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Comprehensive Fuels Treatment Practices Guide for Mixed Conifer Forests: California, Central and Southern Rockies, and the Southwest

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A SUMMARY OF
KNOWLEDGE FROM THE



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The Forest Guild practices and promotes ecologically, economically, and socially responsible forestry—“excellent forestry”—as a means of sustaining the integrity of forest ecosystems and the human communities dependent upon them.

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Executive Summary

The goal of this guide is to provide a resource for managers of mixed conifer forests of the Southwestern plateaus and uplands, the Central and Southern Rocky Mountains, the Sierra Nevada, and the Transverse and Peninsular Ranges in Southern California. Mixed conifer forests have different species, structures, and spatial patterns in these regions but, in general, we focus on forests with a mix of ponderosa or Jeffrey pine, Douglas-fir, true firs, and aspen. The guide includes a comprehensive review of historic conditions, past land use, natural fire regimes, impacts of altered fire regimes, and future prospects, given climate change, for mixed conifer forests. The second half of the guide addresses fuels treatment objectives, techniques, barriers, and successes across a range of ownerships.

Before Euro-American settlement of the West, fires in mixed conifer forests burned on intervals that averaged between eight and 25 years for the Sierra Nevada, Southern Rockies, and Southwestern mixed conifer. Low-severity fires were more frequent in some mixed conifer forests; but, in general, mixed conifer forests have historically tended to be heterogeneous mixtures in which species composition, forest structure, and fuel loads change over short distances. Since Euro-American settlement, many mixed conifer forests have become more homogeneous and can therefore facilitate larger, higher-severity fires than those that occurred historically. Increasing heterogeneity in mixed conifer forests at the landscape scale to approximate historic conditions is important for achieving many management objectives, from fuel reduction to wildlife habitat. Restoration and wildfire hazard reduction are not synonymous, but restoration treatments can reduce the risk of uncharacteristic high-severity fire, i.e., stand-replacing fire covering a large portion of the landscape.

This report discusses prescribed fire, silvicultural treatments, and combinations of cutting and burning. In most mixed conifer forests, thinning that treats both the canopy and understory (crown and low thinnings) combined with prescribed fire is the most effective way to reduce wildfire hazard. However, land management objectives or external constraints can make other tools, such as mastication or prescribed fire alone, more appropriate. Treatments must be maintained for their fuel reduction effect to be sustained, and no single treatment will reverse a long history of fire exclusion. After about ten years, fuels begin building up towards pretreatment levels in many mixed conifer forests.

Interviews with 75 managers and experts helped identify numerous complications and barriers to implementing fuels treatments in mixed conifer forests. Smoke management and wildlife habitat protections are two common issues that can make these treatments more complicated, though not impossible. This report also discusses institutional challenges, such as the loss of local expertise and experience with fire that occurs with retirement. Another institutional challenge to returning natural mixed-severity fire regimes that include patches of high-severity fire to mixed conifer landscapes is the need to build confidence within an organization. Organizations and the public can be wary of prescriptions that include patches of high-severity fire, but landscape-level treatments that reduce wildfire hazard and increase the ability to control fires help build confidence that prescribed mixed-severity fires can be implemented safely.

Another consistent challenge to implementing fuels treatment is funding. Organizations that enjoy community support and strong partnerships through collaboration have allies in the battle for scarce resources and a strong case for grant funding. Though collaboration requires an investment of time and money, it can help avoid even more costly litigation or obstruction. Collaboration helps managers identify objectives that meet broad stakeholder social, economic, and ecological goals. While research questions and management challenges remain, this report documents both the extensive scientific knowledge and the practical management insights that already exist about fuels treatment in mixed conifer forests.



Stephens Lab University of California Berkeley

Prescribed burn at Blodgett Forest, California

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Section I: Introduction

In many western North American forests, a combination of human influences, including fire suppression, grazing, timber harvesting, and habitat fragmentation by roads and cities, have reduced the frequency of fire. For some ecosystems, these changes have permitted ingrowth of many small trees, large accumulations of dead woody material, and increased homogeneity of forest structure at the landscape scale, escalating the threat of uncharacteristic high-severity fires. At the same time, the number of people living in or near the forest has increased dramatically. The colliding trends of increased fire threat and more people at risk create a strong motivation for fuel reduction treatments.

This guide focuses on mixed conifer forests of the Southwestern plateaus and uplands, the Central and Southern Rocky Mountains, the Sierra Nevada, and the Transverse and Peninsular Ranges in Southern California. Section I defines mixed conifer forests in each of these areas. In discussing commonalities across these areas, we refer to these forests as “mixed conifer forests.” Section II deals extensively with fire regimes in mixed conifer forests. Fire regimes in mixed conifer forests are more varied in frequency and severity than in ponderosa pine or longleaf pine (refer to Appendix A for scientific names of species listed in the text). In most mixed conifer forests, a familiar set of influences has reduced the frequency of fire: reduced anthropogenic burning, effective fire suppression, forest management, and reduced forest connectivity because of roads and cities. Mixed conifer forests tend to be denser than they had been under historic fire regimes, and the proportion of shade-tolerant species such as white fir has increased dramatically, with species such as ponderosa pine and aspen declining in dominance. Altered fire regimes and forest densification compound the impact of other stresses, such as air pollution, insects, and disease. The impact of an altered fire regime varies by site characteristics and is described in detail in Section II.



Prescribed burn at Blodgett Forest, California

Stephens Lab University of California Berkeley

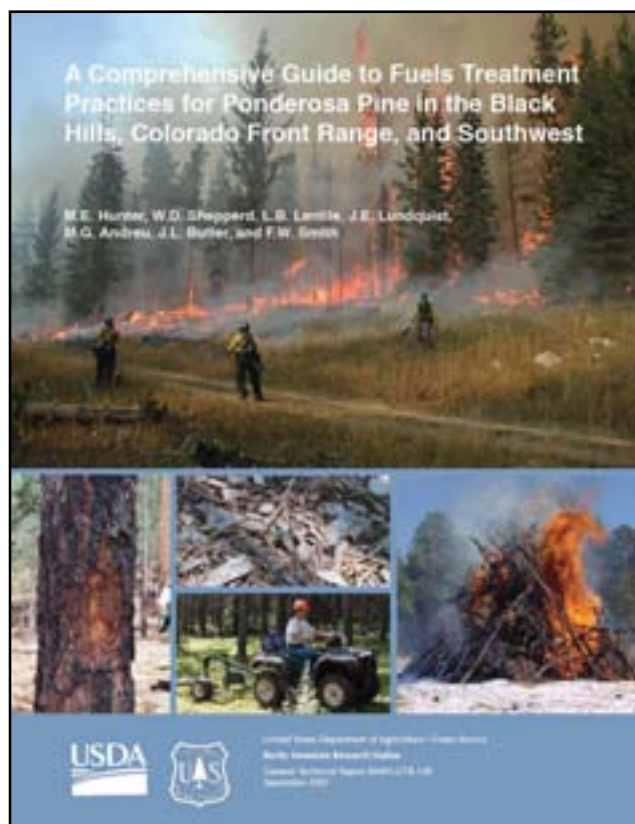
In response to the negative aspects of altered fire regimes in mixed conifer forests, managers are increasing the implementation of a wide range of fuels treatment practices. The objectives for fuels treatment are as varied as the forests themselves, and treatments often combine multiple objectives. Section III discusses treatment objectives such as wildfire hazard reduction,

ecological restoration, and commercial revenue. Objectives are driven entirely by human needs and desires and so differ by landowner and land use. Since a wide range of managers and others contributed to this guide, the fuels treatment practices they discuss and the lessons they share are driven by a wide range of land management objectives. Regardless of the objectives, fuels treatment practices focus on reducing tree densities, surface fuel loads, ladder fuels, and the continuity of tree crowns. Thinning and prescribed fire, the two main tools for changing forest structures, are often used in combination. These as well as other practices, such as mastication, are explained in Section IV.

Section V addresses both the effectiveness of different fuels treatment techniques and the integration of wildlife and forest health issues into those treatments. In most mixed conifer forests, thinning that treats both the canopy and understory (crown and low thinnings) combined with prescribed fire is most effective at reducing wildfire hazard. Other treatments can also effectively change fire behavior, and maintenance of treatments is crucial to sustain those changes. Managers must consider the impacts of treatments beyond their effect on fuels. Mixed conifer forests are home to threatened and endangered species, such as spotted owls, that require particular attention. Similarly, both native and exotic insects and diseases can influence fuels treatment planning.

Section VI covers the impacts of fuels treatment, monitoring, and mitigation. Monitoring is crucial for both identifying undesirable impacts and documenting effective treatments. Equally important are mitigation techniques for the undesirable impacts of fuels treatment. The final section, Section VII, provides an integration of management principles for fuels treatment in mixed conifer forests.

The writing of this guide was initiated by the Joint Fire Science Program (JFSP) to synthesize existing information in a form that is useful to land managers. This guide, which focuses on mixed conifer forests, was conceived and written to complement *A Comprehensive Guide to Fuels Treatment Practices for Ponderosa Pine in the Black Hills, Colorado Front Range, and Southwest* (Hunter et al. 2007), *Synthesis of Knowledge of Hazardous Fuels Management in Loblolly Pine Forests* (Marshall et al. 2008), and *A Comprehensive Guide to Fuels Treatment Practices for Mixed Conifer in the Northern Rocky Mountains* (Battaglia et al. *In preparation*). To provide consistency across guides, we have used essentially the same format and organization as the other guides. Like the other JFSP fuels treatment guides, we have combined an exhaustive



review of published scientific literature on mixed conifer forests with interviews with managers, using a team of researchers from the Forest Guild, the University of California Berkeley, and the U.S. Forest Service.

A central goal for this guide was to collect and synthesize the existing peer-reviewed literature on mixed conifer forests. To that end, we have attempted to create a comprehensive reference list that can serve as a resource for those seeking more detailed information on a particular topic. Equally important is the information gathered from dozens of interviews with managers from all the different mixed conifer forests in this guide. Our interviews included a wide range of managers and researchers from federal land management agencies, Native American tribes, state forestry agencies, universities, private industry, and nongovernmental organizations. While the managers we spoke with did not agree on every aspect of mixed conifer management, in many cases a consensus emerged. Appendix C provides a full list of the people with whom we spoke. Their experience, insights, questions, and recommendations, in combination with published science, informed our recommendations for management practices for restoration and fuels treatment in mixed conifer forests.

Defining Mixed Conifer

The first challenge in describing fuels treatment prescriptions and techniques for mixed conifer forests is that “mixed conifer” is difficult to define. The term “mixed conifer” is used for forests along a broad continuum of climatic zones and includes many different assemblages of species (Dieterich 1983). Unlike forests dominated by a single species, mixed conifer forests have different constituents, which in turn create varying structures and spatial patterns. While forests throughout the western U.S. are labeled mixed conifer, this synthesis focuses on California, Arizona, Utah, New Mexico, Colorado, and Wyoming. Therefore, our definition of mixed conifer is tied to specific areas: Southwestern plateaus and uplands, the Central and Southern Rocky Mountains, the eastern and western Sierra Nevada, and the Transverse and Peninsular Ranges in Southern California. In these areas we focus on mixed conifer forests that include ponderosa and Jeffrey pine; however, we will not include ponderosa pine stands that are too hot or too dry to support mixed conifer forest.

Many managers break mixed conifer into more specific subtypes. All forest type delineations are human-imposed breaks in an ecological continuum. However, for management purposes it is very useful to have relatively homogenous areas where a prescription can be implemented. Mixed conifer forests cover a spectrum of site conditions, from warm, dry ponderosa pine forests to wet, cold spruce-fir forests. Many managers break this continuum into a warm–dry mixed conifer type and a cool–moist mixed conifer type, as described in Table 1.

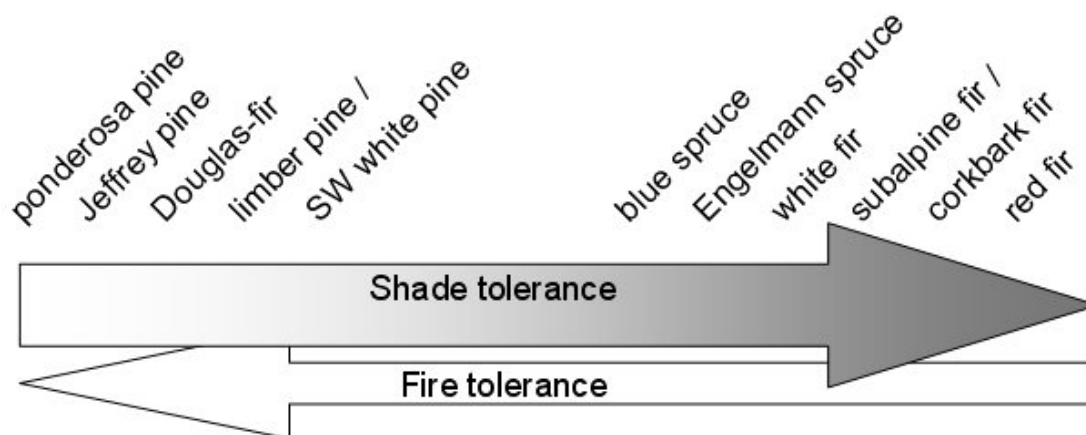
Table 1 Subdivisions of Mixed Conifer Forests (Smith et al. 2008)

Forest Type	Fire Regime	Early Seral Species	Late Seral Species
Warm–Dry Mixed Conifer	Relatively frequent/low to moderate intensity	Ponderosa pine with subdominant aspen and/or oak	Ponderosa pine with subdominant Douglas-fir, white pine, or limber pine
Cool–Moist Mixed Conifer	Relatively infrequent/intensity variable from low to high	Aspen or Douglas-fir	White fir and blue spruce

Some managers find more specific plant association delineations useful for project implementation and predicting treatment effects. For example, in the mountains of southern Arizona mixed conifer can be broken down into habitat types from the Douglas-fir and white fir series (Muldavin et al. 1996). The *U.S. Forest Service Southwestern Region Forest Plant Association Guide* (USDA Forest Service 1997) provides habitat descriptions for Arizona and New Mexico based in part on the work of Moir and Ludwig (1979) and Alexander and colleagues (1984). Other authors have published similar habitat types for the Sierra Nevada (Fites 1993) and the Central Rocky Mountains (Hoffman and Alexander 1980, 1983). Because some of these habitat guides are out of print and difficult to access, new internet-based ecological habitat descriptions, such as NatureServe Explorer (www.natureserve.org/explorer/) (Jennings et al. 2009) or the Fire Effects Information System (www.fs.fed.us/database/feis/), are useful for managers. There are similar databases at the state level, such as the California Natural Diversity Database (www.dfg.ca.gov/biogeodata/cnddb) and the California Wildlife Habitat Relationships (www.dfg.ca.gov/biogeodata/cwhr/). Other tools, such as LANDFIRE (www.landfire.gov), provide geographic data about fire regime (Rollins and Frame 2006). LANDFIRE biophysical settings descriptions also provide estimates of the distribution of succession classes (i.e., stand development stages) within each forest type. For example, the LANDFIRE description of “Southern Rocky Mountain Mesic Montane Mixed Conifer Forest” estimates that historically the early development stage covered 10 percent of the area, the mid development stage covered 60 percent, and the late development stage 30 percent. This distribution of development stages is another example of the heterogeneity of pre-settlement mixed conifer forests.

Tree species found in mixed conifer forests exhibit a wide range of tolerance to shade and low-severity fire; these traits are often related. Those species adapted to establish and grow in low light conditions below other trees often have thin bark and are easily killed by fire. Though species can often establish and grow in a range of conditions, Figure 1 provides a heuristic view of their relative tolerance for shade and fire (Burns and Honkala 1990). Aspen is hard to place on this continuum, since it is intolerant to shade and regenerates well after fire, but it is also susceptible to fire.

Figure 1 Relative shade and fire tolerance of common tree species in mixed conifer forests (Burns and Honkala 1990)



Southwestern Plateaus and Uplands

The mixed conifer forests of Arizona and New Mexico grow at elevations of 8,000 to 10,000 feet and at lower elevations on north-facing slopes and in canyons (Ronco et al. 1983, Dick-Peddie 1993). Generally, mixed conifer is situated between ponderosa pine forests and spruce-fir forests on an elevational gradient. Ponderosa pine forests are too hot and dry to support mixed conifer species, while spruce-fir forests occupy colder sites. Mixed conifer forests of the Southwestern plateaus and uplands are often broken up into warm-dry and cool-moist types (Table 1). The most common species in the warm-dry type include ponderosa pine, Douglas-fir, white fir, blue spruce, limber pine, southwestern white pine, Gambel oak, and occasionally aspen, while the cool-moist type can also include Engelmann spruce, subalpine fir, and corkbark fir (Jones 1974). The warm-dry type tends to be on lower elevations or south-facing slopes and is more open than the cool-moist type. Historically, the warm-dry type experienced low- to moderate-intensity fires frequently. In the cool-moist type, fires were less frequent but generally of a higher intensity and severity. Section III discusses both historic fire regime as well as changes to the fire regime and current conditions. Mixed conifer forests cover large areas of the Sacramento Mountains, White Mountains, Mogollon Rim, Chuska Mountains, Kaibab Plateau, and the sky island forests of the Sierra Madre Occidental.



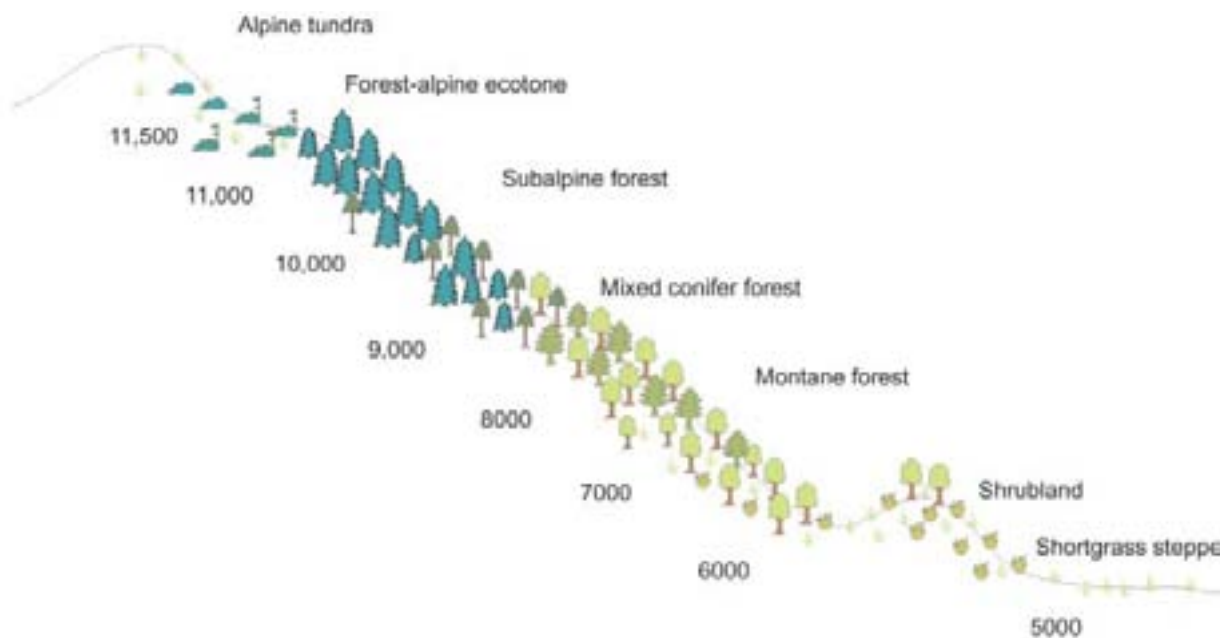
Alexander Evans

San Juan National Forest, Colorado

Central and Southern Rocky Mountains

Mixed conifer forests of the Central and Southern Rocky Mountains include Douglas-fir, ponderosa pine, and true firs as common components. Blue spruce, aspen, lodgepole pine, and Rocky Mountain juniper can be found in some stands (Kaufmann et al. 2000). The composition varies significantly with aspect: cool–moist types are found on the north-facing aspects while the warm–dry type is more prevalent on the south-facing aspects (Romme et al. 2009). In an example from central Colorado, north-facing stands had twice as many Douglas-fir trees than ponderosa pine, while ponderosa pine dominated in the south-facing stands (Kaufmann et al. 2000). Mixed conifer stands in the Front Range include lodgepole pine and limber pine above 8,000 feet (Figure 2) (Huckaby et al. 2003). Mixed conifer forests grade into other forest types where a persistent snowpack starts, between 9,000 and 10,000 feet (Romme et al. 2009). Stands in the Central and Southern Rocky Mountains have been shaped by insects, such as the 1980s western spruce budworm outbreak that killed a large number of Douglas-fir trees (Kaufmann et al. 2000). Insect dynamics and fire regimes for this region are addressed in Section III. Mixed conifer forests are found in the Sangre de Cristo Mountains, Jemez Mountains, San Juan Mountains, Sawatch Range, and the Front Range.

Figure 2 Vegetation communities on an elevational gradient in the Front Range, Colorado. Elevations shown in feet (Huckaby and et al. 2003).



Sierra Nevada

Descriptions of mixed conifer forests in California have varied historically (Sawyer and Keeler-Wolf 1995). The forest type is also referred to as “lower montane forest” and “upper montane forest” (van Wagtenonk and Shaffer 2006). The phrase “mixed conifer” generally describes montane Sierra Nevada and Southern California forests with as many as five different conifer species, but may have as few as two of these species as canopy codominants. Mixed conifer stands are usually characterized by a combination of ponderosa pine, sugar pine, white fir,

incense cedar, and Douglas-fir. Ponderosa pine and incense cedar dominate in warmer sites (lower elevation and south aspect) while white fir dominates on cooler and wetter sites (higher elevation and north aspect) (Helms 1980). Canyon live oak, black oak, giant sequoia, and Jeffrey pine are common species in Sierra Nevada mixed conifer forests, while lodgepole pine and aspen occur less frequently, often in areas with high water tables or in cold air drainages (Gill and Taylor 2009). Sugar pine is indicative of mesic, high-quality sites, Jeffrey pine of upper elevations and serpentine soils, and California red fir of the highest elevations within the forest type (Helms 1980). Reduced precipitation on the eastern slopes of the Sierra Nevada means the mixed conifer belt is higher in elevation. Mixed conifer forests of the Sierra Nevada cover large areas, from the Klamath Ranges in the north to Kern County in the south, and can be as low as 3,000 or over 7,000 feet in elevation (Helms 1980).

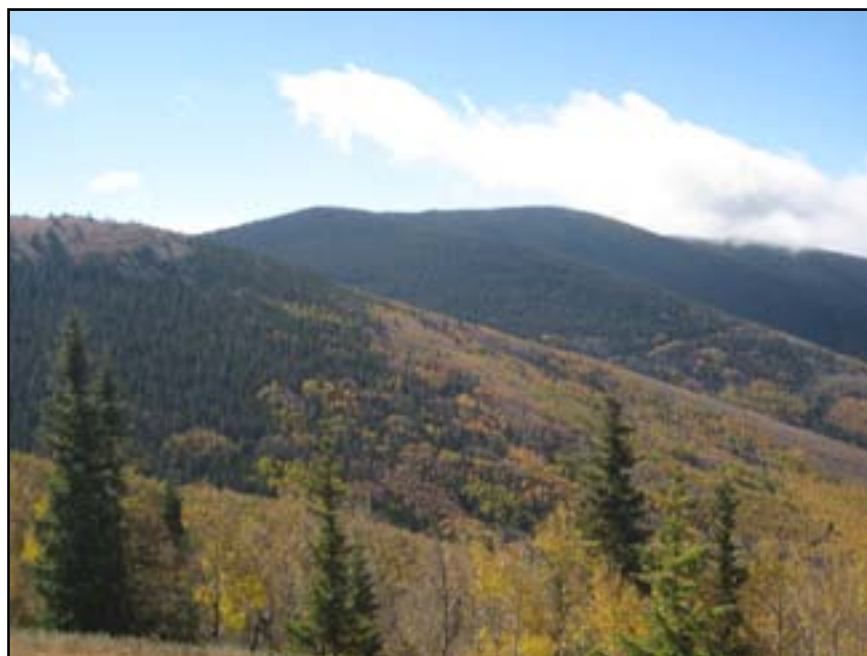
Transverse and Peninsular Ranges in Southern California

In Southern California, the mixed conifer is common in mountains of the Transverse and Peninsular Ranges and portions of northern Baja California. The Transverse Ranges run east-west and include the San Gabriel and San Bernardino Mountains. The Peninsular Ranges include the Santa Ana Mountains, San Jacinto Mountains, Mount Palomar, the Laguna Mountains, Sierra Juárez, and Sierra San Pedro Mártir (SSPM) and run southeast-northwest (Minnich 1983). Mixed conifer forests in these mountain ranges can be found on elevations between 4,500 and 8,500 feet (Minnich et al. 1995). Ponderosa pine, Jeffrey pine, white fir, incense-cedar, and bigcone Douglas-fir are common, but other species, such as western juniper, also occur. Bigcone Douglas-fir may be present in this southerly stand mix instead of Douglas-fir, and Jeffrey pine will take a more dominant role in these ranges on dry, rocky soils (Thorne 1977). Jeffrey pine is extensive on higher elevations in the eastern half of the San Bernardino mountains, particularly on southern aspects and in basins (Minnich et al. 1995).

