Chapter 12
Fire and Forests in the 21st Century: Managing Resilience Under Changing Climates and Fire Regimes in USA Forests

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Abstract Higher temperatures, lower snowpacks, drought, and extended dry periods have contributed to increased wildfire activity in recent decades. Climate change is expected to increase the frequency of large fires, the cumulative area burned, and fire suppression costs and risks in many areas of the USA. Fire regimes are likely to change due to interactions among climate, fire, and other stressors and disturbances; resulting in persistent changes in forest structure and function. The remainder of the twenty-first century will present substantial challenges, as natural resource manag-

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ers are faced with higher fire risk and the difficult task of maintaining ecological function in a rapidly changing biophysical and social landscape. Fuel treatments will continue to be important for minimizing the undesirable ecological effects of fire, and for enhancing firefighter safety; however, treatments must be implemented strategically across large areas. Collaboration among agencies, private landowners, and other organizations will be critical for ensuring resilience and sustainable forest management.

Keywords Wildfire risk · Prescribed fire · Ecological function · Drought · Stress complexes · Adaptation · Fuels treatment · Future range and variation

12.1 Introduction

Fire is perhaps the most influential natural and anthropogenic disturbance affecting the distribution, structure, and function of terrestrial ecosystems around the world (Bond and Keeley 2005; Bowman et al. 2009; Krawchuk et al. 2009; Scott et al. 2014). Many plants and animals depend on fire for their continued existence; those that do not, such as rainforest and tundra plants, are largely intolerant of burning (DeBano et al. 1998; Cochrane 2010). Spatial and temporal patterns of fire and resulting ecosystem effects, termed “fire regimes,” are governed by complex interactions among climate, fuels, vegetation, and ignition pattern and frequency across multiple scales (Agee 1998; Bradstock et al. 2012; Keane 2013; Abatzoglou and Williams 2016; Abatzoglou et al. 2017) (Fig. 12.1). A sound understanding of the interactions between biotic and abiotic ecosystem elements of fire regimes, and the effects of fire regimes on structure and function, facilitates projections of the effects of climate change on fire as an ecological process (Agee 1993; Agee 1997; Mitchell et al. 2014; Scott et al. 2014; Halofsky et al. 2020).

Higher temperatures, lower snowpacks, drought, and extended dry periods have contributed to increased wildfire activity in recent decades, particularly in western USA forests (Freeborn et al. 2016; Holden et al. 2018; Vose et al. 2018). Drier and warmer conditions have also contributed to a longer fire season in some areas (Riley and Loehman 2016). For example, the duration of the wildfire season has increased by 80 days in some parts of the western USA as a result of increased temperature (Jolly et al. 2015; Westerling 2016). A longer burning season combined with dry fuels will promote larger (and longer-duration) wildfires relative to historical wildfire activity (Riley and Loehman 2016). Earlier onset of snowmelt in spring may also contribute to lower fuel moistures at higher elevations (McKenzie et al. 2004). Warmer and drier conditions (annual, intra-annual, and interannual) can also alter vegetation and fuel characteristics (i.e., flammability and spatial distribution) (Flannigan and Van Wagner 1991; Keane et al. 2018; Syphard et al. 2018; Hessburg et al. 2019). For example, wet periods preceding dry periods can increase fuels in dry years in the American Southwest (Grissino-Mayer and Swetnam 2000). Hence,

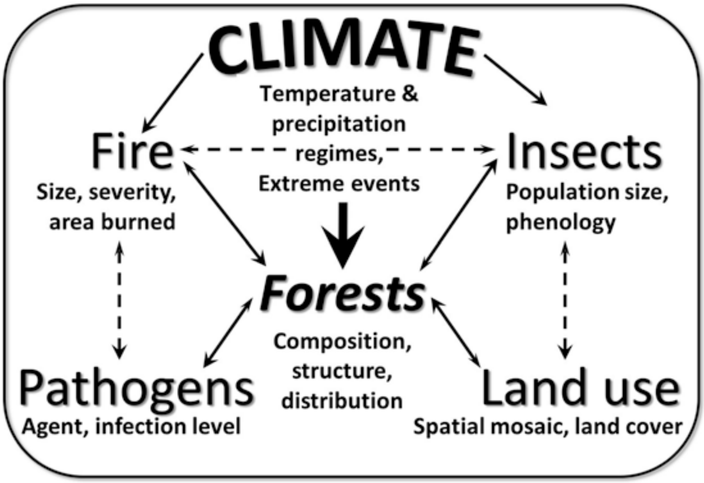

Fig. 12.1 An example of the complex interactions of a small set of ecosystem disturbances on vegetation with feedbacks. Although climate directly influences vegetation, most climate-mediated changes to forested landscapes will result from these interactions

based on recent observations over the past few decades and model projections for the coming century, we anticipate that climate change will increase fire frequency and annual area burned compared with historical wildfire patterns in many areas of the USA (e.g., Bachelet et al. 2000; Krawchuk et al. 2009, Liu et al. 2010).

The remainder of the twenty-first century is likely to present substantial challenges for natural resource managers as they deal with the task of maintaining ecological function and increasing resilience in a rapidly changing biophysical and social landscape. For example, changing drought regimes will make it more difficult to control wildfire and to use prescribed fire as a management tool (Mitchell et al. 2014; Kupfer et al. 2020). Options to increase resilience include activities that address engineering resilience (recovery to a previous condition), ecological resilience (remaining within the prevailing system domain through maintaining important ecosystem processes and functions or shifting to an alternative ecological domain), and socio-ecological resilience (the capacity to reorganize and adapt through multi-scale interactions between social and ecological components of the system) (Seidl et al. 2016). Some approaches for achieving resilience focus on maintaining or re-establishing historical conditions (Keane et al. 2018), but changing biophysical conditions and disturbance regimes may make it difficult to maintain ecosystem structure and function in emerging, no-analog environments (Falk et al. 2019).

In this chapter, we examine how climate change could affect the structure and function of forest ecosystems in the USA through changes in fire regimes. We discuss (1) observed and anticipated changes in climate; (2) the effects of climate change on fire regimes; (3) the effects of changing fire regimes on ecosystem structure and function; (4) interactions of future fire regimes with other stressors and disturbances, and; (5) management options that can increase resilience to future fire regimes. We synthesize existing science and discuss how observed responses and model projections can be used to inform natural resource management and planning.

12.2 The Fire Regime

Fire *regimes* are often described by (1) type of fire (ground, surface, crown); (2) mean and variance in fire frequency, intensity, severity, and seasonality, and; (3) areal extent and pattern of a burn (Agee 1993; Keane 2016; Bond and Keane 2017). Fire frequency describes how often a fire burns in an area, commonly measured by mean fire-return interval (FRI) or fire rotation period associated with a particular landscape (Moritz et al. 2005; Falk et al. 2011). Fire *severity* is a general term describing the effects of a fire on an ecosystem (Morgan et al. 2014) that can be estimated from the amount of plant biomass consumed and/or killed (Keeley 2006). It is often confused with fire *intensity*, which is a physical measure that describes the energy released from a fire.

Fire severity in forests differs greatly across a given landscape, depending on scale, weather conditions, and the prefire condition and composition of live and dead biomass. Although the extent, pattern, and shifting mosaic of fire severity can be complex, fire regimes are commonly classified with percentages of (1) low-severity; (2) mixed-severity, and; (3) replacement- (high) severity fires (in reference to vegetation). In many ecosystems (e.g., dry forests), low-severity fires were historically frequent, comprising surface fires with low intensities and causing low levels of mortality (<10%) of dominant vegetation. Replacement-severity fires kill the majority of the dominant vegetation (grass, shrubs, or trees) (Brown 1995) regardless of fire type (Agee 1993). Mixed-severity fires contain elements of both low-severity and replacement-severity fires (Arno et al. 2000; Perry et al. 2011).

Fire regimes are often characterized as having proportions of low-, mixed-, and replacement-severity events. Passive crown fires, patchy stand-replacement fires, and low-intensity understory burns are common in mixed-severity fire regimes (Marcoux et al. 2015). Typically, mixed-severity is used to describe patterns of patchy burn-severity in an area created during one fire event. However, mixed-severity fire regimes can also describe mixed-severity fires over time (e.g., nonlethal surface fire followed by stand-replacement fire) (Shinneman and Baker 1997). Groundfires burn within surface and subsurface organic layers, typically occurring in forests with high soil organic matter, such as boreal forests (Chap. 7), pocosin swamps of the eastern USA (Chap. 3), and dry forests where fire suppression or exclusion has led to uncharacteristically high accumulations of litter and soil organic matter (Allen et al. 2002; Turetsky and Louis 2006).

