Chapter 4.1—Fire and Fuels

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Summary

Recent studies of historical fire regimes indicate that fires occurring prior to Euro-American settlement were characterized by a high degree of spatial complexity that was driven by heterogeneity in vegetation/fuels and topography and influenced by variability in climate, which mediated the timing, effects, and extents of fires over time. Although there are many important lessons to learn from the past, we may not be able to rely completely on past forest conditions to provide us with blueprints for current and future forest management. Rather than attempting to achieve a particular forest structure or landscape composition that may have existed historically, restoring the primary process that shaped forests for millennia (i.e., fire) may be a prudent approach for hedging against uncertainties around maintenance of fire-adapted forests. This is not to suggest that all forms of fire would be appropriate in these forests. A more suitable goal, albeit a more difficult one, would be to restore the forest stand and landscape conditions that would allow fires to function in what is generally believed to be a more natural way. Given the current state of the frequent-fire adapted forests in the Sierra Nevada and southern Cascade Range, achieving this will be a challenge and will require innovative forest management approaches that focus on large spatial scales. Treating landscapes based on an informed deployment of treatment areas and then moving areas out of fire suppression into fire maintenance is one means of potentially changing current patterns.

Introduction

Fire is an inherent process in most Sierra Nevada, southern Cascade Range, and montane Modoc Plateau forest types, where it has been not only a regulating mechanism, but also the dominant force shaping forest structure within stands as well as patterns across landscapes (Riegel et al. 2006, Skinner and Taylor 2006, van Wagtendonk and Fites-Kaufman 2006). This chapter summarizes recent literature relevant to fire and forest management in several key forest types of the Sierra Nevada and southern Cascade Range: yellow pine (*Pinus ponderosa* and *P. jeffreyi*) and mixed-conifer forest types. Red fir (*Abies magnifica*) forests are addressed in chapter 2.1, "Forest Ecology." The literature summarized and the implications discussed in this chapter apply primarily to forested areas outside of the wildland-urban interface

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(WUI). Social issues related to fire and fuels are examined in further detail in the social chapters (chapters 9.1 through 9.6).

Fire in Sierra Nevada and Southern Cascade Range Ecosystems

Historical Role of Fire

Numerous studies demonstrate the integral role that fire played in shaping historical (i.e., pre-Euro-American settlement) forest structure and composition in the focal area. These studies, which are largely from mixed-conifer, ponderosa pine, and Jeffrey pine forest types, demonstrate frequent occurrence of generally low- to moderate-severity fire over at least the last several centuries. The general consensus from these studies is that frequent fire maintained relatively open, patchy stands composed of primarily large, fire-resistant trees. Although this was likely the case for many areas within these forest types, to surmise that those stand conditions were ubiquitous throughout the Sierra Nevada would be a gross oversimplification. Recent studies of historical fire occurrence have gone beyond solely reporting fire frequency by reconstructing historical forest structure and characterizing spatial patterns resulting from more natural fire-forest interactions in the Sierra Nevada (e.g., Beaty and Taylor 2007, 2008; Knapp et al. 2012; Nagel and Taylor 2005; Scholl and Taylor 2006, 2010) and in the southern Cascade Range (Beaty and Taylor 2001; Bekker and Taylor 2001; Norman and Taylor 2003, 2005; Taylor 2000). These studies indicate a high degree of spatial complexity driven by heterogeneity in vegetation/fuels and topography and influenced by variability in climate, which mediates the timing, effects, and extents of fires over time. Notably, the great difference between the gentle topography of the Cascade Range and the more complex topography of the Sierra Nevada creates considerable differences in how fire functioned historically in the two mountain ranges. As a result of the differences between the two mountain ranges, the following discussion is mostly relevant to the Sierra Nevada and may not be as relevant to the Cascade Range or the Modoc Plateau. Please see Skinner and Taylor (2006) for a discussion of fire in the Cascade Range and Riegel et al. (2006) for the Modoc Plateau area.

The complexity of factors influencing fire regimes in the Sierra Nevada makes it difficult to distill quantitative information relevant to restoring Sierra Nevada forests, but there are several general themes that may inform management activities as they relate to restoration and resilience. Note that these themes are generally applicable to the mixed-conifer, ponderosa pine, and Jeffrey pine forest types; however, there are moisture/productivity gradients within individual forest types, as well as across types, that influence key fire regime characteristics: frequency and severity.

