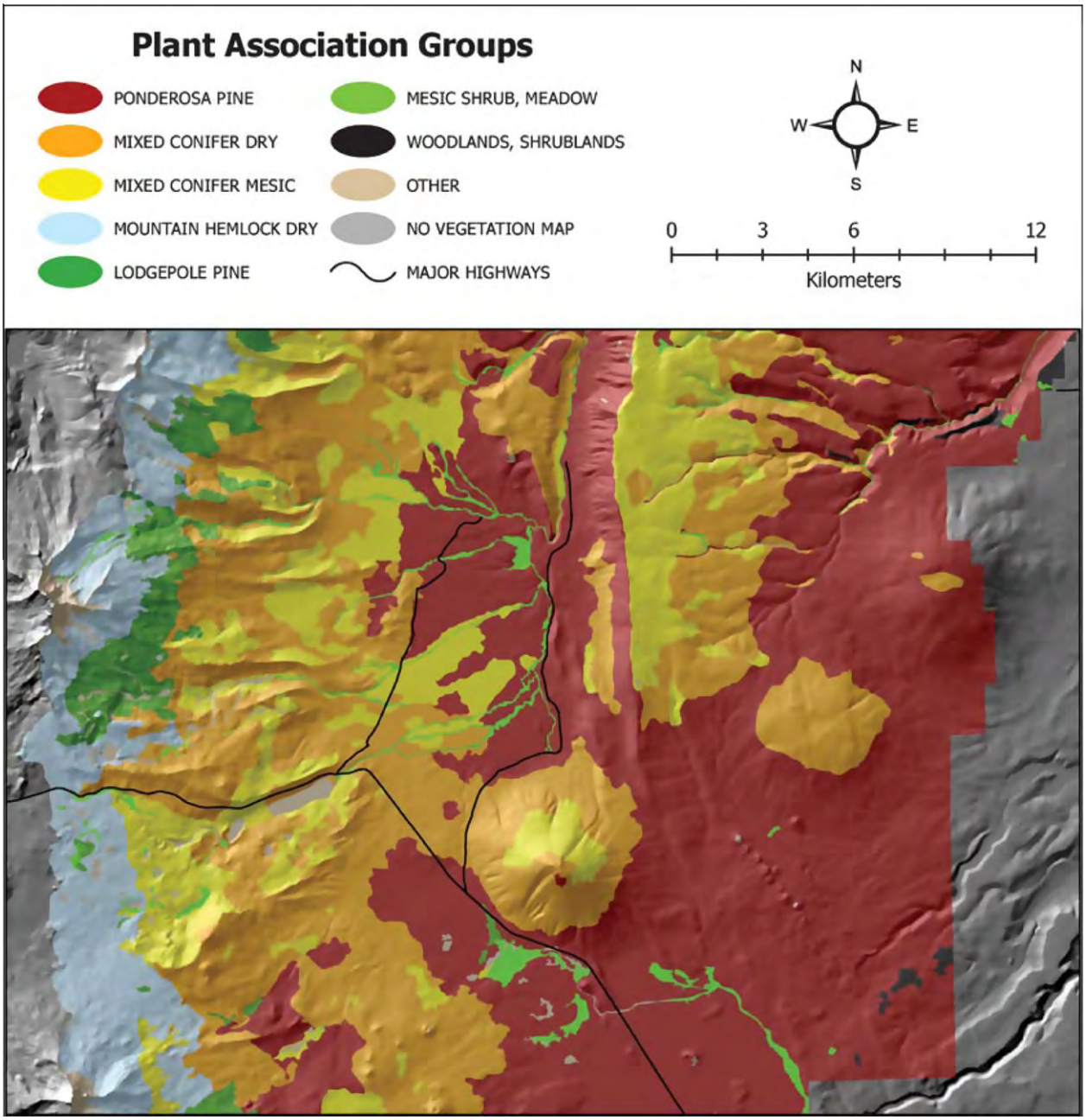

Fig. 4. Relationship between plant association groups and topography on a portion of the Deschutes National Forest, Oregon. Note the intermingling of forest types that occupy different positions on the mixed-severity gradient (refer to Fig. 3). From Spies et al. 2006.