Fire regimes are created by the interactions of bottom-up and top-down controls (Heyerdahl et al. 2001). Bottom-up influences, such as vegetation, wildland fuels, topography, and patch distributions dictate fire spread, intensity, and severity at fine scales (Skinner and Chang 1996). Some bottom-up controls can be manipulated through land management activities (Keane 2015). Top-down controls are mostly driven by climate and weather that dictate fire frequency, duration, and synchrony (Swetnam 1990; Guyette et al. 2012). These top-down controls constrain the extent to which bottom-up controls, such as vegetation and fuel loading, can be manipulated to alter fire regimes.

Climate and weather trends are often embedded in atmospheric teleconnections interacting with fire regimes (Swetnam and Betancourt 1990; Duffy et al. 2005). For example, drought-induced wildfires have been associated with global circulation anomalies, such as the El Niño-Southern Oscillation (Veblen et al. 2000; Chen et al. 2011) and Pacific Decadal Oscillation (Heyerdahl et al. 2002). As a result, a fire regime is actually an aggregation of spatial disturbances that does not follow discrete mapping units. Attempts to characterize fire regimes solely from past fires (Westerling et al. 2006), vegetation (Frost 1998), fuels (Olson 1981), or topography (Keane et al. 2004) have only partially succeeded, primarily because they failed to account for the complex spatial and temporal relationships of fire and the multi-scale interactions that control fire dynamics (Morgan et al. 2014) (Fig. 12.1).

The role of ignition in fire regimes is often misunderstood (Balch et al. 2017). Lightning ignitions are distributed somewhat randomly across landscapes over long temporal scales (Barrows et al. 1977; Fuquay 1980; Balch et al. 2017), whereas human ignitions are associated with locations where human activity is present (Balch et al. 2017; Keeley and Syphard 2018). Lightning ignitions in moist, productive ecosystems can be constrained by fuel moisture (Barrows et al. 1977; Fuquay 1980). However, in these same ecosystems, human ignitions can occur year-round, potentially causing fires during short periods of dry weather (e.g., Chap. 1, Fig. 1.2, Fig. 1.4, Table 1.1).

Landscape fragmentation due to human development can also reduce fire frequencies in some fire-prone ecosystems (Gill and Williams 1996). For example, land abandonment in European countries has led to successional changes producing contiguous, highly flammable vegetation (Moreira et al. 2001; Moreno and Oechel 2012), increasing area burned in recent years (Pausas and Vallejo 1999). In many USA forests, exclusion of fires by active fire suppression and various land uses has led to an increase in understory trees and an accumulation of surface fuels, which can facilitate a transition from low-severity surface fires to mixed- or high-severity crown fires (Keane et al. 2002; Stambaugh et al. 2014).

12.3 Observed and Anticipated Changes in Climate

Most areas of the USA experienced warmer air temperatures over a recent 20-year period (1986–2015) compared to 1901–1960 (Vose et al. 2017). Changes in annual temperature have been highest (>1.5 °F [0.8 °C]) in the western USA, upper Midwest, northeastern USA, and Alaska, with winter temperatures showing even stronger warming patterns than in other seasons. The primary exception is the mid-South (Alabama, Arkansas, Louisiana, Mississippi), where temperatures have generally been unchanged or even slightly cooler, especially during summer (Vose et al. 2017), although nighttime temperatures over much of the southern USA have been increasing (Carter et al. 2018). Temperature extremes are also increasing over most of the USA, with a greater frequency of record high temperatures in the last 20 years (Vose et al. 2017).

Observed changes in precipitation indicate drying in many areas of the western USA, especially the Southwest, where recent (1986–2015) annual precipitation has decreased by 15% compared to 1901–1960 (Easterling et al. 2017). Lower annual precipitation has also been observed in the Interior West and parts of the southern USA, whereas the Midwest, Lake States, and Northeast have generally been wetter in recent years (Easterling et al. 2017). Altered seasonal patterns are also evident, with lower winter precipitation in the western USA and higher winter precipitation in the southeastern USA. Many regions are experiencing greater precipitation extremes (Easterling et al. 2017). For example, in the southern USA, heavy rainfall events (e.g., precipitation events >7.5 cm) have been increasing (Easterling et al. 2017; Carter et al. 2018). Less snow (Wehner et al. 2017) and earlier spring melting of snow is exacerbating summer drought conditions and drying fuels in some Western landscapes (Mote et al. 2018).

Model projections of future climate rely on assumptions about future greenhouse gas emissions and atmospheric concentrations. Consistent with the Fourth National Climate Assessment (USGCRP 2017), we focus on describing projections from a lower (RCP4.5 = peak in CO_2 equivalent concentration around 2040, with a value of ~550 ppm by 2100; where RCP = Representative Concentration Pathway) and higher (RCP8.5 = CO_2 concentration continues to increase, with a value of ~1250 ppm CO_2 by 2100) greenhouse gas concentration scenario (van Vuuren et al. 2011). Projections indicate continued warming across the USA, with increased average annual temperature of 8–10 °F (4.4–5.6 °C) for much of the USA by the end of the century under RCP8.5 (Fig. 12.2). Increases of 2–6 °F (1.1–3.3 °C) are projected by the mid-twenty-first century under RCP4.5. A 2–6 °F temperature increase would have significant effects on wildfire occurrence and fire regimes, in many cases accelerating changes already observed (Freeborn et al. 2016). Furthermore, both emission scenarios project an increase in extreme temperatures, as high as 14 °F (7.8 °C) on the warmest days.

Although the capacity of global climate models to project temperature is more certain than for precipitation, projections suggest generally higher precipitation (by as much as a 30% increase) in the upper Midwest, Northeast, Southeast and Pacific Northwest (especially in the winter months) under RCP8.5 by the end of the twenty-first century. Much drier conditions (30% decrease) are projected for the Interior West and Southwest and moderately drier conditions for much of the USA in summer (Fig. 12.3).

Few studies have examined the periodicity and magnitude of future drought events, due primarily to uncertainties in future climatic and environmental conditions that cause drought (Cook et al. 2015; Ryu and Hayhoe 2017; Wehner et al. 2017). Peters and Iverson (2019) used the Cumulative Drought Severity Index (CDSI) derived from a self-calibrated Palmer Drought Severity Index and future

