Chapter 1

Setting the Stage for Mixed- and High-Severity Fire

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1.1 EARLIER HYPOTHESES AND CURRENT RESEARCH

In the late 19th century and early 20th century, fire—especially patches of high severity wherein most or all of the dominant vegetation is killed—was generally considered to be a categorically destructive force. Clements (1936) hypothesized that the mature/old state of vegetation would result in a stable "climax" condition and described natural disturbance forces such as fire as a threat to this state, characterizing mature forest that experienced high-severity fire as a "disclimax" state. One early report opined that there is no excuse or justification for allowing fires to continue to occur at all in chaparral and forest ecosystems (Kinney, 1900). After a series of large fires in North America in 1910, land managers established a policy goal of the complete elimination of fire from all North American forests (a "one size fits all" policy) through unsuccessful attempts to achieve 100% fire suppression (Pyne, 1982; Egan, 2010). Through the mid-20th century, and in recent decades, views have shifted to broadly acknowledge the importance of low- and low/moderate-severity fire. In this chapter we focus on drier montane forests of western North America as a case study of how diverse, competing, and rather complex sets of evidence are converging on a new story that embraces not just low-severity fire but also mixed- and high-severity fire in these ecosystems. Thus this chapter exemplifies how mixed- and high-severity fire is being better understood and appreciated as scientific evidence accumulates.

A commonly articulated hypothesis is that dry forests at low elevations in western North America were historically open and park-like, and heavily dominated by low-severity and low/moderate-severity fire (Weaver, 1943; Cooper,

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1962; Covington, 2000; Agee and Skinner, 2005; Stephens and Ruth, 2005). Under this hypothesis, high-severity fire patches were rare, or at least were believed to be small to moderate in size, and larger patches (generally hundreds of hectares or larger) that burn today often are considered to be unnatural and ecologically harmful. While this model fits reasonably well in some low-elevation, xeric forest systems (Perry et al., 2011; Williams and Baker, 2012a, 2013), it has been extrapolated far beyond where it seems to apply best. That higher fire severities occurred historically, albeit at a wide variety of spatial and temporal scales, in most or all fire-dependent vegetation types of western North America is becoming increasingly clear (Veblen and Lorenz, 1986; Mast et al., 1998; Taylor and Skinner, 1998; Brown et al., 1999; Kaufmann et al., 2000; Heyerdahl et al., 2001, 2012; Wright and Agee, 2004; Sherriff and Veblen, 2006, 2007; Baker et al., 2007; Hessburg et al., 2007; Klenner et al., 2008; Amoroso et al., 2011; Perry et al., 2011; Schoennagel et al., 2011; Williams and Baker, 2012a; Marcoux et al., 2013; Odion et al., 2014; Hanson and Odion, 2015a).

A key extension of the concept of historical forests characterized by open structure coupled with a low- or low/moderate-severity fire regime is that current areas of dense forest structure—and larger, higher-severity fire patches in such areas—are the result of unnatural fuel accumulation from decades of fire suppression policies, leading to higher-severity fire effects outside the natural range of variability. The most fire-suppressed forests (i.e., those that have gone without fire for periods that exceed their "average" natural fire cycles) are, therefore, expected to experience unnaturally high proportions of higher-severity fire if they burn (Covington and Moore, 1994; Covington, 2000; Agee, 2002; Agee and Skinner, 2005; Stephens and Ruth, 2005; Roos and Swetnam, 2012; Williams, 2012; Stephens et al., 2013; Steel et al., 2015).

We recognize that the historical low-severity fire regime described above has not been applied to all forest types in western North America (e.g., Romme and Despain, 1989; Agee, 1993). The idea has, however, been widely applied in principle to most forest types, and widespread acceptance of the low- and low/moderate-severity fire regime has been the primary basis driving fire management policy in an overwhelmingly large proportion of montane forests in the western United States. Thus many management plans explicitly adopt a low-severity fire regime model without rigorously examining evidence of its applicability to the management of the ecosystem type under consideration. A key research need has been to determine the particular ecosystem types to which the low-severity fire regime applies. Scientists recently rigorously investigated the hypothesis that forests are burning in a largely unnatural fashion and found that historical forest structure and fire regimes were far more variable than previously believed, and that ecosystem responses to large, intense fires often differ from past assumptions (Figure 1.1; see also Chapters 2–5). We discuss these notions in greater depth throughout this book.

FIGURE 1.1 Natural regeneration of native vegetation—including conifers, deciduous trees, and shrubs—in large high-severity fire patches. Top: Star Fire of 2001 (photo by Chad Hanson, 2013); bottom: Storrie Fire of 2000 (photo by Chad Hanson, 2007) (see also Chapter 2).

Do Open and Park-Like Structures Provide an Accurate Historical Baseline for Dry Forest Types in Western US Forests?

Using spatially extensive tree ring field data, historical landscape photographs from the late 19th and early 20th centuries, early aerial photography from the 1930s through 1950s, and direct records from late 19th-century land surveyors, numerous recent studies have been able to reconstruct the historical structure of conifer forests in the western United States. A portion of the historical montane forest landscape in any given region undoubtedly comprised open forest dominated by low-severity fire (e.g., Brown et al., 1999, Fulé et al. 2009, Iniguez et al., 2009; Perry et al., 2011; Williams and Baker, 2012a; Hagmann et al., 2013; Baker, 2014), and some forest types (e.g., ponderosa pine [*Pinus ponderosa*]) often had a preponderance of low-severity fire in many low-elevation or xeric-type forest environments throughout western North America. Nevertheless, landscape-level evidence indicates that vast forested areas also comprised moderate to very dense forests characterized by a mixed-severity fire regime, wherein higher-severity fire patches of varying sizes occurred in a mosaic of low- and moderate-severity fire effects (Veblen and Lorenz, 1986, 1991; Baker et al., 2007; Sherriff and Veblen, 2007; Hessburg et al., 2007; Perry et al., 2011; Baker, 2012; Williams and Baker, 2012a,b; Baker, 2014; Baker and Williams, 2015; Hanson and Odion, 2015a). In general, in historical ponderosa pine and mixed-conifer forests of the western United States, local variability was substantial (Brown et al., 1999, Fulé et al. 2009, Iniguez et al., 2009; Hessburg et al., 2007; Perry et al., 2011; Baker, 2012; Williams and Baker, 2012a,b, 2013; Baker, 2014; Baker and Williams, 2015; Hanson and Odion, 2015a). In sum, these and other studies indicate that historically there was high variability in fire effects (low to high severity) and composition and structure at both small and large spatial scales, and these patterns varied greatly depending on the regional and biophysical setting.

Does Time Since Fire Influence Fire Severity?