Heterogeneity and Spatial Scale

In mixed conifer forests, habitat types are intermingled in relatively small areas, such as opposing aspects of the same hillside. At the landscape scale, warm-dry and cool-moist mixed conifer types intermingle to present a mosaic of structures. One of the important changes since Euro-American settlement in mixed conifer forests has been the increased homogeneity of structure at the landscape scale (as is discussed in detail in Section II). More homogeneous mixed conifer forests can facilitate larger, high-severity fires (Romme et al. 2003, Miller et al. 2009). The National Forest plan revisions in the Southwest will consider three scales: fine, mid, and landscape. The fine scale addresses the distribution of individual trees within a stand, i.e., single, grouped, or aggregates of groups. Mid-scale is a unit of 100 to 1,000 acres and has relatively homogeneous biophysical conditions. Landscape is an assemblage of mid-scale units, typically composed of variable elevations, slopes, aspects, soils, plant associations, and disturbance processes. Heterogeneity is important at each of these scales. For example, in Southwestern warm-dry mixed conifer forests there is heterogeneity at fine scale, where trees historically grew in irregularly shaped groups surrounded by openings. In Sierran mixed conifer forests, varying tree density according to potential fire severity effects on stand structure creates heterogeneity within stands (North et al. 2009a).

Fuel reduction treatments are implemented at the stand scale, but must fit within a landscape plan. The landscape context of these treatments is crucial to their success in modifying wildfire behavior (Schmidt et al. 2008, Moghaddas et al. 2010, Collins et al. *In press*). For example, in one modeled landscape, strategically placed fuel treatments (SPLATs) on 10 percent of the landscape resulted in major reductions in the impacts of wildfire (Ager et al. 2010). The term landscape scale can be ambiguous; generally, it has come to mean an area of at least 50,000 acres (Finney et al. 2007, Omnibus Public Land Management Act of 2009, Moghaddas et al. 2010). Landscape planning is particularly challenging where ownership boundaries split forests (Collins and Stephens 2010). Section III discusses strategies for fuels treatment planning across jurisdictions. Not only do forest and fuels conditions and resources vary between ownerships, but fuels treatment priorities can differ in scale, intensity, and urgency. Limited resources force managers to prioritize treatments based on land management and objectives. For example, wildland-urban-interface zones (WUIs) and wilderness areas are managed with very different objectives; hence, fuels treatment priorities and practices will be different as well.



Alexander Evans

Santa Fe National Forest, New Mexico

Section II: Fire and Fuels Issues

Current conditions in mixed conifer forests are a product of environmental conditions, human use and management of the land, and fire. This section is designed to provide managers a brief background on the influences that have helped create the conditions in which they implement fuels treatments. Also, past conditions provide a baseline forest reference for treatments which aim to restore healthy forest structures and processes.

Past Land Use and Management Activities

Southwestern Plateaus and Uplands

Native peoples have made their homes in the Southwest for more than 12,000 years; during the last 1,000 years their numbers have been large (Allen 2002). Native American populations fluctuated over space and time, as did their impact on forests (Dahms and Geils 1997). Native Americans used mixed conifer forests for resource extraction such as hunting or fuel wood removal, but because most settlements were at lower elevations fire ignition was likely their largest impact. Native Americans burned forests in the Southwest to achieve a range of objectives, such as hunting, crop management, increased plant yield, pest management, fire hazard reduction, and warfare (Cooper 1960, Allen 2002, Stewart et al. 2002). For example, Apaches may have changed fire frequencies and seasonality of fire through their burning in the Chiricahua Mountains, Organ Mountains, and Sacramento Mountains during some periods (Swetnam and Baisan 2003). The other source of fire ignition in the Southwest was lightning, which is very common in the mixed conifer forests of the Southwestern plateaus and uplands (Barrows 1978). For example, in the southern Arizona mountains, lightning can ignite more than five fires per square mile per year (Swetnam and Baisan 2003). Although Native Americans may have altered mixed conifer forest structure in a few places for some periods, at the landscape scale their impact on ecological process was likely small (Allen 2002), although research is not conclusive on this point (Kay 2007). Regional climate drivers such as El Niño-Southern Oscillation (ENSO; El Niño and La Niña are extreme phases of this system) also determined fire occurrence in mixed conifer forests before Euro-American settlement (Brown et al. 2001).



Mark S. Moak

Lightning storm, Bitterroot National Forest, Idaho

The structure and species composition in mixed conifer forests changed greatly during the 20th century. As is common across western forests, many mixed conifer stands are denser than they were before active fire suppression efforts (Dahms and Geils 1997). During the late 1800s the population of Euro-American settlers increased dramatically. Their land management practices, which included logging, fire suppression, road building, and livestock grazing, have changed mixed conifer forests (Cooper 1960, Covington and Moore 1994b, Lynch et al. 2000). Millions of sheep grazed in Arizona and New Mexico in the late 1800s, and they reduced herbaceous cover in many areas (Cooper 1960, Savage and Swetnam 1990, Allen 2002). While grazing was less important in mixed conifer forests, reduction of fuels and surface fire in ponderosa pine forests would have reduced fires in the mixed conifer forests upslope. Though it is still important in some areas, sheep grazing has declined significantly since the late 1800s on a regional level. Cattle grazing remains an important land use. For instance, about 375,000 cattle were grazed in New Mexico in 2007 (Dahms and Geils 1997, USDA 2007).

Along with grazing, Euro-American settlers brought commercial logging to the Southwest. Commercial logging increased as railroads improved access and demand. For example, large-scale timber harvest in northern Arizona expanded when the transcontinental railroad reached the area in 1882 (Fulé et al. 1997). Establishment of a railroad infrastructure allowed for increasingly intense timber harvesting systems. By the later half of the 20th century, forest sales from U.S. Forest Service land was about 300 million board feet per year, of which 80 percent was saw timber (Johnson 1994). Timber harvests from both public and private forests in Arizona and New Mexico peaked in about 1990, with roughly 433 million board feet per year (Covington 2003). After this peak, harvests declined dramatically due to threatened and endangered species habitat mitigation, appeals and litigation of federal timber sales, and declining federal budgets (Morgan



Library of Congress

Rio Grande national forest, Colorado, sometime between 1935 and 1945

et al. 2006). The Mexican spotted owl was listed as threatened in 1993, prompting a federal judge to stop new timber sales on national forests in Arizona and New Mexico in 1995. Harvests on national forests in Arizona and New Mexico dropped from about 425 million board feet in 1990 to 48 million board feet in 1996 (Morgan et al. 2006). In 1996 the timber harvesting injunction was lifted and by 1998 national forest timber sales from Arizona and New Mexico were at approximately 93 million board feet (Morgan et al. 2006).

In the early 1900s, the U.S. Forest Service developed a primary mission of suppressing forest fires, and became very successful over the following decades in reducing the acres burned in the Southwest, including in mixed conifer forests. An increasing human population expanded the road and trail network, which broke the continuity of fuels and kept fires from spreading (Covington and Moore 1994a, Reed et al. 1996). The road network has also improved access for firefighting (Dahms and Geils 1997). At the same time, increasing populations mean that many more people and structures are at risk from fire in the wildland interface (Spyratos et al. 2007).

Central and Southern Rocky Mountains

The Rocky Mountains have a long history of human habitation, though populations were lower than in the Southwestern plateaus and uplands. Tribes such as the Ute, Arapaho, and Jicarilla Apache likely frequented the mixed conifer forests of the Rocky Mountains, though their settlements were probably lower in elevation (Riebsame et al. 1996, Baker 2002). Native Americans in the Central and Southern Rocky Mountains were not the prime drivers of fire in mixed conifer forests, because of relatively low populations, frequent lightning strikes, and the importance of climate as a control on fire (Baker 2002, Grissino-Mayer et al. 2004). Though the Central and Southern Rocky Mountains are not a high lightning area, the number of fires ignited by lightning is still high (Orville 1994, Zajac and Rutledge 2001, Baker 2002, Stephens 2005). The relatively low number of lightning strikes in the Northern Rocky Mountains may have limited fires (Orville 1994, Zajac and Rutledge 2001, Baker 2002). Hence ignitions by Native Americans may have been more important in the northern Rocky Mountains.

In the Central and Southern Rocky Mountains, climate is the main driver of fire in mixed conifer forests. Years with warm, dry spring-summer periods are strongly associated with widespread fire (Bessie and Johnson 1995, Veblen et al. 2000). For example, a five-year drought combined with drier-than-average La Niña conditions



Ouray, Colorado, circa 1901

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were important in setting the stage for the 2002 Hayman Fire (Schoennagel et al. 2004). There is an association between strong ENSO events and widespread fire in the Central and Southern Rocky Mountains (Veblen et al. 2000).

Although climate has always been a primary driver of forest conditions, human influence increased after the mid-1800s as mining and the Homestead Act of 1872 prompted increased settlement in the region (Riebsame et al. 1996). Extensive sheep and cattle grazing reduced grass and forbs cover, a key element for ignitions and spread of fire (Keane et al. 2002). Most pre-1900 logging in the region was for a limited, local market, but markets for products such as railroad ties grew (O'Rourke 1980). In some forests, logging in the 1800s removed most of the larger trees (Kaufmann et al. 2000, Romme et al. 2009). Logging was more intense in the more accessible low-elevation forests (Romme et al. 2003). In the late 1800s, mixed conifer forests were extensively logged and burned, which caused synchronized regeneration across large areas (Romme et al. 2003). The Weeks Act of 1911 signaled the start of coordinated fire suppression, and by the 1930s thousands of people were employed in a well-organized effort to put out fires in the Rocky Mountains (Keane et al. 2002). Timber production in the region increased through the 1980s before declining in 1990s and 2000s. For example, Wyoming timber production was less than 100 million board feet until the mid 1950s and peaked in 1987 at nearly 300 million board feet (Morgan et al. 2005). As with other regions, the contribution of national forests to timber harvest yields has decreased. In Colorado, national forests provided 90 percent of the harvested timber volume in the state for 1974 and in 2002 they provided only 38 percent (Morgan et al. 2006). Human development in the wildland continues to expand in the Central and Southern Rocky Mountains, though public ownership limits some expansion into mixed conifer forests (Riebsame et al. 1996). Roads and other development have reduced forest patch size (Reed et al. 1996, Romme et al. 2003).

Sierra Nevada

Fire has been an important driver of the structure and composition of the Sierra Nevada for thousands of years (McKelvey et al. 1996). The first humans began establishing at least temporary camps there about 10,000 years ago; the first permanent settlements date to about 3,200 years ago (Parker 2002). During that time, Native Americans burned, pruned, sowed, harvested, and tilled in the Sierra Nevada (Anderson and Moratto 1996, Anderson 2005). Tribes in the Sierra Nevada, such as the Washeo, Western Mono, Paiute, and Miwok, used fire to enhance food production, particularly that of acorns (Parker 2002, Anderson 2005). Native American burning likely had a strong influence on forest structure in localized areas where settlements were most dense, such as Yosemite or Hetch Hetchy Valley, but their influence probably decreased with elevation and on the eastern side of the range (Parker 2002).

The Sierra Nevada has few lightning strikes compared to the Rocky Mountains or the Southwest—fewer than two flashes per square mile per year (Orville 1994, Zajac and Rutledge 2001). However, weather still has an important influence on fire. Mixed conifer forests have a Mediterranean climate with a pronounced dry summer season, and rely on snowpack for moisture. ENSO weather patterns help determine the depth of snowpack and hence the water stress of mixed conifer forests (North et al. 2005). Sierra Nevada drought years are associated with increased wildfire activity and larger fires (Beaty and Taylor 2001, Westerling et al. 2003,

Gill and Taylor 2009). During years when climate conditions for fire were favorable, even a small number of lightning or human ignitions would have been sufficient to burn Sierra Nevada mixed conifer forests on a regular basis.



Hydraulic mining near French Corral, Nevada County, California, 1866

The 1849 gold rush was a watershed event in the human history of the Sierra Nevada. Hundreds of thousands of new people came to the Sierra Nevada to mine gold. By the late 1800s, miners switched from traditional placer mining methods to hard-rock and hydraulic engineering to extract deposits (Beesley 1996). Hydraulic mining altered stream channels, caused massive erosion, and motivated timber harvesting for construction (Beesley 1996). By 1880, 680 million cubic yards of material had been washed into California rivers along the western slope of the Sierra Nevada from hydraulic mining (Beesley 1996). At the same time, millions of board feet of timber were cut for the mining industry and for towns in the Central Valley where rail connections

existed (Beesley 1996). Most railroads and roads were in the more accessible areas of the northern Sierra Nevada, and the rail lines themselves consumed large quantities of wood (Beesley 1996). Grazing in the Sierra Nevada experienced a sharp increase because of a severe drought in the early 1860s, when sheep were brought to Sierran meadows to escape the dry Central Valley (Swetnam and Baisan 2003).

Population and development in the Sierra has increased dramatically since the early 1800s. The population of the Sierra Nevada doubled between 1860 and 1960 and doubled again between 1970 and 1990, to more than two million (Duane 1996). By 2000, there were more than 150 thousand houses in the WUI of the Sierra Nevada (Hammer et al. 2007). The population and WUI in the Sierra Nevada is likely to continue growing with the rest of the state, which had a population growth of 9 percent between 2000 and 2009 (U.S. Census Bureau 2010).

Transverse and Peninsular Ranges in Southern California

Before Euro-American settlement, the Native America population in Southern California was comparatively high, averaging two to eight people per square mile (Keeley 2002). Though most permanent settlements were located in valleys and coastal areas, Native Americans used the upper elevation forests extensively during certain seasons (McBride and Laven 1976, Minnich 1988). Many of the numerous tribes in the area burned forests and rangelands to improve hunting and food production (Keeley 2002). The influence of indigenous burning on mixed conifer forests is not certain (Minnich et al. 1995). The opportunity for natural ignition is relatively low because the number of lightning strikes in the Transverse and Peninsular Ranges is lower than in

other regions (Orville 1994, Zajac and Rutledge 2001), though the importance of lightning ignitions increased after 1800 in the southern part of this range (Evelt et al. 2007). However, extreme fire weather, particularly high winds, can ensure that fires spread over large areas (Keeley 2008). Evidence from ecologically similar mixed conifer forests to the south suggests that anthropogenic ignitions play an important role in maintaining a high frequency of fire (Evelt et al. 2007).

Early colonization and missionary efforts reduced burning by Native Americans (Evelt et al. 2007) and started a major change in land use (Minnich 1988). As more Euro-American settlers moved to the area, use of mixed conifer forest resources became more intense. All of the mixed conifer forests that were within the California region of this mountain range were logged before 1900, particularly those accessible from the Los Angeles area (Minnich 1988). In general,



U.S. Geological Survey Photographic Library

Holcomb Valley, San Bernardino Mountains, California, 1905

logging in the Transverse and Peninsular Ranges was much less intense than in the Sierra Nevada because lumber was imported from the Pacific Northwest and the Sierra Nevada (Minnich 2007). Grazing began in the region in the 1700s along the coast, spread inland, and by the mid 1800s stocking levels were high (Bartolome 1989). Both cattle and sheep were grazed in the forests of the Transverse and Peninsular Ranges (Minnich et al. 1995). Population in the region has been dramatic; by 2000 there were more than 2 million homes in the WUI of Southern California's mountains and valleys (Hammer et al. 2007), but very few houses in the Mexican portion of this range. The mixed conifer forests of the Transverse and Peninsular Ranges have become surrounded by high density populations. In fact, many of the surrounding counties have population densities of 300 people per square mile (U.S. Census Bureau 2010).

Fire Regimes and Historic Conditions

Historic forest conditions, and the fire regimes that maintained those conditions, provide a key reference point for fuel reduction treatment. A fire regime is defined by the frequency, extent, intensity, severity, and seasonality of fires within an ecosystem (Helms 1998). Frequency is the number of years between fires for a particular area and extent defines the area affected by the fire. Frequency is often reported in terms of the mean fire return interval (MFRI), i.e., the average time between fires under a given fire regime. Because the time between fires varies over time and space, the MFRI should be viewed as a snapshot. A MFRI drawn from a longer time and larger area of analysis is generally more robust. Therefore, the MFRI is more difficult to

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Angora Fire, Lake Tahoe, California

calculate for ecosystems with less frequent fires because a longer time series is needed to capture sufficient fires to estimate an average. Scientific papers also report a more conservative MFRI based on fires that are recorded by at least 25 percent of the trees in the study area to highlight fires that probably had a greater ecological effect on the stand (Swetnam and Baisan 2003). Most studies use dendrochronology, the analysis of annual growth rings of trees to date past events, to estimate a MFRI. Numerous sources discuss the techniques and challenges of using dendrochronology for estimating MFRI (e.g., Fritts and Swetnam 1989, Speer 2010, Stephens et al. 2010a).

Intensity is determined by the rate of heat released by the flaming front at a specific time (Helms 1998). Intensity is hard to measure because it must be recorded during the fire. In contrast, severity can be measured based on the impact of the fire on plants and soil. Severity is defined as the amount a fire alters a stand; it is the product of fire intensity, fuel consumption, and

residence time (Helms 1998). Severity is often linked to tree mortality in mixed conifer systems, with low-severity fires leaving most trees alive and high-severity burns killing most trees (Miller and Thode 2007).

Mixed conifer forests create two main challenges for researchers trying to estimate fire frequency: spatial heterogeneity and mixed severity. As discussed above, many mixed conifer forests change in composition and structure across small areas. These changes in structure and fuel loads combine with topographic variation to produce variation in fire effects. The importance of spatial heterogeneity and variations in fire severity are detailed for each region below. The following discussion of historic fire regimes focuses on the period before Euro-American immigration affected mixed conifer forests, in most cases before the mid 1800s.

Southwestern Plateaus and Uplands

In the Southwest it is useful to think of a continuum of conditions, from the warm-dry mixed conifer stands that experienced relatively frequent, low-severity fire regimes to the cool-moist mixed conifer stands had a mixed-severity fire regime (Romme et al. 2009). In the warm-dry type relatively frequent fires generally led to less fuel, less dense conditions, and small patches of high-severity fire. In the cool-moist type less frequent fire tended to result in greater fuel buildup, more dense stands, and larger patches of high-severity fire. This pattern led to large, homogeneous patches of mid- and late-seral forest structure.

Historically, fires in Southwestern mixed conifer forests, particularly the cool–moist type, were less frequent and more variable than those of ponderosa pine forests (Touchan et al. 1996). One reason for decreased fire frequency is that in periods of the year when low fuel moistures are conducive to fire spread, lightning occurs less often (Brown et al. 2001). For example, in the White Mountains, thunderstorms that bring lightning that could ignite fires tend to drop more rain on upper elevation mixed conifer stands than lower elevation ponderosa pine stands (Dieterich 1983). However, fires in lower elevation ponderosa pine forest sometimes spread through adjacent mixed conifer forests. For example, in one watershed, though only 24 percent of fires in ponderosa pine spread into mixed conifer forests, those fires made up the majority of fires in the mixed conifer stand (Margolis and Balmat 2009). In the Sacramento Mountains there is a higher fire frequency on the steeper west side than on the east side, which has a more gradual slope (Brown et al. 2001).

Spring fires are common in the Southwestern plateaus and uplands, as recorded in the historic record for the Sacramento Mountains (Geils et al. 1995, Brown et al. 2001, Margolis and Balmat 2009). Historically, spring fires were ignited by dry lightning or Native Americans and spread because of warm temperatures, low humidity, and persistent and gusty winds (Dieterich 1983). Fires also occurred in the late summer or early fall after the monsoon season thunderstorms had ended but temperatures were still high (Dieterich 1983).

Historic fuel loads in one mixed conifer forest was relatively high, at about 13 to 20 tons per acre (Dieterich 1983). Sampling from 16 stands across the Southwest (Table 2) provides a general picture of fuel loads, though they may be higher than pre-settlement fuel loads, since they were recorded in 1975 (Sackett 1979).

Table 2 Dead Fuel Loads from 16 Southwestern Mixed Conifer Stands (Sackett 1979)

Fuel Component	Tons per acre	Std Dev
Surface Fuel (less than 1 inch)	3.5	1.1
Fermentation Layer	5.0	1.8
Humified Layer	13.7	5.6
<i>Subtotal 0 to 1 inch</i>	<i>22.2</i>	<i>6.2</i>
1 to 3 inch woody material	3.3	1.3
Greater than 3 inches rotten material	10.3	7.6
Greater than 3 inches sound material	8.3	7.2
Total dead fuel	44.1	18.0

A more recent study found 12.7 tons per acre of coarse woody material (CWM) in Southwestern mixed conifer stands and noted the high degree of spatial variability (0.8 to 31.7 tons per acre) of CWM (Ganey and Vojta 2010).

In general, before Euro-American settlement, fire in mixed conifer forests of Southwestern plateaus and uplands varied from surface fires to patchy (or passive) crown fires across relatively small geographic areas (Fulé et al. 2003). Large stand-replacing fires were rare, in part because of the heterogeneous composition. Aspen groups would have reduced or extinguished some fires

burning through heterogeneous mixed conifer stands (Dieterich 1983). However large, stand-replacing fires may have occurred in response to multidecadal climate patterns. For example, evidence suggests that large fires in the Sacramento Mountains occurred after a severe drought about 650 years ago (Frechette and Meyer 2009). Recent data from Kendrick Mountain in northern Arizona provides another example of mixed conifer forests on steep slopes where stand-replacing fires occurred on centennial scales (Jenkins et al. 2011).

High-severity patches, as opposed to extensive areas of high-severity fire, have been relatively common. From 1984 to 2004 in the Gila National Forest, high-severity patches accounted for 20 percent of the area burned of all the fires in mixed conifer forests (Holden et al. 2009). The 1867 fire on Rincon Peak included a 150-acre patch of high-severity fire as well as areas of low-severity fire (Iniguez et al. 2009). In general, higher-severity fire is more likely at higher elevations, on steep north-facing slopes, and on locally wet, cool sites (Holden et al. 2009). For example, head fires on short upslope runs can create high-severity patches and cause mortality in larger trees (Dieterich 1983). At least some of the mixed conifer forests of the sky islands in southern Arizona and New Mexico

(such as in the Animas Mountains) had mixed fire regimes, including frequent surface fires and relatively long-interval, patchy crown fires (Swetnam et al. 2001, Iniguez et al. 2009). There is other evidence that high-severity fire in mixed conifer can leave tree densities similar to those of pre-settlement forests. The 1993 Northwest III Fire in Grand Canyon National Park had higher-than-anticipated severity, killed a significant number of trees, and reduced canopy cover and fuel loads to near pre-settlement levels (Fulé et al. 2004). Another line of evidence that suggests high-severity

patches were a natural part of mixed conifer fire regime comes from the endangered Mount Graham red squirrel, an endemic species of spruce-fir and mixed conifer forests. A mixed-severity burn (in which the majority of the area was high severity) showed no impact on reproductive condition, body mass, or survival of Mount Graham red squirrel populations (Leonard and Koprowski 2010). Table 3 shows the range of fire frequencies across the Southwestern plateaus and uplands.

The average of the estimated MFRI is 7.9 years, but the average of the more conservative measure of MFRI that includes only fires that scar at least 25 percent of trees is 14 years. Using the conservative MFRI, the sites labeled “PIPO/MC” (ponderosa pine-mixed conifer) had a mean of 13 years while the sites labeled just “mixed conifer” had a mean of 15 years between fires. It is also important to highlight the range of time between fires, which was up to nearly 80 years in one site and more than 30 years for almost half of the sites (Swetnam and Baisan 1996).