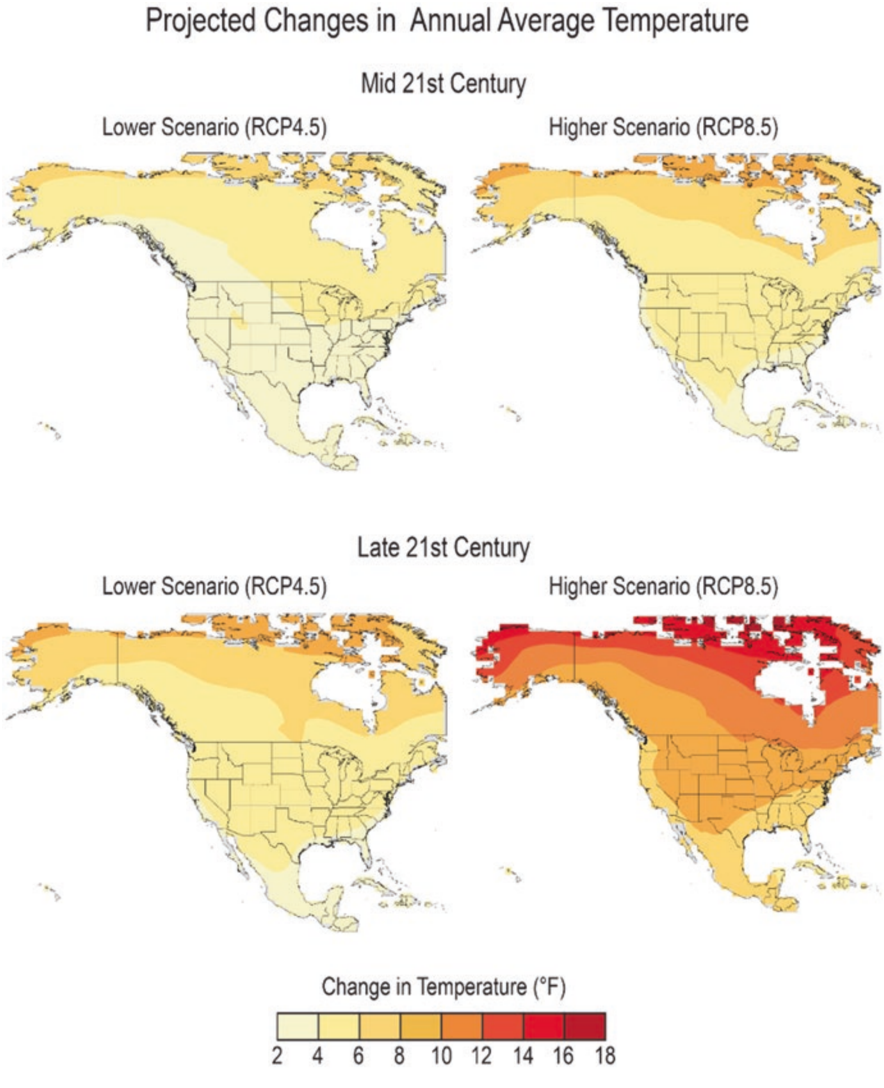

Fig. 12.2 Projected changes in average annual temperatures (°F). Changes are the difference between the average for mid-century (2036–2065; top) or late-century (2070–2099, bottom) and the average for near-present (1976–2005). Each map depicts the weighted multimodel mean. Increases are statistically significant in all areas (>50% of the models show a statistically significant change, and >67% agree on the sign of the change). (Figure source: CICS-NC and NOAA NCEI; USGCRP 2017, Fig. 6.7 in Vose et al. (2017))

temperature to project drought conditions in the USA. The CDSI suggested that more frequent and/or intense droughts are likely in the middle to latter parts of the twenty-first century across all regions of the USA. Similarly, Wehner et al. (2017) projected soil moisture conditions, finding decreased soil moisture over most of the

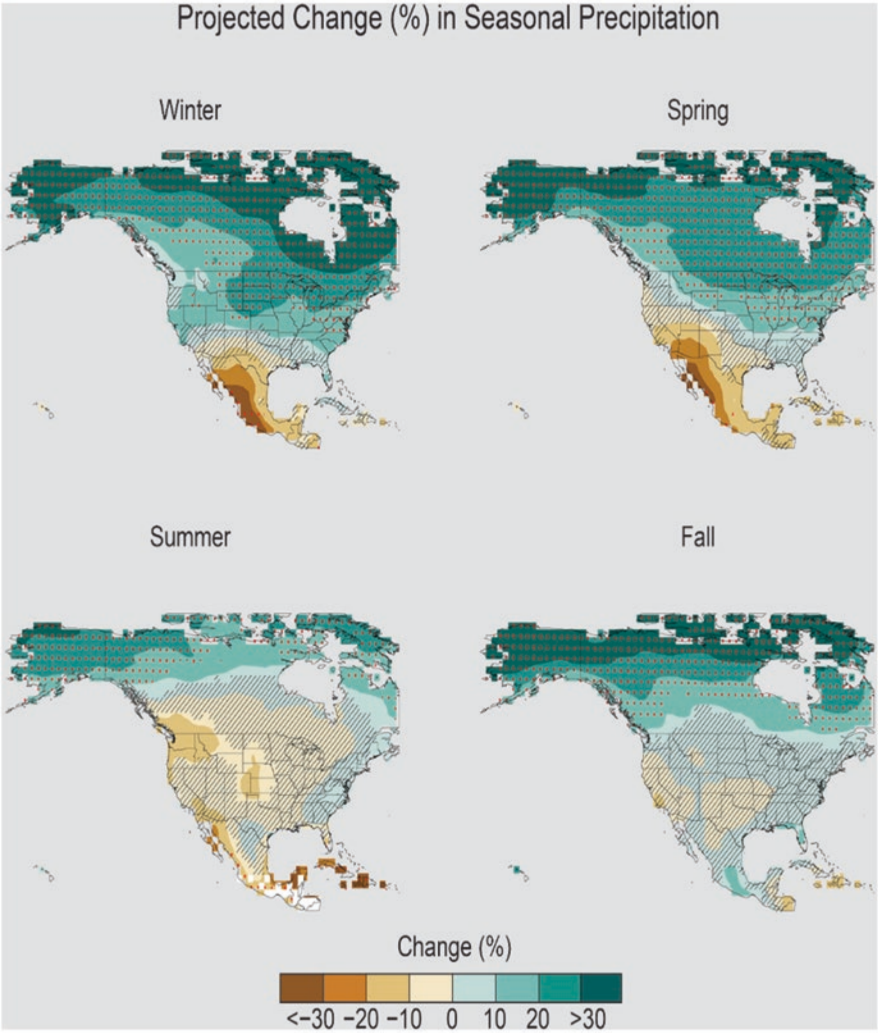

Fig. 12.3 Projected change (%) in total seasonal precipitation from CMIP5 simulations for 2070–2099. The values are weighted multimodel means (RCP8.5) and expressed as the percent change relative to the 1976–2005 average. Stippling indicates that changes are assessed to be large compared to natural variations. Hatching indicates that changes are assessed to be small compared to natural variations. Blank regions are where projections are assessed to be inconclusive. Data source: World Climate Research Program's (WCRP's) Coupled Model Intercomparison Project. (Fig. source: NOAA NCEI; USGCRP 2017; Fig. 7.5 in Easterling et al. (2017))

conterminous USA in all seasons, with the largest decreases in the Southwest and Interior West.

12.4 Effects of Future Fire Regimes on Ecosystems

The combination of higher temperature, reduced precipitation in some areas, and increased drought implies that wildfire risk will increase substantially in the future (Liu et al. 2010), resulting in more and larger fires. Associated changes in fire regimes are likely to affect the composition and structure of vegetation and forest function, a process that will probably require many decades for substantial changes to occur across large landscapes (Halofsky et al. 2020). Repeated fires (lightning- and human-caused) that occur on fire-prone landscapes over time create a shifting mosaic of plant and animal communities and structures that often reflect species ability to survive fire or to colonize burned areas shortly after fire (Agee 1998; McKenzie et al. 2011). In general, plant species with fire-adapted survival traits (e.g., thick bark, high crowns, serotinous cones, buried seeds, epicormic sprouting, and deep roots) tend to dominate landscapes with frequent fires, in the absence of co-occurring stressors such as extreme droughts, high-severity fires, and invasive species that can transform the landscape to non-forested conditions (Allen et al. 2002). In contrast, more frequent fire will likely decrease the abundance of shade-tolerant species, species with thin bark, and slow invaders after fire (Nowacki and Abrams 2008; Chmura et al. 2011; Vose and Elliott 2016). Fire-sensitive vegetation may be lost from a site if unable to regenerate in burned areas (Harvey et al. 2016).

In many forests, tree species respond to differences in fire frequency, seasonality, and fire severity (Gill and Williams 1996). Different fire intervals influence population trends and patterns of succession (Agee 1993; Schoennagel et al. 2003). In forests adapted to crown fire regimes, the effects of fire depend on demographic attributes of the species. For example, population size of nonsprouting species may fluctuate more than that of sprouting species, and local extinction of nonsprouting species may be common after a single fire (Keeley et al. 2011; Pausas et al. 2018). In frequent surface-fire regimes, lengthening fire intervals can quickly affect functional group dominance and long-term survival of resprouting species (Knapp et al. 2015). Species that are slow to mature are particularly vulnerable to more frequent fire, because populations killed by fire before they have flowered and set seed may be unable to regenerate at small spatial scales (Schlaepfer et al. 2014).

Tree populations are also negatively affected where intervals between fires promote dominance of more competitive species or exceed the lifespan of a species or its seedbank. Seasonality of fire can have a profound effect on forests, because phenological state at the time of fire influences the ability of a species to regenerate (Flannigan et al. 2000; Arthur et al. 2012). Where intensity of fires exceeds the capacity of dominant prefire species to survive or regenerate, dominance may shift to other species, such as the increase in deciduous species over conifers observed in some boreal and temperate forests (Barrett et al. 2011; Hollingsworth et al. 2013; Stambaugh et al. 2017).