The predominant view in North American fire science has been that as woody ecosystems age, they steadily increase in their potential for higher-severity fire. Thus in the fire exclusion/fuels buildup model applied to relatively dry conifer forests and woodlands (e.g., Covington and Moore, 1994), long fire-free intervals caused by effective fire suppression result in fuel accumulation and changes in fuel arrangements (e.g., vertical fuel continuity) that lead to increased fire severity. Likewise, even for forest ecosystems known to burn primarily in severe stand-replacing fires, many classical models of fire potential (in this case the instantaneous chance of fire occurrence) assume that fire severity increases with time since the last fire as a result of fuel load accumulation (Johnson and Gutsell, 1994); some research supports this (Steel et al., 2015). Nevertheless, empirical and modeling studies have demonstrated that in many ecosystem types, including temperate forests, flammability is still relatively stable with regard to time since fire (Kitzberger et al., 2012; Perry et al., 2012; Paritsis et al., 2014). We suggest that the predominance of the viewpoint in the western United States that flammability and potential fire severity inexorably increase with time since fire has been an important contributor to the expectation that 20th-century fire suppression—if assumed to have effectively reduced fire frequency—should result in increased, and unnaturally high, fire severity in the modern landscape. This relationship does not seem to hold in various ecosystem types and regions for a wide variety of reasons (Veblen, 2003; Odion et al., 2004; Baker, 2009; van Wagtendonk et al., 2012). Even but even if it held everywhere, this preoccupation with changes from historical proportions of higher-severity fire skirts the key management questions of whether a change in the proportion of higher-severity patches renders a forest incapable of "recovery" after such fires (precious few papers deal with this key management question) or whether the overall spatiotemporal extent of higher-severity fire (i.e., rotation intervals) exceeds historical levels.

A second assumption about fire regimes in the western United States is implied by language commonly used to describe modern fire regimes in terms of "missed fire cycles." While fire cycle may be a useful descriptor of fire regimes, the assertion that a particular place or patch has missed one or more fire cycles implies a regularity to fire return intervals that is not supported by most studies of fire history; there is always variation around a mean. Even in dry forests characterized by relatively frequent fires, the historical fire frequency is typically characterized by such a high degree of variance that descriptors such as means or cycles are misleading. Using the term *missed fire cycle* in mixed-severity fire regimes, among which the frequency and severity of fires are inherently diverse, is particularly problematic. Usage of *missed fire cycles* connotes a consistency and degree of equilibrium in the historical fire regime that is not supported by actual fire history evidence, which shows large variations in fire intervals (e.g., Baker and Ehle, 2001; Baker, 2012). Though it seems to make intuitive sense that, with increasing time since fire, fuels would accumulate to create a higher probability of higher-severity fire effects, numerous countervailing factors modulate fire severity as stands mature since the previous fire.

Notably, many studies of this issue have found that, in some areas, the most long-unburned forests are burning mostly at low/moderate severity and are not experiencing higher levels of high-severity fire than forests that have experienced less fire exclusion (Odion et al., 2004, 2010; Odion and Hanson, 2006, 2008; Miller et al., 2012; van Wagtendonk et al., 2012). Further, forests with the largest amounts of surface fuels (based on prefire measurements) and small trees do not necessarily always experience more severe fire (Azuma et al., 2004). Debate about this issue remains, however. For example, Steel et al. (2015, Table 7 in particular) modeled time since fire and fire severity in California's forests and predicted that, in mixed-conifer forests, high-severity fire would range from 12% 10 years after fire to 20% 75 years after fire, though the modeling for mixed-conifer forests seems to have been based on what appears to be very limited data for forests that experienced fire less than 75 years earlier (Steel et al., Figure 4), weakening inferences about a time since fire/severity relationship. Regardless, the high-severity fire values reported by Steel et al.—even for forests that had not previously burned for 75-100 years—remain well within the range of natural variation of high-severity fire proportions in these forests found by most recent studies (Beaty and Taylor, 2001; Bekker and Taylor, 2001; Baker, 2014; Odion et al., 2014; Hanson and Odion, 2015a).

Although the notion that fire severity would not necessarily increase with time since fire is seemingly counterintuitive, a number of factors help explain it. For example, as forests mature with increasing time since the last fire, canopy cover increases, creating more cooling shade, facilitating moister surface conditions, and slowing wind speeds and thus rates of fire spread. Also, increasing shade in the forest understory can cause a reduction in sun-dependent shrubs and understory trees, making it more difficult to initiate or sustain crown fire (Odion et al., 2004, 2010; Odion and Hanson, 2006). Much more important, however, is that severe fire events are largely driven by weather (Finney et al., 2003) and often have relatively little to do with the amount of fuel available (Azuma et al., 2004).

An analogous assumption about the role of fuels was previously made regarding chaparral, one of the most fire-dependent plant communities in the world; that is, historically, there was less fuel and more moderate fire effects. This idea is also inconsistent with the scientific evidence (Keeley and Zedler, 2009); see also Chapter 7.

What is the Evidence for Mixed- and High-Severity Fire?

In recent decades a growing number of studies has investigated historical fire regimes using a variety of methods to determine the extent and frequency of mixed- and high-severity fire, particularly in the ponderosa pine and mixed-conifer forests of western North America (Table 1.1). Regardless of the method used, most landscape-level studies of dry forest types, for example, tend to find evidence for mixed-severity fire regimes that included low-, moderate-, and high-severity fire (both small and large patches) in most forest types and regional areas across the western United States, with few exceptions (Odion et al., 2014). Here we describe some of the more common methods that researchers have used to determine historical fire regimes, mostly in western North America.

Aerial Photos

Many researchers have used early aerial photos of montane forests to determine the historical occurrence of high-severity fire. Specifically, researchers have used such photos to determine (1) the number of emergent trees that survived previous high-severity fire (Beaty and Taylor, 2001, 2008; Bekker and Taylor, 2001, 2010); (2) broad stand-structure categories consistent with past low-, moderate-, and high-severity fire (Hessburg et al., 2007); and (3) levels of forest canopy mortality consistent with low-, moderate-, and high-severity fire. Such studies concluded that mixed- and high-severity fire effects were generally dominant in both lower- and middle-montane forests, including mixed-conifer forests, as well as upper montane forests. Comparisons of modern and historical aerial photographs have revealed important variability in forest changes along environmental gradients in areas experiencing similar land use and fire exclusion histories. For example, in the Colorado Front Range, comparison of aerial photographs showed no significant increase in tree densities in the upper montane zone of ponderosa pine and mixed-conifer forests from 1938 to 1999 (Platt and Schoennagel, 2009) in an area with mixed- and high-severity

TABLE 1.1 **Summary of Historical Higher-Severity Fire Proportions Found in Various Reconstruction Study Areas at Least 1000 ha Within Mixed-Conifer and Ponderosa Pine Forests of Western North America**

Region	Study	Study Area (ha)	Higher-Severity Fire (%)
Baja California	Minnich et al. (2000)	~75,000	16
Sierra Nevada	Baker (2014) Hanson and Odion, 2015a	330,000 65,296	31-39 26
Klamath	Taylor and Skinner (1998)	1570	12-31
Southern Cascades	Beaty and Taylor (2001) Bekker and Taylor (2001) Baker (2012)	1587 2042 400,000	18-70 52-63 26
Northern/Central Cascades	Hessburg et al. (2007)	303,156	37
Blue Mountains (Oregon)	Williams and Baker (2012a)	301,709	17
Front Range (Colorado)	Williams and Baker (2012a) Sherriff et al. (2014, Figure 6)	65,525 564,413	65 ~72[a]
Southwestern United States (Arizona)	Williams and Baker (2012a)	556,294	15-55
British Columbia, Canada	Heyerdahl et al. (2012)	1105	10

[a]*Includes mixed- and high-severity fire.*

fire (Schoennagel et al., 2011), whereas low-elevation areas near the ecotone with the plains grasslands exhibited a moderate degree of invasion of grasslands by trees during the same period (Mast et al., 1997) in an area known to have had predominantly lower-severity fire (Veblen et al., 2000; Sherriff and Veblen, 2007; Sherriff et al., 2014).