- **Topography**: Several reconstruction studies demonstrate that topography strongly influenced historical fire regimes (Beaty and Taylor 2001, 2008; Taylor and Skinner 1998, 2003). This effect, however, appears to be moderated by the topographic complexity of a particular area; i.e., in landscapes with complex geomorphic structure, topography may have been the dominant influence driving patterns in fire effects, whereas in more gentle landscapes, patterns in fire effects were driven more by the interactions between vegetation/fuel and topography (Skinner and Taylor 2006, Skinner et al. 2006). In more complex landscapes, upper slope positions tended to experience greater proportions of high-severity fire, whereas lower slope positions had lesser proportions. This pattern appears to exist almost independently of the vegetation/fuel structure in a particular landscape. The greater proportions of high-severity fire on upper slopes may have been exacerbated on south- and west-facing slopes, where more exposure and drying of fuels tends to coincide with more pronounced upslope and up-canyon winds. In more gentle landscapes of the Sierra Nevada, it appears that greater proportions of high-severity fire were associated with more mesic forest types (e.g., forests with greater component of fir [*Abies* sp.]). The mesic conditions could be a function of more northerly aspects or higher elevation. It should be noted that there are reconstruction studies that demonstrate no effect of topography on fire regime and forest structure characteristics (e.g., Scholl and Taylor 2010). It is unclear to what extent other factors may be masking more site-level influences (e.g., ignition sources/patterns, cold-air pooling).
- **Riparian areas**: In many riparian sites, reconstruction studies have demonstrated historical regimes of frequent fire that do not appear to differ from adjacent upland areas (Skinner 2003, Van de Water and North 2010). However, results from these studies do suggest that perennial streams, which may have greater influence on understory vegetation, fuel moisture levels, or relative humidity, do have noticeably lower fire frequency than adjacent upland areas. It is suggested that these riparian areas may have acted as filters—not simply barriers—for fire spread, as fires tended to burn through these areas (or burn with enough intensity to scar surviving trees) only when conditions were more favorable for fire spread (e.g., during drought conditions or substantial wind events) (Skinner 2003).
- **East-side (Sierra Nevada) pine forests**: There are far fewer historical reconstruction studies in forests on the eastern side of the Sierran crest than there are for mixed-conifer forests on the west slope. Based on the

few studies in east-side pine, it appears that fire frequency and inferred fire effects were generally similar between east-side pine and west-side mixed-conifer forests (Gill and Taylor 2009, Moody et al. 2006, North et al. 2009b, Taylor 2004, Vaillant and Stephens 2009). There are, however, context-specific distinctions that suggest some differences existed in fire regimes between east-side pine and west-side mixed-conifer: (1) In contrast to the larger expanses of contiguous forests on the west side, east-side forests are sometimes isolated in canyons or on benches in discrete stands (North et al. 2009b); this isolation results in longer fire return intervals for some east-side stands and greater variability in fire frequency and fire effects. (2) Several sampled stands in the east-side pine type maintained frequent fire regimes as late as the early- to mid-1900s (North et al. 2009b), whereas frequent fire in many west-side mixed-conifer forests ceased around the 1880s. The structural changes associated with cessation of fire could be different as a result of these different cessation dates. The contemporary forest conditions in the Jeffrey pine-mixed-conifer dominated area of the Sierra San Pedro Mártir (Baja California) serve as a relevant reference site for east-side pine forests (Stephens and Fulé 2005). This area has experienced very little timber harvesting, and fire suppression dates back only to the 1970s (Stephens et al. 2003). This forest has an open, all-aged structure, with its most salient characteristic being high spatial variability (Stephens and Gill 2005, Stephens et al. 2008). This variability not only pertains to spatial arrangement and sizes of trees, but also to coarse woody debris and tree regeneration patches (Stephens and Fry 2005, Stephens et al. 2007).

- **Cascade Range fire regimes**: The historical reconstructions of fire in these forests depict fire regimes considerably different than those of the Sierra Nevada. Further, many studies have focused on the upper montane forests (Bekker and Taylor 2001, Taylor 1993, Taylor and Halpern 1991, Taylor and Solem 2001) in addition to the mid-elevation pine and mixed-conifer forests (Norman and Taylor 2003, 2005; Taylor 2000). The more gentle topography of the Cascade Range affords conditions where fires are able to spread rather easily over large areas without significant interruption. Especially on the east side of the range in the pine and mixed-conifer forests, pre-suppression era fires were not only primarily frequent, low- to moderate-intensity fires, but were also quite large. Fires of this type covering tens to hundreds of thousands of acres occurred on average once every 20 years (Norman and Taylor 2003). Although they burned less frequently than lower and middle elevation forests, the upper montane forests with mixed-severity fire regimes burned much more frequently than similar forests of the Sierra Nevada (Bekker and Taylor 2001; Taylor 1993, 2000; Taylor and Halpern 1991; Taylor and Solem 2001). The gentle topography of the Cascade Range, combined with continuity of vegetation (fuels) from lower to higher elevations, allowed fires to burn more regularly in the higher elevations (Skinner and Taylor 2006). This is in contrast to the very rocky, vegetationally broken landscapes of the Sierra Nevada, where it is more difficult for fires to move about so freely in the upper montane forest.

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- **Landscape heterogeneity**: Differential fire effects over the landscape, including stand-replacing patches (fig. 1), contribute to coarse-grained heterogeneity across landscapes. This has been demonstrated for historical fire regimes (Beaty and Taylor 2008) and for areas with more intact, contemporary fire regimes (Collins and Stephens 2010). These studies suggest that stand-replacing fire was a component of Sierra Nevada mixed-conifer forests, but at relatively low proportions across the landscape (about 5 to 15 percent), consisting mostly of many small patches (<4 ha [10 ac]) and few

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Figure 1—A stand-replacing patch created by a 1994 fire in the Illilouette basin, Yosemite National Park. The photo was taken in 2010. Jeffrey pine seedlings are beginning to emerge over the *Ceanothus*.

large patches (about 60 ha [150 ac]). Based on these studies, it appears that landscapes with active fire regimes included relatively dense, even-aged stands and shrub patches, as well as the often-referenced open, park-like, multi-aged stands. Actual proportions in each vegetation type/structure are largely unknown owing to the limitations of historical reconstruction studies, although several studies have made estimates based on reconstructed tree ages and density (Beaty and Taylor 2001, 2008; Scholl and Taylor 2010; Taylor 2004, 2010; Taylor and Skinner 1998).