Variation in fire regime characteristics affects vegetation structure at multiple spatial scales (Reilly et al. 2018). At the stand scale, frequent fire tends to reduce forest height, favor shrublands over woodlands, promote flammable or fire-tolerant species and communities, and reduce tree density and biomass (Whelan 1995; Scott et al. 2014). For example, species composition across eastern USA forests has been shifting from fire-tolerant xerophytic species (oaks, hickories, and pines) to less fire-tolerant mesophytic species (maples and yellow – poplar (*Liriodendron tulipifera*)) (Nowacki and Abrams 2008). Reasons for this shift are complex (McEwan et al. 2011; Nowacki and Abrams 2015), but fewer fires in recent decades may be an important driver (Kreye et al. 2013; Hanberry et al. 2020). Climate change may alter future fire regimes in these systems to create forests with lower overall densities and higher proportions of fire-tolerant species (Jenkins et al. 2011; Vose and Elliott 2016).

Fire can also directly influence the spatial mosaic of forest patches across large landscapes (Box 12.1), and climate-mediated changes in disturbance regimes and management interactions may cause landscape heterogeneity to increase (e.g., Marlon et al. 2009). For example, more frequent low- and mixed-severity fires may reduce fuels in drier forest ecosystems (e.g., dry mixed conifer), leading to lower-intensity fires, and a finer-scale patch mosaic (Loudermilk et al. 2019). However, factors that are both directly and indirectly related to climate change could decrease landscape heterogeneity. Continued fire exclusion, coupled with a warmer, drier climate, may promote landscape conditions in which late-seral communities dominate but are stressed from competition and drought (van Mantgem et al. 2009; Stephens et al. 2018). Simulations from global vegetation models suggest that forests might at least double in extent in the absence of fire, particularly in the flammable savanna biome (Daly et al. 2000). Wildfires that eventually burn these landscapes may become large and burn intensely, potentially creating large patches of homogeneous postburn conditions (Haugo et al. 2019) that may convert to savanna, shrub, or grassland communities, especially in areas that are too dry for tree establishment or where seed sources are eliminated (Kuenzi et al. 2008; Stroh et al. 2018).

The complexity of feedbacks among climate change, fire regime, and forest structure and function makes it challenging to project the effects of climate change on future forests. However, examining how historical and contemporary fire regimes influence forest dynamics provides some insight into how forests may respond to climate-altered fire regimes. In the following sections, we examine some of these feedbacks using examples of climate-fire relationships in boreal forests of Alaska and Canada, mesic pine forests of the Southeast, and dry conifer forests of the southwestern USA.

Box 12.1 Landscape Heterogeneity, Restoration, and Resilience

Settlement, development, and land management practices have altered the temporal and spatial characteristics of most USA forests during the past 200 years. Timber management practices have modified vegetation patch shape and structure, and fire exclusion has modified patch size and diversity. Homogeneous Western landscapes generally have low resilience to disturbance (Keane et al. 2002, 2018) and may have little ability to buffer potential climate impacts because of the high tree densities and an abundance of shade-tolerant trees (Vanderwel and Purves 2014). Wildfires and harvest activities over the last decade have returned some heterogeneity to forest landscapes, especially in wilderness areas (Campbell and Shinneman 2017; Hessburg et al. 2019). However, most are well outside of the historical range of variability in landscape structure.

It is challenging for land managers to identify an appropriate level of heterogeneity for a given landscape. Reference conditions can help prioritize and design actions that facilitate landscape heterogeneity and mitigate climate change effects. One method for estimating optimal heterogeneity uses the *historical range and variation* (HRV) of landscape characteristics as a reference (Morgan et al. 1994; Keane et al. 2009). Although the HRV of landscape metrics may not be representative of future conditions (Millar and Woolfenden 1999), it can approximate landscape conditions under which ecosystems have developed over the past thousand years. It is reasonable to assume that these historical conditions produced resilient systems and landscapes (Landres et al. 1999).

The departure of landscapes from historical compositions and structures must be known in order to design effective restoration actions (Keane et al. 2009, 2019; Dickinson 2014). Simulation modeling can be used to generate both HRV time series and future management effects (Loehman et al. 2020). For example, simulation modeling has been used to determine the efficacy of wildland fire suppression activities in the future by comparing with past HRV time series (Keane et al. 2019).

Increased stress associated with climate change—higher temperatures, more droughts, more disturbances—means that restoration (based on past conditions) may need to transition to *resilience building*. This can be accomplished by considering the *future range and variation* (FRV) of landscape characteristics as altered by climate change (Keane et al. 2018, Peterson and Halofsky 2019). FRV can be informed by climate projections integrated with ecosystem modeling to understand the environment in which future forests will regenerate and grow. Land managers can then consider which species, genotypes, structures, and spatial patterns are likely to be resilient several decades into the future.

12.4.1 Boreal Forests

Fire is the dominant disturbance regime in boreal forests (Chapin et al. 2006; Johnstone et al. 2010; Brown and Johnstone 2012; Hollingsworth et al. 2013). Paleorecords indicate that millennial-scale fire-climate relationships are highly influenced by vegetation dynamics (Hu et al. 2006). For example, during cooler, wetter climatic conditions in the Holocene (ca. 4000–7000 ybp), fires occurred relatively frequently compared with preceding millennia, likely because these conditions favored establishment of black spruce (*Picea mariana*) over forests of *Populus* species or white spruce (*P. glauca*) (Lynch et al. 2004; Hoecker et al. 2020). Black spruce forests are more flammable than white spruce or mixed white spruce-deciduous forests due to lower crown-base heights, persistent fine branches, and association with flammable species of understory vegetation (Hu et al. 2006). Little is known about how fire regimes changed in response to major biome shifts before the establishment of black spruce, but following establishment of the modern boreal forest, high-intensity crown fires occurred at intervals of 80–150 years (de Groot et al. 2013).

Under historical climate and fire regimes, boreal forest regeneration following wildfires occurs as (1) self-replacement of the same dominant prefire tree species in the postfire environment, or (2) multiple successions of tree species that assume postfire canopy dominance (Kurkowski et al. 2008). Boreal tree species adaptations for postfire regeneration include cone serotiny (black spruce), rapid resprouting (birch and poplar species), and wind-dispersed seeds (white spruce) (de Groot et al. 2013; Greene and Johnson 2000). Successional pathways depend on soil conditions, prefire species composition, and fire severity and frequency, all of which are affected by variation in temperature and precipitation (Kasischke et al. 2000; Johnstone et al. 2011).

The boreal region of North America is warming twice as fast as the global average (Scheffer et al. 2012; IPCC 2014). Wildfires and seasons with heightened wildfire activity have increased in boreal Alaska over the past 40 years, in association with above-average summer temperatures and prolonged drought (Xiao and Zhuang 2007). Climate change is expected to increase fire season length, increase area burned, and intensify fire weather (Zhang and Chen 2007). Climate-induced shifts in boreal fire regimes can lead to a third successional pathway: species replacement, in which atypically frequent or severe wildfires extirpate prefire canopy species, allowing for new species to dominate (Kurkowski et al. 2008; Johnstone et al. 2010). Increased fire severity may fundamentally reshape future landscapes of Alaska, because high-severity fires can promote conversion of coniferous spruce ecosystems to deciduous hardwoods, leading to broad-scale transformations of forest community composition and structure that may persist for centuries (Johnstone et al. 2010). A shift from conifer forest to less flammable, deciduous-dominated forest could result in a negative feedback to future fire or a shift from canopy- to surface-dominated fire regimes (Johnstone et al. 2011; Hudspith et al. 2015; Hart et al. 2019), unless a warmer climate reduces fuel moisture sufficiently to carry fire in deciduous fuel types (Flannigan et al. 2005).

12.4.2 Fire-Prone Forests in the Southeast

Fire-prone forests in the southeastern USA (Chap. 3) have among the shortest FRIs in the world (1–4 year) (Christensen 1981), but forest change in the past two centuries has been driven largely by forest conversion, development, landscape fragmentation, and reduced fire frequency through fire exclusion (Landers et al. 1995). Fire regimes in this region are now dominated by human ignitions in the form of prescribed fire (Melvin 2018). Maximum temperatures have not risen significantly regionally (Carter et al. 2018), but models project temperatures will increase by the mid-twenty-first century, potentially causing more wildfires (Prestemon et al. 2016).

In this human-dominated landscape, climate change influences on fire regimes are mediated through the filters of human intervention. Managed fire regimes target FRIs of 1–5 years; some studies suggest that managed fire regimes should include months during which lightning ignitions were historically dominant (Platt et al. 1988; Slocom et al. 2007). Climate change is projected to reduce prescribed burning opportunities in the spring and summer due to elevated risks of escape (Kupfer et al. 2020), so the trajectory of ecosystem change in frequently burned communities may increasingly be driven by wildfire. For example, drier and hotter weather conditions will alter fire potential across most ecoregions in the southern USA, which may increase fire season length (Liu et al. 2014) and reduce the number of days when weather conditions are within prescribed fire prescriptions (Kupfer et al. 2020).