Eric La Price

Trimble Point prescribed burn on the San Juan National Forest

Table 3 Fire Frequency in Mixed Conifer Forests of the Southwestern Plateaus and Uplands

Site	State	Elev. (ft)	MFRI	Std. Dev.	Range	25% MFRI	Range	Std. Dev.	Citation	Description
Sacramento	NM	9,022	6.4	4.9	1 to 20				Brown et al. 2001	Westside MC
Sacramento	NM	8,940	4.1	2.6	1 to 13				Brown et al. 2001	Westside MC
Sacramento	NM	9,301	11.3	7.5	2 to 32				Brown et al. 2001	Eastside MC
Sacramento	NM	9,203	4.8	2.5	2 to 14				Brown et al. 2001	Eastside MC
Sacramento	NM	9,203	12	12.1	1 to 50				Brown et al. 2001	Eastside MC
Sacramento	NM	9,416	12.6	9.4	2 to 29				Brown et al. 2001	Eastside MC
Sacramento	NM	8,940	12.6	8.7	3 to 31				Brown et al. 2001	Eastside MC
Sacramento	NM	8,465	11.2	6.5	1 to 24				Brown et al. 2001	Eastside MC
Sacramento	NM	8,760	9.1	4.9	2 to 22				Brown et al. 2001	Eastside MC
Sacramento	NM	8,825	6.2	3.5	3 to 15				Brown et al. 2001	Eastside MC
Sacramento	NM	8,383	13.8	5.5	6 to 20				Brown et al. 2001	Eastside MC
Animas	NM	8,050	7.4	5.0	1 to 21	24.57	4 to 46	13.7	Swetnam and Baisan 1996	PIPO/MC
Chiricahua	AZ	6,110	8.3	7.3	1 to 33	17.08	9 to 50	11.9	Swetnam and Baisan 1996	PIPO/MC
Chiricahua	AZ	6,900	8.0	6.7	1 to 31	13.08	4 to 31	6.7	Swetnam and Baisan 1996	PIPO/MC
Chuska	AZ	8,850	3.9	4.3	1 to 23	9.30	4 to 16	6.0	Swetnam and Baisan 1996	PIPO/MC
Guadalupe	TX	7,874	5.1	3.9	1 to 15	27.40	2 to 13	22.0	Swetnam and Baisan 1996	PIPO/MC
Mogollon	NM	9,300	6.0	5.0	1 to 21	23.29	17 to 79	11.3	Swetnam and Baisan 1996	MC
Mogollon	NM	8,850	3.0	2.6	1 to 15	13.13	4 to 21	9.1	Swetnam and Baisan 1996	PIPO/MC
Mogollon	NM	8,100	12.3	9.4	1 to 31	8.38	1 to 15	6.0	Swetnam and Baisan 1996	PIPO/MC
Organ	NM	7,450	5.2	4.5	2 to 19	10.06	7 to 35	7.3	Swetnam and Baisan 1996	PIPO/MC
Organ	NM	8,000	2.9	2.4	1 to 15	7.77	4 to 31	5.7	Swetnam and Baisan 1996	PIPO/MC
Pinaleño	AZ	8,573	5.8	5.3	1 to 23	12.67	9 to 50	8.8	Swetnam and Baisan 1996	MC
Pinaleño	AZ	9,325	6.1	4.7	1 to 22	12.60	3 to 22	5.2	Swetnam and Baisan 1996	MC
Rincon	AZ	7,661	3.0	1.9	1 to 9	7.32	2 to 13	3.3	Swetnam and Baisan 1996	PIPO/MC
Santa Catalina	AZ	8,855	6.6	4.1	1 to 17	10.42	2 to 23	5.7	Swetnam and Baisan 1996	MC
Santa Catalina	AZ	7,300	5.5	2.9	1 to 15	7.33	2 to 16	3.8	Swetnam and Baisan 1996	PIPO/MC
White	AZ	8,750	2.9	2.3	1 to 9	14.75	1 to 24	8.0	Swetnam and Baisan 1996	MC
Guadalupe	TX	7,218	6.0			17.0	6 to 30		Ahlstrand 1980	MC
White	AZ	8,950	22.0						Dieterich 1983	MC
Sacramento	NM		10.0						Geils et al. 1995	Mesic
Kaibab Plateau	AZ	8,481	8.4	4.7		9.5			Wolf and Mast 1998	MC
Kaibab Plateau	AZ	8,907	10.3	9.1		19.0			Wolf and Mast 1998	MC

Central and Southern Rocky Mountains

In the mixed conifer forests of the Central and Southern Rocky Mountains, the historic fire regime was of mixed severity. Stand structure, fuel loads, topography, and weather conditions during forest fires helped create and maintain a mosaic of stand conditions. Lower elevation ponderosa pine stands experienced frequent fire, but higher elevation mixed conifer forests were characterized by a much lower fire frequency and patches of stand-replacing fire in addition to low-severity surface fires (Veblen et al. 2000, Kaufmann et al. 2007). Mixed conifer forests were composed of a mosaic of stands: some stands were even-aged, created by stand-replacing fire, while others were uneven-aged, maintained by low-severity fire and episodic tree regeneration (Schoennagel et al. 2004). Another element contributing to uneven-aged stand development in the forest mosaic were fire-created openings that persisted for as long as 148 years (Kaufmann et al. 2000). Synergistic, large-scale forest disturbance effects such as strong winds and warm temperatures promote crowning and allow fire to cover large areas. In contrast, increased humidity, cooler weather, or patches of low fuel loads can reduce fire severity (Arno 1980, Schoennagel et al. 2004).

Most fires in the Southern Rocky Mountains occur in May and June before seasonal monsoons come (Baker 2003, Grissino-Mayer et al. 2004). During that time, dry lightning from convective storms provides ample ignition sources (Margolis et al. 2007). After the monsoons end in September and October there is another peak in fire activity, though fires occur throughout the growing season (Grissino-Mayer et al. 2004). Further north in the Rocky Mountains, monsoon weather has less impact and fires are most common in July (Brown et al. 1999).

Typical fuel loads in mixed conifer forests based on inventories of ten national forests in the Northern Rocky Mountains averaged 9.6 tons per acre (Brown and See 1981). The Fire and Fire Surrogate Study reported a similar quantity of standing and downed dead wood (8.6 tons per acre) for the Northern Rocky Mountains (Stephens



NASA Visible Earth

Central and Southern Rocky Mountains, Terra MODIS, September, 2002

et al. 2009a). Fuel loads are spatially variable, and disturbances such as wind storms can create locally dense concentrations of fuel (Robertson and Bowser 1999, Baker et al. 2007). Brown and colleagues (2003) recommend retaining 5 to 10 tons per acre of CWM in warm-dry ponderosa pine-Douglas-fir types and 10 to 20 tons per acre of CWM in cool Douglas-fir types.

Fires in mixed conifer forests varied significantly in extent, but unlike fires in lower elevation ponderosa pine forest most fires in mixed conifer stands burned through most of the stand (Grissino-Mayer et al. 2004). In one Rocky Mountain case study, fires greater than 4 square miles occurred about every 50 to 60 years, though the range was between 27 and 180 years (Brown et al. 1999, Kaufmann et al. 2000). In the Upper Rio Grande Basin, 14 different stand-replacing fires were identified between 1847 and 1901 (Margolis et al. 2007).

In general, the warm-dry mixed conifer stands experienced non-lethal fires every 20 to 50 years, while higher-severity fires occurred much less frequently (Romme et al. 2009). In contrast, these lethal fires were the dominant process in cool-moist mixed conifer stands, although occasional small, low-severity fires also occurred (Romme et al. 2009). Table 4 shows fire return intervals for 16 sites in the Southern Rocky Mountains. The average of the estimated MFRI is about 16 years, but the average of the more conservative measure of MFRI that only includes fires that scar at least 25 percent of trees is 24 years. Fires in the Central and Southern Rocky Mountains were less frequent than in the Southwestern uplands and plateaus but similar to intervals in the Northern Rockies (13 to 26 years; Arno 1980).



Alexander Evans

San Juan National Forest, Colorado

Table 4 Fire Frequency in Mixed Conifer Forests of the Central and Southern Rocky Mountains

Site	State	Elev. (ft)	MFRI	Std. Dev.	Range	25% MFRI	Std. Dev.	Range	Citation
Jemez	NM	9,870	25.2	32.9	1 to 89	16.00	2.8	7 to 29	Swetnam and Baisan 1996
Jemez	NM	9,780	19.5	15.7	4 to 52	20.00	5.6	3 to 34	Swetnam and Baisan 1996
Jemez	NM	8,400	9.5	6.7	1 to 21	14.27	6.7	13 to 46	Swetnam and Baisan 1996
Jemez	NM	8,850	4.5	2.9	1 to 12	11.27	3.8	2 to 23	Swetnam and Baisan 1996
Jemez	NM	9,375	15.8	10.2	1 to 33	26.14	11.4	5 to 19	Swetnam and Baisan 1996
Jemez	NM	9,710	12.0	8.8	3 to 32	19.50	4.7	12 to 66	Swetnam and Baisan 1996
Jemez	NM	7,400	6.8	5.6	1 to 24	11.50	6.2	5 to 25	Swetnam and Baisan 1996
Jemez	NM	9,387	4.8	3.5	1 to 17	12.67	6.7	1 to 18	Swetnam and Baisan 1996
San Juan	CO	8,499	24.3	11.5	3 to 50	32.30	23.4	7 to 79	Fulé et al. 2009
San Juan	CO	9,154	30		6 to 51	36			Grissino-Mayer et al. 2004
San Juan	CO	8,497	19		5 to 28	22			Grissino-Mayer et al. 2004
San Juan	CO	8,399	21		4 to 50	22			Grissino-Mayer et al. 2004
Front Range	CO	8,694	18.6	24.6	1 to 88	34.3*	32	1 to 88	Veblen et al. 2000
Front Range	CO	8,406	17.2	25.0	1 to 92	40.8*	29.8	1 to 92	Veblen et al. 2000
Front Range	CO	8,084	22.4	36.2	1 to 125	43.4*	48.1	7 to 125	Veblen et al. 2000
San Juan	CO		12.6		3 to 50	19.8*		3 to 35	Korb et al. 2007

*these measurements are based on a 10% MFRI

Sierra Nevada

As with other mixed conifer forests, west-side mixed conifer stands dominated by ponderosa pine are likely to have had frequent low- to medium-severity fires (Skinner and Chang 1996). Eastern Sierra forests tend to have fire return intervals similar to western Sierra forests; however, stand isolation and local rain shadow conditions can affect fire regime (North et al. 2009b). Douglas-fir dominated mixed conifer stands tend to be more mesic and have both low- and mixed-severity fires. In mixed conifer stands dominated by white fir in the southern Sierra, fires were mostly low to moderate severity, with occasional small patches of high-severity fires (Skinner and Chang 1996, Scholl and Taylor 2010). Fire severity in eastern slope Sierra Nevada mixed conifer forests tends to follow the pattern of increasing severity with increasing elevation (Gill and Taylor 2009). Fire history in mixed conifer stands dominated by giant sequoia is the most well documented of Sierra Nevada mixed conifer forests (e.g., Swetnam 1993, Swetnam et al. 2009). In these stands, fires burned at low- to moderate-severity with occasional patches of high-severity (Skinner and Chang 1996). In the LANDFIRE model for Sierra Nevada mixed conifer, 15 percent of fires are stand-replacing while about 70 percent of fires are surface fires (Barrett et al. 2004). Fire severity in mixed conifer forests of the Sierra Nevada is, in part, the result of feedback between fire regimes and forest structure, with high-severity patches driving forest structure in certain areas such as upper slopes and southwesterly aspects (Beatty and Taylor 2008b, North et al. 2009a). In a Jeffrey pine-dominated mixed conifer forest in the central Sierra that has been repeatedly burned by managed wildfire over the last 30 years, high-severity patches covered 15 percent of the burned area (Collins and Stephens 2010) and the percentage of high-severity fire has remained relatively stable over this period (Collins et al. 2009).

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The skidway over which the great logs are dragged to the mill, Millwood, California, 1902

Before Euro-American settlements, most fires in Sierra Nevada mixed conifer forests occurred during the late season (late summer and early fall) (Moody et al. 2006, Van de Water and North 2010). During this period lightning is more common and fuels are drier (Monroe and Converse 2006). In a northern Sierra study, mixed conifer fires burned during the dormant season, though about a quarter of fires burned during the growing season (Stephens and Collins 2004, Beatty and Taylor 2007). In a Jeffrey pine-dominated forest on the east side of the Sierra Nevada, fires were most common during the late summer and fall months (Vaillant and Stephens 2009). Because of high fuel moisture, early-season burns tend to consume less of the available

fuel and leave a more heterogeneous burn pattern (Knapp et al. 2005, Knapp and Keeley 2006). The most widespread fires in Sierra Nevada mixed conifer forests occur during drought years (Beaty and Taylor 2001). Most patches of stand-replacing fires were small (less than 10 acres) while a few were large (more than 148 acres) (Beaty and Taylor 2001, Collins and Stephens 2010). Pre-settlement mixed conifer forest had large, persistent gaps that were not closed by regeneration (North et al. 2002).

The fire frequency in the mixed conifer of the Sierra Nevada is similar to the Central and Southern Rocky Mountains, with an average MFRI of 13 years and more conservative MFRI of 25 years (Table 5). In the LANDFIRE model for Sierra Nevada mixed conifer, surface fires have a mean return interval of about 15 to 20 years while mixed-severity fires occur every 30 to 50 years on average (Barrett et al. 2004). As Table 5 shows, there is important variation in fire frequencies, often driven by local factors. For example, in the Lake Tahoe Basin fire return intervals are longer on north-facing than on south-facing slopes (Beaty and Taylor 2008a). Riparian areas in Sierra mixed conifer forests burn at similar frequency to upland forests (Van de Water and North 2010) and fire effects to riparian areas may have been moderate for most fires (Bêche et al. 2005).



Stephens Lab University of California Berkeley

Prescribed burn at Blodgett Forest, California

Table 5 Fire Frequency in Mixed Conifer Forests of the Sierra Nevada

Site	Elev. (ft)	Aspect	MFRI	Range	25% MFRI	Range	Citation	Description
Diamond Mts.	5,577		10.8	4 to 32	13.4	7 to 32	Gil and Taylor 2009	Pine/MC
Diamond Mts.	5,906		15.9	6 to 30	18.0	12 to 30	Gil and Taylor 2009	Fir/MC
S. Sierra	6,398	SW	8.4	3 to 14			Kilgore and Taylor 1979	PIPO/MC
S. Sierra	6,398	SW	10.0	3 to 22			Kilgore and Taylor 1979	Sequoia/MC
S. Sierra	6,398	SE	16.4	4 to 35			Kilgore and Taylor 1979	Sequoia/MC
S. Sierra	6,398	NW,NE	14.2	3 to 27			Kilgore and Taylor 1979	Sugar/MC
Lassen	5,333		27.0		32.2		Van de Water and North 2010	MC
Lassen	5,333		16.0		27.7		Van de Water and North 2010	RiparianMC
Onion Creek	6,108		14.4		22.9		Van de Water and North 2010	MC
Onion Creek	6,108		15.6		29.4		Van de Water and North 2010	RiparianMC
Lake Tahoe	6,419		16.9		31.4		Van de Water and North 2010	MC
Lake Tahoe	6,419		17.1		50.1		Van de Water and North 2010	RiparianMC
N. Sierra			13.0*	3 to 35			Skinner and Chang 1996	Fir/MC
C. Sierra			11.0*	4 to 32			Skinner and Chang 1996	Fir/MC
N. Sierra	4,480	W	11.4	1 to 53	12.5	4 to 29	Moody et al. 2006	MC
N. Sierra	5,200	W	10.2	1 to 36	29.3	23 to 36	Moody et al. 2006	MC
N. Sierra	5,500	NE	12.3	1 to 36	21.0	16 to 36	Moody et al. 2006	MC
N. Sierra	6,000	NE	16.9	6 to 46	17.8	6 to 36	Moody et al. 2006	MC
N. Sierra	6,100	SSW	12.9	4 to 45	35.5	30 to 41	Moody et al. 2006	MC
N. Sierra	5,000	SE	9.9	4 to 21	12.7	4 to 22	Moody et al. 2006	MC
N, Sierra	4,400	SW NE	10	6 to 40			Stephens and Collins 2004	MC
S. Sierra	5,577		4.2	1 to 12			Phillips 2002	MC
C. Sierra	4,593		6.3				Collins and Stephens 2007	MC
S. Sierra	4,593		9.3				Collins and Stephens 2007	MC
S. Sierra			4				Caprio and Lineback 1997	PIPO/MC
S. Sierra			10				Caprio and Lineback 1997	White fir/MC
S. Sierra			30				Caprio and Lineback 1997	Red fir/MC
E. Sierra	7,434		2.9	1 to 8	8.1**	1 to 32	Vaillant and Stephens 2009	Pine/MC

*medians **10% MFRI

Transverse and Peninsular Ranges in Southern California

At the regional level, Southern California experienced both high- and low-severity fires before fire suppression. While fires in upper-elevation mixed conifer forests were low or mixed severity (Minnich et al. 1995), lower elevation chaparral tended to be high severity, and some of these high-severity fires may have burned into the mixed conifer zone (Keeley and Zedler 2009). The Sierra San Pedro Mártir (SSPM) is a mixed conifer ecosystem similar to the Transverse and Peninsular Ranges in Southern California, but the SSPM has been the subject of greater scientific scrutiny than have the Southern California ranges, in part because systematic fire suppression did not begin until 1970 (Stephens et al. 2003) and logging has only occurred in a very small area (Stephens et al. 2008). Because of the intense study of the SSPM and its similarity to the Southern California ranges, we include data from the SSPM in this discussion. SSPM mixed conifer forests experienced smaller, low intensity burns and moderately intense understory fires (Minnich et al. 2000).

In general, the chaparral of Southern California burns in the fall, driven by the Santa Ana winds, which are strong dry winds that blow from the east and towards the coast (Keeley et al. 1999). Fires driven by Santa Ana winds burn with enough intensity to burn through even young fuels and can send fire brands far beyond the fire front (Keeley 2008). The majority of fires in the San Jacinto Mountains were late-season fires (Everett 2008), but the SSPM was dominated by early-season fires (Stephens et al. 2003). Fine fuels in Southern Californian mixed conifer forests may have been spatially correlated with species, as they are in the SSPM (Fry and Stephens 2010). In the Southern California ranges, fires prior to the onset of the 20th century were generally less than 16 acres in size: only a small percentage of fires exceeded 45 acres in size (Everett 2008). Fires in the SSPM ranged in size up to nearly 25 square miles in extent in the mid 20th century; however, most fires were much smaller (Minnich et al. 2000).

Before Euro-American settlements the fire return interval in the mixed conifer of the San Jacinto Mountains was on the order of 2.5 years (Everett 2008). McBride and Laven (1976) measured a fire return interval of 10 years for ponderosa pine and 12 years for Jeffrey pine in the San Bernardino Mountains. On the Los Padres National Forest small localized fire burned about every 12 years while larger fires were much less frequent (Lombardo et al. 2009). Stephens and colleagues (2003) estimate that the fire return interval in the SSPM ranges from 5.7 to 6.9 years, though Minnich and colleagues (2000) suggest a much longer interval between more intense understory fires. Chaparral covers much of the lower elevations in the region and its fire regime is important to mixed conifer because many fires in chaparral burn up into mixed conifer stands (Lombardo et al. 2009). Prior to Euro-American settlement, chaparral had a fire return interval on the order of 30 to 50 years (Keeley et al. 1999). Some researchers maintain that the evidence shows that fire regimes in chaparral have shortened since fire suppression and fire behavior can be changed by altering the age structure of chaparral vegetation (Minnich 1983). On the other side of the scientific debate, other researchers maintain that the chaparral fire regime has not changed significantly and that the age structure of chaparral vegetation does not affect fire behavior (Keeley et al. 1999). Given our focus on upper elevation mixed conifer forests, we will not delve further into fuels treatment in chaparral.



Blackhall Fire in Wyoming

Impact of Altered Fire Regimes and Forest Health

Though the fire regimes and species composition for each of the regions in this report are different, the impacts of altered fire regimes are very similar. In most mixed conifer forests, altered fire regimes have produced changes in species dominance and tree density. Altered fire regimes have also increased the homogeneity of both age and structure of many mixed conifer landscapes. Fire suppression has increased the impact of stresses such as drought and air pollution (Savage 1997). Although not directly related to altered fire regimes, new pests and pathogens have taken hold in mixed conifer forests since Euro-American settlement. These exotic species, such as spruce aphid (Lynch 2004) and white pine blister rust (Conklin 2004), affect forest health and should be considered in fuels treatment planning.

The increased tree density, fuel loads, and stand homogeneity due to fire suppression have increased the likelihood of uncharacteristic extensive crown fires in ponderosa pine forests (Covington and Moore 1994a, Skinner and Chang 1996, Schoennagel et al. 2004). Mixed conifer forests had a mixed-severity fire regime in which crown fire patches were not uncommon. The historic fire regime included much rarer, larger stand-replacing fires driven by multidecadal climate patterns. Nevertheless, altered fire regimes have changed the landscape context in which mixed conifer fires occur (Collins and Stephens 2010). The increased threat of catastrophic fire in lower elevation ponderosa pine amplifies the potential for extensive crown fire in mixed conifer forests because of the high likelihood of fire spread upslope. The increased homogeneity of mixed conifer forests, particularly the increased density of the warm-dry mixed conifer type, could add to the extent and impact of severe fires. Finally, the trend towards a warmer, drier

climate is likely to increase the likelihood of extensive severe fires (as discussed in more detail in the section “Climate Change” below). The likelihood of extensive severe fires is important because of the impact they can have on ecosystem processes and services. Hunter and colleagues (2007) detail a range of negative effects of extensive, severe fires in ponderosa pine forests; these include community and firefighter risk, soil damage, invasion of exotic species, and insect and disease outbreaks.

Southwestern Plateaus and Uplands

After frequent fires in the Southwest were interrupted between 1870 and 1900, the structure and composition of mixed conifer forests began to change (Swetnam and Baisan 2003). One key change has been the increase of shade-tolerant species such as white fir and a reduced importance of ponderosa pine and other shade-intolerant species such as southwestern white pine, aspen, and Douglas-fir (Johnson 1994, Covington et al. 1998). In some stands in the Sacramento Mountains where fire frequency has been reduced, Douglas-fir and white fir are becoming dominant, replacing ponderosa pine (Brown et al. 2001). At the upper elevations of

mixed conifer forests on the Kaibab Plateau, the absence of surface fires has permitted the encroachment of subalpine species, particularly Engelmann spruce (Mast and Wolf 2006). In mixed conifer forests on the San Francisco Peaks in Arizona, tree density has increased four times since 1876 (Cocke et al. 2005). In many stands, the gaps created by fire or other mortality agents where shade-intolerant species (particularly ponderosa pine, aspen, or Douglas-fir) regenerated before fire suppression have filled in (Dieterich 1983). Meadows within the mixed conifer mosaic have also been altered by fire suppression. For example, the area of open montane grasslands in the Jemez Mountains decreased 55 percent between 1935 and 1981 (Allen 1989). The area of aspen stands in the Southwest has also decreased significantly: 46 percent between 1962 and 1986 (Johnson 1994). Ingrowth of shade-tolerant species has also increased the density of many mixed conifer stands (Dahms and Geils 1997). Studies in a mixed conifer stand in the Grand Canyon National park showed that after more than a century of fire suppression the basal area was 35 to 45 percent greater, and the number of trees had increased even more (Fulé et al. 2002, Fulé et al. 2004). Species shifts have also occurred in ponderosa pine forests and increased the density of shade-tolerant species

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Santa Fe National Forest, New Mexico

in these stands formerly classified as a ponderosa pine cover type. This change in ponderosa pine stands has resulted in many of them being classified as mixed conifer cover type in later inventories. For example, mixed conifer forest cover type increased by 1,040,000 acres (81 percent) from 1962 to 1986 (Johnson 1994). The effect of altered fire regimes has been more moderate on cool–moist mixed conifer stands, which were historically denser and experienced longer fire return intervals (Romme et al. 2009).

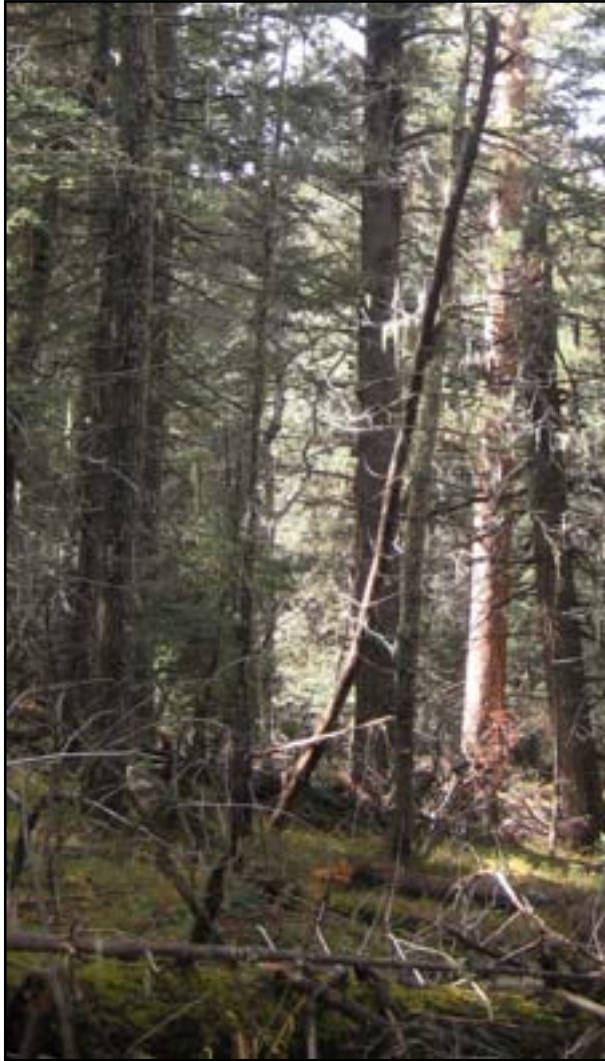
Altered fire regimes affect forest health and, in turn, forest health influences fire threat and behavior. For example, fire and dwarf mistletoe are interrelated: dwarf mistletoes are a natural part of mixed conifer forests and endemic levels of dwarf mistletoe were probably high in many locations (Pollock and Suckling 1995, Conklin and Fairwather 2010). However, fire, even lower-severity surface fires, can reduce the severity of mistletoe infections by killing infected trees and by scorch pruning infested trees (Conklin and Geils 2008). By the same token, lack of fire can encourage the development of homogenous, stressed stands that facilitate expansion of mistletoe populations (Conklin and Fairwather 2010). Dwarf mistletoe infections can encourage fires by killing trees and changing available fuel, creating brooms that serve as ladder fuel, and facilitating the buildup of flammable resins (Geils et al. 1995, Conklin and Fairwather 2010). Douglas-fir mortality in stands infected with dwarf mistletoe can be as high as four times greater than uninfected stands, with growth losses as high as 65 percent (Mathiasen et al. 1990).