- **Climate**: Variability in historical fire occurrence is linked to both short- and long-term fluctuations in regional and synoptic climate (Beaty and Taylor 2009; Gill and Taylor 2009; Stephens and Collins 2004; Swetnam 1993; Swetnam and Baisan 2003; Taylor and Beaty 2005; Taylor and Scholl 2012; Taylor et al. 2008; Trouet et al. 2006, 2009, 2010).
    - **Short-term climatic variation (e.g., annual to decadal scale):** Although climatic fluctuations do not appear to have moderated fire effects, climate (particularly variation in precipitation) has been shown to drive fire extent (e.g., widespread fire years coincided with regional drought years, and were sometimes preceded by regionally wet years).
    - **Long-term climatic variation (decades to century scale):** Fire frequency, or chance of fire occurrence, appears to be associated with variation in air temperature (Swetnam 1993, Swetnam and Baisan 2003), with higher temperature associated with more frequent fires and longer fire seasons (Westerling et al. 2006). Precipitation appears to be associated with fire extent (Swetnam 1993, Swetnam and Baisan 2003). Thus, moist years produce vegetation that is available to burn in the inevitable drier years that occur during otherwise moist periods.
    - The **rain shadow effect** on the east side of the Sierra Nevada and the tendency for greater stand isolation, primarily in the southern portion, appears to somewhat de-couple fire in east-side pine forests from synoptic climatic conditions (North et al. 2009b).

Although there are many important lessons to learn from the past, we may not be able to rely completely on past forest conditions to provide us with blueprints for current and future management (Millar et al. 2007, Wiens et al. 2012). In particular, the nature and scale of past variability in climate and forest conditions, coupled with our inability to precisely reconstruct those conditions, introduce a number of conceptual and practical problems (Millar and Woolfenden 1999). Detailed reconstructions of historical forest conditions, often based on dendroecology, are very useful but represent a relatively narrow window of time and tend to coincide with tree recruitment in the period referred to as the Little Ice Age, which was much cooler than the present (Stephens et al. 2010). Therefore, manipulation of current forests to resemble historical forest conditions may not be the best approach when considering future warmer climates (Safford et al. 2012a). Rather, restoring the primary process that shaped forests for millennia (i.e., fire) may be a prudent approach for hedging against uncertainties around the maintenance of fire-adapted forests (Fulé 2008). This is not to suggest that all forms of fire would be appropriate in these forests. A more suitable goal, albeit a more difficult one, would be to restore the forest stand and landscape conditions that would allow fires to function in what is generally believed to be a more natural way.

Altered Ecosystems

Past harvesting practices and livestock grazing, coupled with over a century of fire suppression, have shifted forest structure and composition within the ponderosa pine, Jeffrey pine, and mixed-conifer types of the Sierra Nevada. This shift is generally characterized by increased tree densities, smaller average tree diameters, increased proportions of shade-tolerant tree species, and elevated surface fuel loads relative to historical or pre-European settlement forest conditions (Collins et al. 2011a, Scholl and Taylor 2010, van Wagtendonk and Fites-Kaufman 2006). In addition to the stand-level changes within these forest types, fire exclusion and other management practices have led to considerable homogenization across landscapes (Hessburg et al. 2007, Perry et al. 2011, van Wagtendonk and Fites-Kaufman 2006). This homogenization is a product of several interacting influences: (1) widespread timber harvesting, primarily involving removal of larger trees left during railroad or mining-related logging in the 19th and early 20th centuries; (2) infilling of trees into gaps that were historically created or maintained by variable-severity fire; and (3) forest expansion into shrub patches and meadows that were formerly maintained by fire. In addition to a loss of beta-diversity, these stand- and landscape-level changes have increased vulnerability of many contemporary forests to uncharacteristically high disturbance intensities and extents, particularly from fire and drought-induced insect/disease outbreaks (Allen 2007, Fettig 2012, Guarin and Taylor 2005). Following such disturbances, these forests and the species that depend on them have limited capacity to return to predisturbance states. This issue may be exacerbated if climate changes according to predictions in the next several decades, as large, high-intensity fires may become catalysts for abrupt changes in vegetation and associated species (i.e., type conversion).

Trends

Recent research has demonstrated an increased proportion of high-severity fire in yellow pine and mixed-conifer forests in the Sierra Nevada between 1984 and 2010 (Miller and Safford 2012, Miller et al. 2009). In addition, these studies demonstrated that fire sizes and annual area burned have also risen during the same period. The authors point out that these increases co-occur with rising regional temperatures and increased long-term precipitation. Westerling et al. (2006) also demonstrated increased area burned over a similar time period, which they attributed to regional increases in temperature and earlier spring snow melts. Despite these documented increases over the last few decades, California and the Western United States as a whole are in what Marlon et al. (2012) described as a large "fire deficit." This is based on reconstructed fire occurrence over the last 1,500 years using sedimentary charcoal records. Marlon et al. (2012) argue that the current divergence between climate (mainly temperature) and burning rates is unprecedented throughout their historical record. In other words, with temperatures warming as they have been over the last several decades, we would expect to see much higher fire activity, based on historical fire-climate associations. This divergence is due to fire management practices, which, as the authors point out, may not remain effective over the long term if warming trends continue. It is likely, given increasing temperature and the precipitation patterns since the onset of fire suppression, that fire activity would have increased over the 20th century rather than decreased had fire suppression not been implemented (Skinner and Taylor 2006, Stine 1996), further exacerbating the current fire deficit.