Altered prescribed fire frequency has feedbacks on fuel loading, which will increase as the interval between burning widens. Compared to current prescribed fire behavior, prescribed fires in the future may be more severe, kill more and larger trees, and produce more smoke (Mitchell et al. 2014). Although fuel accumulations and consecutive drought years have increased wildfire in some areas (e.g., Georgia) (Terando et al. 2016), other areas in the Southeast may become less flammable as fire is removed from the landscape (Glitzenstein et al. 2012). This alternate stable state in the absence of prescribed fire produces a more mesic forest with longer fire-return intervals and altered ecosystem properties that may perpetuate reduced flammability (McKay and Parker 2001; Kreye et al. 2013; Carpenter et al. 2020) under future climates, although this transition is uncertain (Vose and Elliott 2016).

12.4.3 Fire-Prone Forests in the Southwest

Fire regimes in fire-prone forests of the southwestern USA (Chap. 11) have been altered by a combination of land management and drought. Historically, low-severity fires in dry forests occurred frequently, spreading in grassy surface fuels coincident with an alternating pattern of increased cool-season precipitation (resulting in fuel growth) followed by cool-season drought (resulting in fuel flammability) (Swetnam and Betancourt 1998; Margolis et al. 2017). A hundred-plus years of livestock grazing, logging, and fire exclusion have led to less frequent fire in dry conifer forests compared with the presettlement period, creating increased surface fuel loads, dense forests, and reduced structural and spatial heterogeneity of vegetation (Allen et al. 2002; Reynolds et al. 2013). Fires in these forests can be more intense with larger patches of high-severity fire than occurred historically (Allen 2016; Singleton et al. 2019), reducing biodiversity, eliminating tree seed sources, and altering ecological function and resilience (Guiterman et al. 2018; Haffey et al. 2018).

Climate change has increased the length of fire seasons and lowered fuel moistures, making large portions of the landscape flammable for longer periods of time (Riley and Loehman 2016), and widespread, regional fire years have been associated with prolonged droughts (Heyerdahl et al. 2008). However, at some point in the future, increased fire activity in these low-productivity ecosystems may be self-limiting, such that fuel consumption limits frequency, extent, and effects of subsequent fires (Collins et al. 2007; Parks et al. 2015).

12.5 Interactions of Future Fire Regimes and Other Stressors and Disturbances

Direct and indirect interactions among fire and other stressors and disturbances will result in persistent changes in forest composition and structure under climate change (Fig. 12.1). In the context of future fire regimes, these changes are especially important for how they influence fuel loads, fire severity, and flammability.

12.5.1 Native Insects and Diseases

Insects and diseases influence forest ecosystem structure and function in complex ways, regulating primary production, nutrient cycling, ecological succession, and the distribution and abundance of plants and animals (Mattson and Addy 1975; Castello et al. 1995). The effects of insects and diseases are most influenced by abiotic factors (e.g., weather, wildfires, avalanches, windstorms) and management activities.

Climate change is expected to increase the effects of some insects and diseases in USA forests and reduce the effects of others (Vose et al. 2018). The effects of insects and diseases can be low level/chronic or acute/episodic. Many USA states have at least 10% of their forested landscapes at risk to forest insects and diseases, in which at least 25% of standing live basal area >2.5 cm in diameter will be killed in the next 15 years in the absence of remediation (management actions applied to increase resistance and resilience). Based on the latest National Insect and Disease Forest Risk Assessment, four tree species are expected to suffer substantial mortality across their extent, including redbay (*Persea borbonia*) (90% of total basal area),

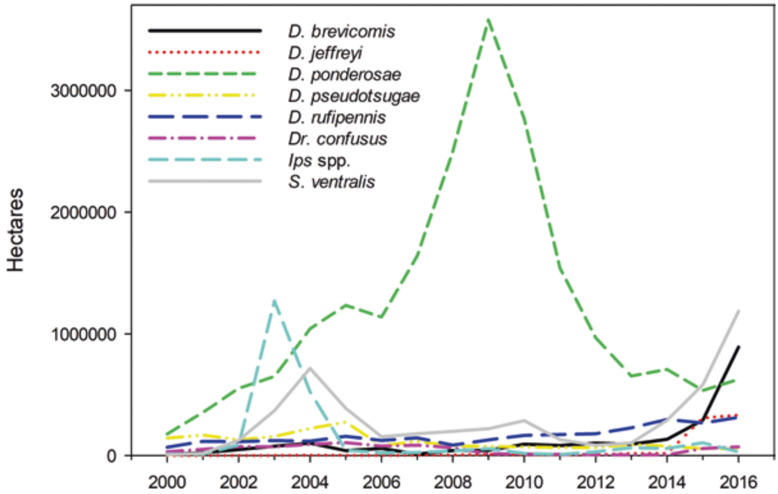

Fig. 12.4 Bark beetles and wildfire are principal drivers of change in conifer forests. This figure shows the area affected by native bark beetles in the western USA, where impacts are most severe, 2000–2016. About 10.3 million ha were affected by mountain pine beetle (*Dendroctonus ponderosae*) alone, which represents almost half of the total area for all bark beetles combined in the western USA (Fettig et al. 2021). Values represent the effect observed each individual year and should not be summed across years (i.e., there may be overlap in areas affected from year to year). Note that the line for *D. jeffreyi* is overlapped by the line for *D. pseudotsugae*. Fig. developed using data from the National Insect and Disease Survey database, USDA Forest Service (Fettig *unpubl. data*)

whitebark pine (*Pinus albicaulis*) (58%), limber pine (*P. flexilis*) (44%), and lodgepole pine (*P. contorta* var. *latifolia*) (39%) (Krist et al. 2014).

Bark beetles cause most of this tree mortality, and several species have undergone notable outbreaks in recent years (Fig. 12.4; Chaps. 8, 9, 10). This is primarily due to climate and human-caused changes in forest structure and composition (Fettig et al. 2007; Bentz et al. 2010; Kolb et al. 2016), which influence host susceptibility, and beetle survival and population growth. For example, increased densities of grand fir (*Abies grandis*) and white fir (*A. concolor*) in the Cascade Range and Sierra Nevada, which are largely due to fire exclusion and selective harvest of pines, have led to increased impacts by fir engraver (*Scolytus ventralis*), especially during periods of elevated temperatures and drought (Fettig et al. 2021).

Increased wildfire and bark beetle outbreaks raise concerns about the effects of wildfires on bark beetles, and the effects of bark beetle outbreaks on fuels and fire regimes (Hicke et al. 2012; Fettig et al. 2020). Following fire, tree mortality can be immediate, or it can occur over multiple years as a result of fire injuries to the crown, bole, or roots (Hood et al. 2018). Levels of delayed tree mortality caused by bark beetles depend on tree species, tree size, tree phenology, degree of fire-caused injuries, initial and postfire levels of tree vigor, the postfire environment, and the scale, severity and composition of bark beetle populations and other tree mortality agents. Trees moderately injured by fire are generally most susceptible to bark

Fig. 12.5 Much of California experienced a severe drought in 2012–2015, inciting a large tree mortality event in the central and southern Sierra Nevada. Much of the tree mortality was attributed to western pine beetle (*Dendroctonus brevicomis*) (Fettig et al. 2019). Bark beetle outbreaks can alter fuels sufficiently to affect fire regimes and firefighter safety (photo L.M. Mortenson, USDA Forest Service, Pacific Southwest Research Station)

beetles. High-severity fires kill large numbers of potential host trees, often reducing bark beetle populations (Fettig et al. 2020). Bark beetles routinely cause additional tree mortality after low- to moderate-severity fires.

Altered fire behavior in forests that have experienced beetle outbreaks are manifest through changes in fuel loadings (Fig. 12.5), depending on the severity of the outbreak (the proportion of trees colonized and killed by bark beetles) and the amount of time since the outbreak occurred. Altered foliar moisture following beetle colonization affects flammability, but this change is short-lived. For example, the needles and twigs of lodgepole pine killed by mountain pine beetle lose 80–90% of their water content within one year. Loss of moisture increases flammability by shortening time to ignition, lowering temperature at ignition, and raising heat yields when burned (Page et al. 2012). Over time, canopy bulk density and canopy cover decline as beetle-killed trees shed needles, then small twigs and larger branches (Fig. 12.5). Bark beetle outbreaks can increase wildfire rates of spread, with crown fire potential peaking 1–3 years after outbreaks (Fig. 12.6), although outbreaks have little effect on the extent of area burned (Hart et al. 2015) or likelihood of wildfire occurrence (Bebi et al. 2003; Meigs et al. 2015).