The spread of white pine blister rust (WPBR) to the Southwest adds another stress to mixed conifer forests. WPBR can damage or kill southwestern white, limber, and bristlecone pines (Schwandt et al. 2010). WPBR is well-established in the Sacramento Mountains, has been detected in northern and western New Mexico and eastern Arizona, and is likely to affect white pines throughout the Southwest in the coming decades (Conklin et al. 2009). In general southwestern white pine is less damaged by fire, bark beetles, and root disease; however, WPBR may reduce white pine's ability to buffer mixed conifer forests from these agents of disturbance (Geils et al. 1995). Another exotic, the spruce aphid, has the potential to increase fuel loads in mixed conifer forests. Though many trees can recover from defoliation, individuals stressed by other factors such as drought will die. Spruce aphid may reduce species diversity in mixed conifer forests because defoliation is much more severe on Engelmann spruce than other spruce (Lynch 2004). Corkbark fir may increase in dominance where drought, dwarf mistletoe, or other stressors combine with spruce aphid defoliation to cause mortality of Engelmann spruce. The potential for spruce aphid to change species composition is augmented by the fact that it defoliates seedlings as well as mature trees.

Central and Southern Rocky Mountains

The mixed conifer forests of the Central and Southern Rocky Mountains exhibit an increase in tree densities as a result of altered fire regimes that is common across the western U.S. (Kaufmann et al. 2007). For example, in one southwestern Colorado mixed conifer forest modern basal area was 145 percent greater than a reconstruction of pre-settlement density and the number of trees had increased even more (377 percent) (Fulé et al. 2009). White fir has increased in density because of altered fire regimes, while regeneration of ponderosa pine and Douglas-fir has decreased (Romme et al. 2009). The greatest increase in tree density has occurred on south-

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San Juan National Forest, Colorado

facing slopes, low elevations, and ponderosa pine dominant stands that previously had relatively low tree densities (Veblen et al. 2000, Platt and Schoennagel 2009). Trees have also encroached into gaps and openings within forests, increasing canopy connectivity (Zier and Baker 2006). Lower-elevation ponderosa pine stands in the region, particularly those logged prior to 1960, exhibit higher stand densities, more homogenous stand structure, and an increased abundance of shade-tolerant trees compared to pre-settlement conditions (Naficy et al. 2010). In contrast, tree density in cool-moist mixed conifer stands were historically dense because of favorable topographic and soil conditions, and are unlikely to have increased as much (Schoennagel et al. 2004). Many of these cool-moist mixed conifer stands had longer fire return intervals, and climate played a larger role in the timing of fires before Euro-American settlement. Hence altered fire regimes have had less of an impact (Romme et al. 2009).

At the landscape level, there has been an increase in homogeneity of forest structure because the previously lower density mixed conifer stands increased in density while previously higher density mixed conifer stands remained dense. In addition, forest age structure has become more mature overall due to decreased small- and mid-scale sized

disturbances during the past century. The increased homogeneity may facilitate larger patches of crown fire than occurred historically (Keane et al. 2002, Schoennagel et al. 2004). Similarly, there is evidence in some parts of the Southern Rockies that western spruce budworm outbreaks have become more widespread and intense in the late 20th century because of greater forest homogeneity, but this is not the case in other areas (Ryerson et al. 2003). Budworm defoliations can predispose host trees to Douglas-fir beetle and root rots (Keane et al. 2002). As in Southwestern mixed conifer forests, exotic pests and pathogens are a concern in the Rocky Mountains. In the central Rocky Mountains, 55 percent of the white pine has WPBR. While mortality of white pine has been relatively low, incidence and intensity of the exotic disease have increased significantly since the 1960s (Smith and Hoffman 2000).

Sierra Nevada

In the mixed conifer forests of the Sierra Nevada, altered fire regimes have facilitated the establishment and growth of a cohort of white fir and other fire-intolerant species, increasing the density of many stands (Beaty and Taylor 2007, Beaty and Taylor 2008a). For example, in a southern Sierra Nevada mixed conifer forest, 84 percent of white fir and incense cedar had established since the last widespread fire of 1986 (North et al. 2005). Many mixed conifer stands have become multiple-canopy, dominated by shade-tolerant species (Weatherspoon et al. 1992). Fire suppression has also permitted a buildup of both surface and ladder fuels (Skinner and Chang 1996, Stephens and Moghaddas 2005c). During the same period, fire suppression has diminished the herbaceous layer that previously included a well-developed community of forbs, perennial bunchgrasses, and dispersed shrubs (Chang 1996). At least in some areas, such as the Teakettle Experimental Forest, gaps within mixed conifer forests have not closed since fire suppression (North et al. 2002). However, without new fires to create new gaps or openings, their prevalence on the landscape scale is likely to have decreased (North et al. 2009a). At the landscape scale, Sierra Nevada mixed conifer forests have become more homogenous because of altered fire regimes (Weatherspoon et al. 1992, Chang 1996). The decreased fire frequency during Euro-American settlement encouraged increased fire size (Chang 1996, Stephens et al. 2007). At the same time there has been an increase in the extent of high-severity fire and high-severity patch size (Miller et al. 2009).

In moisture-limited forests, uncharacteristic increases in tree density commonly facilitate bark beetle, mistletoe, and root disease mortality. In Sierra Nevada mixed conifer forests, fire suppression means insects and diseases in association with periodic drought events have become the most important mortality agents (Maloney and Rizzo 2002). Mistletoe infection may have increased as well, because before Euro-American settlement fire had some sanitation effect on mistletoe (Maloney et al. 2008). The impact of pests and pathogens is aggravated by the introduction of exotic species. For example, WPBR affects white pine throughout the Sierra Nevada range and in one study infected about 67 percent of host trees (Smith and Hoffman 2000). It is worth noting that some insect outbreaks act as a negative feedback on increased tree densities. For example, Douglas-fir tussock moth caused almost complete defoliation of understory firs during an outbreak from 1997 to 1998 (Schowalter 2000 p. 459, North et al. 2002).



Stephens Lab University of California Berkeley

Prescribed burn at Blodgett Forest, California

Transverse and Peninsular Ranges in Southern California

As with the other regions covered in this report, since Euro-American settlement the mixed conifer forests of Southern California have experienced an increase in stand density, a shift to younger age classes, and a shift in dominance from ponderosa pine to white fir (Savage 1997). Other shade-tolerant species, such as incense cedar, have also flourished under altered fire regimes. In the San Bernardino Mountains, tree densities increased 79 percent between the 1930s and 1990s (Minnich et al.



Angeles National Forest

Angeles National Forest, California

1995). The increased density of small trees provides ladder fuels and hence potential for crown fire (Franklin et al. 2006). Fire suppression and logging have reduced spatial heterogeneity and caused increased mortality during periods of drought (Stephens and Fulé 2005). Air pollution is another stress factor that has increased since Euro-American settlement. Because of the proximity of dense urban development and the prevailing weather patterns, mixed conifer forests of Southern California have suffered disproportionately from air pollution (Savage 1994).

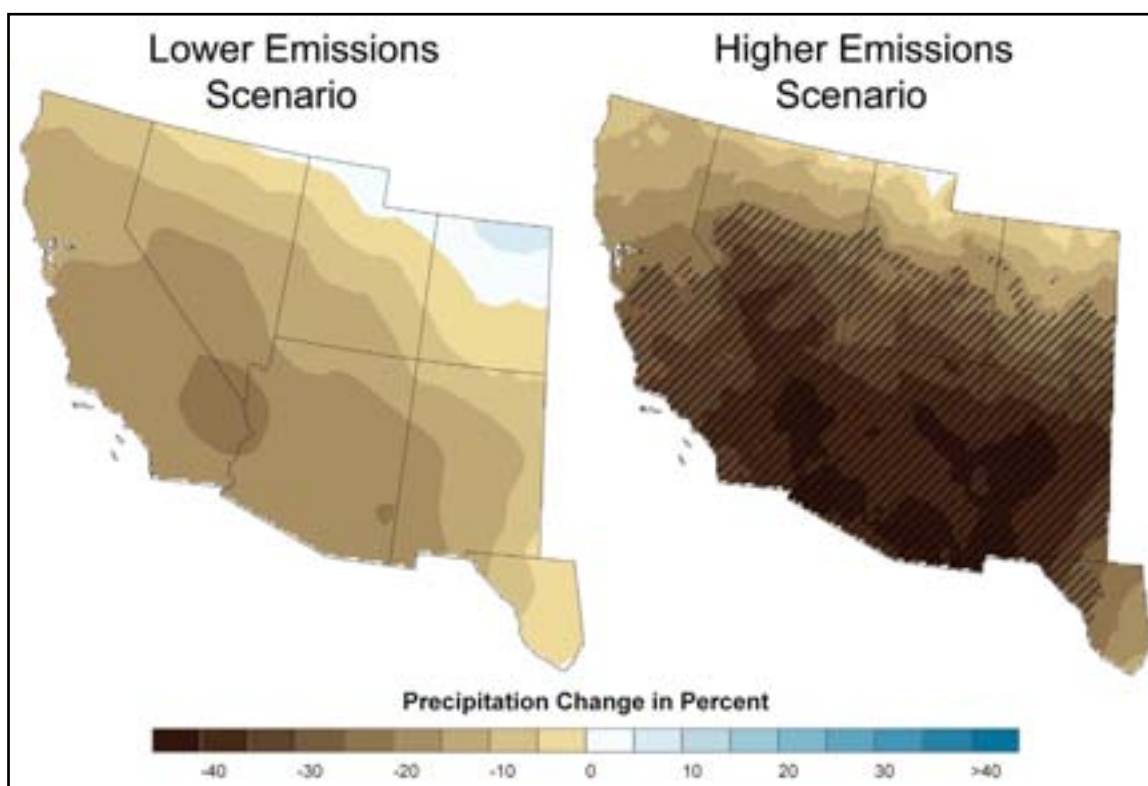
Altered fire regimes have affected chaparral ecosystems that are interwoven with the mixed conifer forests of Southern California. For example, patch size has increased in adjacent chaparral vegetation (Minnich 1987). Altered fire regimes have also facilitated conversion of native chaparral to alien-dominated grasslands (Keeley 2006). Similarly, exotic species benefited from an intense wildfire in mixed conifer forests (Franklin et al. 2006).

Climate Change

Mixed conifer forests are likely to continue to change during the 21st century. On average, the climate in mixed conifer forests is likely to be warmer and drier by the end of the 21st century than it was during the 20th century, with warmer spring and summer temperatures, reduced snowpack and earlier snowmelts, and longer, drier summer fire seasons (Westerling et al. 2006, IPCC 2007, Dominguez et al. 2010). Three lines of evidence predict that warming and drying conditions in mixed conifer forests are likely to cause increased fire activity: reconstructions of fire and climate in the past (Swetnam 1993, Frechette and Meyer 2009), trends over the last few decades (Westerling et al. 2006), and predictive models (Westerling and Bryant 2008). Other predicted effects of a warmer, drier climate include reduced growth and increased mortality in mixed conifer forests (van Mantgem and Stephenson 2007, van Mantgem et al. 2009). For

example, modeling predicts declines in stem volume growth in Sierran mixed conifer due to increased summer temperatures (Battles et al. 2008). A warming climate and altered precipitation regimes will cause other ecosystem changes, such as increased success for bark beetles (Bentz et al. 2010).

Scientists are only just beginning to provide the certainty and detail in climate forecasts that managers can use to plan prescriptions, but some guidance is emerging (Millar et al. 2007, Smith et al. 2008, Stephens et al. 2010b). Although the future climate may not be within the historic range of conditions for mixed conifer forests, reestablishing fire as an active ecological process is important because restored forests are more likely to be resilient to the negative effects of an altered climate (Fulé 2008). In addition, management that increases heterogeneity of forest structure and fuels in mixed conifer forests will help maintain resilience (Stephens et al. 2010b). The impact of and management responses to a changing climate are important areas for future research and may justify updates to this guide.



Percentage change in March-April-May precipitation for 2080 to 2099 compared to 1961 to 1979 for a lower emissions scenario (left) and a higher emissions scenario (right). Confidence in the projected changes is highest in the hatched areas (Karl et al. 2009).

Section III: Fuels Treatment Objectives

By definition, fuels treatment focuses on modifying forest attributes to affect how fires burn. However, most treatments in mixed conifer forests include additional objectives such as ecological restoration, commercial sale of forest products, enhancement of wildlife habitat, protection of water resources, or maintenance of cultural resources. Often these multiple objectives are intertwined.

The overlapping nature of land management objectives is reflected by the combination of funding from timber, fuels, and fire programs for national forest projects. Although overlapping objectives can help programs or departments share resources, differing objectives can create barriers. Managers we interviewed talked about how different objectives, or at least different priorities among objectives, can make matching fuels treatments across ownership boundaries hard work. Obviously, working across ownerships is particularly important in achieving landscape goals and where ownerships are small or fractured.

A fuels treatment is any manipulation or removal of wildland fuels to reduce the likelihood of ignition or to lessen potential damage and resistance to control (Helms 1998)

Objectives and Ownership Boundaries

Some strategies for aligning or keeping objectives aligned include the new landscape-level initiatives and local level collaboratives. There are three new national programs relevant to managers working on fuels treatments: the Collaborative Forest Landscape Restoration Program (CFLRP), the Landscape Conservation Cooperatives, and the Joint Fire Science Knowledge Exchange Consortia. The Forest Landscape Restoration Act of 2009 established CFLRP to foster ecosystem restoration that encourages economic and social sustainability, leverages local resources with national and private resources, reduces wildfire management costs, and addresses the utilization of forest restoration byproducts to offset treatment costs and benefit local economies (www.fs.fed.us/restoration/CFLR/index.shtml). CFLRP provides funds for National Forest System lands but requires participation across jurisdictions. A number of managers of mixed conifer forests interviewed for this study are participating in CFLRP or similar state-based programs. In one case, on the Grand Mesa, Uncompahgre and Gunnison (GMUG) National Forests, managers feel that CFLRP funding would give them a chance to implement treatments on a large enough scale to allow for the reintroduction of fire. Without additional funding such as CFLRP, treatments would occur more slowly, perhaps not fast enough to establish appropriate fuel conditions over a large enough area to allow for the reintroduction of fire.

The LCCs are the result of a Department of the Interior initiative to create applied conservation science partnerships focused on a defined geographic area that inform on-the-ground strategic conservation efforts at landscape scales (www.doi.gov/lcc/). Although LCCs are not focused directly on fuels treatment, they are designed to provide scientific and technical expertise and include forest resilience and wildfire as central issues. As they develop, LCCs may provide

managers with answers to cross-boundary challenges such as climate change impacts, water availability, and invasive species that affect fuels treatments in mixed conifer forests. The Joint Fire Science Knowledge Exchange Consortia were created with a premise similar to that of the LCCs (i.e., the need for science to answer managers' questions), but with a focus on fire (www.firescience.gov/JFSP_Consortia.cfm). The Consortia are designed to accelerate the adoption of wildland fire science information by federal, tribal, state, local, and private managers within ecologically similar regions. The Consortia also afford managers a mechanism to help direct which fire science questions receive government funding.

These types of national policy directions change frequently enough that the programs listed here may not be relevant in 2020. However, the general trend towards more competition for funding such as CFLRP and emphasis on landscape cooperation is here to stay. Though the managers we interviewed often mentioned a preference for fieldwork, participating in or at least being aware of national programs has become a new job requirement.

Another set of strategies to help harmonize objectives has arisen from local-level collaboration. Managers at a number of federal agencies indicated that it is crucial to ensure local stakeholders agree with fuels treatment objectives. For example, a Bureau of Indian Affairs forester described frequent meetings with the tribal council and field trips every few months that included a bilingual speaker as helpful in maintaining support for the objectives of the fuels treatment program. Even private companies such as Sierra Pacific Industries have found that cooperation and communication is necessary, particularly on issues that cross property lines, such as smoke. In another example, the San Juan National Forest has developed the Upper San Juan Mixed Conifer Working Group to improve the health and long-term resilience of mixed conifer forests and the communities located near them in southwest Colorado. These types of groups allow land managers and other stakeholders to talk and work together, which tends to forge greater agreement on treatment objectives. A good example of expanding the agreement across a diverse set of stakeholders is the Uncompahgre Partnership. The Partnership has been able to support the treatment of thousands of acres, aggressive weed management programs, and development of a strong native seed program. It has received a ten-year, \$8.5-million-dollar grant to treat 160,000 acres. These local-level collaboratives are related to the national programs discussed above because areas with local collaboratives in place receive priority for some programs such as the CFLRP.



Arnie Friedt

Fuels treatment near Angel Fire, New Mexico

Reduced Wildfire Hazard

In order to reduce wildfire hazard, treatments must change or remove wildland fuels in a way that lessens the likelihood of fire ignition, potential damage, or resistance to control. Fire risk, the chance that fire may start and cause damage, is driven by frequency of lightning strikes and human ignition, neither of which is affected by fuels treatment. In contrast, fire hazard, the state of the fuel exclusive of weather or topography, is determined by the volume, condition, arrangement, and location of fuels (Hardy 2005). It is these parameters that treatment can affect, and so the overarching objective of reducing wildfire hazard is often broken down into subsidiary goals that include

- reducing surface fuels,
- increasing height to live crown (i.e., canopy base height),
- reducing canopy continuity (developing or maintaining canopy gaps), and
- decreasing crown density (Agee and Skinner 2005, Peterson et al. 2005, Stephens et al. 2009a).

Often the goal of fuel reduction treatments is to reduce uncharacteristic fires; in other words, fires which occur outside the time, space, and severity parameters of the historical, natural fire regime (Hardy 2005). Treatment placement is discussed in more detail in Section IV, but

Nick Smokovich



Mastication treatment, New Mexico

objectives also play a role in landscape planning. Wildland Urban Interface (WUI) zones are a clear example of a situation in which objectives drive the placement of treatments. Often WUI areas are dominated by other forest types, such as ponderosa pine or piñon-juniper, but mixed conifer also occurs in many WUI areas.

Treatment placement is also related to the discussion of working across boundaries to harmonize objectives. Where objectives across land owner boundaries are at least similar, treatments can be placed to optimize their effect on fire hazard. For example, in the Lake Tahoe basin a Multi-Jurisdictional Fuel Reduction and Wildfire Prevention Strategy brings together 16 federal, state, and local organizations that share the objective of reducing the probability of a catastrophic fire

in the basin (USDA Forest Service 2007). Under the strategy, managers have been able to link forest fuels treatments on federal lands and non-federal properties with homeowner defensive space work as part of a comprehensive strategy. Another example of collaboration across boundaries is the partnership between the U.S. Forest Service and the Tri-State Generation and Transmission Association to protect transmission lines by treating adjacent lands with burning, commercial harvest, and hand crews.

Two other aspects of the objective of reducing wildfire hazard are timing and maintenance. Managers cannot simply reduce fuels in a stand and consider the objective met. Forests grow, fuels build up, and fuel reductions must be maintained or repeated. As discussed in more detail in Section IV, returning fire as a natural process to stands can be the most cost-effective way of ensuring hazard reduction objectives are maintained following initial fuels treatments.

Restoration

There are some key differences between restoration and wildfire hazard reduction as treatment objectives. At times restoration and wildfire hazard reduction objectives may overlap, but at other times they may be at odds. Neither wildfire hazard reduction nor restoration is a “better” objective; each may be appropriate in different areas. However, it is important to identify where they differ, particularly in mixed conifer forests, where there may be greater differences between wildfire hazard reduction and restoration than in other forest types.

Restoration focuses on returning ecosystems or habitats to their original structure and species composition based on the idea that ecosystems are most healthy when they are within the range of conditions to which their component species have adapted (Swanson et al. 1994). A key element of restoration is the use of reference or benchmark conditions which describe the properly functioning ecosystem (Fulé et al. 1997). Often these reference conditions are framed in terms of the historic range of variability

(HRV). Because ecosystems are not static, there is a range of conditions which can still be considered healthy or natural (Morgan et al. 1994). For example, conditions immediately post-fire may be quite different from conditions two decades after a fire, but both could be natural states for a mixed conifer stand at the local scale. Additionally, the

Ecological restoration is the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed. (SER 2004)

proportion of these states on the landscape also may or may not reflect a natural range of variability. Restoration can recognize the inherent variability of ecosystems and aim to return ecosystems to conditions that are within HRV. “Fire Regimes and Historic Conditions,” in Section II, provides a general discussion of the HRV for mixed conifer stands, but site-specific data is needed to provide a clear guide for restoration (Landres et al. 1999). Pre-settlement conditions are often used to determine the HRV because of the impact increased populations and intensive resource utilization have had on ecosystem structure and processes, most notably fire (Fulé et al. 1997). Even as climate change alters basic environmental conditions, ecosystems are more likely to be resilient and resistant when they are within HRV (Smith et al. 2008, Keane et al. 2009, Stephens et al. 2010b).

Ecological restoration in ponderosa pine forests has become a common objective and often dovetails well with wildfire hazard reduction objectives. Because ponderosa pine ecosystems have historically experienced frequent low-severity fires, restoring natural fire regimes generally reduces the probability of high-severity, stand-replacing fires (Hunter et al. 2007). Therefore, restoration of ponderosa pine forests often lessens the potential damage and resistance to control

of wildfires (Fulé et al. 2000, Graham et al. 2004). However, many forest restoration efforts may do little to reduce wildfire hazard, e.g., restoration of a spruce-fir forest adapted to a high-severity fire regime would not reduce wildfire hazard.

Since fire return intervals in mixed conifer have averaged between eight and 25 years for the Southwestern plateaus and uplands, Southern Rockies, and Sierra Nevada (Tables 3, 4, and 5), fuel buildup and fire severity has historically been greater than that of ponderosa pine forests. In fact, high-severity patches, as opposed to extensive areas of high-severity fire, were relatively common in some mixed conifer forests during extended droughts. However, low-severity fire was the driving process in some mixed conifer forests, such as those in Yosemite National Park, California (Scholl and Taylor 2010). High-severity fires that are within the HRV may not be acceptable under a wildfire-hazard-reduction objective. For example, research suggests that historically dense upper-elevation mixed conifer forests “have long been naturally fire-prone, are dangerous places to live, and will remain so after restoration” (Baker et al. 2007). However, there is overlap between restoration and fuel reduction for mixed conifer forests, particularly at the landscape scale. For example, a restoration treatment in Grand Canyon National Park that included a high-severity fire created more heterogeneity by killing many trees in some areas of the site but few in other areas (Fulé et al. 2004). The new open areas contributed to reduction of wildfire hazard by reducing both the likelihood of ignition and the resistance to control.

Commercial Value

As with wildfire hazard reduction and restoration, there is overlap between generating commercial value from forest treatments and wildfire hazard reduction, but not complete alignment. There are two basic issues: (1) sales of wood products to help pay for treatments designed to meet other objectives and (2) generating commercial value as an objective in and of itself. Many projects use the sale of wood products to offset the cost of fuel reduction treatments. These range from individuals who remove fuel wood from designated areas to large sales of timber to traditional mills (Hartsough et al. 2008). It is important to note that market demand, not tree size, creates value. For example, a manager in Colorado received more money from selling the bark of ponderosa pine trees than the trees themselves. In some locations, such as the Pecos District of the Santa Fe National Forest and the Bureau of Land Management Bishop Field Office in California, managers have had success engaging local firewood collecting to remove material. A more industrial example comes from a restoration treatment in a Colorado mixed conifer stand that generated \$1,272 per acre from the sale of white fir logs for studs,



Thinning treatment, Vermejo Park Ranch, New Mexico

aspen logs for paneling, and excelsior (fine wood shavings) (Lynch and Mackes 2003). The ability to sell larger-diameter logs for higher-value products can be the factor that determines whether or not a treatment generates revenue. For example, ponderosa pine treatments that could generate \$496 or \$615 per acre if larger diameter logs were sold for timber in the Southwest or Sierra Nevada respectively would cost more than \$1,000 per acre without that revenue (USDA Forest Service 2005). A simulation based on inventory data from dry mixed conifer stands in New Mexico indicated that fuel reduction treatments would require a subsidy even given optimistic market assumptions (Fight et al. 2004). In contrast, private land managers at W.M. Beaty and Associates and Collins Pine Company in the Sierra Nevada have been able to pay for fuel reduction treatments because of an existing biomass market. Many managers of mixed conifer forests said that access to markets had a big impact on the number of acres they were able to treat. As one manager put it, “Take the mill out of the equation and you’re done for.”