Notable increases in fire activity are predicted for California, and they are driven largely by projected increases in temperature and decreases in snow pack and, to a lesser extent, increased fuel production from carbon dioxide (CO_2) "fertilization" (Flannigan et al. 2000; Lenihan et al. 2003, 2008; Westerling et al. 2011). It remains unclear how these increases in fire activity would be manifested in Sierra Nevada forests (Safford et al. 2012a). Increased area burned does not necessarily result in increased proportions of high-severity fire (Miller et al. 2012b). However, one of the potential ramifications of decreased snowpack forcing longer fire seasons is that the probability of fire occurring on a given spot increases, potentially resulting in shorter intervals between successive fires. This may not be a problem if fire severity is generally low to moderate, with lesser proportions of high severity occurring in small patches. However, if high-severity proportions and patch sizes are elevated (Miller and Safford 2012), decreased time between successive fires could lead to type conversion or local loss of a particular plant association (Safford et al. 2012a). Further, even if proportions are not elevated but remain similar, this would translate into greater area burned at high severity as total burned area increases.

Effects of Ecosystem Management Strategies

Passive Management (No Action, With Continued Fire Suppression)

There is little evidence to suggest that passive management in Sierra Nevada forests will result in increased resilience to stressors (e.g., drought) or disturbance (e.g., fire, insects); in fact, there is evidence to the contrary (Agee 2002). A recent study demonstrates that crown fire potential in untreated stands continues to increase over time (Stephens et al. 2012). Modeling studies at the landscape scale also predict much greater losses from wildfire in untreated scenarios than in fuels-treated scenarios (Ager et al. 2007, 2010a; Collins et al. 2011b; Finney et al. 2007; Schmidt et al. 2008). Stephens and Moghaddas (2005a), however, reported that relatively untreated mixed-conifer stands with little understory and ladder fuels had generally low torching potential. These stands, which were 80 to 100 years old, regenerated naturally after early railroad logging and were subjected to minimal or no silvicultural treatments throughout their development (except full fire suppression). However, stands with similar structure (closed stem exclusion phase, sensu O'Hara et al. [1996], with relatively low surface and ladder fuels) and management history are probably rare in the Sierra Nevada. The prevailing evidence, both from studies of fire effects following actual wildfires and from studies reporting modeled wildfire effects, demonstrates that untreated stands (no action) are more prone to crown fire initiation and high fire-induced mortality (Ritchie et al. 2007; Safford et al. 2009, 2012b; Stephens and Moghaddas 2005b; Symons et al. 2008).

Vegetation Management

Fuels reduction is becoming the dominant forest management activity in dry forest types throughout the Western United States. The primary objectives of these activities are to modify wildland fire behavior in order to protect private property and public infrastructure, minimize negative impacts on forests (Agee and Skinner 2005), enhance suppression capabilities (Agee et al. 2000), and improve firefighter safety (Moghaddas and Craggs 2007). In drier Sierra Nevada forest types, objectives for fuel reduction treatments can often be aligned with those aimed at increasing ecosystem resilience through restoration treatments (McKelvey et al. 1996, Weatherspoon and Skinner 1996). One key potential difference between a fire hazard versus restoration focus is the incorporation of variability in both residual stand structure and surface fuels, which for a restoration-focused treatment would involve creating more horizontal and vertical spatial variability that would include retaining clumps of trees and woody debris (North et al. 2009a). This clumpiness could result in local tree torching, and thus overstory tree mortality, under wildfire conditions. Torching potential within the denser clumps would likely exceed that in stands treated for fire hazard reduction, in which the goal is to more uniformly raise canopy base height and reduce surface fuels. Although research is underway to more directly assess the relative differences between the two treatment strategies (Knapp et al. 2012), there are no recent published results. However, early publications recognized the importance of spatial variability, which was described by Show and Kotok (1924: 31) in this way: "The virgin forest is uneven-aged, or at best even-aged by small groups, and is patchy and broken; hence it is fairly immune from extensive devastating crown fires."