Historical Reports

Scientists have reviewed evidence in early historical US government forest reports, finding widespread occurrence of small and large higher-severity fire patches in all forest types, including ponderosa pine and mixed-conifer forests

(Shinneman and Baker, 1997; Baker et al., 2007; Williams and Baker, 2012a [Appendix S1], 2014; Baker, 2014; Hanson and Odion, 2015a). Evidence in these reports includes detailed descriptions of low-, mixed-, and high-severity fires; maps of slightly to severely burned forests; estimates of total area burned at mixed and high severity; and photographs of the landscapes after these fires.

Direct Records and Reconstructions from Early Land Surveys

Field data from unlogged forests collected by the US General Land Office in the 19th century before fire suppression has been extensively analyzed across large landscapes, and historical stand structure has been correlated to fire severities that facilitated or stimulated those forest structures. Based on these analyses, substantial areas of ponderosa pine and mixed-conifer forests across the western United States were dominated by a mixed-severity fire regime that includes evidence of high-severity fire (Baker, 2012, 2014; Williams and Baker, 2012a,b, 2013) typically intermixed with areas of predominantly low/moderate-severity fire. Importantly, note that nearly all tree ring reconstructions that found open, park-like historical forests in some areas have been supported by these land survey reconstructions for those same areas, but the land surveys show definitively that these park-like forests grew only in portions of most dry forests in the western United States. Historical mixed- and high-severity fires shown by the land surveys led to diverse landscapes at scales of a few townships (e.g., 25,000 ha) within each region. These landscapes contained intermixed patches of open forests, dense forests, complex early seral forests, old-growth forests, dense shrub fields, and large patches of snag habitat important to wildlife. This landscape diversity was missed by tree ring reconstructions because using tree ring methods without abundant extant large, old trees is difficult, and thus more heavily burned historical forests were avoided or missed (Baker and Ehle, 2001; Williams and Baker, 2012a).

The land survey records also show (Baker and Williams, 2015) that historical dry forests were numerically dominated by small trees (e.g., <40 cm in diameter) but also included abundant large trees, which together provided "bet-hedging" resilience against a variety of forest disturbances that produce high levels of tree mortality (e.g., insect outbreaks, severe droughts, mixed- and high-severity fires). Large surviving trees are particularly important after severe fires, but smaller trees can differentially survive insect outbreaks and droughts (Baker and Williams, 2015).

Though some (e.g., Fule et al., 2014) have recently questioned some findings of Williams and Baker (2012a), an in-depth analysis of the critique found that most of its points were founded on mistakes, misunderstandings, and omission and misuse of evidence by critics (Williams and Baker, 2014). The accuracy of land survey methods has undergone extensive checking and cross-checking, and the findings are strongly corroborated by other published sources, including historical US government fire-severity mapping and reports (Williams and Baker, 2010, 2011, 2012a, 2014; Baker, 2012, 2014). These checks show that land survey reconstructions can achieve accuracies almost as high as those from tree ring reconstructions but can do so across very large land areas (e.g., ≥400,000 ha).

Tree Ring Reconstructions of Stand Densities and Fire History

Many scientists have used stand-age data from unlogged forests, often in combination with fire-scar dating of past fires, to reconstruct historical fire regimes and changes in the rate of new stand initiation from mixed- to high-severity fire. In mixed-conifer and ponderosa pine forests of western North America, researchers have found regional stand-age distributions consistent with a mixed-severity fire regime that maintained a mix of age classes and successional stages (e.g., Taylor and Skinner, 1998; Heyerdahl et al., 2012; Odion et al., 2014). Reconstructions of stand structures and fire history are most effective when supported by diverse evidence, gathered independently, that converges to the same overall interpretations. For example, in the Colorado Front Range, tree ring evidence, historical landscape photographs, and General Land Office surveys converge to the same conclusions demonstrating that the historical (i.e., before 1920) fire regime of ponderosa pine and mixed-conifer forests included low-severity fires (i.e., not lethal to large, fire-resistant trees) as well as high-severity fires (i.e., killing >70% of canopy trees) (Veblen and Lorenz, 1986, 1991; Mast et al., 1998; Schoennagel et al., 2011; Williams and Baker, 2012b). The conclusion that most of the montane zone forests dominated by ponderosa pine in the Front Range were characterized by a mixed-severity fire regime is further supported by independently conducted studies by noncollaborating researchers based on tree ring evidence of past fires and their ecological effects (Brown et al., 1999; Kaufmann et al., 2000; Huckaby et al., 2001).

Clear delineation of the spatial extent of past fire regimes is a major concern of ecosystem managers in the context of ecological restoration and management of wildfire. Fire histories and stand structures reconstructed using tree ring data are most useful for guiding management decisions where data sets are sufficiently robust to produce high-resolution spatial layers to compare historical and modern landscape conditions. As an example, in the Colorado Front Range a data set consisting of 7680 tree cores and 1262 fire-scarred tree samples collected at 232 field sites allowed for a spatially explicit comparison of historical fire severity (before fire exclusion in 1920) with observed modern fire severity and modeled potential fire behavior across 564,413 ha of montane forests (Sherriff et al., 2014). Forest structure and tree ring fire history were used to characterize fire severity at the 232 sites. Then, historical fire severity was spatially modeled across the entire study area using biophysical variables that had successfully predicted (retrodicted) fire severity at the 232 sampled sites.

Only 16% of the study area recorded a shift from historical low-severity fire to a higher potential for crown fire today. A historical fire regime of more frequent, low-severity fires at elevations below 2260 m is consistent with the view among land managers that these forests be thinned both to restore historical structure and to reduce fuels in this area of widespread exurban development. By contrast, at higher elevations in the upper montane zone (i.e., 2260-3000 m), mixed-severity fires were predominant historically and continue to be so today. Thus thinning treatments at higher elevations of the montane zone are inappropriate if the management goal is ecological restoration. Comparison of the severity of nine large fires that occurred between 2000 and 2012 with the severity of fires before the 20th century revealed no significant increase in fire severity from the historical to the modern period except for a few fires that occurred within the lowest elevations (16%) of the montane study area (Sherriff et al., 2014). This spatially extensive tree ring–based reconstruction is strongly corroborated by land survey records of higher-severity fire patches across the same area (Williams and Baker, 2012b).