One of the motivations for making commercial value an objective is to generate jobs and income for local communities. In 2005, fuels reduction programs on five national forests in the Southwest and the Southern Rockies produced \$40 million of output and helped generate 500 jobs (Hjerpe and Kim 2008). A study in Oregon estimated that watershed restoration contracting generates between 16 and 24 jobs for each million dollars invested (Nielsen-Pincus and Moseley 2010). Even the use of biomass for energy can generate jobs. For example, the National Renewable Energy Laboratory estimated that biomass power plants create 4.9 full-time jobs for each megawatt of generating capacity (Morris 1999).

Wildlife, Water, Recreation, and Other Objectives

Protecting or enhancing wildlife habitat, forest health, water quality, or cultural resources is not often the primary objective of fuels treatments in mixed conifer forests, but these other objectives are often a secondary goal or influence the main project focus.

Wildlife

Mixed conifer forest provide habitat for a number of important wildlife species, including threatened and endangered species such as the Mexican spotted owl (MSO), as well as more common species such as elk and deer. The habitat requirements and interactions with fuels treatment for these and other species are discussed in Section V. In some cases, the inclusion of wildlife habitat as an objective is a legal mandate. For example, MSO is listed as threatened under the Endangered Species Act, so treatments around MSO sites are governed by recovery plans (USFWS 1995). In contrast, elk are common in many mixed conifer forests—too common for many forest managers, since they suppress aspen regeneration. A number of management plans in mixed conifer forests include objectives to address the interactions between elk and aspen regeneration. For example, the Hart Prairie



Bill Radke

Mexican spotted owl

Fuels Reduction and Forest Health Restoration Project on the Coconino National Forest includes jackstrawing cut conifers and fencing to protect aspen regeneration from severe elk browsing. To prevent ungulate browse, fences must be at least 7 to 8 feet high; an alternative is to use partially felled overstory trees to create a natural fence (Kota and Bartos 2010). Increases in understory growth for grazing forage of domestic cattle can also be an objective of fuels treatment (Allen and Bartolome 1989).

Insects and Diseases

Another objective that can be included in fuels treatment projects is to reduce the impacts of insect and disease outbreaks. In mixed conifer forests a number of insects are part of the natural disturbance regime, including Douglas-fir tussock moth, western spruce budworm, and assorted bark beetles. Insects and disease explain nearly 80 percent of the background mortality in the relatively pristine Sierra San Pedro Mártir mixed conifer ecosystem (Maloney and Rizzo 2002). Dwarf mistletoe can infect many of the trees that dominate mixed conifer forests, including ponderosa pine, Douglas-fir, spruce, and white fir, and can have a significant drain on tree growth (Conklin and Fairwather 2010). A number of managers described treatments

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Beetle damage, Lake Tahoe Basin, California

implemented in the 1980s and 1990s where the main objective was control of mistletoe through even-aged management. Treatment prescriptions that develop or maintain even-aged forest structure are applicable to heavily infested stands (those in which more than 50 percent of trees being managed for are infected) (Conklin and Fairwather 2010). In ponderosa pine forests of the Southwest, the current recommendation is to use prescriptions to develop or maintain uneven-aged forest structure only when at least 75 percent of the area is free of mistletoe. Experts recommend even-aged management when more than 25 percent of the area is infected; and when 90 percent of the area is infected, stands are often best replaced or deferred from treatment (Conklin and Fairwather 2010). Even where commercial value is a low priority, reducing mistletoe infestations may be an important objective because of its interaction with fire and effects on overall forest health and resilience to disturbances (Hoffman et al. 2007).

In addition to the natural suite of insects, managers must also deal with exotic pests and pathogens such as white pine blister rust (WPBR) or spruce aphid. Objectives for exotic pests and pathogens can include efforts to protect potentially resistant individuals or increase ecosystem resistance and resilience (Stephens et al. 2010b). For example, a number of managers of mixed conifer stands that include species susceptible to WPBR have modified their fuels treatment plans to include an objective to protect potentially resistant individuals. Similarly, an exotic spruce aphid has potential to change management options and objectives in the mixed conifer forests of the Southwest (Lynch 2004).

Water

Two-thirds of the clean water supply in the U.S. comes from water that has been filtered through forested land, and the direct value of U.S. national forest headwaters is estimated to be over \$27 billion per year (Smail and Lewis 2009). There is potential for fuels treatment in mixed conifer forests to affect water yield. For example, aspen stands in one study had a greater peak snow accumulation and a greater water yield than conifer stands (LaMalfa and Ryle 2008). However, increases in water yield for runoff and groundwater recharge in aspen stands were partially offset by greater evapotranspiration than conifer stands (LaMalfa and Ryle 2008). Conifer canopies intercept a large portion of snowfall, and snow caught in canopies sublimates at higher rates than ground-level snow (Essery et al. 2003). Dispersed retention results in greater snow accumulation than grouped retention in lodgepole pine stands (Woods et al. 2006). Though few managers we interviewed currently include water as an objective, water yield may become a more important treatment objective over time, given the projection for a hotter, drier climate in the future.

Cultural Resources

Cultural resources, including archeological, ceremonial, and cultural sites, can change treatment objectives for managers of mixed conifer forests, particularly those who work with Native American tribes or the Bureau of Indian Affairs. For example, the main objective for a mixed conifer fuels treatment project near Taos Pueblo in New Mexico was the protection of ponderosa pines harvested annually for ceremonial purposes. Similarly, Douglas-fir is a culturally important tree to a number of Native American pueblos in northern New Mexico. Archeological sites are not as common an issue in mixed conifer as in some forest types, because there were fewer long-term settlements in mixed conifer forests (McBride and Laven 1976, Minnich 1988, Allen 2002, Baker 2002, Parker 2002). Standard archeological surveys allow for the protection of sites during fuels treatments. Including traditional forest uses in treatment projects can be a particular challenge because specific information on religious sites or ceremonial forest uses may be difficult to obtain. In fact, tribal ceremonial sites or activities can be closely guarded secrets that not all tribal members share. Strong



Alexander Evans

Fuels treatment, Taos Pueblo, New Mexico

and consistent collaboration between tribal leaders (both political and religious) and managers of tribal forest resources provides the best approach to ensuring Native American cultural resources are integrated into fuels treatment objectives and implementation.

Wilderness

Wilderness areas could be considered another type of cultural resource. Objectives in wilderness areas or in watersheds that include some portion of wilderness have a different mix of objectives because of the constraints on treatment in wilderness. Since the overriding goal for wilderness areas is to allow nature to operate unrestrained and unaltered (Wilderness Act of 1964, Public Law 88-577), thinning treatments are inappropriate. Still, wilderness area objectives can be harmonized with adjacent lands that are more directly managed. For example, the overall objective for the Santa Fe Municipal Watershed Plan is to minimize the risk of high-severity fire by reducing fuel loads (Derr 2009). This goal translates to fuel reduction treatments in some portions of the watershed, but in the wilderness area it means establishing conditions that allow for naturally ignited fire to be used for resource benefit (Collins and Stephens 2007).

Summary

It comes as no surprise for managers of mixed conifer forests that there is a wide range of objectives that can influence fuels treatment. A review of common objectives for treatments in mixed conifer highlights four lessons. Increasingly, collaboration to harmonize objectives or at least align treatments is a high priority, and national policies, programs, and funding support collaboration. Second, wildfire hazard reduction is not necessarily the same as restoration, but restoration treatments can reduce wildfire hazard, particularly at the landscape level. Generating commercial value can help meet other treatment objectives. Fuel reduction treatment can, and in some case must, include the objectives of protecting wildlife habitat, forest health, water quality, or cultural resources.

Section IV: Fuels Treatment Techniques

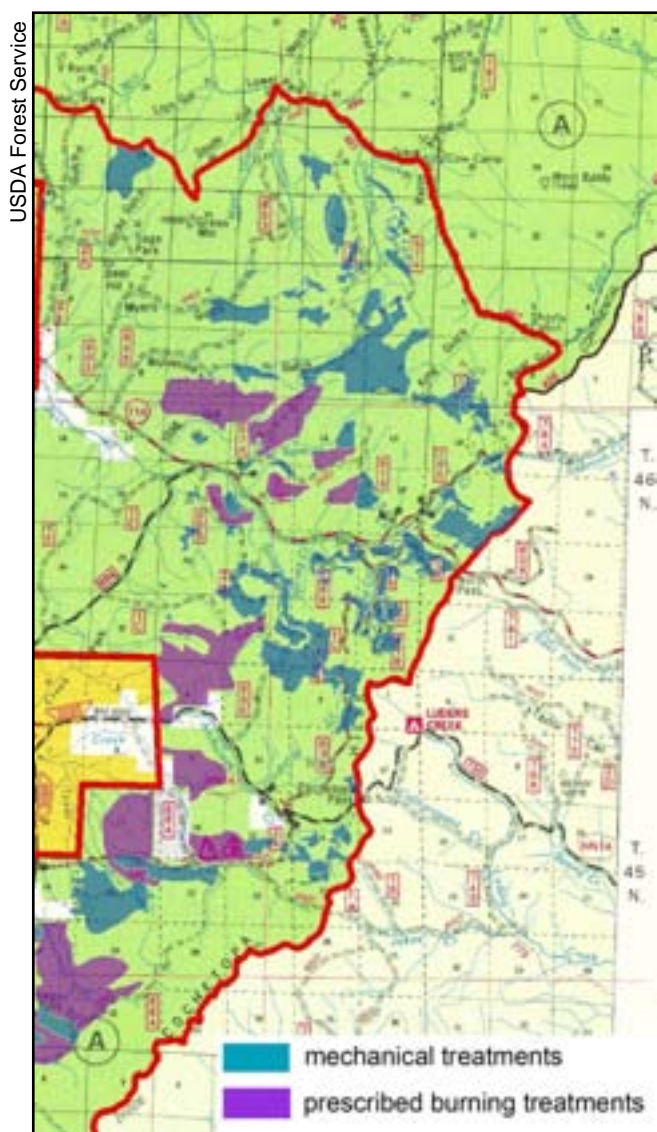
The goal of this section is to provide details on techniques to reduce fuels in mixed conifer forests. Section V covers the effectiveness and resources needed to implement these techniques.

Developing Landscape-Wide Fuels Treatment Strategies

A landscape-level perspective on fuels treatment in mixed conifer forests may be even more important than in most forest types because of the spatial variability of historic fires in mixed conifer. Historically, variability in fire severity created mosaics across the landscape (see “Fire

Regimes and Historic Conditions” in Section II). For land managers, the historic mosaic of fire has implications for fuel reduction. Since fires burn in a spatially variable pattern historically, fuels have been spatially variable as well (e.g., Keane et al. 2002, Fulé et al. 2003, Stephens 2004, Beaty and Taylor 2008b, Stephens et al. 2008). Forest structure generally mirrored fire pattern, with densities highest in riparian areas, lower in midslope, and lowest on ridgetops (North et al. 2009a). Southwest aspects and steep slopes also tend to have relatively low densities compared to northeast aspects or gentle slopes (North et al. 2009a). This heterogeneous spatial pattern of fire and fuels in mixed conifer forests contrasts with those forest types dominated by low-severity fire. For example, fire and fuels in Southwestern ponderosa pine have been more homogeneous historically. In mixed conifer forests, heterogeneity in stand structure is important in reducing the risk of wildfires that burn at high severity over large areas. For example, patches of aspen can impede the spread of fire (Fechner and Barrows 1976, Shepperd et al. 2006). Similarly, heterogeneous patches of varying age and structure can break up high-severity fire effects on the landscape.

Another reason to think at the landscape scale is that a landscape approach helps ensure fuel reduction treatments affect wildfire behavior (Schmidt et al. 2008, Moghaddas et al. 2010). Larger individual treatments have a greater



Map of treatments for the Cochetopa Hills area of the Gunnison Ranger District, GMUG National Forests, Colorado

impact on fire behavior, but the size of treatments is limited by competing objectives and resource limitations (Collins and Stephens 2010, Collins et al. 2010). Moreover, given a similar total treatment area, dispersed treatments can have a greater impact on fire behavior at the landscape scale (Finney et al. 2007, Ager et al. 2010, Moghaddas et al. 2010). At the same time, if treatments are too small they will be overwhelmed in a high-severity fire (Moghaddas et al. 2010). Planning treatments at a landscape scale can also yield economic benefits. Landscape planning provides the opportunity to increase supply stability and combine financially strong units with weak units, both of which can help make treatments more economically viable (Lynch et al. 2000). There are also organizational reasons to tackle planning over larger areas. As summarized in the plan for the Cochetopa Hills area of the Gunnison Ranger District, GMUG National Forests: a large planning area will “increase planning efficiency to better utilize limited agency resources and the public involvement process.”

Treatment placement in mixed conifer forests is generally designed to interrupt fuel continuity at the landscape scale and control efforts by providing defensible zones (Weatherspoon and Skinner 1996). Simulations suggest that treatment pattern, such as the orientation and proximity to other treatments, can have a significant impact on fire spread (Finney 2001). For example, a simulation of randomly placed treatments suggest that they require two to three times the rate of implementation as strategically placed treatments to produce the same fire hazard reduction (Finney et al. 2007). Also, creating and maintaining canopy gaps within stands will often reduce fire severity and transition crown fire to surface fire.

Models Used to Derive Landscape Strategies

In addition to knowledge of the landscape, geographic information systems (GIS) and modeling tools can be important for designing landscape strategies. Computer maps and models are no substitute for familiarity built up by walking and working in the woods and learning from others who have done so. However, geographic information systems and fire models have come a long way and can help even the most seasoned expert gain new insights.

This report is not the place to detail all of the computer maps and models available for planning fuels treatment at the landscape level (instead see Stratton 2006). However, it is worth highlighting a few resources as signpost towards more detailed information. Some key resources for land managers include the following:

- BehavePlus is a fire modeling system that combines a collection of models that describe fire behavior, fire effects, and the fire environment at the stand scale. BehavePlus is a point system with input supplied interactively by the user (www.firemodels.org/index.php/behaveplus-introduction).
- FlamMap is fire behavior mapping and analysis software that models potential fire behavior characteristics (including spread rate, flame length, and fireline intensity). FlamMap is a point representation of the FARSITE model and uses the same input data as FARSITE (www.firemodels.org/index.php/flammap-introduction) (Finney 2006).
- FARSITE computes growth and behavior of wildfire for long time periods under heterogeneous conditions of terrain, fuels, and weather. FARSITE adds a temporal and contagion component not available in FlamMap (www.firemodels.org/index.php/farsite-introduction) (Finney 1998).

- LANDFIRE provides geospatial data that can be viewed online or imported into a GIS. LANDFIRE data is most appropriate for analysis of large areas such as significant portions of states or multiple federal administrative entities. LANDFIRE includes data layers from biophysical setting to fire regime condition class (www.landfire.gov).

LANDFIRE can help overcome the data integration issues that often occur at ownership boundaries. For example the recent *New Mexico Statewide Natural Resources Assessment* (2010) used LANDFIRE to map forest resources and restoration needs across federal, tribal, state, and private forestlands in the state. A survey of federal fuels treatment specialists by JFSP indicated that Behave is the most commonly used system, while FlamMap, FARSITE, and LANDFIRE are common tools. The Forest Vegetation Simulator (FVS) also has a fire and fuels extension (FFE; www.fs.fed.us/fmnc/fvs/description/ffe-fvs.shtml) that models fire behavior, fire effects, fuel loading, and snag dynamics. Some managers like using FVS-FFE because it fosters collaboration, in part because it can incorporate stand exams. FVS-FFE can help managers understand longer-term effects of fuels treatments because of the link to the FVS growth model. However, FVS-FFE is not spatially explicit and does not capture heterogeneous fuel conditions or stand structures.

The profusion of fire modeling tools and the difficulty of mastering and integrating these tools have led to an effort to build an Interagency Fuels Treatment Decision Support System (IFT-DSS; frames.nbj.gov/jfsp/sts_study). The IFT-DSS is not a new fuels treatment model; rather, it is designed to provide an internet-based user interface with multiple software tools. As of this writing, IFT-DSS Version 0.3 can calculate fire behavior variables for a single point location, calculate fuel consumption using CONSUME, perform landscape-level analysis of fire behavior and hazard using FlamMap with LANDFIRE data, and display the output in the IFT-DSS map viewer or in Google Earth.

In-depth research projects provide an example of the opportunities for landscape analysis to aid treatment design. Using detailed remote sensing imagery and FARSITE models, Moghaddas and colleagues (2010) demonstrated the ability of fuel reduction treatments to reduce burn probabilities. In their example, treatments in a landscape dominated by Sierran mixed conifer were able to reduce burn probabilities even on the untreated stands designated for management of spotted owls, riparian and aquatic resources, and future reserve lands (Moghaddas et al. 2010). However, if the untreated areas were to burn they would still likely burn with high severity.

Prescribed Fire Techniques

Prescribed fire is an effective tool to restore vegetative communities and to protect values threatened by wildfire. In addition, prescribed fire effects approximate the effects of natural fires in mixed conifer ecosystems (Nesmith et al. 2011). A prescribed burn can reduce loads of fine fuels, duff, large woody fuels, rotten material, shrubs, and other live surface and ladder fuels, and hence change the potential spread rate and intensity of a future wildfire (Graham et al. 2004). Implementing a prescribed burn requires more information and training than can be provided in this guide and other resources focus on the topic (Wade and Lunsford 1989, Biswell 1999, USDA and USDOJ 2008, Heumann 2010).

Prescribed fire can help restore heterogeneity to mixed conifer forests that have become more homogeneous because of the absence of fire. Even when fuel loads are higher than they have been historically, fire severity is still heterogeneous, because of topographic and biotic factors (Knapp and Keeley 2006). In wilderness areas, prescribed fire can also be one of the few appropriate management tools (Keifer et al. 2000). Prescribed fire is often the lowest-cost treatment per acre, particularly where thinnings do not generate products or markets are not available (Hartsough et al. 2008). However, low-intensity prescribed fire is unlikely to reduce canopy density, and hence crown fire potential, in mature stands where overstory trees are relatively fire resistant.



Strip firing under ponderosa pines on the Fort Valley Experimental Forest, Coconino National Forest

Controlling Fire Intensity

The intensity of a prescribed burn is one of the main drivers of its impact on fuels and other ecological attributes. The timing of a prescribed fire plays a role in controlling intensity. Burns conducted during periods of high relative humidity, high fuel moisture, and low wind speeds will tend to be lower intensity. Steep slopes are common in mixed conifer forests and can significantly increase burn severity (e.g., Holden et al. 2009). Ignition and firing patterns also influence intensity. For example, backing fires set to burn downslope or against a steady breeze tend to burn at a low to moderate intensity, while a head fire burning upslope or with the wind will be more intense (Biswell 1999). One way to control the intensity of head fires is lighting strip head fires that burn for a short distance into a control line or previously burned strip. The distance the strip head fire burns in combination with fuel loads and fuel moisture will control the intensity of the burn (Hunter et al. 2007). Some prescribed burns in mixed conifer take advantage of aerial ignition because it can cover large areas more rapidly. Clear communication between aerial and ground personnel is crucial during aerial ignitions. For example, the Redwood Mountain prescribed fire plan from Sequoia and Kings Canyon National Parks highlights the need for coordination between aerial and ground crews to ensure blacklining stays well ahead of interior ignitions.

Seasonality

For mixed conifer forests, one of the points of discussion is the seasonality of burning. In the Sierra Nevada the historic fire regime was dominated by late-season fires, while in the Southern

Rocky Mountains and Southwest there was a mix of both early and late-season fires (see “Fire Regimes and Historic Conditions” in Section II). In general, fire intensity is a better predictor of ecosystem effects than season, though often season has an effect on intensity (Knapp et al. 2009). In the Sierra Nevada, early-season prescribed burns in mixed conifer forests occur before fuels have completely dried out and hence have less intensity and lower severity than late-season burns (Knapp et al. 2005). The opposite may be true in areas such as the Southern Rocky Mountains and Southwest, where fuel is drier in the spring than later in the year because of summer monsoon rains. Where fuel loads are higher than the historic conditions, prescribed fire under higher fuel-moisture conditions may have effects similar to historical burns, because the amount of fuel consumed and fire intensity are closer to the natural fire regime (Knapp et al. 2005, Knapp et al. 2009). In one Sierra Nevada study, early-season burns consumed less fuel and had less soil heating because fuels were still moist (Knapp and Keeley 2006). In another, shrub resprouting in the Sierra Nevada was much more common after an early spring burn compared to an early fall burn (Kauffman and Martin 1990). In another, late-fall burn after a substantial rainfall had only a moderate effect on stand conditions because of its low severity (North et al. 2007). Finally, a study in the Sierra Nevada suggests that an early-season prescribed burn had similar impacts as late-season fires on small mammals (Monroe and Converse 2006). In summary, it is worth reiterating a key point from a recent review of the effects of burn season:

A single prescribed burn (or even a few prescribed burns) outside of the historical fire season appear(s) unlikely to have strong detrimental effects. Substantial shifts in community composition often require multiple cycles of prescribed burning. In many ecosystems, the importance of burning appears to outweigh the effect of burn season. (Knapp et al. 2009)

Silvicultural Methods

Thinning

Managers have developed a range of thinning techniques used to remove or change fuels in order to reduce the likelihood of ignition or change fire behavior. Thinnings maintain or improve existing stand conditions without the immediate goal of tree regeneration. In silvicultural terms, thinning treatments fit in one of four categories: crown thinning (i.e., thinning from above), low thinning (i.e., thinning from below), geometric thinning, and free thinning (Smith et al. 1997, Graham et al. 1999, Peterson et al. 2005). A crown thinning removes dominant and codominant trees from the canopy while giving more growing space to the residual dominant and codominant trees. In a fuel reduction context, a crown thinning reduces canopy continuity and bulk density. However, if thinning only reduces bulk density in the upper canopy, crown fire spread rates may remain high (Hunter et al. 2007). A low thinning



Alexander Evans

Thinning treatment, Vermejo Park Ranch, New Mexico

removes intermediate or suppressed trees in favor of dominant and codominant trees. Low thinnings are often used to remove ladder fuels and can also reduce canopy continuity and bulk density. A geometric thinning removes trees based on a predetermined spacing rather than crown position. Often geometric thinnings are used in plantations where rows of trees are removed at a set interval (Smith et al. 1997). In fuel reduction treatments in mixed conifer forests, the most common application of the geometric thinning concept is in shaded fuel breaks. For example, a WUI fuels treatment project near Angel Fire, New Mexico, reduced the basal area of a south-facing mixed conifer stand from 150 to 60 square feet per acre by leaving only one tree every 16 feet. A defensible fuel profile zone (DFPZ) is similar in concept to a shaded fuel break but generally covers a wider area (a quarter mile in width) and makes up part of a landscape plan for fuel reduction (Weatherspoon and Skinner 1996). Free thinnings are usually designed to release individual crop trees from the competition of surrounding trees (Smith et al. 1997). All of these thinning techniques can be modified to accommodate stands that are irregular in age or density. Thinnings can be combined with group or patch reserves of uncut trees in an approach sometimes called variable density thinning (Carey 2003, Peterson et al. 2005).

Even-Aged Regeneration Methods

Although many fuel reduction treatments are thought of as thinnings, in fact managers often use even-aged or uneven-aged regeneration harvests as parts of landscape wildfire hazard reduction strategies in mixed conifer forests. The use of regeneration harvests to change fire behavior is particularly appropriate for mixed conifer forest in which the historic fire regime has included patches of high-severity fire that has initiated tree regeneration.

In mixed conifer forests, patch cuts or clearcuts that remove all trees over a relatively large area have a significant impact on fire behavior. Patch cuts or clearcuts aim at regenerating species that require open conditions for regeneration, such as ponderosa pine or Douglas-fir (Smith et al. 1997). These large-area cuts tend to initiate a new cohort of similarly aged trees and hence are fundamental to even-aged management. Even-aged management is also an effective approach to severe infestations of dwarf mistletoe because it eliminates infestation of young trees from old trees. Managers also use patch cuts or clearcuts to regenerate aspen within mixed conifer forests; although technically this treatment should be called a clear-fell coppice since aspen would be expected to regenerate from existing root systems, not seed (Shepperd et al. 2006).