The activities carried out in fire hazard reduction- or restoration-focused treatments include fire (either prescribed or managed wildland fire), mechanical treatments (e.g., thinning, mastication, chipping), or a combination of the two. In field-based experiments, Stephens and Moghaddas (2005b), Schmidt et al. (2008), and Stephens et al. (2009) all found that prescribed fire alone effectively reduces surface fuels, thus reducing modeled spread rates, fire line intensities, and flame lengths under a range of weather conditions. In addition, these studies also demonstrated substantial reductions in ladder fuels in areas treated with prescribed fire. However, as fire-killed trees fall and contribute to surface fuel pools, the overall effectiveness in reducing potential fire behavior can be short lived (Keifer et al. 2006, Skinner 2005). It is likely that in dense, fire-excluded stands, multiple burns will be needed to achieve more long-lived effects (Stephens et al. 2009). Thinning effectiveness depends on the type of thinning performed and the subsequent treatment of activity fuels (Agee and Skinner 2005). In fire-excluded forests, fuel reduction prescriptions often aim to both reduce ladder fuels (increase canopy base height) and increase crown spacing (reduce crown bulk density), in combination with removing activity and existing surface fuels (e.g., piling and burning or broadcast underburning) (Agee and Skinner 2005, Stephens et al. 2009). Whole-tree harvests have also been shown to effectively reduce modeled fire behavior (Schmidt et al. 2008, Stephens et al. 2009) and actual fire effects (Ritchie et al. 2007, Symons et al. 2008). Data on tree mortality in thinned areas burned by wildfires, which demonstrate greater survivability in areas underburned following thinning, serve as real-world tests on the importance of treating activity fuels following thinning (see Raymond and Peterson 2005, Ritchie et al. 2007, Safford et al. 2012b, Symons et al. 2008). It is worth noting the instances in which extreme fire behavior (e.g., plume collapse, extreme wind) can overwhelm even well-designed fuels treatments, and lead to high tree mortality (Finney et al. 2003, Werth et al. 2011).

One concern regarding treatments that reduce tree densities and increase canopy base heights is that more open stands could experience greater windspeeds and reduced fuel moistures (Countryman 1956). It has been suggested that these potential microclimatic changes could contribute to increased fire spread rates and surface fire intensities under wildfire conditions. However, a recent study by Bigelow and North (2011) demonstrated only modest increases in wind gust speeds and no significant differences in fuel moisture between treated and untreated stands. Findings from Estes et al. (2012) also demonstrated little to no effect of thinning on fuel moistures, particularly during peak fire season in northern California. The results from these studies suggest that there is little evidence that microclimatic changes associated with fuels treatments will result in noticeably increased fire behavior, at least not in Mediterranean climates, where long, dry periods desiccate fuels irrespective of stand conditions. Furthermore, reductions in fire hazard through well-designed fuels treatment are likely to compensate for any potential increases in fire behavior (Weatherspoon and Skinner 1996).

Plantations present a unique concern in the Sierra Nevada. Plantations are generally dense, have uniformly low canopy base heights, and can often have shrub understories. These characteristics make plantations particularly susceptible to lethal fire, whether by high-intensity fire in tree canopies or from excessive heat produced by moderate-intensity surface fires (Kobziar et al. 2009, Thompson and Spies 2010, Weatherspoon and Skinner 1995). Recent research has demonstrated that prescribed fire treatments, either before plantation establishment (Lyons-Tinsley and Peterson 2012, Weatherspoon and Skinner 1995) or following establishment (Kobziar et al. 2009), can be effective at increasing tree survivability in wildfire. Note that even under prescribed fire conditions, trees in plantations are fairly vulnerable to cambial kill or crown scorch (Knapp et al. 2011). Post-establishment mastication in plantations (shrubs and small trees) may be able to reduce fire behavior (e.g., flame length, rate of spread) under wildfire conditions sufficiently to aid in fire-suppression activities, but it does not appear to be very effective at reducing tree mortality (Knapp et al. 2011, Kobziar et al. 2009). However, if masticated fuelbeds are allowed to decompose for a decade or so, fire hazards can be substantially reduced (Stephens et al. 2012).

Landscape-Scale Considerations

The large wildfires that are occurring annually throughout the Sierra Nevada demonstrate the pressing need to scale up insights gained at the stand level to larger landscapes. The effort required for planning and analysis of alternatives tends to force larger project areas, encouraging actions at the landscape scale. Yet implementing fuels treatments across an entire landscape may not be consistent with desired conditions or may not be operationally feasible (because of such issues as funding, access, and land designations [e.g., wilderness, etc.]) (Collins et al. 2010). In response, fire scientists and fire managers have conceptually developed and are refining methods for the strategic placement of treatments across landscapes (Finney 2001, 2004; Finney et al. 2007; Stratton 2004; Weatherspoon and Skinner 1996). The basic idea is that an informed deployment of treatment areas (i.e., a deployment that covers only part of the landscape) can modify fire behavior and effects for the entire landscape. Owing to the complexity of modeling fire and fuels treatments across landscapes (e.g., data acquisition, data processing, model execution, etc.), fuels treatment project design is often based on local knowledge of both the project area and past fire patterns. Recent studies in the northern Sierra Nevada and southern Cascade Range suggest that these types of landscape-level fuels treatment projects (where treatment arrangement is based more on local knowledge and fairly simple fire behavior modeling rather than intensive modeling associated with an optimization approach) can be quite effective at reducing potential fire behavior at the landscape scale (Collins et al. 2011b, Moghaddas et al. 2010, Schmidt et al. 2008).