at the landscape scale would allow some areas to develop greater resistance (e.g. thicker bark, higher crowns) and thereby lower severity.[3]

The two clearest examples of self-reinforcing memory in the Pacific Northwest are montane *Ceanothus/Arctostaphylos* shrub fields (chaparral) and forests dominated by large, fire resistant tree species. Skinner and Taylor (2006) hypothesize that because of the frequency with which it burns, chaparral disrupts the normal successional processes that act to bring forest back. Topographic position, edaphic conditions, and less frequent, but more intense fires contribute to long-term persistence of chaparral in the landscape (Nagel and Taylor, 2005; Odion et al., 2009), especially on the upper third of slopes and ridgetops, topographic positions prone to more severe fires (Weatherspoon and Skinner, 1995; Taylor and Skinner, 1998; Beaty and Taylor, 2001). The chaparral growth habit of mostly live material with little surface fuel hinders fire from burning under all but the most severe conditions, but when it does burn it tends to crown and kill intermixed conifers (Thompson and Spies, 2009). In contrast, the neighboring conifer stands produce ample needle cast and small dead material to carry fires more frequently under more benign burning conditions.

Patchy but abundant conifer regeneration has been documented within early successional shrub/hardwood communities in the Klamath Mountains (Shatford et al., 2007; Donato et al., 2009), at least some of the patchiness determined by whether or not the broadleaves and conifers share mycorrhizal species (Horton et al., 1999). However, the time required for conifers to

[3] We are indebted to an anonymous reviewer for pointing this out.

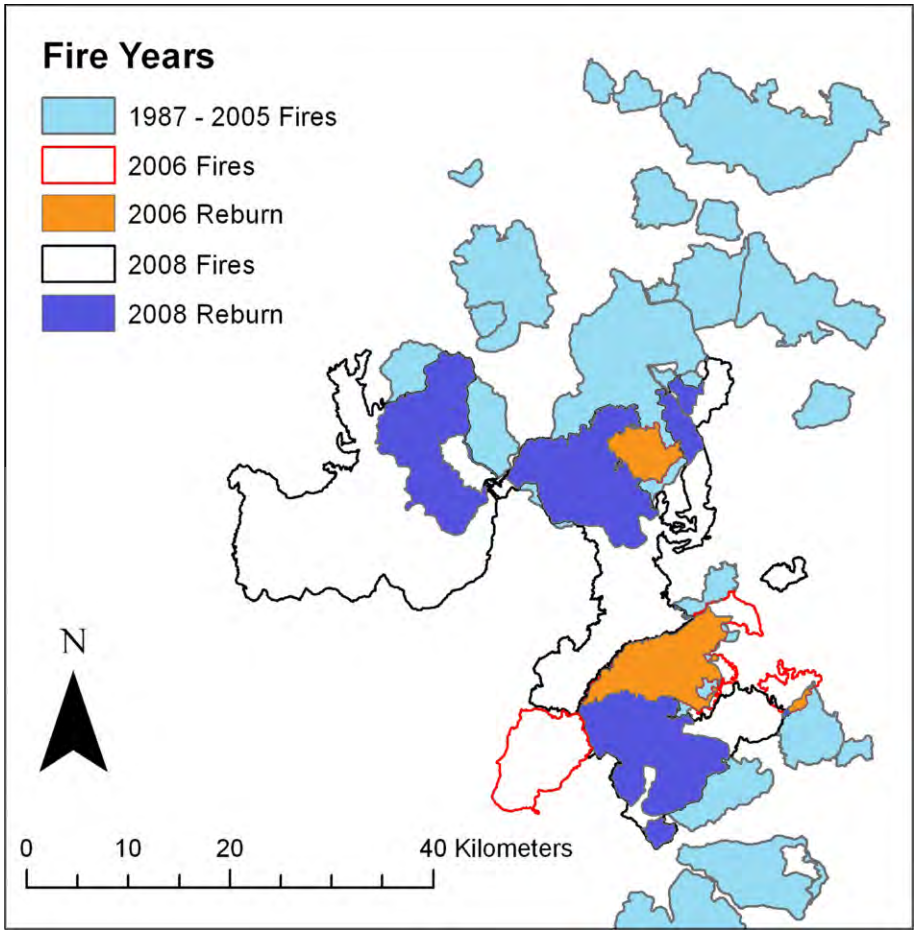

Fig. 5. Map of recurring fires in the Klamath Mountain portion of the Shatford et al. (2007) study. Light blue are areas burned from 1987-2005. The orange areas reburned in 2006, while the dark blue areas reburned in 2008. The dark blue area near center is the location of several of their data sites (see Shatford et al., 2007, Fig. 1. p. 140). Base map by B. Estes.