Defoliating insects can influence fire behavior and severity (Hummel and Agee 2003), but these relationships are still being explored. Fire potential in balsam fir (*A. balsamea*) forests affected by eastern spruce budworm (*Choristoneura*

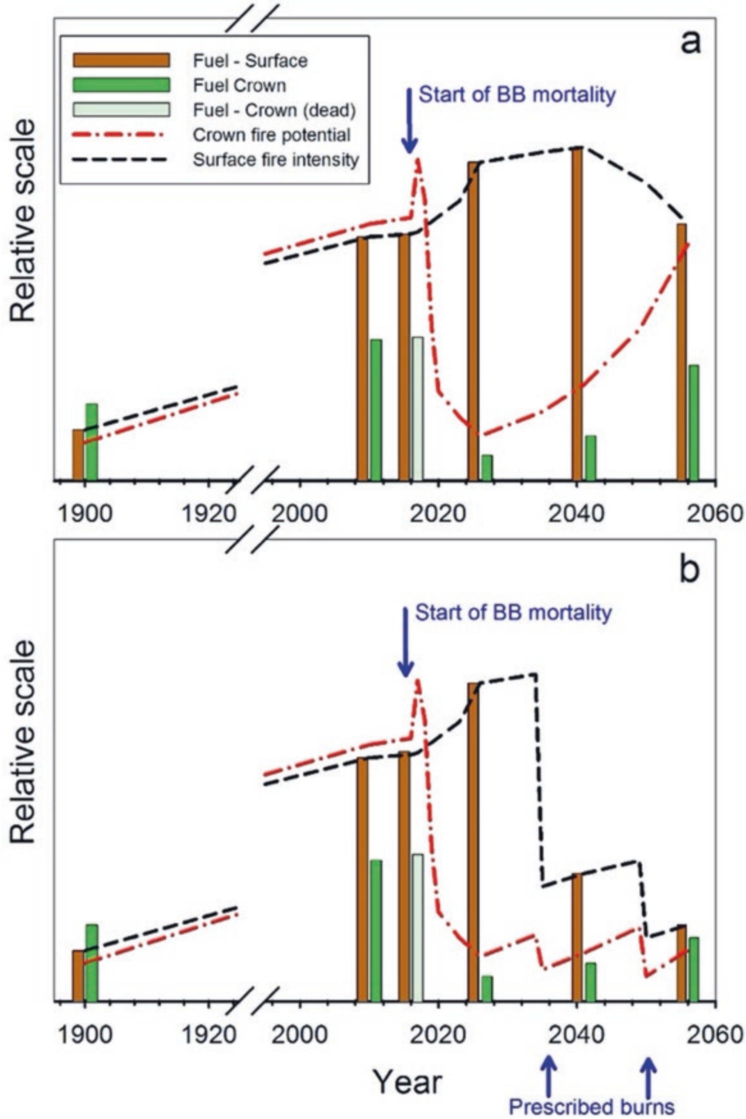

Fig. 12.6 Conceptual diagram showing changes in fuel profiles and wildfire behavior following severe drought and a bark beetle outbreak in a Sierra Nevada mixed-conifer forest without (**a) and with (b**) periodic fuel reduction treatments. Similar trends may be observed for other disturbances that cause similar patterns and levels of tree mortality. A greater potential for "mass fires" now exists in the central and southern Sierra Nevada, driven by the amount, size, and continuity of dry combustible woody fuels. Adapted from Stephens et al. (2018)

fumiferana) was shown to be significantly higher for several years following tree mortality (Stocks 1987). Fire potential peaked 5–8 years after tree mortality and decreased gradually as surface fuels began to decompose. Flower et al. (2014) proposed that if there is a relationship between defoliators and fire, it is a subtle synergistic relationship in which climate determines the probability of occurrence of each disturbance, and each disturbance dampens the severity but does not alter the probability of occurrence of other disturbances. Forest diseases can also affect fire regimes, although this relationship is not well studied. In Oregon, fire reduced the area infested by laminated root rot (*Phellinus weirii*) by favoring less susceptible host species and reducing the modal size of dead roots and logs, but infestations enhanced the probability of stand-replacing fires (Dickman and Cook 1989).

12.5.2 Invasive (Nonnative) Plants

Although most studies indicate that climate change will accelerate establishment of non-native plants in the USA (Finch et al. 2021), projecting how individual taxa will respond to different climate change scenarios and interact with fire regimes is difficult. Invasive plants, which may exert important effects on fire regimes, generally respond more favorably to elevated CO_2 than native plants (Bradley et al. 2010a), whereas responses to altered temperature and precipitation are variable (Dukes et al. 2009). Many invasive species arrive as stowaways in cargo ships. In the USA, current inspection of cargo ships for invasive species involves examining a low percentage of imports (<2%) for a small subset of federally-listed species (Finch et al. 2021). Hence, it is likely that the influx of invasive species will continue, although their effects on future fire regimes is uncertain.

Cheatgrass (*Bromus tectorum*) is an annual species that dies and dries out in spring, increasing the flammability and continuity of surface fuels (Brooks et al. 2004). Its rapid growth rate, ability to photosynthesize at low temperatures (Rice et al. 1992; Chatterton et al. 1993), and competitiveness under drought stress (Melgoza and Nowak 1991) contribute to its dominance. Cover by cheatgrass is now >15% on over 210,000 km^2 in the Intermountain West (Bradley et al. 2018). These lands are twice as likely to burn as those with a low abundance of cheatgrass, and four times more likely to burn multiple times within a 15-year period. Even small amounts of cheatgrass (1–5% cover) increase fire risk (Balch et al. 2013; Bradley et al. 2018), which is concerning given that the density and distribution of cheatgrass are projected to increase with climate change (Zelikova et al. 2013; Finch et al. 2021).

In the southeastern USA, cogongrass (*Imperata cylindrica*) is an invasive grass that can alter fire regimes by increasing fine-fuel loads, expanding horizontal fuel continuity, and increasing fuel depth, causing higher fire temperature at greater heights (Lippincott 2000). The current distribution of cogongrass is limited (at least in part) by cold; however, higher temperatures may expand its range northward and westward, increasing its role in altering fire regimes in more areas of the eastern USA (Bradley et al. 2010b).

A related concern is the increased demand for nonnative plants with greater tolerances to warmer temperatures and drought for use in landscaping (Bradley et al. 2012). The risk of these plants escaping and becoming invasive is increasing with urbanization (Marco et al. 2010). These concerns are most relevant to wildfire in the West, especially in areas with a Mediterranean climate where fires can be frequent (Chap. 9). In the eastern USA (especially in Southern forests) where the majority of prescribed burning occurs, nonnative and native invasive trees, shrubs, and grasses can confound restoration activities that use prescribed fire.

12.6 Managing for Resilience

Managing for ecological resilience, defined broadly as the ability of a system to recover after disturbance, is a strategy intended to mitigate the environmental challenges caused by climate change, shifting fire regimes, nonnative flora and fauna, industrialization, urbanization, and their interactions (Seidl et al. 2016; Keane et al. 2018). Some strategies for achieving resilience focus on maintaining or re-establishing historical conditions (Keane et al. 2018). However, rapid climate change and altered disturbance regimes will make it difficult to maintain ecosystem functionality in emerging, no-analog environments (Falk et al. 2019).

Resilience management focuses on improving the long-term response of a system by focusing on specific attributes or drivers, and developing principles for human actions (Benson and Garmestani 2011). Management actions that can increase resilience in forest landscapes typically balance multiple and often competing objectives. Managed fire regimes are a critical element of these activities for conservation outcomes and ecosystem resilience (Stephens et al. 2018; Freeman et al. 2019). Specifically, management options can be crafted to conserve fire-dependent species, reduce wildfire risks, and address climate-related effects and influences (Carter et al. 2018; Vose et al. 2018).