Although only a few studies have explicitly modeled effectiveness of landscape fuels treatments using different proportions of treated area, there are some common findings: (1) noticeable reductions in modeled fire size, flame length, and spread rate across the landscape relative to untreated scenarios occurred with 10 percent of the landscape treated, but the 20-percent treatment level appears to have the most consistent reductions in modeled fire size and behavior across multiple landscapes and scenarios (Ager et al. 2007, 2010b; Finney et al. 2007; Schmidt et al. 2008); (2) increasing the proportion of area treated generally results in further reductions in fire size and behavior; however, the rate of reduction diminishes more rapidly when more than 20 percent of the landscape is treated (Ager et al. 2007, Finney et al. 2007); (3) random placement of treatments requires substantially greater proportions of the landscape to be treated compared to optimized or regular treatment placement (Finney et al. 2007, Schmidt et al. 2008); however, Finney et al. (2007) noted that the relative improvement of optimized treatment placement breaks down when larger proportions of the landscape (about 40 to 50 percent) are excluded from treatment because of land management constraints that limit treatment activities. It should be emphasized that this is not to preclude treating more than 20 percent of a landscape to achieve restoration, resilience, or other resource objectives. These studies suggest that when beginning to deal with fire hazard in a landscape, the initial objective would be to strategically reduce fire hazard on between 10 and 20 percent of the area to effectively limit the ability of uncharacteristically high-intensity fire to easily move across the landscape. This would buy time to allow restoration activities to progress in the greater landscape.

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In designing landscape-level fuels treatment or restoration projects, there are often conflicts between reducing potential fire behavior and protecting/conserving other resources (Collins et al. 2010). One common conflict is habitat for wildlife species of concern (e.g., California spotted owl [*Strix occidentalis occidentalis*] and Pacific fisher [*Martes pennanti*]). Often these species prefer multistoried stands or closed canopies for nesting or denning habitat (Solis and Gutiérrez 1990, Weatherspoon et al. 1992). Although it has been argued that fire suppression and past harvesting practices have created much of the habitat that is being called "desirable" for many of these species (see Spies et al. 2006), the species-specific approach toward managing forests continues to prevail (Stephens and Ruth 2005). This approach limits the timing and intensity of fuels treatments. Consequently, the ability to modify potential fire behavior, particularly fast-moving, high-intensity fire, in forests with prolonged fire exclusion is restricted. Furthermore, regulations on forest management within and around nesting centers or natal dens (e.g., protected activity centers, or PACs) and riparian buffer zones affect the size and placement of fuels treatments across landscapes. Therefore, there is limited opportunity to apply "optimal" placement of fuels treatments to maximize the reduction in spread of intense fire across the landscape. Additionally, these protected areas are often highly productive and contain large amounts of live and dead fuel. Thus, these areas may be prone to exacerbated fire behavior, creating effects not only within these protected areas (Spies et al. 2006), but also carrying into adjacent stands.

The dynamic nature of forest ecosystems imposes an important temporal consideration on landscape fuels management planning. A suite of fuels treatments deployed strategically across the landscape will have a characteristic life cycle. As time since treatment increases, vegetation growth will contribute to fuel pools and rebuild fuel continuity (Agee and Skinner 2005, Collins et al. 2009). Thus, as stand-level treatments mature and become less effective at reducing fire behavior, the performance at the landscape level will also decline (Collins et al. 2011b). Therefore, the design of landscape-level fuels treatments involves a tradeoff between maximizing the fraction of the landscape area treated (if only once) and treating a limited area repeatedly to maintain treatment effectiveness (Finney et al. 2007). Empirical studies from wildfires (Collins et al. 2009; Martinson and Omi 2013) and studies based on modeled fire (Collins et al. 2011b, Stephens et al. 2012) suggest that treatments can be expected to reduce fire behavior for 10 to 20 years. Obviously, a number of factors contribute to this longevity: type and intensity of treatment, site productivity, forest type, etc. Ultimately, this balance between treatment longevity and landscape-scale effectiveness is going to be location-specific, but it will require continual consideration in fire-adapted forest landscapes.

Fire Management

North et al. (2012) performed an analysis comparing current levels of fuels treatment across the Sierra Nevada to the estimated levels of historical burning throughout the range. They estimated that current treatment rates, which include wildfire area, account for less than 20 percent of the area that may have burned historically. Given that re-treatment intervals may need to be every 20 to 30 years depending on forest type, the authors argued that the current pattern and scale of fuels reduction and restoration treatments is unlikely to ever significantly advance restoration efforts, particularly if Forest Service (FS) budgets continue to decline. Furthermore, because the estimate of treatment rates includes wildfire, regardless of severity, it is likely that North et al. (2012) overestimate current restoration efforts. Treating and then moving areas out of fire suppression into fire maintenance is one means of potentially changing current patterns. However, this approach would require a fundamental change in the objectives and scale of fuels treatments. Rather than treating areas to enhance fire suppression efficacy and continue to limit the spread of fire, which would only perpetuate the current hazardous fuels/fire deficit problem, the intent would be to implement fuels treatments that allow fire to occur such that fire effects are within a desired range across the landscape (Reinhardt et al. 2008). This type of strategy would not necessarily seek to achieve ubiquitous low-severity fire effects across a landscape. Instead, the aim would be to restore a fundamental ecosystem process that involves a range of fire effects consistent with the historical range of variability. Spatial fire modeling/fuels treatment tools have recently been developed to assist planning for transitioning toward a managed fire-dominated landscape (Ager et al. 2012, 2013). Minimizing ecological impacts associated with fire suppression activities (Backer et al. 2004) would be an additional benefit of transitioning toward increased use of managed fire.

The aim should not be to simply limit fire spread or achieve ubiquitous low-severity fire effects across a landscape, but rather to restore a range of fire effects consistent with historic range of variability.