Charcoal and Sediment Reconstructions

Paleoecologists have explored fire-induced sediment layers in alluvial fans (e.g., Pierce et al., 2004) and charcoal sediments (e.g., Whitlock et al., 2008; Colombaroli and Gavin, 2010; Jenkins et al., 2011; Marlon et al., 2012) to reconstruct historical fire occurrence. They found numerous periods of large and severe fire activity over the past several centuries and millennia in North American mixed-conifer and ponderosa pine forests (see Chapter 9 for many additional citations). Thus paleoecological methods and evidence further corroborate findings based upon other methods, discussed above, regarding historical mixed- and high-severity fire in these forests.

1.2 ECOSYSTEM RESILIENCE AND MIXED- AND HIGH-SEVERITY FIRE

Along with the surge in scientific investigation into historical fire regimes over the past 10-15 years has come enhanced understanding of the naturalness and ecological importance of mixed- and high-severity fire in many forest and shrub ecosystems. Contrary to the historical assumption that higher-severity fire is inherently unnatural and ecologically damaging, mounting evidence suggests otherwise. Ecologists now conclude that in vegetation types with mixed- and high-severity fire regimes, fire-mediated age-class diversity is essential to the full complement of native biodiversity and fosters ecological resilience and integrity in montane forests of North America (Hutto, 1995, 2008; Swanson et al., 2011; Bond et al., 2012; Williams and Baker, 2012a; DellaSala et al., 2014). Ecological resilience is essentially the opposite of "engineering resilience," which pertains to the suppression of natural disturbance to achieve stasis and control of resources

(Thompson et al., 2009). Ecological resilience is the ability to ultimately return to predisturbance vegetation types after a natural disturbance, including higher-severity fire. This sort of dynamic equilibrium, where a varied spectrum of succession stages is present across the larger landscape, tends to maintain the full complement of native biodiversity on the landscape (Thompson et al., 2009). Forests that are purported to be burning at unprecedented levels of high-severity fire are generally responding well in terms of the forest succession process and native biodiversity (see Chapters 2–5), so the widespread fear of too much severe fire seems to be unfounded in the vast majority of cases (see, e.g., Kotliar et al., 2002; Bond et al., 2009; Donato et al., 2009; Burnett et al., 2010; Malison and Baxter, 2010; Williams and Baker, 2012a, 2013; Buchalski et al., 2013; Baker, 2014; Odion et al., 2014; Sherriff et al., 2014; Hanson and Odion, 2015a). We acknowledge that more research is needed for some forest regions, such as some areas of the southwestern United States experiencing increasing fire severity (Dillon et al., 2011), to determine the effects of climate change on forest resilience.

As discussed above, in mixed-severity fire regimes, higher-severity fire occurs as patches in a mosaic of fire effects (Williams and Baker, 2012a; Baker, 2014). In conifer forests of North America, higher-severity fire patches create a habitat type, known as complex early seral forest (DellaSala et al., 2014), that supports levels of native biodiversity, species richness, and wildlife abundance that are generally comparable to, or even higher than, those in unburned old forest (Raphael et al., 1987; Hutto, 1995; Schieck and Song, 2006; Haney et al., 2008; Donato et al., 2009; Burnett et al., 2010; Malison and Baxter, 2010; Sestrich et al., 2011; Swanson et al., 2011; DellaSala et al., 2014). Many rare, imperiled, and declining wildlife species depend on this habitat (Hutto, 1995, 2008; Kotliar et al., 2002; Conway and Kirkpatrick, 2007; Hanson and North, 2008; Bond et al., 2009; Buchalski et al., 2013; Hanson, 2013, 2014; Rota, 2013; Siegel et al., 2013; DellaSala et al., 2014; Baker, 2015; see also Chapters 2–6). The scientific literature reveals the naturalness and ecological importance of multiple age classes and successional stages following higher-severity fire, as well as the common and typical occurrence of natural forest regeneration after such fire (Shatford et al., 2007; Donato et al., 2009; Crotteau et al., 2013; Cocking et al., 2014; Odion et al., 2014). These and other studies suggest that mixed-severity fire, including higher-severity fire patches, is part of the intrinsic ecology of these forests and has been shaping fire-dependent biodiversity and diverse landscapes for millennia (Figure 1.2).

1.3 MIXED- AND HIGH-SEVERITY FIRES HAVE NOT INCREASED IN FREQUENCY AS ASSUMED

Fire history studies show that for many montane forests, including mixed-conifer and ponderosa pine forests, fire frequencies in most forested regions were substantially less during the 20th century (and the early 21st century) compared with the previous few centuries (e.g., Odion et al., 2014). Nonetheless,

FIGURE 1.2 Though high-severity fire patches in montane forests may initially seem to be relatively lifeless landscapes, within the first weeks and months after fire, by the first spring after fire, and for many springs thereafter, native shrubs, conifers, and deciduous trees naturally regenerate, creating an ecologically rich habitat for numerous wildlife species. (a) Star Fire of 2001, Eldorado National Forest, Sierra Nevada *(photo by Chad Hanson, 2013)*. (b) McNally Fire of 2002, Sequoia National Forest, Sierra Nevada *(photo by Chad Hanson, 2014)*.

factors responsible for this decline in fire vary from region to region and include fire suppression, changes in forest structure as a result of timber harvesting, removal of fine fuels by livestock grazing, and climate change. The result is that all fire types, including high-severity fire, have been reduced substantially in broad regions since the early 20th century (Veblen et al., 2000, Odion and Hanson, 2013; Odion et al., 2014; Hanson and Odion, 2015a). Nevertheless, some forest types or local areas within regions may have more high-severity fire than they did historically (e.g., some low-elevation or other particular environments of xeric montane forests; Perry et al., 2011; Sherriff et al., 2014).

While some chaparral/shrub ecosystems (and some forests) are in close proximity to large human populations and associated unplanned human-caused ignitions resulting in an excess of fire relative to historical rates (see Chapter 7), these are the exception, not the rule—at least for conifer forests with mixed-severity fire regimes.

Recent climate-induced increases in fire frequency (Kasischke and Turetsky, 2006; Westerling et al., 2006; Dennison et al., 2014) have led to increases in total area burned in most regions of western North America, but most areas have not experienced trends in high-severity fire (for example, see Hanson et al., 2009; Dillon et al., 2011; Miller et al., 2012; Hanson and Odion, 2014; Hanson and Odion, 2015b; also see Chapter 9), though some have. For example, while the severity of fires (the proportion of high-severity fire effects) is not increasing in forests in the southwestern United States, the overall high-severity fire area has increased in recent decades (Dillon et al., 2011). In the southern Rocky Mountains, both high-severity fire area and proportion have increased in recent decades (Dillon et al., 2011).