achieve a size sufficient to survive subsequent fires reduces the chances they will eventually replace chaparral (e.g. Fig. 5), although shrub seed banks and *Arctostaphylos* skeletons in mature conifer stands indicate that succession from shrub dominated communities to closed conifer forests is not uncommon (and has probably become more so in the era of fire suppression). The pathway followed by a given site is likely to depend on a variety of factors, including environment, history (e.g. the timing of a reburn), composition of the shrub community, and initial shrub density. This diversity in pathways provides for heterogeneity of habitats across the landscape (Nagel and Taylor, 2005).

It is well known that short-term fluctuations in weather can strongly influence fire behavior, however both Thompson and Spies (2010) and Collins and Stephens (2010) found that, while weather was clearly a factor, self-reinforcing dynamics resulting from fire history and vegetation type were more important determinants of fire severity. It may take changes in climate over relatively long intervals to produce shifts in self-reinforcing components of the mosaic. Long periods with low or no fire activity would increase the probability of conifers replacing shrub fields and in many forest types long fire-free periods could also allow shade-tolerant fuel ladders to develop and increase the probability of stand-replacement fire in closed forests. For example, in their analysis of factors influencing stand-replacing patches created by mixed-severity fires in Yosemite National Park (fires had not been suppressed in certain areas of the park since 1975), Collins and Stephens (2010) found that the largest patches occurred in forests where *Abies* spp. were intermixed with lodgepole pine, perhaps indicating succession to the more shade tolerant firs. Taylor and Solem (2001) found that *Abies* spp. are replacing lodgepole, ponderosa, and Jeffery pines in California's Caribou Wilderness.

However, fire-free period is not by itself a good general metric for susceptibility to severe fire. Collins and Stephens (2010) found that larger stand-replacing patches in pine and shrub-dominated vegetation types occurred in areas that had burned 17–30 years previously. They attributed the relatively small patch sizes in older stands of those types in part at least to discontinuous fuel beds and the presence of natural fire breaks. Both Odion et al. (2004) and Thompson and Spies (2009) found that stands with the longest fire-free periods in the Klamaths burned with the lowest severity. Stand-development pathways in mixed evergreen forests of the Klamaths are likely to have different implications for fire susceptibility than in other regions, in part because of the large hardwood component. For example, many of the stands studied by Odion et al. (2004) had subcanopies dominated by tanoak. Various factors may play a role in the degree to which vegetation strata function as fuel ladders or fire suppressors (e.g. through foliar characteristics or by suppressing solar radiation and wind speed), and given the wide range of environments occupied by mixed-severity forests these functions probably vary significantly throughout the region. Proximity to seed sources for shade-tolerant tree species is one likely factor. It is possible that more productive sites with relatively high closure in the upper canopy retard the establishment and growth of even shade-tolerant tree species, however

on the east slopes of the Oregon Cascades understory *Abies* spp. were abundant in stands with basal areas up to 26 m^2 ha^{-1}in large (>50 cm DBH) early seral tree species (Perry unpublished). There, *Abies* stocking was influenced primarily by mean annual precipitation. Nevertheless, selective logging or natural processes that open the upper canopy and allow light to penetrate potentially stimulate development of fuel ladders in some stands.

In summary, extended periods with relatively low fire activity have the potential to trigger shifts in the self-reinforcing landscape mosaic, but the dynamic is likely to be complex and such shifts are not a foregone conclusion.