A recent comparison of contemporary fire patterns (extent and severity) between lands managed by the FS and National Park Service (NPS) in the Sierra Nevada revealed a significant distinction in fire severity patterns between the two agencies (Miller et al. 2012a). Across the forest types that were analyzed, Miller et al. (2012a) demonstrated that the proportion of high-severity fire and high-severity patch size were smaller for NPS fires than for FS fires. In addition, their results showed that overall fire extent was less on NPS lands. The authors point out that although in recent years the FS has begun to manage more wildfires for resource benefit, a policy of full suppression was in effect on most fires that occurred during their study period. In contrast, the NPS areas that they analyzed (all within Yosemite National Park) have a policy of suppressing only lightning-ignited fires when they occur outside their fire use zone or out of prescription, which resulted in most fires being managed for resource benefit. Miller et al. (2012a) suggested that by allowing most lightning fires to burn relatively unimpeded under a planned range of fire weather conditions, Yosemite has been able to achieve fire patterns that are closer to what may have occurred historically. This is not the case for the FS fires that were analyzed, which tended to burn under more extreme fire weather conditions, as these are the conditions under which fires generally escape initial fire suppression efforts (Finney et al. 2011). The authors did note that the NPS and FS lands included in the study have different land management histories, particularly with respect to timber harvesting, which have resulted in different contemporary stand structures, and could contribute to differential fire patterns. The NPS and FS lands in the study also had very different landscape contexts, with FS lands exhibiting a wider range of topographic and geomorphic configurations that would ultimately affect fire behavior. However, we should also note that many of the FS fires analyzed were in the Cascade Range and Modoc Plateau, where gentler, less complex landscapes more easily facilitate large fires and major fire runs (e.g., Fountain Fire 1992, Huffer Fire 1997) compared with the rocky, interrupted landscapes of Yosemite National Park that were the focus of the Miller et al. (2012b) study.

Efforts to restore fire as an ecological process may be guided by metrics that help to quantify the effects of fire relative to reference conditions. One important set of metrics is the fire regime interval departure (FRID) geodatabase, which focuses on fire frequency (see box 4.1-1). However, it is important to note that burning to achieve a particular interval between successive fires may not result in desired forest conditions. Clearly, fires were frequent in yellow pine and mixed-conifer forests of the synthesis area prior to Euro-American settlement. However, frequency alone did not appear to have generated the fine- and coarse-grained heterogeneity that has been associated with historical forest conditions. Rather, it seems that a range of fire effects over time, with a distribution skewed to the low- and moderate-severity, but including some stand-replacing effects, contributed to overall heterogeneity. The development of robust fire severity estimates derived from satellite imagery serves as a useful tool to quantify the distribution of fire effects both within individual fires (Collins et al. 2007) and across multiple fires throughout a region (Thode et al. 2011). It is important to emphasize that low-severity fire alone, even when applied multiple times, may not restore historical forest conditions (Collins et al. 2011a, Miller and Urban 2000). Reestablishing distributions of fire effects similar to historical conditions may prove difficult to achieve in the short term in fire-suppressed forests, but it is a useful long-term goal for promoting socioecological resilience (SNEP Science Team 1996).

Box 4.1-1

Fire Return Interval Departure Metrics

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Fire return interval departure (FRID) is a measure of how much the frequency of fire has changed in recent years versus the time before Euro-American settlement (Van de Water and Safford 2011). These data are fundamental for planning fuels treatments and restoring fire regimes, because they allow forest managers to identify areas at high risk of passing ecological thresholds resulting from altered fire regimes and their interactions with other factors (Van de Water and Safford 2011). High positive FRID values indicate areas that were characterized by frequent fire but have not experienced fire for many decades (fig. 2). FRID analyses can be combined with strategic considerations of fire behavior, topography, and values at risk to help identify priorities for fuels reduction and restoration of fire.

FRID maps are available for California from the U.S. Forest Service Remote Sensing Lab. Unlike the national fire regime condition class program, which primarily measures departure from modeled conditions of vegetation structure, the California FRID data directly measure fire frequency departure. The FRID geodatabase includes several different metrics ("PFRID" metrics, based on percent departure) that account for the cumulative fire history of the national forests and adjoining areas since 1908. Another metric, the National Park Service (NPS) FRID index, compares the time since last fire against the pre-Euro-American fire frequency. The NPS FRID index is not structured to deal with areas experiencing more frequent fire today than under reference conditions (red areas in fig. 2), as is the case in much of low- and middle-elevation southern California and other areas where human activity and vegetation changes have made fire more frequent over time. The PFRID metrics extend into negative numbers to permit departure measurements under any scenario. Because the NPS FRID index weighs only the time since the most recent fire, it is most useful as a short-term performance measure, whereas the percentage-based metrics comparing long-term frequencies are better measures of actual fire restoration. Measures like mean, minimum, and maximum PFRID, which evaluate the influence of fire over a longer time scale, will be more helpful in targeting and tracking a longer term strategy to promote resilience to disturbance, a warming climate, and other stressors.