Not only is the habitat created by higher-severity fire biodiverse and—in many forest regions—rare compared with historical conditions, it also is often severely threatened by the inertia of historical misconceptions about the effects of high-severity fire and the responses of ecosystems and biodiversity to such fire (Bond et al., 2012; DellaSala et al., 2014; Hanson, 2014; also see Chapter 11 and 13). This results in forest management policies that continue to focus on aggressive fire suppression, postfire logging, postfire shrub eradication and plantation establishment, homogenous low-severity prescribed burning designed to prevent mixed- and high-severity fire, and prefire mechanical thinning operations implemented across landscapes to further curb complex early seral forest habitat (Lindenmayer et al., 2004; Hutto, 2006; Hanson and North, 2008; Bond et al., 2009; DellaSala et al., 2014; Hanson, 2014).

1.4 CONCLUSIONS

Historical forest structure and fire regimes in mixed-conifer and ponderosa pine forests of western North America were far more variable than current management regimes assume, and mixed- and high-severity fires are a natural and ecologically beneficial part of many forests and shrublands. Yet the unique and ecologically rich habitat created by such fire remains demonized and, in nearly all places, is a habitat threatened by fire suppression, postfire logging (Chapter 11), and prefire management designed to reduce further the creation of postfire habitat. Ecologists are increasingly urging a shift in policies that would allow more mixed- and high-severity fire in the wildlands away from homes, while focusing on fuel reduction and fire suppression activities adjacent to homes to provide for public safety (Gibbons et al., 2012; Calkin et al., 2014; Moritz et al., 2014; see also Chapter 13). A paradigm shift in land management policies is needed to restore mixed-severity fire by allowing wildland fires to burn safely in the backcountry while protecting postfire habitat from the ecologically damaging practices of postfire logging, shrub removal, and artificial plantation establishment (Lindenmayer et al., 2004; Bond et al., 2012; DellaSala et al., 2014; Hanson, 2014).

REFERENCES

Agee, J.K., 1993. Fire ecology of Pacific Northwest forests. Island Press, Washington, DC.

Agee, J.K., 2002. The fallacy of passive management: managing for fire safe forest reserves. Conserv. Practice 3, 18–26.

Agee, J.K., Skinner, C.N., 2005. Basic principles of forest fuel reduction treatments. For. Ecol. Manage. 211, 83–96.

Amoroso, M.M., Daniels, L.D., Bataineh, M., Andison, D.W., 2011. Evidence of mixed-severity fires in the foothills of the Rocky Mountains of west-central Alberta, Canada. For. Ecol. Manage. 262, 2240–2249.

Azuma, D.L., Donnegan, J., Gedney, D., 2004. Southwest Oregon Biscuit fire: An analysis of forest resources and fire severity. U.S. Forest Service Research Paper PNW-RP-560, Pacific Northwest Research Station, Portland, OR, USA.

Baker, W.L., 2009. Fire Ecology in Rocky Mountain Landscapes. Island Press, Washington DC.

Baker, W.L., 2012. Implications of spatially extensive historical data from surveys for restoring dry forests of Oregon's eastern Cascades. Ecosphere 3, 23.

Baker, W.L., 2014. Historical forest structure and fire in Sierran mixed-conifer forests reconstructed from General Land Office survey data. Ecosphere 5, 79.

Baker, W.L., 2015. Historical Northern spotted owl habitat and old-growth dry forests maintained by mixed-severity wildfires. Landsc. Ecol. 30, 655–666.

Baker, W.L., Ehle, D., 2001. Uncertainty in surface-fire history: the case of ponderosa pine forests in the western United States. Can. J. Forest Res. 31, 1205–1226.

Baker, W.L., Veblen, T.T., Sherriff, R.L., 2007. Fire, fuels and restoration of ponderosa pine-Douglas fir forests in the Rocky Mountains, USA. J. Biogeogr. 34, 251–269.

Baker, W.L., Williams, M.A., 2015. Bet-hedging dry-forest resilience to climate-change threats in the western USA based on historical forest structure. Front. Ecol. Evol. (in press).

Beaty, R.M., Taylor, A.H., 2001. Spatial and temporal variation of fire regimes in a mixed conifer forest landscape, Southern Cascades, USA. J. Biogeogr. 28, 955–966.

Beaty, R.M., Taylor, A.H., 2008. Fire history and the structure and dynamics of a mixed-conifer forest landscape in the northern Sierra Nevada, Lake Tahoe Basin, California, USA. For. Ecol. Manage. 255, 707–719.

Bekker, M.F., Taylor, A.H., 2001. Gradient analysis of fire regimes in montane forests of the southern Cascade Range, Thousand Lakes Wilderness, California, USA. Plant Ecol. 155, 15–28.

Bekker, M.F., Taylor, A.H., 2010. Fire disturbance, forest structure, and stand dynamics in montane forest of the southern Cascades, Thousand Lakes Wilderness, California, USA. Ecoscience 17, 59–72.

Bond, M.L., Lee, D.E., Siegel, R.B., Ward Jr., J.P., 2009. Habitat use and selection by California Spotted Owls in a postfire landscape. J. Wildlife Manage 73, 1116–1124.

Bond, M.L., Siegel, R.B., Hutto, R.L., Saab, V.A., Shunk, S.A., 2012. A new forest fire paradigm: the need for high-severity fires. Wildlife Prof. 6, 46–49.

Brown, P., Kaufmann, M., Shepperd, W., 1999. Long-term, landscape patterns of past fire events in a montane ponderosa pine forest of central Colorado. Landsc. Ecol. 14, 513–532.

Buchalski, M.R., Fontaine, J.B., Heady III, P.A., Hayes, J.P., Frick, W.F., 2013. Bat response to differing fire severity in mixed-conifer forest, California, USA. PLoS One 8, e57884.

Burnett, R.D., Taillie, P., Seavy, N., 2010. Plumas Lassen Study 2009 Annual Report. U.S. Forest Service, Pacific Southwest Region, Vallejo, CA.

Calkin, D.E., Cohen, J.D., Finney, M.A., Thompson, M.P., 2014. How risk management can prevent future wildfire disasters in the wildland-urban interface. Proc. Natl. Acad. Sci. U. S. A. 111, 746–751.

Clements, F.E., 1936. Nature and structure of the climax. J. Ecol. 24, 252–284.

Cocking, M.I., Varner, J.M., Knapp, E.E., 2014. Long-term effects of fire severity on oak-conifer dynamics in the southern Cascades. Ecol. Appl. 24, 94–107.

Colombaroli, D., Gavin, D.G., 2010. Highly episodic fire and erosion regime over the past 2,000 y in the Siskiyou Mountains, Oregon. Proc. Natl. Acad. Sci. 107, 18909–18915.

Conway, C.J., Kirkpatrick, C., 2007. Effect of forest fire suppression on buff-breasted flycatchers. J. Wildlife Manage. 71, 445–457.