Riparian zones have been little studied with regard to fire; however they represent a significant landscape feature that in at least some cases affects fire behavior at larger scales. In the Klamath Mountains, Skinner (2003) found that, while the range of fire return intervals (FRI's) were similar between riparian (along perennial streams) and upslope areas, median FRI's were approximately twice as long in the former as in the latter. Skinner et al. (2006) concluded that, in the Klamaths, ''..riparian areas along perennial watercourses served as effective barriers to spread of many low-intensity and some moderate-intensity fires and strongly influenced patterns of fire occurrence beyond their immediate vicinity''. Olson and Agee (2005) found a similar pattern on the western slopes of the Cascades in southern Oregon, however median FRI's differed much less between riparian and upslope than they did in the Klamaths and were statistically insignificant. More work is needed on this topic.

3. Biodiversity threats associated with contemporary conditions

Logging and fire suppression during the 20th century have increased the density of young conifers and in many cases triggered a shift from shade-intolerant to shade-tolerant species, putting some components of biological diversity at risk (Perry et al., 2004; Hessburg et al., 2004; Haugo et al., 2010; Naficy et al., 2010). EuroAmerican settlement and management exacted an enormous toll on the large tree structure of the forests of the historical mixed severity regime. Not only were old forests clearcut, but in many areas large and very large remnant emergent trees that made up the upper crown classes of forest patches were selectively harvested (Hessburg et al., 2000a, Hessburg and Agee, 2003).The area of old forest habitat in eastern Oregon and the interior Columbia Basin has been sharply reduced (Henjum et al., 1994; Hessburg et al., 2000a, Wisdom et al., 2000), and the remainder is threatened by wildfires, drought, and insects. For example, about 3 per cent (>5500 ha) of older forest on the east slopes of the Cascades was burned by stand-replacing fires between 1994 and 2003 (Moeur et al., unpublished data). During the same period in the Oregon Klamath province about 11 per cent (>32, 500 ha) of older forest burned at high severity (Moeur et al., unpublished data).

Particularly in the dry portions of the mixed conifer zone, decades of fire suppression have accelerated successional processes and set the stage for resistant forest structures and landscape patterns to be weakened and overcome. Young conifers are poised to dominate many early successional shrub-hardwood communities and have established in relatively high numbers beneath older