Uneven-Aged Regeneration Methods

A number of managers interviewed for this report use group selection cuts, or gaps, from a quarter acre to three acres in size to reduce canopy continuity and develop uneven-aged forest structures. The use of a selection system to regenerate a mixed conifer forest helps create heterogeneity by creating a range of ages and densities. In a true group selection system, the goal would be to regenerate the entire stand by creating groups over a long period of time and multiple entries (Smith et al. 1997). For example, Collins Pine Company uses group selection to regenerate mixed conifer forests in the Sierra Nevada with groups of about one and half to two and a half acres in size. Group selection can be combined with low thinnings to produce a stand where ladder fuels and canopy bulk density are lower throughout the stand and newly created gaps reduce canopy continuity. On occasion, high grading—removing high value-trees while leaving smaller, low-quality trees behind—is passed off as individual tree selection. Managers should be wary of harvests that would encourage the growth of shade-tolerant species such as fir

while removing the economic value and seed source for future regeneration of Douglas-fir and ponderosa pine (Graham et al. 1999). In the warm-dry mixed conifer forests of the Southwest and the Central and Southern Rocky Mountains, group selection cuts are designed to regenerate shade-intolerant species by creating openings at least two times the height of mature trees and generally not larger than one acre in size to ensure full sun in the center of the gap. In the cool-moist type, selection cuts are generally designed to be larger in order to develop patches of different ages and structures (such as aspen, mature Douglas-fir and ponderosa pine, or intolerant conifers) in an effort to mimic the openings created by the mixed-severity fire regime. Some managers use the patch or group size of aspen clones in mixed conifer forests as a guide for the natural scale of disturbance and hence an appropriate scale of silvicultural openings.

Jay Francis



Group selection, Sierran mixed conifer, California

Researchers have developed some very specific uneven-aged management systems based on restoration goals. For example, restoration treatments in Southwestern ponderosa pine can replicate pre-settlement stand structure by retaining all live pre-Euro-American settlement-age ponderosa pine trees, oak trees, and snags (Covington et al. 1997). To reconstruct the spatial patterns of pre-settlement forests, these thinnings identify all pre-settlement evidence (stumps, stumps holes, snags, logs) and retain three trees within 60 feet of the pre-settlement evidence if possible (Friederici 2003).

Treatment Combinations: Addressing Slash

Thinning without treatment of the residual slash can increase wildfire hazard, as the Hayfork Fires in California in 1987 illustrated when fuels that were left after a selective harvest resulted in high mortality (Agee and Skinner 2005). Common approaches to slash removal include pile burning, broadcast burning, mastication, and slash removal.

Piling slash and burning it under controlled conditions is often a preferred treatment, because the chance of fire escaping is low and prescription windows are wide (Hunter et al. 2007). Guidelines for the construction of slash piles differ, but guidelines from Larimer County include many common recommendations:

Pile slash immediately after cutting (while still green), and before winter snowfall. Remove all wood products such as firewood prior to piling. Pile branches and tops with the butt ends towards the outside of the pile, and overlapping so as to form a series of dense layers piled upon each other. Use a

mixture of sizes and fuels throughout the pile. This prevents snow from filtering into the pile and extinguishing the fire while it is starting. Piles should be approximately 8 feet across in diameter and 6 feet in height, again to prevent drifting snow from entering the pile. Piles should be kept compact, with no long extensions, to reduce snow filtration and improve ignition. Do not place large stumps and sections of logs in the piles, as they will burn for extended periods and will frequently need to be mopped-up.

www.co.larimer.co.us/burnpermit/slash_burning_guidelines.htm

The combination of thinning and prescribed fire is a particularly useful approach to fuel reduction because, while thinning can alter forest structure (density, canopy base height, canopy continuity, and canopy bulk density), prescribed fire can reduce surface fuel loads and increase canopy base height (Vaillant et al. 2009b). DFPZs often rely on a combination of thinning from below and prescribed fire treatments to reduce surface, ladder, and crown fuel loads (Moghaddas et al. 2010).

Mastication—the mulching, chipping, or grinding of trees, brush, or slash into small pieces—is becoming a more common approach to fuel reduction throughout the West. Vertical or horizontal shaft mastication heads are mounted on an excavator boom or directly on the front of a tracked vehicle (Windell and Bradshaw 2000, Harrod et al. 2009a, Battaglia et al. 2010). In some locations, such as the Cleveland National Forest, mastication fuels treatments focus on reducing areas of brush that are interspersed with a timber overstory by targeting species such as ceanothus and manzanita.

One solution to the fire threat posed by thinning and harvest residues is to take them off-site. In ideal conditions, removing slash from treated stands can both reduce surface fuel and generate income through the sale of biomass. Unfortunately, these ideal conditions rarely occur. One difficulty is that in many locations woody biomass costs more to remove from the forest than it is worth in the marketplace (Evans and Finkral 2009).



Orlando Romero

Pile burning, Carson National Forest, New Mexico

Section V: Fuels Treatment Effectiveness and Requirements

A manager's decision about how to address wildfire hazard must take into account first the objectives for the forests and then the relative effectiveness of each treatment option, their impacts, and the requirements for implementing them. The effectiveness, impacts, and requirements of fuels treatment alternatives differ with each site, but research and managers' experience suggest trends for mixed conifer forests. Managers' decisions about how to address fuels treatment must next be put through the planning process. On many land ownerships, the planning process ensures compliance with an array of regulations and requires consultation with wildlife, archeology, and hydrology specialists. Smoke management may require permits or at least dialogue with air quality regulators. On federal lands, National Environmental Policy Act (NEPA) analyses and similar assessments require managers to consider the effects of fuels treatments on fire hazard and natural resources. Neighboring landowners and the general public are key stakeholders in fuels treatment planning, and encouraging their support for a project can be a key to its success. This section and the next assist in planning fuels treatments. Section V discusses the effectiveness of different treatment techniques and Section VI addresses the potential impacts of fuels treatments on air quality, wildlife habitat, and other forest values.

Effectiveness of Prescribed Fire

Prescribed fire in mixed conifer forests reduces surface fuels effectively without additional treatments (Stephens and Moghaddas 2005a, Schmidt et al. 2008, Stephens et al. 2009b). In one study of prescribed fire in a Sierran mixed conifer stand, prescribed burning significantly reduced the total combined fuel load of litter, duff, and 1-, 10-, 100-, and 1000-hour fuels by as much as 90 percent, thereby reducing modeled fireline intensities, rate of spread, and mortality (Stephens and Moghaddas 2005a). In mixed conifer forests, prescribed fire alone can substantially change forest structure after multiple burns (Keifer et al. 2000, van Mantgem et al. 2011). In parts

of Yosemite National Park and Sequoia and Kings Canyon National Parks, wildland-use fire programs that facilitate multiple burns have successfully reestablished conditions close to pre-settlement forest structure (Collins and Stephens 2007). However, in other areas stand structures with relatively fire-resistant, mature trees are difficult to alter with fire alone. Where there is a high density of mature, relatively fire-resistant trees, thinning combined with fire may initially be the only feasible step. A single fire will not reestablish pre-settlement conditions, in part because



Noah Daniels

Prescribed fire, Grand Canyon National Park, Arizona

of tree mortality caused by the fire. Trees killed by prescribed fires and associated stressors are likely to add to fuel loads over time, and in one example fire-induced mortality occurred more than eight years after the burn (Collins et al. 2010, van Mantgem et al. 2011). In a mixed conifer forest in Sequoia and Kings Canyon National Parks, prescribed fire initially reduced fuel load to 15 percent of pretreatment levels; after ten years fuel loads returned to 85 percent of pretreatment levels (Keifer et al. 2006). Use of low-intensity prescribed fire alone is unlikely to reduce canopy bulk density and raise crown base height sufficiently to reduce the potential for crown fire in a stand, because fire alone is unlikely to kill larger trees or affect canopy conditions. In contrast, high-intensity fire alone can kill canopy trees and move a forest to within the historic range of variability (Miller and Urban 2000). For example, a high-intensity and high-severity fire in the mixed conifer forest of the Grand Canyon National Park returned tree density to within the historic range of variation (Fulé et al. 2004). High-intensity burns that occur when fuels are drier will also consume more surface fuel (Kauffman and Martin 1989, Knapp et al. 2005).

The rate of spread will be slower when surface fuels have higher moisture (Kauffman and Martin 1989). Where mixed conifer understories are dominated by shrubs, prescribed fire may have a slower spread rate than forests dominated by herbaceous cover because shrubs tend to dry more slowly (Korb et al. 2007). In a Sierran mixed conifer forest, both early- and late-season burns effectively reduced fine surface fuels and ladder fuels, but late-season burns reduced large downed woody fuels more than early-season burns (Fettig et al. 2010).

In general, prescribed fire is considered one of the lowest costs per acre for treatment (USDA Forest Service 2005). Nevertheless, there are substantial resource costs involved. Mixed conifer prescribed burns may require more resources because they are often mixed-severity fires. Additionally, prescribed fires in mixed conifer forests are likely to require more planning and resources than lower-severity burns because of relatively high fuel loads. There is also the potential for litigation resulting from an escaped fire, which can be costly (Yoder et al. 2003). In most cases, the cost of prescribed fire on a per acre basis drops as the block size increases (Wood 1988, Rideout and Omi 1995). Similarly, wildland fire-use events have a lower cost per acre than management-ignited prescribed fires (Hunter et al. 2007). As with all treatments, the financial cost of prescribed fire should be compared to the costs (financial and otherwise) of wildfire, which are often much greater (Mason et al. 2006).

Table 6 Average Treatment Costs with Range in Parentheses (Hartsough 2008)

	Prescribed Fire	Thinning	
	Cost	Cost	Revenue
Central Sierra Nevada, CA	\$490 (360 to 923)	\$1,040 (486 to 1578)	\$2,201 (850 to 3,035)
Southern Sierra Nevada, CA	\$413 (368 to 461)	–	–
Southwestern Plateau, AZ	\$125 (101 to 154)	\$700 (769 to 850)	\$704 (486 to 971)

Fuels Treatment for Mixed Conifer Forests



Before treatment



Mastication treatment



One year after mastication



Prescribed burn



One year after prescribed burn



Three years after prescribed burn

Photos of Devil Creek Unit 2 by Sara Brinton, San Juan National Forest

Effectiveness of Thinning and Harvests

In the absence of additional treatments, both thinnings and harvests tend to increase surface fuels even as they decrease canopy bulk density and canopy continuity. The degrees to which thinnings and harvests reduce wildfire hazard are directly related to the silvicultural prescription, i.e., the number of trees cut. Thinning and regeneration harvests both generate significant slash; if left untreated, these surface fuels can result in fire behavior that is more extreme than in untreated areas (Stephens 1998, Innes et al. 2006). For example, after a combination of crown and low thinning in a Sierran mixed conifer stand, 1,000-hour fuels increased four times; because of this increase, the predicted rate of spread, fire line intensity, and flame length also increased significantly (Stephens and Moghaddas 2005a). The same combination of crown and low thinning increased crowning index compared to controls at all percentile weather conditions (Stephens and Moghaddas 2005a). In another Sierran mixed conifer study, single tree selection, overstory removal, thinning from below, and unmanaged stands all averaged approximately 67 tons per acre of surface and ground fuels where slash was left on-site (Stephens and Moghaddas 2005c). A study from the mixed conifer forests of the Sacramento Mountains in New Mexico indicates that a commercial harvest resulted in a crown fire potential two times lower than the control, but increased 1,000-hour fuels (Mason et al. 2007). The 2006 Tripod Complex fires tested the effectiveness of fuels treatment in a mixed conifer forest in Washington: only 36 percent of trees greater than 8 inches DBH survived in the thin-only units compared to more than 73 percent survival where surface fuels had been treated (Prichard et al. 2010). If slash is left on-site, thinning can be relatively inexpensive; as low as \$100 per acre in Colorado (Lynch and Mackes 2003). However, where reduction of wildfire hazard is a goal, it is likely that thinning or harvests will be combined with some treatment of surface fuels.

Effectiveness of Treatment Combinations

Mechanical Treatment and Prescribed Fire

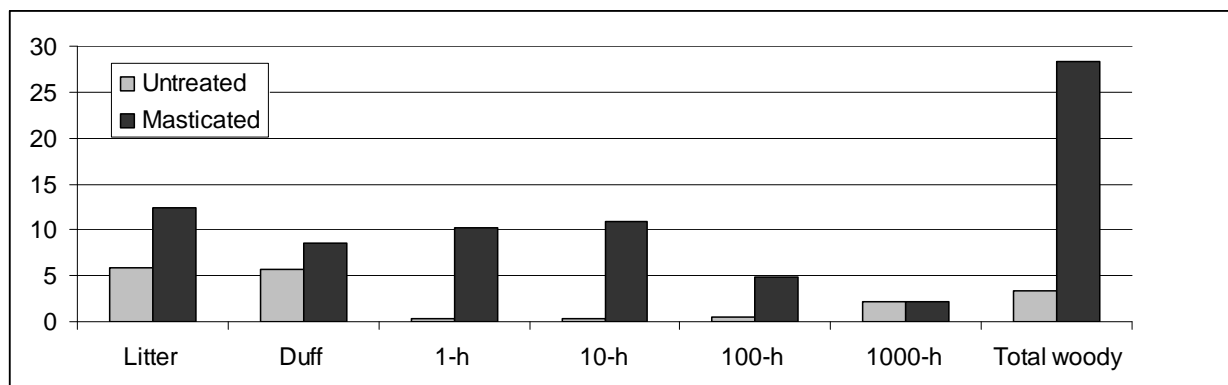
A review of seven sites across the western U.S. as part of the National Fire and Fire Surrogate Study found that mechanical treatment combined with prescribed fire was the most effective at reducing the modeled severity of wildfire effects under extreme weather conditions (Hartsough et al. 2008). Crown thinnings and harvests can reduce canopy bulk density and the potential for active crown fire, and prescribed fire provides a good complement by decreasing surface fuels (Innes et al. 2006, Mason et al. 2007). In a Sierran mixed conifer stand, a combined crown and low thinning followed by prescribed fire significantly reduced predicted tree mortality due to the combination of reduced surface fuels and increased height to crown base ratio (Stephens and Moghaddas 2005a). A simulation based on Sierran mixed conifer data suggests that the increase in canopy base height is more important than the decrease in canopy bulk density in reducing spread rate (Dicus 2009). Thinning, particularly low or free thinning, followed by prescribed fire has been successful in reducing wildfire hazard and returning forest structural conditions to within HRV (Fulé et al. 2002, North et al. 2007). In a simulation of treatments in Sierran mixed conifer, harvests followed by fire provided the quickest path to restoring at least three aspects of forest structure and composition to historic conditions (Miller and Urban 2000). During the 2007 Angora Fire in the Lake Tahoe Basin, California, combined thinning and pile burning treatments

reduced bole char height, crown scorching, torching, and mortality (Safford et al. 2009). Notably, the Lake Tahoe treatments were effective in changing fire behavior from an active crown fire to a surface fire (Safford et al. 2009). On the Lassen National Forest, managers have found that mechanical piling is more efficient than hand piling on larger treatments, and grapple piling adds less soil to the pile than a Bobcat or bulldozer piling. However, in the WUI, neighbors prefer seeing hand piling rather than machines and near streams hand piling has less risk of sediment runoff. Many managers report burning piles when there is snow cover to help with control. The effectiveness of mechanical treatments combined with pile burning is similar to that of broadcast burning. Burning slash piles can be labor intensive where fuels are hand piled, costing \$150 to \$850 per acre to implement, depending on the amount of fuel (Han et al. 2010).

Mastication

Mastication can include the chipping of small standing trees without additional thinning or the chipping of slash and fuel already on the ground after another mechanical treatment. Where mastication includes chipping small standing trees, it has the same effect as a low thinning combined with mastication of slash. Mastication does not remove fuel from the stand as prescribed fire does; rather, it changes fuel characteristics. A study of mastication in a ponderosa pine plantation documents an increase in surface fuel and decrease in canopy fuel with just mastication and a decrease in both surface and canopy fuel loads when mastication was combined with prescribed fire (Reiner et al. 2009). Mastication can increase surface fuel depth and continuity, allowing fires to spread more easily and burn hotter at the soil surface (Stephens and Moghaddas 2005a, Harrod et al. 2008, Reiner et al. 2009). Managers on the Truckee National Forest have found that masticated fuels can be difficult to ignite but, once ignited, can result in high levels of tree mortality, which can hinder future prescribed fire. Soil heating during post-mastication fires has the potential to cause biological damage, particularly in dry soil with a mulch depth of 3 inches or greater (Busse et al. 2005). Mastication can inhibit herbaceous species growth and tree regeneration because of reduction in available light, soil temperatures, and seed access to mineral soil (Resh et al. 2005). Mastication in a mixed conifer stand in Colorado increased surface fuels three times and resulted in 28 tons of surface fuel per acre (Figure 3) (Battaglia et al. 2010). Median fuelbed bulk density after mastication was approximately 8.6 pounds per cubic foot (Battaglia et al. 2010).

Figure 3 Fuel by Time-Lag Classes for Mixed Conifer Stands in Colorado (Battaglia et al. 2010)



In a 25-year-old pine plantation, surface fuel loads were 15 tons per acre and bulk density was 7.8 pounds per cubic foot after thinning and mastication (Reiner et al. 2009). A survey of ten sites across Northern California and Southern Oregon found that woody surface fuels ranged from 6.8 to 28 tons per acre and bulk densities from 2.9 to 7.2 pounds per cubic foot (Kane et al. 2009). As with any treatment, costs of mastication vary significantly with site conditions and treatment goals, but in one study the cost per acre averaged \$452 (Harrod et al. 2008).

Removal

The removal of material from a site can reduce the amount of fuel and, potentially, the wildfire hazard. For example, cut-to-length harvesting in a Sierran mixed conifer stand doubled the total fuel loads, but whole tree harvesting had little effect on post-treatment surface fuels (Walker et al. 2006). A number of managers mentioned that whole-tree harvesting is a very effective tool for fuel reduction, where markets make it possible. Similarly, slash removal requires an outlet for the material, usually a commercial market. For example, material removed from ponderosa pine and mixed conifer stands in the Nutrioso Wildland Urban Interface Fuels Reduction Project on the Apache-Sitgreaves National Forests supplies small power plants and a wood-heating-pellet manufacturing facility (Neary and Zieroth 2007). In general, slash or small tree removal is relatively costly because the small piece size of the slash has high handling costs, and most forest harvesting systems were originally designed for larger-diameter timber (Han et al. 2004, Ralevic et al. 2010). The cost of transporting forest biomass is also often high, because the distance to markets is often long (Becker et al. 2009). These prices are similar to estimates from 2005 for the cost of bringing woody biomass to the roadside, which ranged from \$400 to \$1,630 per acre depending on forest type and terrain and had a median value of \$680 for gentle slopes (USDA Forest Service 2005).



Jon Arnold

Biomass removal, Lassen Volcanic National Park, California

Maintenance of Treatments

Forests grow, fuels accumulate, and managers must repeat or maintain treatments in order to sustain their effects on wildfire hazard. Of course, treatments create different conditions, as described above, and so the interval between treatments to maintain a particular level of fire resilience will differ. Usually fire-only will require follow-up sooner than combined mechanical and prescribed-fire treatments (Hartsough et al. 2008). It is likely that treatment intervals should be similar to the historic fire return interval, since treatments are a surrogate for natural fire. Estimates for the longevity of prescribed burn effects range from ten to 14 years in the Sierra Nevada (van Wagtenonk and Sydoriak 1987, van Wagtenonk 1995, Graham et al. 2004, Keifer et al. 2006). Restoration of historic forest structural patterns such as dead wood accumulations requires repeated prescribed burns (Innes et al. 2006, van Mantgem et al. 2011). Ten years post-harvest, surface loads remained low in a clearcut in Sierran mixed conifer where slash had been piled and burned, though high horizontal fuel continuity and high hazards had developed (Stephens and Moghaddas 2005c). In general, fire severity increases with time since treatment, but decreases with number of prescribed burns (Finney et al. 2005). Fuel accumulation rates can be estimated based on species, crown height, and live crown ratio (van Wagtenonk and Moore 2010). In some circumstances, the first post-harvest prescribed fire may not consume all the 100-hour fuels created by the thinning or harvest (Schmidt et al. 2008). Mechanical treatments combined with prescribed fire are effective at reducing wildfire hazard, but no single treatment will completely mitigate nearly a century of fire exclusion and fuel accumulation (Youngblood 2010).

Integrating Wildlife Objectives

Managers of mixed conifer forests must integrate wildlife objectives (maintaining, improving, or protecting habitat) into fuels treatment. While it is beyond the scope of this guide to describe all the wildlife habitat issues related to mixed conifer forests, spotted owls provide a good example. Spotted owls have a particularly large influence on management because of recovery plans and other guidance (Verner et al. 1992, USFWS 1995). There are three different subspecies with similar habitat requirements (Northern, Californian, and Mexican) within the mixed conifer forests covered in this report. With all three, there is a perception that conservation of spotted owl through habitat protection conflicts with fuels treatments (Prather et al. 2008). However, at the landscape scale, the area in which high priority treatments and spotted owl habitat overlap is relatively small—only about one third of the area (Prather et al. 2008, Gaines et al. 2010). One of the reasons for the perception of conflict between habitat and fuels treatment is the spotted owl's requirement for dense forests with high canopy closure and multiple canopy layers (USFWS 1995, Gaines et al. 2010). For the Mexican spotted owl, the recovery plan calls for the retention of high canopy density in nesting and roosting habitat called protected activity centers (PACs) (USFWS 1995), while the California spotted owl guidelines recommend no stand-altering activities within PACs other than light underburning (Verner et al. 1992). Outside the PACs in spotted owl habitat, the California guidelines call for retention of trees 30 DBH and greater, as well as 40 percent canopy cover (Verner et al. 1992). Similarly, the Mexican spotted owl recovery plan recommends management of 25 percent of the landscape for future nesting and roosting habitat; this area should consist of stands that have greater than 130 square feet of basal area per acre and include more than 20 trees per acre that are greater than 18 inches DBH

(USFWS 1995). The recovery plan recommends maintaining the remaining 75 percent of the landscape for foraging habitat, which generally has much low tree densities and is often managed for uneven-aged structure and retention of large downed logs (USFWS 1995).

In general, spotted owl nesting and roosting habitat in mixed conifer forests is dense, with a relatively high potential for fire; some authors have suggested fuels treatments be implemented to reduce the risk of habitat loss from wildfires (Everett et al. 1997, Lehmkühl et al. 2007). In prescribed fires on the Cibola National Forest, managers have been able to use ignition methods such as backing fire to keep fire intensities low and minimize impacts to spotted owl habitat. The approach to spotted owl habitat on the Coconino National Forest is to use group selection to create stand conditions favored by owls (i.e., open gaps for regeneration), reduce densities in the rest of the stand to encourage growth of larger diameter trees, and retain old trees of species that are more fire resistant in groups of up to four acres in size.

The effects of fire on spotted owl habitat are complex, but high-severity fire can not only kill individual owls but also leave low-quality breeding habitat (Bond et al. 2002, Bond et al. 2009). Low- to moderate-severity fires appear to have little to no effect on spotted owl survival and may even increase reproductive rates post fire (Bond et al. 2002). Similarly, a combination of population data, canopy cover measurements, and forest simulation models indicate that mechanical thinning with prescribed fire would not degrade canopy conditions in productive owl territories (Lee and Irwin 2005). Moreover, increasing heterogeneity within mixed conifer forests can fit both spotted owl habitat goals and fuels treatment goals (Weatherspoon and Skinner 1996, Prather et al. 2008, Gaines et al. 2010).

Moonlight Fire

The Moonlight Fire started on September 3, 2007, and by the time it was contained on September 15, it had burned nearly 65,000 acres, mostly on the Plumas National Forest in northern California. Dry conditions, steep slopes, heavy fuel loads, and frontal winds contributed to high-severity fire behavior over large areas. The Moonlight Fire provides a glimpse into the effectiveness of fuels treatments and fire behavior in spotted owl PACs because it burned through 22 PACs and 25 core areas. There was a 75 to 100 percent reduction in canopy cover on 64 percent of the PAC acreage, while there was a 75 to 100 percent canopy cover reduction on 68 percent of the owl core areas. In general, fire behavior was more severe with higher canopy cover crown reduction in untreated areas, including protected owl habitat, compared to treated areas. The Moonlight Fire corroborated research cited in this report that thinning combined with prescribed burning is most effective at reducing burn severity. The reduction in canopy cover was significantly greater in protected owl habitat than treated areas, and it may even have been greater than other untreated areas (though this difference was not statistically significant). Perhaps most importantly, the Moonlight Fire likely reduced the utility of this area for spotted owls (Dailey et al. 2008).



USDA Forest Service

Jemez Mountains Salamander

The Jemez Mountains salamander (JMS) is another example of a rare endemic animal that lives in mixed conifer forests. JMS is already on the endangered species list in New Mexico. In September 2009, a U.S. Fish and Wildlife review indicated that JMS warrants being listed as an endangered species under the Endangered Species Act, although currently its listing is precluded by higher priority actions. JMS lives under and in fallen logs and old, stabilized talus slopes, especially those with a good covering of damp soil and plant material, which are key habitat elements for the species (Reagan 1972).

One of the biggest threats to JMS habitat is large, stand-replacing crown fire in its mixed conifer forest habitat. Unfortunately, there is little scientific information on JMS tolerance of thinning or prescribed fire fuels treatment. One concern is that managers will avoid treatments in JMS habitat because of this uncertainty, and the risk of stand-replacing fire will increase. Therefore, a collaborative group of scientists and managers are investigating the fire history and historic stand structure along with current stand conditions in salamander habitat.