Cooper, C.F., 1962. Pattern in ponderosa pine forests. Ecology 42, 493–499.

Covington, W.W., 2000. Helping western forests heal: the prognosis is poor for U.S. forest ecosystems. Nature 408, 135–136.

Covington, W.W., Moore, M.M., 1994. Southwestern ponderosa forest structure: changes since Euro-American settlement. J. Forestry 92, 39–47.

Crotteau, J.S., Varner III, J.M., Ritchie, M.W., 2013. Post-fire regeneration across a fire severity gradient in the southern Cascades. For. Ecol. Manage. 287, 103–112.

DellaSala, D.A., Bond, M.L., Hanson, C.T., Hutto, R.L., Odion, D.C., 2014. Complex early seral forests of the Sierra Nevada: what are they and how can they be managed for ecological integrity? Nat. Areas J. 34, 310–324.

Dennison, P.E., Brewer, S.C., Arnold, J.D., Moritz, M.A., 2014. Large wildfire trends in the western United States, 1984–2011. Geophys. Res. Lett. 41, 2928–2933.

Dillon, G.K., Holden, Z.A., Morgan, P., Crimmins, M.A., Heyerdahl, E.K., Luce, C.H., 2011. Both topography and climate affected forest and woodland burn severity in two regions of the western US. 1984 to 2006. Ecosphere 2, 130.

Donato, D.C., Fontaine, J.B., Robinson, W.D., Kauffman, J.B., Law, B.E., 2009. Vegetation response to a short interval between high-severity wildfires in a mixed-evergreen forest. J. Ecol. 97, 142–154.

Egan, T., 2010. The Big Burn: Teddy Roosevelt and the Fire that Saved America. Houghton Mifflin Harcourt, New York.

Finney, M.A., Bartlette, R., Bradshaw, L., Close, K., Collins, B.M., Gleason, P., Min Hao, W., Langowski, P., McGinely, J., McHugh, C.W., Martinson, E., Omi, P.N., Shepperd, W., Zeller, K., 2003. Fire behavior, fuels treatments, and fire suppression on the Hayman Fire. U.S. Forest Service Gen. Tech. Rpt. RMRS-GTR-114, Rocky Mountain Research Station, Missoula, Montana, USA.

Fule, P.Z., Swetnam, T.W., Brown, P.M., Falk, D.A., Peterson, D.L., Allen, C.D., Aplet, G.H., Battaglia, M.A., Binkley, D., Farris, C., Keane, R.E., Margolis, E.Q., Grissino-Mayer, H., Miller, C., Hull Sieg, C., Skinner, C., Stephens, S.L., Taylor, A., 2014. Unsupported inferences of high severity fire in historical western United States dry forests: Response to Williams and Baker. Glob. Ecol. Biogeogr. 23, 825–830.

Fulé, P.Z., Korb, J.E., Wu, R., 2009. Changes in forest structure of a mixed conifer forest, southwestern Colorado, USA. For. Ecol. Manage. 258, 1200–1210.

Gibbons, P., van Bommel, L., Gill, A.M., Cary, G.J., Driscoll, D.A., Bradstock, R.A., Knight, E., Moritz, M.A., Stephens, S.L., Lindenmayer, D.B., 2012. Land management practices associated with house loss in wildfires. PLoS One 7, e29212.

Hagmann, R.K., Franklin, J.F., Johnson, K.N., 2013. Historical structure and composition of ponderosa pine and mixed-conifer forests in south-central Oregon. For. Ecol. Manage. 304, 492–504.

Haney, A., Apfelbaum, S., Burris, J.M., 2008. Thirty years of post-fire succession in a southern boreal forest bird community. Amer. Midland Nat. 159, 421–433.

Hanson, C.T., North, M.P., 2008. Postfire woodpecker foraging in salvage-logged and unlogged forests of the Sierra Nevada. Condor 110, 777–782.

Hanson, C.T., 2013. Pacific fisher habitat use of a heterogeneous post-fire and unburned landscape in the southern Sierra Nevada, California, USA. Open Forest Sci. J. 6, 24–30.

Hanson, C.T., 2014. Conservation concerns for Sierra Nevada birds associated with high-severity fire. Western Birds 45, 204–212.

Hanson, C.T., Odion, D.C., DellaSala, D.A., Baker, W.L., 2009. Overestimation of fire risk in the Northern Spotted Owl Recovery Plan. Conserv. Biol. 23, 1314–1319.

Hanson, C.T., Odion, D.C., 2014. Is fire severity increasing in the Sierra Nevada mountains, California, USA? Int. J. Wildland Fire 23, 1–8.

Hanson, C.T., Odion, D.C., 2015a. Historical forest conditions within the range of the Pacific Fisher and Spotted Owl in the central and southern Sierra Nevada, California, USA. Natural Areas Journal.

Hanson, C.T., Odion, D.C., 2015b. Sierra Nevada fire severity conclusions are robust to further analysis: a reply to Safford et al. Int. J. Wildland Fire 24, 294–295.

Hessburg, P.F., Salter, R.B., James, K.M., 2007. Re-examining fire severity relations in premanagement era mixed conifer forests: inferences from landscape patterns of forest structure. Landsc. Ecol. 22, 5–24.

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

Heyerdahl, E.K., Lertzman, K., Wong, C.M., 2012. Mixed-severity fire regimes in dry forests of southern interior British Columbia, Canada. Can. J. Forest Res. 42, 88–98.

Huckaby, L.S., Kaufmann, M.R., Stoker, J.M., Fornwalt, P.J., 2001. Landscape Patterns of Montane Forest Age Structure Relative to Fire History at Cheesman Lake in the Colorado Front Range. U.S.D.A. Forest Service Proceedings RMRS-P-22, Rocky Mountain Research Station, Missoula, Montana, USA.

Hutto, R.L., 1995. Composition of bird communities following stand-replacement fires in Northern Rocky Mountain (U.S.A.) conifer forests. Conserv. Biol. 9, 1041–1058.

Hutto, R.L., 2006. Toward meaningful snag-management guidelines for postfire salvage logging in North American conifer forests. Conserv. Biol. 20, 984–993.

Hutto, R.L., 2008. The ecological importance of severe wildfires: some like it hot. Ecol. Appl. 18, 1827–1834.

Iniguez, J.M., Swetnam, T.W., Baisan, C.H., 2009. Spatially and temporally variable fire regime on Rincon Mountain, Arizona, USA. Fire Ecol. 5, 3–21.

Jenkins, S.E., Hull Sieg, C., Anderson, D.E., Kaufman, D.S., Pearthree, P.A., 2011. Late Holocene geomorphic record of fire in ponderosa pine and mixed-conifer forests, Kendrick Mountain, northern Arizona, USA. Int. J. Wildland Fire 20, 125–141.

Johnson, E.A., Gutsell, S.L., 1994. Fire frequency models, methods and interpretations. Adv. Ecol. Res. 25, 239–283.