Planning and sequencing of management activities at relevant scales are critical for managed fire regimes, enhancing resilience in fire-prone ecosystems (Krofcheck et al. 2018). For example, forest thinning can increase tree vigor as well as reduce fuels, thus minimizing the intensity and severity of future wildfires (Reinhardt et al. 2008; Hurteau et al. 2019). Thinning guidelines may need to be revised with fewer residual trees for some cover types (e.g., ponderosa pine [*Pinus ponderosa*]) to maintain adequate levels of resilience to drought and disturbances (e.g., bark beetle outbreaks) that influence fire regimes (Peterson et al. 2011a, b). Fuel reduction treatments may need to be more aggressively applied in the future at large spatial scales to yield effective outcomes in treated stands. In addition, more extreme fire weather conditions associated with climate change (Collins 2014) will likely require higher biomass removals to achieve resilience and fuel management objectives.

The near-term challenge in using restoration or fuel treatments is to apply thinning and managed fires in landscapes of highest priority. For example, aggregation and prioritization of treatments in the Lake Tahoe Basin (California) have reached a stage that, according to model results, will alter large wildfire potential (Loudermilk et al. 2014). However, documentation of treatment effectiveness for modifying wildfire potential at large spatial scales in Western landscapes is minimal (with the exception of protecting communities with focused treatments), and few landscapes have been treated at sufficient levels to affect long-term resilience.

In the eastern USA, the intersection of prescribed fire, conservation, and restoration is well developed, particularly in the southeastern USA Coastal Plain (Mitchell et al. 2009; Chap. 3). Prescribed fire use in the southern Great Plains and the Southeast is comparable to or greater than the areal extent of wildfire in the West (Melvin 2018; Hiers et al. 2020). However, climate change is expected to challenge current practice and threaten established restoration management practices (Mitchell et al. 2014; Kupfer et al. 2020). Although prescribed fire is an effective mitigation strategy for reducing wildfire risk (Addington et al. 2015), increasing the capacity of resource managers to safely use this treatment is often constrained by degraded air quality and smoke emissions (Kobziar et al. 2015; Chiodi et al. 2018).

Reducing the effects of existing non-climatic stressors on ecosystems, such as landscape fragmentation and invasive plants, will also increase ecosystem resilience to climatic changes (Joyce et al. 2008). For example, tactics to minimize establishment and spread of invasive species include early detection/rapid response for new invasions, implementing weed-free policies, preventing invasive plant introductions during projects, and planting locally-adapted species of native vegetation to compete with invasive plants (Table 12.1).

12.6.1 Managing Wildfire

Wildfire management involves a broad range of ecological considerations and interactions (Table 12.1, Fig. 12.5, Box 12.1). Wildfires can be actively suppressed (uncontrolled wildfires) or they can be allowed to burn under a prescribed set of weather conditions while being observed (wildfires for resource benefits). The latter option is rarely used, so fuels tend to accumulate well beyond what would occur under a "natural" fire cycle, thus perpetuating elevated fuels and fire hazard, especially in dry forests. In many landscapes, the unconsumed fuels remaining after a wildfire may be insufficient to support the spread of future fires, so previously burned areas act as a firebreak and impede fire spread (Ricotta et al. 1999; Agee et al. 2000; Peterson 2002; McKenzie et al. 2011). As a result, the continuity of flammable vegetation at large spatial scales influences the spread of future fires, and the pattern and amount of area burned influence fuel continuity (McKenzie et al. 2011; Keane et al. 2012).

Wildfire suppression tactics, such as retardant and water drops, backfiring, and fireline construction, can also be used to protect infrastructure, property, and

Table 12.1 Key climate change vulnerabilities of forests related to fire and associated adaptation strategies and on-the-ground tactics. These adaptation options were developed by resource managers in adaptation workshops across the western USA, and are summarized in the Climate Change Adaptation Library (http://adaptationpartners.org/library.php)

Climate change vulnerability	Adaptation strategy	Adaptation tactic
Increased temperatures, drier summers, and lower snowpack will result in more fire (larger areal extent and more high-severity patches).	Plan and prepare for more area burned.	Incorporate climate change into fire management plans.
		Anticipate more opportunities to use wildfire for resource benefit.
		Plan postfire response for large fires.
		Consider using prescribed fire to facilitate transition to a new fire regime in drier forests.
		Consider planting fire-tolerant tree species after fire in areas with increasing fire frequency.
		Manage forest restoration for future range of variability.
	Increase resilience of existing vegetation by reducing hazardous fuels and forest density and maintaining low densities.	Thin and burn to reduce hazardous fuels in the wildland-urban interface.
		Increase intentional use of lightning-ignited fires and management of re-ignition of lightning-ignited fires.
		Consider using more prescribed fire where scientific evidence supports change to more frequent fire regime.
		Use prescribed fire to maintain structure and promote fire-tolerant conifer species.
		Increase interagency coordination and shared risk.
		Conduct thinning treatments (precommercial and commercial).
		Use regeneration and planting to influence forest structure.
	Increase resilience through postfire management.	Consider climate change in postfire rehabilitation.
		Determine where native seed may be needed for postfire planting.
		Anticipate greater need for seed sources and propagated plants.
		Experiment with planting native grass species to compete with invasive species after fire.
		Increase postfire monitoring in areas not currently monitored.

(continued)

Tabel 12. 1 (continued)

Climate change vulnerability	Adaptation strategy	Adaptation tactic
		Identify, prioritize, and protect values at risk; initiate programs to assess values and determine best protective actions.
		Conduct prefire planning to improve response time and efficiency, prioritizing key areas at risk to geologic hazard.
		Conduct postfire vegetation management and prevent invasives.
	Manage forest vegetation to reduce severity and patch size; protect refugia (e.g., old trees).	Map fire refugia.
		Include gaps in silvicultural prescriptions.
		Identify processes and conditions that create fire refugia.
	Use high-severity wildfires as opportunities to modify ecosystem structure.	Use postfire timber harvest to prevent uncharacteristic reburns.
		Allow some burned areas to regenerate naturally.
	Manage forest landscapes to encourage fire to play a natural role.	Implement fuel breaks at strategic locations.
		Create incentives to encourage managed wildland fire.
		Implement strategic density management through forest thinning.
		Push boundaries of prescribed burning (e.g., burn earlier in spring, later in summer).
Disturbances will alter ecosystem structure, species distribution, and species abundance across large landscapes.	Increase knowledge of patterns, characteristics, and rates of change in species distributions.	Expand long-term monitoring programs.
	Create landscape patterns that are resilient to expected disturb-ance regimes.	Continue research on expected future disturbance regimes; evaluate potential transitions and thresholds.
		Improve communication across boundaries.
		Manage for diversity of structure and patch size with fire and mechanical treatments.
Lack of disturbance has caused shifts in species composition and structure in dry mixed-conifer forest, creating a risk of high-severity fire with climate change.	Maintain and restore species and age class diversity.	Identify and map highest risk areas across large landscapes to provide context for prioritization.
		Reduce stand density and shift composition toward species that are more fire adaptive and drought-tolerant.
		Restore age class diversity while protecting legacy trees.

(continued)

Tabel 12. 1 (continued)

Climate change vulnerability	Adaptation strategy	Adaptation tactic
The frequency and scale of disturbance will likely increase with climate change.	Promote disturbance-resilient species.	Thin to favor disturbance-resilient species.
		Plant disturbance-resilient species.
		Promote disturbance-resilient species with prescribed fire and/or managed wildland fire.
Areas with limited species and genetic diversity will likely be more susceptible to climate change stressors.	Promote species and genetic diversity.	Plant potential microsites with a mix of species.
		Maintain species diversity during thinning.
		Interplant to supplement natural regeneration and genetic diversity.
Current dry forest conditions (overstocked stands with more shade-tolerant species as a result of fire suppression) increase vulnerability to drought and wildfire.	Actively manage dry forest areas that are susceptible to fire and drought.	Conduct more intensive thinning.
		Introduce frequent fire.
		Promote ponderosa pine by favoring frequent fires.
Higher-elevation forests may burn more frequently with climate change.	Increase resilience of vegetation types at high elevations.	Increase landscape heterogeneity with prescribed fire.
		Use fire behavior and spatial modeling to identify high-priority areas to reduce or maintain fuels.
		Use silvicultural practices (e.g., prescribed fire, thinning) to reduce fire hazard.
Climate change may increase disturbance interactions, compounding effects.	Increase post-disturbance planning, management, and treatment implementation.	Create a strategy and develop criteria to prioritize areas that are more likely to recover after disturbance (e.g., critical habitats, population served by disturbed habitat).
		Promote climate-adapted species (species resistant and resilient to disturbance) and genotypes.
		Identify sites more susceptible to compounding disturbances (e.g., with high fuel loads, beetle-caused tree mortality, invasives); monitor disturbance occurrence; prioritize seed sources to preserve some sites; map sites across landscapes; conduct proactive treatments in areas more resistant to disturbance.

wildlife habitat (Reinhardt et al. 2008). Postfire activities that mitigate adverse wildfire impacts include erosion control, site stabilization, and planting trees when fire has killed the majority of seed sources. These actions can be implemented to protect ecosystem components that were burned by fire and to restore or maintain resilience. The application of new coupled fire-atmospheric modeling tools to managed wildfires may offer firefighters and communities greater opportunities to use wildfires to increase resilience (Hiers et al. 2020; Linn et al. 2020).