Mark Watson

Jemez Mountains salamander

Surveys for salamanders in burned and unburned habitat combined with the forest stand structure information will improve our understanding of how the species responds to fire disturbance. Collaborators hope to develop a set of possible management approaches to mixed conifer JMS habitat. In general, treatments will maintain or improve habitat for JMS by increasing the amount of downed logs and reducing the risk of stand-replacing fires.

A final example from the numerous other animals whose habitat managers consider in mixed conifer forest fuels treatments comes from Gunnison Ranger District of the GMUG National Forests. In the Taylor Canyon prescribed burn, the primary goal was to open up migration corridors for bighorn sheep between summer and winter ranges. The other goal for the burn was to increase the quality and quantity of summer range forage and improve forest health. The project covered 31,640 acres, of which a little more than half was designated primary and hand or aerially ignited with no control. The remainder of the area was a buffer around the primary ignition units where spotting and spread from the primary area was allowed to burn under specific conditions.

Insects and Diseases

Aspen Decline

Sudden aspen decline (SAD) has caused significant concern, particularly in southwestern Colorado, where it has caused the rapid decline of entire aspen clones. By 2008, SAD had affected at least 544,000 acres, about 17 percent of the aspen cover type in Colorado (Worrall et al. 2010). SAD appears to be related to moisture status and not to overstory age or diameter (Worrall et al. 2010). The 2002 warm drought may have been the trigger for current outbreak of SAD (Rehfeldt et al. 2009, Worrall et al. 2010). Managers report that aspen stands on southern

aspects and lower elevations are more likely to be affected by SAD. Preliminary data suggests that overstory removal treatments in stands affected by SAD on the San Juan National Forest, Mancos-Dolores Ranger District, have resulted in good aspen regeneration.

William M. Ciesla



Declining aspen stands, Carson National Forest, New Mexico

Even before the identification of SAD, there were concerns about aspen decline (Bartos and Campbell 1998, Shepperd et al. 2006). For example, areas covered by aspen in the Southwest had declined by 62 percent (Johnson 1994), but aspen is not in decline in all areas (Kulakowski et al. 2004). One reason that aspen is on the wane is lack of fire. Aspen is fire adapted, and fire can provide the black soil and open-canopy conditions that encourage sprout growth (Bartos et al. 1994, Kaye et al. 2005, Shepperd et al. 2006). Although there is no single solution to aspen decline, the

phenomenon highlights the need for returning fire to mixed conifer forests. The pressure of elk and other herbivores means that small projects to encourage aspen can easily be overwhelmed (Kaye et al. 2005, Beschta and Ripple 2010). Elk browse can result in failure of aspen to regenerate at levels as low as 13 elk per square mile (Suzuki et al. 1999, White et al. 2003). Aspen regeneration requires a landscape approach to ensure sufficient acres are regenerated in order to overwhelm herbivores.

Mountain Pine Beetle

Managers of mixed conifer forests are also struggling with the impact of the mountain pine beetle (MPB). In the Rocky Mountains, an outbreak of MPB has killed lodgepole pine on millions of acres in Colorado and Wyoming. In Colorado, MPB was active on 1,046,000 acres in 2009 and 878,000 acres in 2010 (Colorado Department of Natural Resources 2010). MPB also attacks ponderosa pine and its effect on ponderosa pine in the Front Range has increased in 2010 to 229,000 acres (Colorado Department of Natural Resources 2010). In most stands



USDA Forest Service

Mountain pine beetle damage Bear Mountain Basin, Colorado

lodgepole pine will regenerate and replace the beetle-killed stand (Klutsch et al. 2009). However, in some cases MPB is removing the lodgepole component from mixed conifer stands or allowing aspen to regenerate once the lodgepole overstory is killed. Managers in the region are working to promote increased diversity of species and age class in the MPB affected areas. Without such intervention, the region may develop a large homogeneous cover of lodgepole pine that will be susceptible to another region-wide MPB outbreak in the future. In Rocky Mountain National Park, after the MPB outbreak subalpine fir, Engelmann spruce, and aspen are relatively more abundant, but lodgepole pine continued to dominate 85% of the landscape (Diskin et al. 2011). Within mixed conifer stands, the MPB-induced mortality of lodgepole and, to a lesser degree, ponderosa pine underscores the importance of species diversity, because other species are available to fill the growing space. For instance, managers on the Medicine Bow-Routt National Forest and other areas report seeing aspen regeneration in stands formerly dominated by lodgepole pine. Another reason the MPB affects management of mixed conifer stands is not ecological, but rather administrative: the MPB outbreak has consumed much of the attention and resources of most forest management organizations in the region.

White Pine Blister Rust

WPBR is caused by a rust fungus that infects five-needle pine species in North America as well as *Ribes* species such as currants and gooseberries (Maloy 1997). The rust requires both pine and *Ribes* species to complete its lifecycle. The bark of infected pines swells and initially shows a yellowish discoloration; then a canker forms, and the branch or trunk eventually dies (Lachmund 1926). WPBR has spread through western North America at a rate of about 6 miles a year since it was introduced in 1923 (Evans and Finkral 2010). Five-needle pines are an important part of the species diversity in mixed conifer stands.

For example, white pines tend to be less susceptible to root diseases, bark beetles, and windthrow than other species in mixed conifer forests (Samman et al. 2003, Tomback and Achuff 2010). There is significant genetic resistance in five-needle pine populations, and by protecting those individual pines that appear resistant to WPBR management can help maintain the species on the landscape (Jacobs et al. 2009, Schwandt et al. 2010). A number of managers reported that they include protecting genetic diversity among five-needle pines in fuels treatment plans and establish a preference for retaining healthy five-needle pine in marking guides (Conklin et al. 2009, Zeglen et al. 2010). Management efforts to encourage and protect five-needle pines are linked to returning fire to its natural role in mixed conifer stands, because pines are shade-intolerant and establish on the bare mineral soil created by fire (Tomback and Achuff 2010).



USDA Forest Service R3-FH-09-01

Fruiting stem canker of white pine blister rust on southwestern white pine

Section VI: Fuels Treatment Impacts, Mitigation, and Monitoring

Any treatment will have both positive and negative impacts on a forest, many of which are difficult to measure. The following section details the available research on measured impacts of the treatments described in Section V.

Mortality

Mortality induced by fire is likely to be most common in more fire-susceptible species, such as white fir, but high-severity fire can kill large trees of any species (Fulé et al. 2004). There is some debate about the effect of raking surface fuels away from large trees, with recent studies showing little benefit to raking (Fowler et al. 2010, Noonan-Wright et al. 2010). This may be due in part to the fact that the effectiveness of raking varies with fire intensity. When fire intensity was very low or very high raking did not affect mortality, but at moderate fire intensity it increased survival by nearly 10 percent (Nesmith et al. 2010). In addition to raking, thinning trees adjacent to high-value trees can reduce mortality over the long term (Kolb et al. 2007). Burning slash in piles can have more severe but localized effects because of the increased heat and long residence times on soils (Seymour and Teclé 2005), plant establishment (Korb et al. 2004), and adjacent vegetation (Hillstrom and Halpern 2008). The pile location (i.e., away from high-value trees), size, and burning conditions affect the amount and rate of pile combustion (Johnson 1984, Hardy 1996).

Alexander Evans



Blowdown, Santa Fe National Forest, New Mexico

Another mortality issue with treatments is blowdown. A number of managers mentioned that wind events and wet soils contribute to the uprooting and snapping of trees after a thinning. Thinnings that create relatively open environments for trees in previously dense stands can increase wind damage to remaining trees (Kolb et al. 2000). Thinning can increase wind speeds and soil moisture (Ma et al. 2010), and these in turn can increase susceptibility to wind damage. In general, trees with a higher percentage of live crown, stands with greater post-harvest stand density, and group reserves rather than individual tree reserves are more resistant to wind damage (Scott and Mitchell 2005). In the Sacramento Mountains of New Mexico,

managers retain at least 80 square feet of basal area per acre to avoid blowdown. Managers mentioned that even where MPB had killed lodgepole pine within mixed conifer forests, there was blowdown within remaining trees.

Insects and Diseases

Fire-caused mortality continues for a number of years after fire, in part because of the activity of bark beetles in weakened trees. The impact of beetles after treatment is heavily influenced by beetle population levels during and after fire (Fettig and McKelvey 2010). After an early-season burn in a Sierran mixed conifer forest, beetles did not cause extensive mortality in large-diameter trees: western pine beetle killed less than 1 percent of trees, mountain pine beetle less than 1 percent, red turpentine beetle about 9 percent, and *Ips* species combined killed about 3 percent (Fettig et al. 2010). Most beetle-induced mortality occurs within 2 years after fire, and crown injury is a key predictor of post-fire beetle-induced mortality (Hood et al. 2010). Piling slash can also facilitate the increase in bark beetles, which in turn can cause mortality in residual trees (Six et al. 2002). Where background populations of bark beetles are low, the impact of bark beetles after treatment is short term and limited to fire-damaged trees (Six and Skov 2009). Raking surface fuels away from large trees has been shown to reduce beetle activity after prescribed fire (Nesmith et al. 2010). In general, lower-density stands where trees compete less for water are most resistant to bark beetles (Fettig and McKelvey 2010). For instance, mortality caused by fir engraver beetles in a Sierran mixed conifer stand during a multiyear drought was lower in thinned stands, and mortality increased with the proportion of white fir (Egan et al. 2010). Thinning treatments can exacerbate root disease, but in a study of Sierran mixed conifer forest fewer than 12 percent of cut stumps were infected with root pathogens such as *Armillaria gallica* and *Heterobasidion annosum* (Maloney et al. 2008). By opening up growing space for understory species, treatments can also increase the density of *Ribes*, the alternate host of WPBR, and potentially increase infection in five-needle pines (Maloney et al. 2008).

Stands that are heavily infested with dwarf mistletoe are likely to have higher surface fuel loads and hence a high wildfire hazard (Hoffman et al. 2007). Prescribed fire can reduce these surface fuels, and with sufficient intensity (generating 30 to 60 percent average crown scorch) underburning can help control dwarf mistletoe in ponderosa pine stands, in part by scorch-pruning infected branches (Conklin and Geils 2008). Dwarf mistletoe infection, particularly severe infections, reduces survival of scorched ponderosa pine trees (Conklin and Geils 2008). Thinning is not a recommended method for



USDA Forest Service R3-FH-10-01

Mature Douglas-fir with heavy dwarf mistletoe infection

reducing moderate to severe dwarf mistletoe infestations, because it will stimulate the remaining mistletoe (Conklin and Fairwather 2010). Thinnings in mixed conifer stands designed to remove ponderosa pine dwarf mistletoe can accelerate the conversion to fir, while in stands with infected Douglas-fir and a healthy pine component dwarf mistletoe will favor the pine (Conklin and Fairwather 2010).

Habitat

The habitat impacts of fuels treatments depend on the species of interest, so this discussion focuses on general trends and the most common concerns. Other potential habitat impacts related to fuel reduction treatments, such as road construction (Forman and Alexander 1998), should be considered but are beyond the scope of this report.

Although in the past treatments have avoided riparian areas, recent research suggests that the impact of prescribed fire on riparian areas within mixed conifer forests is small. In a mixed conifer forest in Idaho, there were no detectable changes in periphyton, macroinvertebrates, amphibians, or fish for three years after prescribed fire treatment (Arkle and Pilliod 2010).

Alexander Evans



Apache Canyon, Santa Fe National Forest, New Mexico

Similarly, a range of fuel reduction treatments in a Sierran mixed conifer forest increased habitat heterogeneity at the compartment level, providing additional habitat for rare species (Apigian et al. 2006). Prescribed fire after thinning in riparian areas had only a small effect on bird density and a near-term effect on reproductive success (Stephens and Alexander 2011). One possible reason for the minimal impact of prescribed fire on riparian systems is that historically they burned at similar frequency to the surrounding mixed conifer forest in many cases. Though Sierran mixed conifer upland forests exhibited a greater degree of fire-climate synchrony, a survey of 36 sites indicates they have similar fire return intervals (Van de Water and North 2010). Similar links between fire in upland mixed conifer and riparian areas have been identified in the Cascade Range of Washington (Everett et al. 2003).

Vegetation generally recovers quickly after prescribed fire, in part because more growing space is available for herbaceous and understory plants. Forbs and graminoids returned to pre-treatment

abundance after all combinations of thinning and burning in a Sierran mixed conifer forest (Collins et al. 2007). Thinning and burning can be effective for increasing understory diversity and reducing shrub cover (Wayman and North 2007). The timing of burning does have an impact on herbaceous and understory plants, and there was a temporary but significant drop in cover and a decline in species richness the year following a late-season burn in a Sierran mixed conifer forest (Knapp et al. 2007). Again in a Sierran mixed conifer forest, shrubs and hardwood survival was lower (12 percent) after early fall burns than after early spring burns (79 percent) (Kauffman and Martin 1990). Crown thinnings that promote shrub growth in previously uncharacteristically dense stands may correlate with an increased abundance of nesting birds (Siegel and Desante 2003). A meta-analysis of 22 studies showed that low thinnings and prescribed fire treatments had positive effects on most small mammals and passerine bird species, while selective harvests had no detectable effect (Kalies et al. 2010). In contrast, overstory removal and wildfire both resulted in an overall negative effect on most small mammals and passerine bird species (Kalies et al. 2010). For example, neither early- nor late-season prescribed fire in Sierran mixed conifer had an effect on deer mouse populations, chipmunk populations, or total small mammal biomass (Monsanto and Agee 2008).

Any forest management program risks the introduction of exotic species (Keeley 2006). Since thinning and burning combinations tend to cause larger changes in forest structure than burning alone, combined treatments can open growing space for exotic species (Collins et al. 2007). Similarly, more intense harvest (i.e., shelterwood versus group selection) opens up more growing space and can have a greater proportion of exotic species (Battles et al. 2001). Fuel breaks can provide corridors of invasion and facilitate the spread of exotic species into wildlands (Keeley 2006). In the southern Sierra Nevada, unburned coniferous forests had few if any exotic species, but some of the burned forests sampled had significant populations of exotic species (Keeley et al. 2003). A study in a Sierra Nevada mixed conifer forest indicates that allowing greater surface fuel buildup inhibits cheatgrass, both by physically blocking establishment and by creating more intense fires that kill more of the seed bank (Keeley and McGinnis 2007). However, for Southwestern ponderosa pine forests, maintaining low-intensity burns may be a more appropriate way to limit cheatgrass (James 2007). In addition, repeated burning can increase the exotic grass population (McGinnis et al. 2010). Of course, wildfires also open up growing space for exotic species. For example, both the 2002 Hayman Fire in Colorado and the 2003 Cedar Fire in the mixed conifer forests of the Peninsular Ranges stimulated an increase in exotic species, particularly in severely burned areas (Franklin et al. 2006, Fornwalt et al. 2010).



Chris Evans

Cheatgrass

Dead Wood

Dead wood is an important habitat element in mixed conifer forests. Thinning treatments combined with prescribed fire have been shown to increase snag density and eventually result in downed dead wood buildup (Boerner et al. 2008, Harrod et al. 2009b). In Sierran mixed conifer forests, both thinning and prescribed fire have been shown to reduce downed dead wood quantities, while burning reduces piece size (Knapp et al. 2005, Stephens and Moghaddas 2005b, Innes et al. 2006). In contrast, thinning without follow-up treatment increases downed dead wood, particularly fine material (Stephens 1998, Stephens and Moghaddas 2005a, Innes et al. 2006). Even low-intensity fires can remove a large portion of downed dead wood. For example, a burn in an Arizona ponderosa pine forest consumed 99 percent of large, rotten wood (Covington and Sackett 1984), though in most cases downed dead wood consumption is less, i.e., between 44 and 69 percent (Covington and Sackett 1984, Sackett and Haase 1996, Youngblood et al. 2006). The impact of prescribed fire on dead-wood-dependent arthropods may be minimized if refugia of litter and coarse woody debris are retained (Niwa et al. 2001). Many mixed conifer forests of the Southwest currently have lower snag densities than the U.S. Forest Service management target of three per acre (Ganey and Vojta 2005). The average large snag (greater than 18 inches in diameter) density in mixed conifers forests in northern Arizona is 1.4 per acre in managed forests and 2.8 in unmanaged forests (Ganey and Vojta 2005). In contrast, most mixed conifer forests sampled in northern Arizona exceeded U.S. Forest Service guidelines for retention of large downed logs, and 30 percent exceed guidelines for overall downed dead wood (USDA Forest Service 1999, Ganey and Vojta 2010). Recommended management targets for downed dead wood include eight to 16 tons per acre (USDA Forest Service 1999), five to ten tons per acre for the warm-dry ponderosa pine and Douglas-fir forest types (Brown et al. 2003), and ten to 20 tons per acre for cool Douglas-fir types (Brown et al. 2003).

Soils

A meta-analysis of 26 studies shows that, in general, forest harvesting has little or no effect on soil carbon and nitrogen (Johnson and Curtis 2001). Sawlog harvesting can increase soil carbon and nitrogen (18 percent increase) and whole-tree harvesting can result in decreases (6 percent decrease) (Johnson and Curtis 2001). Similarly, a meta-analysis of 12 studies shows that fire resulted in no significant short-term effects on either carbon or nitrogen, but there was an increase in both soil carbon and nitrogen after 10 years compared to controls (Johnson and Curtis 2001). In a mixed conifer forest in the Sacramento Mountains of New Mexico, low-to-moderate soil disturbance by mechanical operations did not result in increased runoff or sedimentation compared non-disturbed sites, even on steep slopes (Cram et al. 2007). Studies in the Lake Tahoe basin suggest that thinning followed by burning



Alexander Evans

Burned log, Carson National Forest, New Mexico

Fuels Treatment for Mixed Conifer Forests

Surface fuel from the Colorado Front Range Photo Series, Douglas-fir type (Battaglia et al. 2005, www.treesearch.fs.fed.us/pubs/9766)



10.7 tons per acre



19.7 tons per acre



37.9 tons per acre

Surface fuel from the Northern Sierra Photo Series, mixed conifer-pine type (Blonski and Schramel 1981, www.treesearch.fs.fed.us/pubs/27007)



12.6 tons per acre



28.2 tons per acre



46.6 tons per acre

can increase overland flow and litter interflow nutrient loading in the short term; however, wildfires such as the 2002 Gondola Wildfire have a much larger impact (Miller et al. 2010). A U.S. Forest Service study estimated that 70 acres of thinning in western forests yield about the same amount of sediment as one acre consumed in wildfire (USDA Forest Service 2005). Thinning both with and without follow-up prescribed fire increased soil moisture in a Sierran mixed conifer stand (Ma et al. 2010). With good logging practices, thinning projects have minimal effects on soil compaction (McIver et al. 2003, Ares et al. 2007). Compaction is a particular concern at high levels of soil moisture, and under these conditions cut-to-length systems cause less compaction than whole tree harvests (Han et al. 2009). Avoiding sensitive soils, using designated or existing harvesting traffic lanes, and leaving some slash in high traffic areas can reduce soil compaction (Page-Dumroese et al. 2010).

Smoke

Smoke from prescribed fires can be a significant impact, and many managers described public and regulatory reactions to smoke as an impediment to burning in mixed conifer. Smoke from prescribed fire can affect public health, visibility, and traffic safety (Sandberg et al. 2002). Smoke can be a particularly important issue in the WUI or where air patterns move smoke into urban areas and other smoke sensitive areas (Wade and Mobley 2007). Smoke from prescribed fire falls under the overarching regulatory framework of the Clean Air Act, but is often further regulated at the state or local level (Hardy et al. 2001). A number of counties in California and

NASA



Plumes of smoke from the fires (outlined in red) in the Sierra Nevada, California, November 2002

Southern Arizona with mixed conifer forests are non-attainment areas under the National Air Quality Standards (www.epa.gov/air/data/). In these non-attainment areas, emissions from prescribed fire are of particular concern because pollution levels are already above the limits that may affect human health (Riebau and Fox 2001). In addition, national parks and wilderness (more than 6,000 and 5,000 acres respectively) are Class I airsheds and subject to tight pollution restrictions. Smoke production is driven by factors that include the quality of fuel and fuel moisture. In general, high moisture content will increase smoke because more of the fuel will be consumed during the residual and smoldering phases (Wade and Mobley 2007).

Most managers reported generally good relationships with air quality boards and other smoke regulators (though some managers indicated local air quality boards were tough to work with). Wildfires may be opportunities to engage regulators in a conversation on the benefits of prescribed fire in reducing wildfire threat. For example, since the Angora Fire in Tahoe Basin, the air quality board has been more permissive for burn days. This increased permissiveness has been good for prescribed fire implementation. Even in places where fire managers have good relationships with air quality regulators, smoke management and smoke transport is a central challenge. For example, on the Placerville Ranger District of the Eldorado National Forest, urban developments and roads or highways at the bottom of steep drainages are a particularly difficult smoke management situation because air flows down canyons at night and inversions hold smoke close to the ground. In some cases, innovative strategies are required to avoid impacting population centers or other smoke-sensitive areas. For example, the Coconino National Forest in Arizona found that igniting prescribed fires during windy periods before approaching cold fronts helps disperse smoke, avoiding smoke-sensitive areas (Hunter et al. 2007).

Removal of biomass from mixed conifer forests presents the opportunity to reduce smoke and carbon emissions from burning in the forest while still reducing fuel loads (Jones et al. 2010). Smoke emissions are reduced because less fuel translates directly into less smoke. For example, in a Bureau of Land Management fuel reduction project in the WUI of Clancy, Montana, managers searched for and found an off-site utilization for biomass because of smoke concerns (Evans 2008).

Carbon

The increase of tree density in mixed conifer forests has made them a sink for carbon and an offset to the rising concentrations of greenhouse gases in the atmosphere (Sohngen and Haynes 1997, Houghton et al. 2000). While fuel reduction treatments release carbon into the atmosphere, they also decrease the likelihood that wildfire will cause even greater releases. For example, fuel reduction treatments in a Sierran mixed conifer forest reduced carbon stored in live trees, but without treatment 90 percent of the live trees had a high (greater than 75 percent) chance of being killed and eventually releasing carbon if a wildfire burned the area; risk for treated stands was significantly lower (Stephens et al. 2009b). Similar results have been shown for other forests (Finkral and Evans 2008, Sorensen et al. 2011). A study from Montana demonstrated that in a warm-dry mixed conifer forest treatments decreased fire severity, reduced subsequent wildfire emissions, and increased carbon storage; but in a cool-moist mixed conifer stand the untreated area had greater wildfire emissions but stored more carbon (Reinhardt and Holsinger 2010).



East Magdalena prescribed burn, New Mexico

Lynn Moore

The carbon impact of removal and utilization of forest biomass from thinnings depends on the fate of the material removed. If material is burned to generate heat or electricity, and thereby offsets fossil fuel use, then net carbon emissions are reduced (Finkral and Evans 2008, Eriksson and Gustavsson 2010). However, the tradeoffs of carbon storage and fuel reduction are still being debated. Some authors argue that the avoided carbon release from wildfires makes up for short-term carbon emissions from treatment (e.g., North and Hurteau 2011). Others argue that treatments increase overall carbon emissions even in comparison to wildfire (e.g., Mitchell et al. 2009). The discussion of carbon costs and benefits of fuels treatment is in part based on methods of analysis. For example, the carbon storage of ponderosa pine stands where treatments occurred was only greater than untreated wildfire-burned stands if the carbon in long-lived wood products and avoided fossil fuel use was included in the analysis (Sorensen et al. 2011).

Mastication

The impacts of mastication treatments, particularly in the absence of post-treatment burning, are different from other treatments because of the quantity and type of material left on-site (Kane et al. 2009). For example, mastication in a Sierran mixed conifer forest resulted in different effects on arthropod communities than prescribed fire treatments (Apigian et al. 2006). Mastication supported a significantly different understory plant community composition compared to plots that were thinned but not masticated (Wolk and Rocca 2009). Mastication without burning can reduce shrub cover (Collins et al. 2007). Herbaceous species growth and tree regeneration is inhibited by reduction in available light, soil temperatures, and seed access to mineral



Alexander Evans

Masticated slash, Lake Tahoe Basin, California

soil (Resh et al. 2005, Kane et al. 2006). Mastication can remove large downed logs that provide wildlife habitat (Harrod et al. 2008). Because masticated fuels burn hotter at the soil surface, there is potential for increased tree mortality (Stephens and Moghaddas 2005a, Harrod et al. 2008, Reiner et al. 2009). Raking masticated material back from trees can reduce tree mortality (Reiner et al. 2009, Vaillant et al. 2009a). In open forests such as piñon-juniper, mastication can lower soil temperature and increase soil moisture, which may help increase plant cover and richness compared to untreated plots (Owen et al. 2009). Mastication can increase plant available nitrogen in mixed conifer forests (Battaglia et al. 2009). In another study from Colorado, the physical exclusion of plants was more important in determining the effect on understory composition than the nitrogen status (Miller and Seastedt 2009). Mastication can attract *Ips* bark beetles because of the release of monoterpenes, so avoiding beetle flight season by chipping in the late summer through early winter is optimal (Fettig et al. 2006). Ensuring that chips do not pile up at the base of remaining trees may also reduce post-treatment mortality (Fettig et al.