As with any simple metric, users should be cautious when interpreting the significance of FRID data or using them to plan treatments. FRID data do not account for nonfire silvicultural treatments and do not provide a measure of overall fire risk. Although FRID would be expected to be correlated to vegetation burn severity, FRID analyses do not directly account for expected fire intensity or burn severity. Consequently, strategies need to consider other components of the fire regime, such as fire size, severity, and spatial pattern. Furthermore, a restoration strategy would take into account other factors, including forest productivity, aquatic ecosystems, wildlife habitat, social values, and other values at risk (Franklin and Agee 2003), as well as understanding of how fire may affect a landscape. As one example, mixed-conifer forests in areas of high productivity may be at higher risk of uncharacteristically severe fire after missing only three or four fires than are lower productivity ponderosa or Jeffrey pine forests that have missed more than four fires. However, because of the importance of fire frequency, FRID metrics can serve a useful role in measuring progress toward restoring a more natural role of fire as a dominant ecological process.

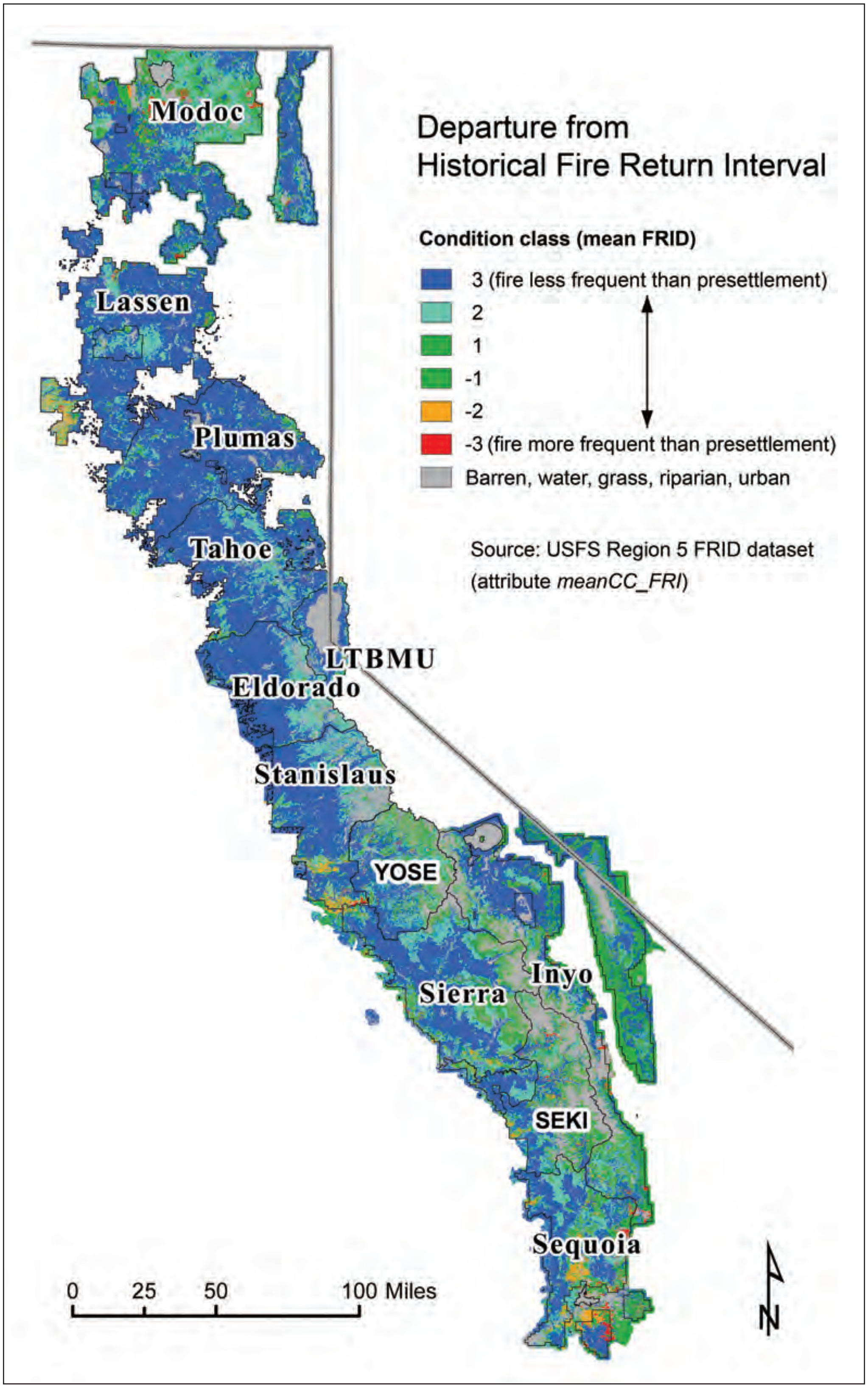

Figure 2—Mean percent fire return interval departure (mean PFRID) coded by condition class for the mountains of the Sierra Nevada and southern Cascade Range. Negative condition classes indicate areas where fires have been burning more often than under presettlement conditions, whereas positive condition classes indicate areas where fires have been burning less often. Condition classes 1 and -1 are depicted with the same color because they both indicate conditions that are not greatly departed from the mean presettlement value. See Van de Water and Safford (2011) for more details regarding how the metric is calculated. LTBMU = Lake Tahoe Basin Management Unit. YOSE = Yosemite National Park. SEKI = Sequoia-Kings Canyon National Parks.

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