Managing the resilience of fire-prone ecosystems in the USA will likely require more prescribed fire (Hiers et al. 2020). Unlike in the Southeast where prescribed burning is widespread, managed wildfire provides a flexible option for managing fire regimes and increasing resilience in the West. Expanding the use of managed wildfire in the West will be challenging due to risk aversion in decision making (relative to protecting human communities and values) and due to sociopolitical pressure to suppress fires (Hiers et al. 2020).

12.6.2 Wildland Treatments and Effects

In the USA, prescribed fires dominate fire management east of the Rocky Mountains, with nearly five million ha burned annually for a variety of objectives (Melvin 2018; Chap. 8). In the western USA, fuel treatments designed to reduce wildfire intensity and the severity of effects are rarely implemented at scales that are large enough to mitigate undesirable wildfire impacts (Hiers 2017; Kolden 2019). Efforts to reduce fuel loads at meaningful scales are critical but may promote invasive plants in some cases (Schwilk et al. 2009).

Despite the challenges of large-scale implementation of fuel treatments, proactive treatments will undoubtedly play an increasingly important role in reducing wildfire impacts on living trees and in restoring the role of fire in forest ecosystems. However, treating fuels is only one of many objectives for planning and implementing these treatments. Care should be given to maintain stand structures and compositions that facilitate the survival of fire-adapted plant species after wildfires and promote other desired outcomes, such as habitat quality for specific animals.

Eliminating vegetation that competes with fire-adapted trees can improve tree vigor. This will be increasingly important as the climate warms, because low soil moisture, insect outbreaks, and other stressors (including wildfire) will be more prevalent. Increased vigor may allow trees to remain on the landscape for a longer period of time, as well as increase the quality and possibly the quantity (e.g., frequency of cone crops) of reproductive propagules.

Removal of fuels with mechanical treatments and prescribed fire can target specific portions of large landscapes, optimizing the social and ecological value of treatments (Table 12.1). Wildfires are often considered undesirable from a social perspective, but they can in some cases quickly create large-scale heterogeneity that would be difficult to achieve with prescribed fire and mechanical fuel treatments.

Fuel treatments involve reducing canopy fuels by cutting, masticating, or burning living trees, and reducing surface fuels by burning or mechanical removal (Stephens et al. 2012; McIver et al. 2013). Reducing fuels in or near stands that contain ecologically valuable trees can be an important hedge against losing them to future wildfires. However, some fuel treatments, such as mastication, are not designed with ecological relationships in mind. Conversely, restoration treatments that do not remove a large quantity of fuels (including some live trees) may result in unnecessary overstory tree mortality and loss of seed sources when wildfires occur.

Tree harvests are used to remove competing species and to reduce stand density (Table 12.1). It is important that a large number of shade-tolerant trees be removed to optimize the growing environment of fire-tolerant, overstory species. Some cutting treatments are followed with prescribed burning to kill residual (especially small in size and resprouting species) shade-tolerant trees and leave the more fire-tolerant individuals. However, prescribed fires are often highly variable across a given site, missing parts of the stand and severely burning other parts (including desirable trees). Although prescribed fires may be a more ecologically beneficial and less expensive treatment, mechanical removal of trees and fuels is more precise. Combining prescribed burning and mechanical removal is an effective method of achieving multiple desired outcomes (Stephens et al. 2012), and appropriate scheduling can enhance the effectiveness of treatments.

12.6.3 Planting Trees

To mitigate loss of fire-adapted tree species due to climate-mediated changes in disturbance regimes, planting key species might be appropriate when seed sources have been lost because of large, severe fires (Stevens-Rumann et al. 2018) (Table 12.1). Hotter, drier conditions will likely result in postfire regeneration failures in some locations in the future (Littlefield 2019). Increasing postfire seed sources by reducing fire severity (through fuel treatments) and increasing the number of live residual trees can increase natural postfire regeneration in some dry forests (Dodson and Root 2013). Regeneration in the driest topographic locations will generally be slower in a warmer climate than in the past (Boag et al. 2020). Some areas are likely to convert from forest to non-forest vegetation, particularly in ecotones and drier ecoregions.

Resource managers may want to supplement natural regeneration after fire in some locations, for example, in locations farther than 200 m from living trees, and where costs are not prohibitive because of remoteness or topography (North et al. 2019). Potential adaptation strategies include varying planting densities by site microclimate, and creating spatial discontinuity in fuels with variable tree densities.

Fire and large-scale tree mortality events provide opportunities to plant diverse species and genotypes (including genotypes adapted to drought) and to modify forest structure. Through postfire management, managers may be able to help transition ecosystems to warmer conditions by promoting species that will be adapted to the likely future conditions in a particular forest type and setting (Halofsky et al. 2020) or alter fuel characteristics that affect fire regimes (Stephens et al. 2012; McIver et al. 2013). Managers may want to consider moving seed zones to elevation- and temperature-appropriate locations (within a current species distribution), and modifying genetic movement guidelines to allow more flexibility (Kilkenny et al. 2013). Care should be taken to identify species hybridization and its potential influence on the assumed fire adaptations and tolerance of planted trees (Tauer et al. 2012). Tools such as the Seedlot Selection Tool (https://seedlotselectiontool.org/sst) can help identify seedling stock that will be adapted to a given site in the present and future.

12.6.4 Working Across Boundaries

Fire and other stressors associated with climate change cross boundaries, so implementing climate-smart practices across different ownerships is needed for effective adaptation (Peterson et al. 2011a,b) (Table 12.1). Agencies can coordinate by aligning budgets, priorities, and work, and by better communicating about projects on adjacent lands (Halofsky and Peterson 2016). Agencies and landowners can also coordinate and share monitoring data, which can detect changes in plant species regeneration, growth, and mortality, and help determine adaptation treatment effectiveness (Joyce et al. 2008). With the uncertainty of climate change and its influence on managed fire regimes, collaborative learning will be critical for adapting management strategies to increase resilience.

12.7 Conclusions

Climate change is expected to increase the frequency of large fires, cumulative area burned, and fire suppression costs and risks in many areas of the USA (McKenzie et al. 2004; Vose et al. 2018). Hence, the remainder of the twenty-first century is likely to present substantial challenges, as natural resource managers are challenged by higher fire risk (Fig. 12.7) and the difficult task of maintaining ecological function and increasing resilience in a rapidly changing biophysical and social landscape. Furthermore, no-analog climatic conditions are likely to strain the applicability of current climate-fire regime concepts (e.g., Flatley and Fulé 2016), creating novel ecosystems and forest conditions (Vose et al. 2018; Falk et al. 2019). Uncertain feedbacks may result from the historical legacy of altered fire regimes (e.g., McKay and Parker 2001; Kreye et al. 2013; Carpenter et al. 2020).

The notion of "managing wildfire" to achieve desired outcomes is an emerging concept (Hiers et al. 2020); however, social acceptance of allowing wildfire and associated risks may limit widespread application, especially in the western USA. As a result, fuel treatments will likely play an increasingly important role for minimizing the undesirable ecological effects of fire, and for enhancing firefighter safety (Reinhardt et al. 2008). However, except in the Southeast where prescribed

Fig. 12.7 Extreme droughts and extended dry periods are likely to increase fire risk in many ecosystems in the USA. The photo is from a 2011 wildfire in the Okefenokee Swamp in southeast Georgia/northern Florida. The fire was ignited by lightning during a period of extreme drought (photo credit: US Fish and Wildlife Service)

burning is institutionalized and relatively inexpensive, financial resources for fuel treatments will likely never be sufficient to reduce fuels across large landscapes with elevated fuel loadings. Therefore, fuel treatments must be implemented strategically across large spatial and temporal scales to optimize spatial patterns that protect valued resources and confer resilience to climate change. Collaboration among agencies, private landowners, and other organizations will be increasingly important to ensure large-scale resilience and long-term sustainable management of forest resources in the USA.

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