2006). It is important to note that the effects of mastication are often variable at the stand level because treatments can produce a mosaic of chip depths, from high concentration to complete absence (Wolk and Rocca 2009).

Monitoring

Monitoring of treatment impacts is crucial to adaptive management. There will never be enough scientific studies to assess the effectiveness of all possible combinations of treatments and ecosystem attributes. Monitoring can help document programmatic successes and help navigate the uncertainties of fuels treatment. Most importantly, monitoring can detect changes, both positive and negative, at a variety of scales, towards or away from management goals. Monitoring is particularly important as experienced managers retire or change locations, because it documents the lessons learned that are often stored as memories rather than reports. Ideally, a monitoring program should be established before treatment begins, include untreated controls, and evaluate the response of ecosystem components at multiple scales (Allen et al. 2002). Baseline data can also be used to develop treatment targets for a variety of indicators. These targets, often a range of values, can be very useful to measure success and inform adaptive management efforts with quantitative site data. In practice, monitoring is often a lower priority than other activities, and it is often difficult to secure funding for monitoring (Moote et al. 2007). Programs such as CFLRP require monitoring plans and direct funding to ecological and socioeconomic monitoring.

Monitoring helps build collaboration and stakeholder support for fuels treatment. Multiparty (i.e., participatory or collaborative) monitoring can help stretch limited resources, build trust among stakeholders, improve community relations, limit conflict and litigation, support community development, address public concerns, and incorporate traditional knowledge (Pilz et al. 2006). Multiparty monitoring is a cornerstone for local-level collaboratives that facilitate mixed conifer treatments throughout the western U.S. In Colorado's Uncompahgre Partnership, the multiparty monitoring group identified the central questions about effectiveness and impacts raised by the project. One lesson learned from multiparty monitoring in New Mexico is that, because of the diverse activities and goals that can be considered monitoring, it is essential to establish a clear purpose within the multiparty monitoring group (Moote et al. 2007). Establishment of a clear purpose can keep the diverse collaborative from fraying over time; it makes efficient use of stakeholders' time

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Fire effects monitoring, Blodgett Forest, California

and resources and is essential when pursuing adaptive management through the multiparty collaborative.

Fuels treatment monitoring systems to characterize changes in ecosystem attributes over time are available; these include FIREMON and the National Park Service's *Fire Monitoring Handbook*. FIREMON assists managers in developing a specific monitoring protocol, collecting data, storing results, and analyzing fire effects (Lutes et al. 2006). The *Fire Monitoring Handbook* (2003) was developed to help managers document basic information, detect trends, and ensure that each park meets its fire and resource management objectives. Created more recently, a new monitoring tool called FFI (FEAT/FIREMON Integrated) can assist managers with collection, storage, and analysis of ecological information (Lutes et al. 2009). In FFI, managers can enter data on plot location, surface fuels, tree data, point intercept, density, line intercept, rare species, cover/frequency, species composition, fire behavior, disturbance history, Fuel Characterization Classification System, post-burn severity, and composite burn index. A new addition to FFI not previously available is a "Biomass-Fuels" protocol for storing ocular or photographic estimates of biomass like those based on the Natural Fuels Photo Series (depts.washington.edu/nwfire/dps/) (see page 66 also). One of the goals of FFI is to be flexible and accommodate data from a wide variety of plot-based sampling schemes in addition to FIREMON.



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Prescribed burn at Blodgett Forest, California

Another example of fuels treatment monitoring is the New Mexico CFRP, which is based on multiparty collaboration and focuses on basic data to guide management. CFRP requires the monitoring of live and dead tree density, live and dead tree size, crown base height, overstory canopy cover, understory cover, and surface fuels (Moote et al. 2009). The CFRP program encourages use of other indicators and more in-depth monitoring that focuses on particular attributes of interest, such as wildlife. Although in 2008 only about half of the CFRP projects had implemented even the basic monitoring program in a way that could be used to inform adaptive management; over time, the quality of monitoring has significantly improved (Derr et al. 2008).

While most fuels treatment monitoring is familiar to managers, new treatments present new challenges. For example, mastication treatments create novel fuel beds that are not well characterized by protocols for surface fuels such as the standard planar intercept method (Kane et al. 2009). One recommendation to quantify masticated fuel loads uses a hybrid methodology in which 1-hour and 10-hour fuel loadings are estimated using a plot-based method, and 100-h and 1000-h fuel loadings are estimated using the standard planar intercept method (Kane et al. 2009). Another alternative is to estimate masticated fuel load from measures of fuelbed depth or fuel coverage using the equations Battaglia and colleagues (2010) developed from plots in mixed conifer forests in Colorado. Another aspect of monitoring in mixed conifer forests that may differ from that in other forest types is the heterogeneity within the forest type. Managers have reported increased efficiency by delineating different types of mixed conifer such as warm–dry and cool–moist in the monitoring program. Delineating by tree species composition, aspect, slope, or other characteristics prior to collecting pre-treatment baseline data allows managers to tailor prescriptions and monitoring to homogeneous areas.

Limitations and Examples of Overcoming Them

Often managers have a clear picture of the treatments that should be implemented to meet the land management objectives for a particular stand or watershed, but there are impediments to treatment. In other cases, managers must accommodate competing objectives that can seem irreconcilable.

Organizational restriction on natural fire in mixed conifer forests can limit managers. For example, some fire or forest management plans do not allow for wildland fire for resource benefit within mixed conifer

The reality of allowing or generating frequent fire is a real challenge
—U.S. Forest Service silviculturist in Region 3

forests. A number of managers expressed hope that revisions of forest and fire management plans would remove the limitation on fire in mixed conifer forests. Another universal organizational limitation is funding. Of course more funding would help managers plan and implement treatments, but managers also highlighted that consistency of funding is important even if funding increases are not possible. Funding uncertainty makes it difficult to plan and address long-term challenges. Managers expressed concern that the new Hazardous Fuels Priority Allocation System within the Department of the Interior may decrease funding for fuels treatments on some lands. Managers also described how limited budgets for infrastructure improvements, such as updating old roads, bridges, and culverts, can reduce treatment options by excluding harvest machinery from certain areas.

Another limitation managers highlighted is the loss of expertise because of retirement or other job changes. When managers and administrators leave a forest or a region they can leave a knowledge gap. The widespread retirement of baby boomers from land management agencies means loss of local knowledge is a serious issue. Some managers described mentoring programs as one way to help transfer expertise. Another trend which can hamper prescribed fire programs is the decrease in the number of qualified burn personnel because of more restrictive

qualifications for prescribed fire positions and liability fears. Many burn bosses now feel the need to purchase personal insurance to protect themselves from liability risks. Liability, or the fear of potential liability, can also encourage an overly conservative approach to prescribed fire.

Often, outside help is needed to implement large treatment when the local work force (e.g., fire crews) is small and therefore comparatively slow. Smaller crews are better suited for treating small areas, such as near riparian zones.

Though contract crews can cost two to three times more than a local workforce, they can help get large treatments done faster. Managers highlighted that crew availability can be an impediment when burn windows are particularly short. Stands may be within prescription at times of year that crews are not committed to a primary mission of fire suppression. Another area in which outside help has proven useful for managers of mixed conifer forests is NEPA. Some land management budgets are weighted very heavily towards firefighting. As a result, there is limited staff available to support the NEPA or other environmental review processes. Private contractors or U.S. Forest Service Enterprise Units can provide help getting NEPA and other review processes done rapidly.



Alexander Evans

Collaborative meeting, San Juan National Forest, Colorado

Building Confidence

Returning natural fire regimes that include patches of high-severity fire to mixed conifer landscapes often requires building confidence within an organization. Land management agencies and other landowners are becoming comfortable with low-severity fires, or at least accepting their importance to ecosystem health. Prescriptions that include patches of high-severity fire present new challenges and risks, even though they may better replicate natural processes in mixed conifer forests. Lack of experience in implementing prescribed burns in mixed conifer is a barrier. One manager said that prescribed fire in mixed conifer forests was deferred in favor of ponderosa pine forest burns, because much of the mixed conifer in that area occurs on steeper slopes and canyons. Managers also reported that fires that create gaps large enough to allow for ponderosa pine establishment can create containment problems.

One approach to controlling the risks is to “box in” the higher-risk area with natural fire barriers and fuels treatments (Rytwinski and Crowe 2010). Thinning along roads and other boundaries can create defensible boundaries while allowing interior areas to burn at higher severity (Fulé et al. 2002). The San Juan National Forest has been able to implement mixed-severity burns in mixed conifer with aerial ignition of fires that burn to the natural barrier of snow at higher elevations. These burns would not be possible if treatments had not already significantly reduced

the fire hazard below the burn. Landscape-level treatments that reduce wildfire hazard and increase the ability to control fires over a large area help build confidence that mixed-severity prescribed fire can be implemented safely.

Deliberate use of intense burning will likely always pose a greater challenge for managers than underburning, because of the greater risk of escape and the perception that the burning may damage the forest. But administrative and public support can be enhanced if careful measurement of ecological effects shows that such burning can meet restoration goals. In remote settings like Grand Canyon, large-scale management tests of intense prescribed fire within secure boundaries may prove to be a more fruitful direction for adaptive management experimentation than continuing attempts to underburn dense, fire-excluded forests. (Fulé et al. 2004)

Section VII: Comprehensive Management Principles

Our exhaustive review of scientific literature on fuels treatment in mixed conifer forests combined with discussions with managers of mixed conifer forests highlights the diversity of ecology, objectives, and outcomes. However, some general management principles do emerge. The following discussion draws on previous sections, and the reader should refer to those sections for more detail and scientific references. In this section, as throughout the report, we use the phrase “mixed conifer forests” to refer to the mixed conifer forests of the Southwestern plateaus and uplands, the Central and Southern Rocky Mountains, the Sierra Nevada, and the Transverse and Peninsular Ranges in Southern California.

Fire in Mixed Conifer Forests

Though few managers need to be reminded of the fact, this review underscores that fire is a fundamental process in mixed conifer forests. Moreover, patches of high-severity fire are a natural part of most mixed conifer forests, particularly those occupying moister, high-elevation sites. Before Euro-American settlement of the West, fires in mixed conifer burned in intervals that averaged between eight and 25 years for the Southwestern plateaus and uplands, Southern Rockies, and Sierra Nevada (Tables 3, 4, and 5). Low-severity fires were more frequent in some mixed conifer forests, such as those in the Transverse and Peninsular Ranges in Southern California and the warm-dry type in the Southwest and Southern Rocky Mountains. Historically, even where low-severity fires were relatively frequent, mixed conifer forests tended to be heterogeneous mixtures where species composition, forest structure, and fuel loads changed over short distances. Since Euro-American settlement, mixed conifer forests have become increasingly homogeneous, and many mixed conifer forests, particularly those of the warm-dry type, have increased in density. More homogeneous mixed conifer forests can facilitate larger high-severity fires than those that occurred historically. Increasing heterogeneity at the landscape scale to approximate historic conditions is important to achieve many management objectives, from fuel reduction to wildlife



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Prescribed burn at Blodgett Forest, California

habitat. Restoration and wildfire hazard reduction are not synonymous, but restoration treatments can reduce the risk of uncharacteristic high-severity fire, i.e., stand-replacing fire covering large portions of the landscape.

Fuel Reduction Treatments

In most mixed conifer forests, thinning that treats both the canopy and understory (crown and low thinnings) combined with prescribed fire is most effective at reducing wildfire hazard. Crown thinning reduces canopy continuity, lowers canopy bulk density, and can create canopy gaps which in turn reduce the ability of a stand to sustain a crown fire. Low thinnings reduce ladder fuels, but they can also reduce canopy continuity and bulk density. Both treatments produce significant additions to surface fuel loads, which must be dealt with to effectively reduce wildfire hazard. Prescribed fire is effective at reducing these surface fuels, and even more importantly



Eric Siemer

A strip-head fire at dusk on the Butte Prescribed Fire at Lassen Volcanic National Park, California

returns a fundamental ecological process to mixed conifer forests. Land management objectives can dictate reliance on or avoidance of prescribed fire. In wilderness areas or other areas where thinning is inappropriate, repeated prescribed fires can reduce wildfire hazard and return forests to conditions that are within the historic range. However, some stands with relatively fire-resistant, mature trees are difficult to alter with fire alone. On the other side of the spectrum, biomass removal can reduce surface fuel where prescribed fire is inappropriate, such as next to houses. Mastication does not remove fuel but it does change fuel characteristics. By making fuels more homogeneous, mastication can facilitate prescribed fire. New research into the fire behavior of masticated fuel beds, including new fuel models, will improve our understanding of its effectiveness in fuel reduction. Treatments must be maintained for their fuel reduction effect to be sustained, and no single treatment will reverse a long history of fire exclusion. After about ten years, fuels begin building up towards pretreatment levels in many mixed conifer forests.

Building Confidence

Returning natural mixed-severity fire regimes that include patches of high-severity fire to mixed conifer landscapes often requires building confidence within an organization. Organizations and the public are becoming comfortable with low-severity fire, or at least accepting its importance to ecosystem health. Prescriptions that include patches of high-severity fire present new challenges and risks, even though they may better replicate natural processes in mixed conifer

forests. Landscape-level treatments that reduce wildfire hazard and increase the ability to control fires help build the confidence that prescribed mixed-severity fire can be implemented safely. In some cases, this landscape approach can include intensive fuel reduction treatments, such as a fuel break with wide spacing in the WUI, with prescribed fire in more remote areas. Though drastic density reductions and the removal of all understory in mixed conifer forests may not have any natural analog, these treatments may be necessary for managers to feel comfortable with high-severity fire elsewhere within the watershed. Similarly, ensuring that fuel reduction has already been implemented in adjacent forest types can reduce the fear of escape of mixed-severity prescribed fire in mixed conifer forests.

Transferring Knowledge and Mentoring

The loss of local expertise is an impediment to building confidence. When managers and administrators leave a forest or a region, they can leave a knowledge gap. Experience gained over years of observation and trial and error are difficult to capture and pass on to replacement personnel. Given the demography of the nation in general, and land management organizations specifically, retirement of experienced land managers is a significant issue for management of mixed conifer forests. When organizations lose fire experts through retirement or job changes, they might not have enough qualified people to implement prescribed burns. Organizations need to focus on ways of transferring knowledge to new land managers, whether they are new to the landscape or to the profession. Mentoring—a partnership between an experienced and a less experienced manager—can provide a crucial link between generations and maintain local

knowledge. Of course, knowledge exchange should work both ways: managers new to the profession or location may bring new skills and insights. In an era of tight budgets and increased job responsibilities, organizations need to provide the time and resources to allow for mentoring. In the short term, mentoring may mean a project takes longer; but the cost of losing years of local ecological and social knowledge can be much greater. Mentoring can also help record lessons from the local landscape and it is an essential element in adaptive management. Monitoring data can document a successful program, help justify additional funding, and build trust with collaborators.

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Prescribed burn at Blodgett Forest, California

Collaboration

Collaboration has become a necessary part of land management. New national programs such as CFLRP, Landscape Conservation Cooperatives, and the Joint Fire Science Knowledge Exchange Consortia encourage alliances; but they have also become a key to ensuring adequate funding for land management. Collaboration helps managers identify objectives that meet broad stakeholder social, economic, and ecological goals. Organizations that have community support and strong partnerships through collaboration have allies in the battle for scarce resources and a strong case for grant funding. Though collaboration requires an investment of time and money, it can help avoid even more costly litigation or obstruction. Relationships with wood utilization businesses are another form of collaboration. Project goals are much easier to achieve where demand for wood products matches the material targeted for removal by land management objectives. Although demand for wood and management objectives rarely match exactly, dialogue and long-term planning can help align these two sides and facilitate fuel reduction or restoration.



Alexander Evans

Collaborative meeting, San Juan National Forest, Colorado

Future Directions

It has become trite to highlight the need to address climate change with future management and research. However, the likelihood of increased fire activity due to warming and drying conditions in mixed conifer forests over the coming decades really does present a challenge for managers and researchers. Warmer and drier conditions have already caused an increase in wildfire activity that adds new urgency to fuels treatment. These changing environmental conditions force scientists to reevaluate assumptions and models. The changing climate also increases the importance of monitoring as a way of detecting environmental and biotic community changes early enough to be able to respond. However, even in an uncertain future, reestablishing fire as an active ecological process and increasing heterogeneity in mixed conifer forests are linked goals; each helps build resilience to the effects of climate change, droughts, and other environmental stressors.

Appendix A – Species List

Trees

Aspen (*Populus tremuloides*)
Bigcone Douglas-fir (*Pseudotsuga macrocarpa*)
Blue spruce (*Picea pungens*)
California red fir (*Abies magnifica*)
California black oak (*Quercus kelloggii*)
Canyon live oak (*Quercus chrysolepis*)
Corkbark Fir (*Abies lasiocarpa* var. *arizonica*)
Douglas-fir (*Pseudotsuga menziesii*)
Engelmann spruce (*Picea engelmannii*)
Gambel oak (*Quercus gambelii*)
Giant sequoia (*Sequoiadendron giganteum*)
Incense cedar (*Calocedrus decurrens*)
Jeffrey pine (*Pinus jeffreyi*)
Limber pine (*Pinus flexilis*)
Lodgepole pine (*Pinus contorta*)
Longleaf pine (*Pinus palustris*)
Ponderosa pine (*Pinus ponderosa*)
Rocky Mountain juniper (*Juniperus scopulorum*)
Southwestern white pine (*Pinus strobiformis*)
Subalpine fir (*Abies lasiocarpa* var. *lasiocarpa*)
Sugar pine (*Pinus lambertiana*)
Western red cedar (*Juniperus occidentalis*)
White fir (*Abies concolor*)

Plants

Cheatgrass (*Bromus tectorum*)

Animals

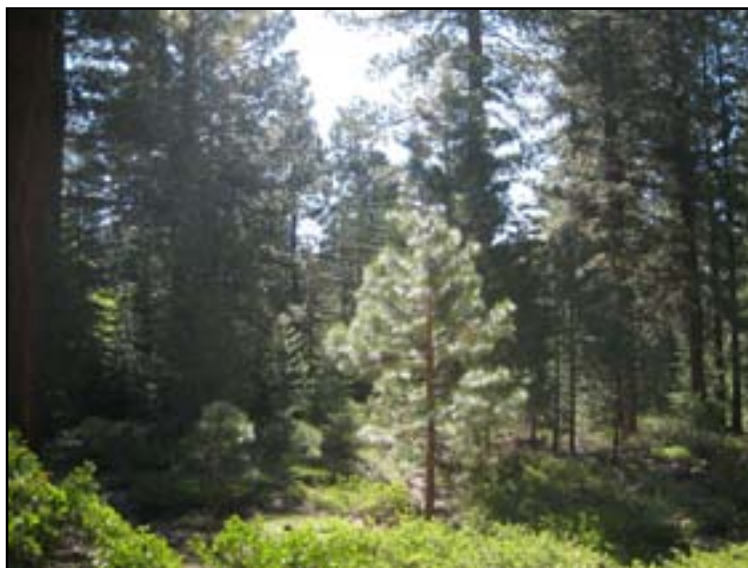
Elk (*Cervus canadensis*)
Jemez Mountains salamander (*Plethodon neomexicanus*)
Mount Graham red squirrel (*Tamiasciurus hudsonicus grahamensis*)
Spotted owl
 California (*Strix occidentalis occidentalis*)
 Mexican (*S. occidentalis lucida*)
 Northern (*S. occidentalis caurina*)
Tassel-eared squirrel (*Sciurus aberti*)

Insects

Douglas-fir beetle (*Dendroctonus pseudotsugae*)
Fir engraver beetle (*Scolytus ventralis*)
Mountain pine beetle (*Dendroctonus ponderosae*)
Red turpentine beetle (*Dendroctonus valens*)
Spruce aphid (*Elatobium abietinum*)
Western pine beetle (*Dendroctonus brevicomis*)
Western spruce budworm (*Choristoneura occidentalis*)

Diseases

White pine blister rust (*Cronartium ribicola*)



Lake Tahoe Basin, Nevada

Appendix B – References

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Appendix C – Interviews

This report would not have been possible without the cooperation and insights of the following managers and scientists:

Rich Adams, California State Parks, CA
Deb Allen-Reid, U.S. Forest Service, Region 3 Forest Health, NM
Leslie Allison, Banded Peak Ranch, NM
Bruce Bauer, Santa Clara Pueblo, NM
Ken Belcher, Bureau of Land Management, Kremmling Field Office, CO
Bryan Bird, Wild Earth Guardians, NM
Monica Boehning, Apache-Sitgreaves National Forests, AZ
Anne Bradley, The Nature Conservancy, NM
Sara Brinton, San Juan National Forest, NM
John Bristow, Lassen National Forest, CA
Jan Burke, White River National Forest, CO
Bruce Buttrey, Arizona Department of Emergency and Military Affairs, AZ
Joe Carrillo, New Mexico State Forestry, NM
Matt Cerney, Lassen National Forest, CA
J. Michael Chavarria, Santa Clara Pueblo, NM
Jerry Chonka, Grand Mesa, Uncompahgre & Gunnison National Forests, CO
Dave Conklin, U.S. Forest Service, Region 3, NM
Scott Conway, Tahoe National Forest, CA
Paul Czeszynski, Carson National Forest, NM
Christy Daugherty, CALFIRE / Tahoe Conservancy, CA
Terry DeLay, formerly Lincoln National Forest, NM
Jerry Drury, Apache-Sitgreaves National Forests, AZ
Pete Duncan, Plumas National Forest, CA
Matt Etzenhouser, Grand Mesa, Uncompahgre & Gunnison National Forests, CO
Stephen Fillmore, Cleveland National Forest, CA
Jay Francis, Collins Pine Company, CA
Arnie Friedt, New Mexico State Forestry, NM
Todd Gardiner, San Juan National Forest, CO
Tim Garvey, Grand Mesa, Uncompahgre & Gunnison National Forests, CO
Andy Graves, U.S. Forest Service, Region 3 Forest Health, NM
Art Haines, Grand Mesa, Uncompahgre & Gunnison National Forests, CO
Gary Harris, Forestry Services of Chama, NM
Dave Hattis, Arapaho & Roosevelt National Forests, CO
Dave Heft, Cibola National Forest, NM
Bill Hornsby, Bureau of Indian Affairs, Mescalero, NM

Fuels Treatment for Mixed Conifer Forests

Dan Huisjen, Bureau of Land Management, Montrose Interagency Fire, CO
Ben Jacobs, Sequoia and Kings Canyon National Parks, CA
Dale Johnson, Bureau of Land Management, Bishop Field Office, CA
Mike Kerrigan, Carson National Forest, NM
Mark Krabath, San Juan National Forest, CO
David Lawrence, Santa Fe National Forest, NM
Chuck Lewis, Lassen National Forest, CA
Jere McLemore, Bureau of Indian Affairs, Fort Apache Agency, AZ
Mark Meyers, New Mexico State Land Office, NM
Jason Moghaddas, Feather River Land Trust, CA
Lann Moore, Bureau of Land Management, Albuquerque District, NM
Ruben Morales, Coronado National Forest, NM
Duane Nelson, Eldorado National Forest, CA
Rick Ondrejka, Medicine Bow-Routt National Forest, CO
Craig Ostergaard, Sierra Pacific Industries, CA
Keith Pajkos, Arizona Division of Forestry, AZ
George Panek, Grand Mesa, Uncompahgre & Gunnison National Forests, CO
Sharon Paul, Mescalero Tribe, NM
Larry Peabody, Tahoe National Forest, CA
Jim Pitts, Apache-Sitgreaves National Forests, AZ
Lindsey Quam, New Mexico State Forestry, NM
Kenneth Reese, Santa Fe National Forest, NM
Jon Regelbrugge, Inyo National Forest, CA
Sarah Reif, Arizona Game and Fish, AZ
Patty Ringle, Coconino National Forest, AZ
Renee Romero, Taos Pueblo, NM
Bob Rynearson, W.M. Beaty and Associates, CA
Anne Sandoval, Taos Pueblo, NM
Christie Schneider, Medicine Bow-Routt National Forest, WY
Carl Skinner, U.S. Forest Service, Pacific Southwest Research Station, CA
Gus Smith, Yosemite National Park, CA
Nick Smokovich, NM State Forestry, NM
Ryan Tompkins, Plumas National Forest, CA
Laura Vallejos, Gila National Forest, NM
Andy Vigil, Santa Fe National Forest, NM
Scott Wagner, San Juan National Forest, NM
Kathy Wallace, Lincoln National Forest, NM
Mark Watson, New Mexico Game and Fish, NM
Rob York, University of California Center for Forestry, CA
Kevin Zimlinghaus, Arapaho & Roosevelt National Forests, CO