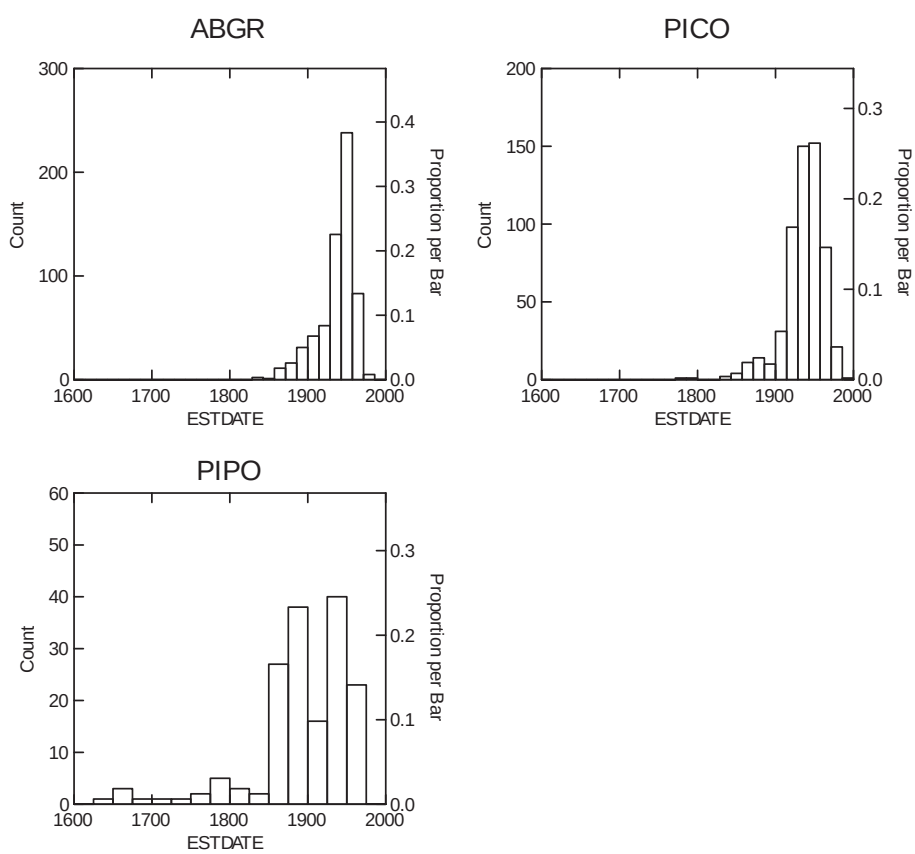

Fig. 6. Establishment date (at breast height) for ponderosa pine (PIPO), grand fir (ABGR), and lodgepole pine (PICO) in old growth forests within the mixed-conifer zone on the Deschutes National Forest, Oregon. Note different Y-scales. Adapted from Perry et al. (2004).

Fig. 7a. Top: Jeffrey pine-white fir and red fir stands in Lassen Volcanic National Park in 1925. Patches of mature trees of variable size are intermixed with areas dominated by shrubs. This vegetation pattern is the result on mixed severity fire effects that burn areas at low, moderate, and high severity. Shrub cover is much lower in 2009 (bottom) and the shrub fields have been invaded by mixture of white fir, Jeffrey pine, red fir, and western white pine. Overall, the forest is now more dense and the forest cover is more homogenous than in 1925. Fires burned frequently in this landscape until fire suppression became effective in 1903 (Taylor, 2000).

stands of early successional, fire resistant trees (Figs. 6 and 7). The latter may be accompanied by a dramatic shift in tree species composition. For example, a study in the mixed conifer zone on the eastern slopes of the central Oregon Cascades found that, while 89 percent of trees older than 150 years were either ponderosa pine, western white pine, or Douglas-fir (all relatively fire resistant after the sapling stage), 90 per cent of trees younger than 100 years were either grand fir or lodgepole pine (Perry et al., 2004).

Relationships between conifers and shrubs are not solely antagonistic. Through their ability to recover quickly (from sprouts or seed banks) and stabilize soils, shrubs and hardwood trees play essential roles in ecosystem resilience (Perry et al., 1989). Ectomycorrhizal species (e.g. manzanita, oaks, madrone) stabilize mycorrhizal fungi and perhaps other soil biota that are important to conifer recovery (Amaranthus and Perry, 1989; Borchers and Perry, 1990; Perry et al., 1989), and *Ceanothus* spp. replenish soil nitrogen. In the Klamaths, both the density and relative growth rate of conifer regeneration correlate positively with density of broadleaves in recovering burns within the Douglas-fir and Douglas fir/tanoak vegetation series (Shatford et al., 2007; Irvine et al., 2009). These associations are negative in the white fir series; however, Shatford et al. (2007) found abundant conifer regeneration within shrub fields in that zone.

Increased stocking densities resulting from invasion by Douglas-fir, grand fir, white fir and lodgepole pine have increased fire risk in some forest types, especially the dry eastern slopes of the Cascades, western slopes of the southern Cascades, and lower to midslopes of the Sierra Nevada. Significant increases in stocking density due to fire suppression are less likely in the more mesic to wet western slopes of the central and northern Cascades and eastern slopes of the Coast Range in Oregon and Washington. In the highly diverse Klamath Range, topographic and vegetation complexity make generalizations difficult. However there, as elsewhere, the effect of ingrowth is locally modified by topography and associated environmental conditions. Moreover, in some cases trees that are potential fuel ladders may have the opposite effect and reduce fire intensity by creating shaded cooler conditions and blocking wind, a poorly understood effect not necessarily restricted to the Klamath region.

The fire risks posed by ingrowth vary with species and forest type. We have discussed broad-leaved hardwoods in this respect. On the eastern slopes of the Oregon Cascades stocking density of *Abies* spp., which increases with mean annual precipitation, is a significant predictor of crown fire risk, but density of *Pinus* spp. is not (Perry et al., 2004, Perry unpublished),[4] reflecting in part the long crowns maintained by shade-tolerant *Abies.* By virtue of hydration and foliar chemistry, some species are less flammable than others (e.g. Weatherspoon and Skinner, 1995). Improved understanding of interactions among plant species, fire behavior, and climatic context are key research questions.