Kasischke, E.S., Turetsky, M.R., 2006. Recent changes in the fire regimes across the North American boreal region—Spatial and temporal patterns of burning across Canada and Alaska. Geophys. Res. Lett. 33, L09703.

Kaufmann, M.R., Regan, C.M., Brown, P.M., 2000. Heterogeneity in ponderosa pine/Douglas-fir forests: age and size structure in unlogged and logged landscapes of central Colorado. Can. J. Forest Res. 30, 698–711.

Keeley, J.E., Zedler, P.H., 2009. Large, high-intensity fire events in southern California shrublands: debunking the fine-grain age patch model. Ecol. Appl. 19, 69–94.

Kinney, A., 1900. Forest and water. Post Publishing Company, Los Angeles, California, USA.

Kitzberger, T., Araoz, E., Gowda, J., Mermoz, M., Morales, J., 2012. Decreases in fire spread probability with forest age promotes alternative community states, reduced resilience to climate variability and large fire regime shifts. Ecosystems 15, 97–112. http://dx.doi.org/10.1007/s10021-011-9494-y.

Klenner, W., Walton, R., Arsenault, A., Kremsater, L., 2008. Dry forests in the Southern Interior of British Columbia: Historical disturbances and implications for restoration and management. For. Ecol. Manage. 256, 1711–1722.

Kotliar, N.B., Hejl, S.J., Hutto, R.L., Saab, V.A., Melcher, C.P., McFadzen, M.E., 2002. Effects of fire and post-fire salvage logging on avian communities in conifer-dominated forests of the western United States. Stud. Avian Biol. 25, 49–64.

Lindenmayer, D.B., Foster, D.R., Franklin, J.F., Hunter, M.L., Noss, R.F., Schmiegelow, F.A., Perry, D., 2004. Salvage harvesting policies after natural disturbance. Science 303, 1303.

Malison, R.L., Baxter, C.V., 2010. The fire pulse: wildfire stimulates flux of aquatic prey to terrestrial habitats driving increases in riparian consumers. Can. J. Fish. Aquat. Sci. 67, 570–579.

Marcoux, H., Gergel, S.E., Daniels, L.D., 2013. Mixed-severity fire regimes: How well are they represented by existing fire-regime classification systems? Can. J. Forest Res. 43, 658–668.

Marlon, J.R., Bartlein, P.J., Gavin, D.G., Long, C.J., Anderson, R.S., Briles, C.E., Brown, K.J., Colombaroli, D., Hallett, D.J., Power, M.J., Scharf, E.A., Walsh, M.K., 2012. Long-term perspective on wildfires in the western USA. Proc. Natl. Acad. Sci. 109, E535–E543.

Mast, J.N., Veblen, T.T., Hodgson, M.E., 1997. Tree invasion within a pine/grassland ecotone: An approach with historic aerial photography and GIS modeling. For. Ecol. Manage. 93, 187–194.

Mast, J.N., Veblen, T.T., Linhart, Y.B., 1998. Disturbance and climatic influences on age structure of ponderosa pine at the pine/grassland ecotone, Colorado Front Range. J. Biogeogr. 25, 743–755.

Miller, J.D., Skinner, C.N., Safford, H.D., Knapp, E.E., Ramirez, C.M., 2012. Trends and causes of severity, size, and number of fires in northwestern California, USA. Ecol. Appl. 22, 184–203.

Minnich, R.A., Barbour, M.G., Burk, J.H., Sosa-Ramirez, J., 2000. Californian mixed-conifer forests under unmanaged fire regimes in the Sierra San Pedro Martir, Baja California, Mexico. J. Biogeogr. 27, 105–129.

Moritz, M.A., Batllori, E., Bradstock, R.A., Gill, A.M., Handmer, J., Hessburg, P.F., Leonard, J., McCaffrey, S., Odion, D.C., Schoennagel, T., Syphard, A.D., 2014. Learning to coexist with wildfire. Nature 515, 58–66.

Odion, D.C., Frost, E.J., Strittholt, J.R., Jiang, H., DellaSala, D.A., Moritz, M.A., 2004. Patterns of fire severity and forest conditions in the Klamath Mountains, northwestern California. Conserv. Biol. 18, 927–936.

Odion, D.C., Hanson, C.T., 2006. Fire severity in conifer forests of the Sierra Nevada, California. Ecosystems 9, 1177–1189.

Odion, D.C., Hanson, C.T., 2008. Fire severity in the Sierra Nevada revisited: conclusions robust to further analysis. Ecosystems 11, 12–15.

Odion, D.C., Moritz, M.A., DellaSala, D.A., 2010. Alternative community states maintained by fire in the Klamath Mountains, USA. J. Ecol. 98, 96–105.

Odion, D.C., Hanson, C.T., 2013. Projecting impacts of fire management on a biodiversity indicator in the Sierra Nevada and Cascades, USA: the Black-backed Woodpecker. Open For. Sci. J. 6, 14–23.

Odion, D.C., Hanson, C.T., Arsenault, A., Baker, W.L., DellaSala, D.A., Hutto, R.L., Klenner, W., Moritz, M.A., Sherriff, R.L., Veblen, T.T., Williams, M.A., 2014. Examining historical and current mixed-severity fire regimes in ponderosa pine and mixed-conifer forests of western North America. PLoS One 9, e87852.

Paritsis, J., Veblen, T.T., Holz, A., 2014. Positive fire feedbacks contribute to shifts from *Nothofagus pumilio* forests to fire-prone shrublands in Patagonia. J. Veget. Sci. 26, 89–101. http://dx.doi.org/10.1111/jvs.12225, Online.

Perry, D.A., Hessburg, P.F., Skinner, C.N., Spies, T.A., Stephens, S.L., Taylor, A.H., Franklin, J.F., McComb, B., Riegel, G., 2011. The ecology of mixed severity fire regimes in Washington, Oregon, and northern California. For. Ecol. Manage. 262, 703–717.

Perry, G.L.W., Wilmshurst, J.M., McGlone, M.S., McWethy, D.B., Whitlock, C., 2012. Explaining fire-driven landscape transformation during the Initial Burning Period of New Zealand's prehistory. Glob. Chang. Biol. 18, 1609–1621.

Pierce, J.L., Meyer, G.A., Jull, A.J.T., 2004. Fire-induced erosion and millennial-scale climate change in northern ponderosa pine forests. Nature 432, 87–90.

Platt, R.V., Schoennagel, T., 2009. An object-oriented approach to assessing changes in tree cover in the Colorado Front Range 1938–1999. For. Ecol. Manage. 258, 1342–1349.

Pyne, S.J., 1982. Fire in America: A Cultural History of Wildland and Rural Fire (Cycle of Fire). University of Washington Press, Seattle.

Raphael, M.G., Morrison, M.L., Yoder-Williams, M.P., 1987. Breeding bird populations during twenty-five years of postfire succession in the Sierra Nevada. The Condor 89, 614–626.

Romme, W.H., Despain, D.G., 1989. Historical perspective on the Yellowstone Fires of 1988. BioScience 39, 695–699.