Threats to the biodiversity of mixed-severity ecosystems are exacerbated by various factors that homogenize landscapes, reducing beta diversity and potentially synchronizing fires and insect outbreaks. The spread of fuel ladders out from topographically-protected areas sets the stage for fires to burn with more uniform high severity than had probably been the case in the past (Schoennagel et al., 2004). A possible example of that effect is the B & B fire in central Oregon, where dry and moist mixed conifer burned with similar severity. Increased evapotranspiration by densely stocked stands reduces water available to streams, which in turn likely effects fuel moisture in riparian zones and could shift riparian areas from fire barriers to fire corridors.[5] Large wildfires homogenize the landscape, and uniform fuels reduction potentially does so as well. While mixed severity forest types contained dense patches of young trees in the past and experienced some high severity fire, both the abundance of young trees and the likelihood of large, high severity fires have increased. The predominance of densely stocked plantations in some areas has in all likelihood altered the landscape disturbance dynamic, although exactly how is unclear.

Even where fires are patchy, the increased predominance of late successional tree species has altered the composition of the seed rain and, depending on survival probabilities, may shift species composition of the regenerating forest from a dominance by early-successional to a dominance by late successional tree species. Fire suppression has resulted in fewer and smaller naturally-recovering, early successional gaps and patches being created or maintained (Skinner, 1995).

Fire risk is not the only issue. Drying and warming due to climate change is poised to dramatically alter the environments many forests now experience (Neilson et al., 2005, 2007; McKenny

[4] Multiple regression relating the critical wind speed for producing a crown fire (predicted by NEXUS) to crown bulk density, crown base height, stocking density of *Abies*, and stocking density of *Pinus* accounted for 97% of the variation among plots (plot variables were described by Perry et al. 2004). All variables correlated negatively with the critical wind speed (i.e. as they increased the threshold for crown fire decreased), and all except stocking density of *Pinus* entered the model with a probability of at least .005 (*Pinus* had a probability of .227). The difference between species was not related to differences in overall stocking density.

[5] Frank Lake, personal communication.

Fig. 7b. Young grand fir beneath old growth ponderosa pine. Deschutes National Forest, Oregon. Plots within old-growth forests in the mixed conifer zone of the Bend Ranger District show that 90 percent of trees older than 150 years are ponderosa pine, Douglas-fir, or western white pine, while 90 per cent of trees younger than 100 years are grand fir or lodgepole pine (Perry et al., 2004).

Fig. 7c. Young Douglas-fir beneath older ponderosa pine. Rogue River National Forest, Oregon.

et al., 2007; Brown, 2008; Marlon et al., 2009). For example, in Washington State, Littell et al. (2010) concluded that "climate will be inconsistent with the establishment of Douglas-fir, ponderosa pine, and lodgepole pine in many areas by the middle of the twenty-first century". Models show that northeastern British Columbia, currently occupied by boreal forest, will have a climate more suitable for ponderosa pine by 2080 (Hamann and Wang, 2006). In all likelihood fire behavior will be altered along with climate (Bachelet et al., 2007; Marlon et al., 2009; Littell et al., 2010); in fact, an analysis of Canadian fires since 1970 shows that climate warming already is producing increased fire activity (Gillett et al., 2004). One implication is that the characteristic boundaries between high-, mixed-, and low severity fire regimes will shift.

All trees in densely-stocked stands are threatened by drought, which has been and is predicted to continue increasing in the western US (Dai 2010).A recent study found that 72 percent of gauging stations in the Pacific Northwest experienced significant declines in 25th percentile flows between 1948 and 2006, i.e. dry years are getting drier (Luce and Holden, 2009). A significant increase in non-fire mortality of old growth trees over the past several decades is probably due to increasing water deficits (Guarin and

Taylor, 2005; van Mantgem et al., 2009 Overstocking undoubtedly contributes significantly to this problem. On Blacks Mountain Experimental Forest in Northern California, an average of 63 percent of large trees (>60 cm DBH) were rated at high risk to mortality in unthinned plots, compared to 16 per cent in heavily thinned plots (Ritchie et al., 2007).

It is not only trees that are threatened by overstocking. Densely stocked stands alter the light environment and significantly impact understory diversity and cover. On the Stanislaus National Forest (Northern California), studies on permanent plots showed that between logging in 1929 and remeasurement in 2008, the cover of understory shrubs went from 28.6% to 2.5%, with *Ceanothus* and *Arctostaphylos* almost completely dropping out (Eric Knapp personal communication). The number of stems of herbaceous species dropped from an average of 4.0 to 0.9 m^{-2}, but there was great deal of variability around those averages. In 2008, tree density was approximately double the historic norm on these sites. Historically, such high densities may well have occurred as patches in mixed severity regimes, but not uniformly across the landscape.

Replacement of early successional shrub-hardwood communities by closed forests in the absence of fire significantly impacts landscape diversity. Shatford et al. (2007) recorded 47 species of shrubs and hardwoods in early successional communities of the Klamaths. Fontaine et al. (2009) found that broad-leaved hardwoods and shrubs played a major role in structuring bird communities in the Klamaths, and concluded that "extended periods of early seral broadleaf dominance and short-interval high severity fires may be important to the conservation of avian biodiversity". Not surprisingly, however, they also found that closed forests and early-successional communities were characterized by different avian guilds. As we pointed out earlier, it is the diversity of successional stages across a landscape that creates the high species richness typical of mixed-severity types, not any one particular community type.

4. Research needs

(1) More needs to be known about the biodiversity costs associated with loss of open canopied forest or early successional patches, and the landscape mix that best accommodates diverse habitat needs.
(2) Better estimates are needed of the relation between stand density/ species mix and water use, and particularly the degree to which older trees are water stressed by the presence of younger trees within a stand.
(3) Topographically-related patterns of burning vary significantly from north to south across the region. Although history is no longer necessarily a reliable guide to the future (Millar et al., 2007), it seems likely that historic topographic relations to fire will be preserved in a warming climate. Because they provide insights into landscape strategies, more needs to be known about historic patterns in specific locales.
(4) The ecological functions of hardwoods in the Klamath region should be clarified, especially with regard to their role in fire (their importance as both habitat and rapid-response soil-stabilizers seems clear). While there is considerable evidence that hardwoods reduce fire severity when intermixed with conifers, results from the Biscuit fire appear not to fit that pattern. Why the difference?
(5) Regional variation in mixed severity fire regimes and their ecological effects needs to be characterized more systematically and in greater ecological and spatial detail. We have highlighted some of the major differences in this paper but further work is needed to create a solid foundation for conservation and restoration. For example, we do not know how to spatially stratify fire regimes and restoration needs at subregional-landscape scales based on topography, potential vegetation, and current vegetation.

5. Summary

1. The defining element of mixed severity regimes is spatial and temporal variability in fire effects and ecological responses. Regimes vary regionally in the mean proportion of high severity patches and the frequency with which fires occur. However, some aspects of spatial patterning seem relatively constant, particularly the occurrence of a Pareto-type distribution of many small and few large high severity patches.
2. Consistent with the intermediate disturbance hypothesis, the diversity of patch types typical of mixed-severity regimes results in high levels of beta diversity in both plants and animals.
3. Wildfire behavior in mixed severity types is influenced by both top-down and bottom-up factors. From the top-down, regional climate, shaped by large scale atmospheric processes and modified by large scale geomorphic features, plays a key role. From the bottom up, local topography, vegetation type, and disturbance history influence burning patterns. Vegetation types (species composition and structure) that are either relatively vulnerable or resistant to stand-replacing fire can result in a self-reinforcing dynamic and consequent partial decoupling from the top-down effects of climate.
4. A combination of logging older forests and fire suppression has produced landscapes with many more young conifers than was likely to have been true prior to the 20th century.
5. Two or more canopy layers do not always mean higher risk for crown fire. It depends on region and species composition. Hardwoods respond differently to fire than conifers. In many cases they can help to reduce the intensity of a fire. In other cases, when they occur as dense young vegetation, they can help carry fire into crowns of adjacent conifers under extreme weather.

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