Roos, C.I., Swetnam, T.W., 2012. A 1416-year reconstruction of annual, multidecadal, and centennial variability in area burned for ponderosa pine forests of the southern Colorado Plateau region, Southwest USA. The Holocene 22, 281–290.

Rota, C.T., 2013. Not all forests are disturbed equally: population dynamics and resource selection of Black-backed Woodpeckers in the Black Hills, South Dakota. Ph.D. Dissertation, University of Missouri-Columbia, MO.

Schieck, J., Song, S.J., 2006. Changes in bird communities throughout succession following fire and harvest in boreal forests of western North America: literature review and meta-analyses. Can. J. Forest Res. 36, 1299–1318.

Schoennagel, T., Sherriff, R.L., Veblen, T.T., 2011. Fire history and tree recruitment in the Colorado Front Range upper montane zone: implications for forest restoration. Ecol. Appl. 21, 2210–2222.

Sestrich, C.M., McMahon, T.E., Young, M.K., 2011. Influence of fire on native and nonnative salmonid populations and habitat in a western Montana basin. Trans. Am. Fish. Soc. 140, 136–146.

Shatford, J.P.A., Hibbs, D.E., Puettmann, K.J., 2007. Conifer regeneration after forest fire in the Klamath-Siskiyous: how much, how soon? J. Forestry 105, 139–146.

Sherriff, R.L., Veblen, T.T., 2006. Ecological effects of changes in fire regimes in *Pinus ponderosa* ecosystems in the Colorado Front Range. J. Veget. Sci. 17, 705–718.

Sherriff, R.L., Veblen, T.T., 2007. A spatially explicit reconstruction of historical fire occurrence in the Ponderosa pine zone of the Colorado Front Range. Ecosystems 9, 1342–1347.

Sherriff, R.L., Platt, R.V., Veblen, T.T., Schoennagel, T.L., Gartner, M.H., 2014. Historical, observed, and modeled wildfire severity in montane forests of the Colorado Front Range. PLoS One 9, e106971.

Shinneman, D.J., Baker, W.L., 1997. Nonequilibrium dynamics between catastrophic disturbances and old-growth forests in ponderosa pine landscapes of the Black Hills. Conserv. Biol. 11, 1276–1288.

Siegel, R.B., Tingley, M.W., Wilkerson, R.L., Bond, M.L., Howell, C.A., 2013. Assessing home range size and habitat needs of Black-backed Woodpeckers in California: Report for the 2011 and 2012 field seasons. Institute for Bird Populations.

Steel, Z.L., Safford, H.D., Viers, J.H., 2015. The fire frequency-severity relationship and the legacy of fire suppression in California's forests. Ecosphere 6, 8.

Stephens, S.L., Agee, J.K., Fule, P.Z., North, M.P., Romme, W.H., Swetnam, T.W., Turner, M.G., 2013. Managing forests and fire in changing climates. Science 342, 41–42.

Stephens, S.L., Ruth, L.W., 2005. Federal forest fire policy in the United States. Ecol. Appl. 15, 532–542.

Swanson, M.E., Franklin, J.F., Beschta, R.L., Crisafulli, C.M., DellaSala, D.A., Hutto, R.L., Lindenmayer, D.B., Swanson, F.J., 2011. The forgotten stage of forest succession: early-successional ecosystems on forest sites. Front. Ecol. Environ. 9, 117–125.

Taylor, A.H., Skinner, C.N., 1998. Fire history and landscape dynamics in a late-successional reserve, Klamath Mountains, California, USA. For. Ecol. Manage. 111, 285–301.

Thompson, I., Mackey, B., McNulty, S., Mosseler, A., 2009. Forest Resilience, Biodiversity, and Climate Change. A synthesis of the biodiversity/resilience/stability relationship in forest ecosystems. Secretariat of the Convention on Biological Diversity, Montreal. Technical Series no. 43, 67 pages.

van Wagtendonk, J.W., van Wagtendonk, K.A., Thode, A.E., 2012. Factors associated with the severity of intersecting fires in Yosemite National Park, California, USA. Fire Ecol. 8, 11–32.

Veblen, T.T., Kitzberger, T., Donnegan, J., 2000. Climatic and human influences on fire regimes in ponderosa pine forests in the Colorado Front Range. Ecol. Appl. 10, 1178–1195.

Veblen, T.T., Lorenz, D.C., 1986. Anthropogenic disturbance and recovery patterns in montane forests, Colorado Front Range. Phys. Geogr. 7, 1–24.

Veblen, T.T., Lorenz, D.C., 1991. The Colorado Front Range: A Century of Ecological Change. University of Utah Press, Salt Lake City, 186 pp.

Veblen, T.T., 2003. Historic range of variability of mountain forest ecosystems: concepts and applications. The Forestry Chronicle 79, 223–226.

Weaver, H.A., 1943. Fire as an ecological and silvicultural factor in the ponderosa pine region of the Pacific slope. J. Forestry 41, 7–15.

Westerling, A.L., Hidalgo, H.G., Cayan, D.R., Swetnam, T.W., 2006. Warming and earlier spring increases western US forest wildfire activity. Science 313, 940–943.

Whitlock, C., Marlon, J., Briles, C., Brunelle, A., Long, C., Bartlein, P., 2008. Long-term relations among fire, fuel, and climate in the north-western US based on lake-sediment studies. Int. J. Wildland Fire 17, 72–83.

Williams, J., 2012. Exploring the onset of high-impact mega-fires through a forest land management prism. For. Ecol. Manage. 294, 4–10.

Williams, M.A., Baker, W.L., 2010. Bias and error in using survey records for ponderosa pine landscape restoration. J. Biogeogr. 37, 707–721.

Williams, M.A., Baker, W.L., 2011. Testing the accuracy of new methods for reconstructing historical structure of forest landscapes using GLO survey data. Ecol. Monogr. 81, 63–88.

Williams, M.A., Baker, W.L., 2012a. Spatially extensive reconstructions show variable-severity fire and heterogeneous structure in historical western United States dry forests. Glob. Ecol. Biogeogr. 21, 1042–1052.

Williams, M.A., Baker, W.L., 2012b. Comparison of the higher-severity fire regime in historical (A.D. 1800s) and modern (A.D. 1984-2009) montane forests across 624,156 ha of the Colorado Front Range. Ecosystems 15, 832–847.

Williams, M.A., Baker, W.L., 2013. Variability of historical forest structure and fire across ponderosa pine landscapes of the Coconino Plateau and south rim of Grand Canyon National Park, Arizona, USA. Landsc. Ecol. 28, 297–310.

Williams, M.A., Baker, W.L., 2014. High-severity fire corroborated in historical dry forests of the western United States: Response to Fulé et al. Glob. Ecol. Biogeogr. 23, 831–835.

Wright, C.S., Agee, J.K., 2004. Fire and vegetation history in the eastern Cascade mountains, Washington. Ecol. Appl. 14, 443–459.