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Climate change, fire management, and ecological services in the southwestern US

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ABSTRACT

The diverse forest types of the southwestern US are inseparable from fire. Across climate zones in California, Nevada, Arizona, and New Mexico, fire suppression has left many forest types out of sync with their historic fire regimes. As a result, high fuel loads place them at risk of severe fire, particularly as fire activity increases due to climate change. A legacy of fire exclusion coupled with a warming climate has led to increasingly large and severe wildfires in many southwest forest types. Climate change projections include an extended fire season length due to earlier snowmelt and a general drying trend due to rising temperatures. This suggests the future will be warmer and drier regardless of changes in precipitation. Hotter, drier conditions are likely to increase forest flammability, at least initially. Changes in climate alone have the potential to alter the distribution of vegetation types within the region, and climate-driven shifts in vegetation distribution are likely to be accelerated when coupled with stand-replacing fire. Regardless of the rate of change, the interaction of climate and fire and their effects on Southwest ecosystems will alter the provisioning of ecosystem services, including carbon storage and biodiversity. Interactions between climate, fire, and vegetation growth provide a source of great uncertainty in projecting future fire activity in the region, as post-fire forest recovery is strongly influenced by climate and subsequent fire frequency. Severe fire can be mitigated with fuels management including prescribed fire, thinning, and wildfire management, but new strategies are needed to ensure the effectiveness of treatments across landscapes. We review the current understanding of the relationship between fire and climate in the Southwest, both historical and projected. We then discuss the potential implications of climate change for fire management and examine the potential effects of climate change and fire on ecosystem services. We conclude with an assessment of the role of fire management in an increasingly flammable Southwest.

1. Introduction

Fire is an integral ecosystem process across tens of millions of hectares of forest in the southwestern United States. The region historically had a diversity of fire regimes, ranging from frequent surface fires to infrequent crown fires (Stephens et al., 2007). Over the past 3000 years, there has been a slight decline in western US fires, culminating in a sharp decline during the 20th century (Marlon et al., 2012). This recent policy-driven reduction in fire frequency has resulted in many of the forest types in this region experiencing changes in forest structure and increased fuel accumulation. As a result of fire exclusion and its effects on forest structure and fuel loads, wildfires are burning with increasing severity (Fulé et al., 2003; Miller et al., 2009), and large wildfires are increasingly frequent due to changing climate (Westerling et al., 2006).

The increase in large wildfire frequency has come with substantial economic and ecosystem service costs (Wu et al., 2011). The scope of the current wildfire problem already poses significant management challenges (Donovan and Brown, 2007; North et al., 2012). Globally, increasing temperature is projected to drive future fire regimes (Pechony and Shindell, 2010). Thus, a legacy of forest structural changes and fuel accumulation due to fire exclusion, coupled with increasing climate-driven flammability is likely to exacerbate the current wildfire issue in the southwestern US. Fire management is likely to play a crucial role in maintaining forests

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across the region, but development in the wildland–urban interface and concerns about air quality create impediments to reintroducing fire across forested landscapes. An understanding of the historical role of fire across different forest types must be combined with projections of future fire-forest type interactions to sustain ecosystem function.

Here we provide a broad synthesis of the role of fire in the diverse forest types of the southwestern US and how changing climatic conditions may influence future wildfire occurrence. The effects of changing climate on wildfire pose risks to ecosystem services that are integral for climate regulation and ecosystem function, such as carbon sequestration and biodiversity. We address some potential climate–fire interaction effects on the provision of ecosystem services. We conclude with an assessment of the role of fire management in an increasingly flammable Southwest.

2. Forest diversity in the SW

A diversity of forest types occur across the climatic and elevation gradients of the Southwest. The region has three distinct climatic types which influence vegetation distribution and fire regimes. The arid and semi-arid eastern portions of the Southwest in Arizona and New Mexico experience a bimodal precipitation distribution, with precipitation falling during winter months and the summer monsoon period. The 14.6 million hectares of forest and woodlands in Arizona and New Mexico are dominated by woodlands (*Pinus edulis*, *Juniperus* spp.) and drier forest types: ponderosa pine (*Pinus ponderosa*) alone or mixed with Gambel oak (*Quercus gambelii*) (19%) and mixed conifer (5%). Mixed conifer forests include white fir *(Abies concolor*), Douglas-fir (*Pseudotsuga menziesii*), and southwestern white pine (*Pinus strobiformis*) in addition to ponderosa pine. Mesic forests of aspen (*Populus tremuloides*), Engelmann spruce (*Picea engelmannii*), and subalpine fir (*Abies lasiocarpa*) are restricted to the highest elevations and comprise only 3% of the forested area. In Nevada, the intermountain west climate is characterized by cold winters, warm summers, and the majority of precipitation falling in winter and spring. Forests and woodlands occupy approximately 4 million hectares in Nevada. Forest types are a function of the basin and range geology of the state, with conifer forests and madrean oak and pine forests occupying the higher elevation ranges and the basins occupied by piñon-juniper woodlands, shrublands, and grasslands. In contrast, California has a Mediterranean climate with cool, wet winters and warm, dry summers. Characterized by a diversity of vegetation types, approximately one-third of the California's 40 million ha are forested. Forest types range from oak savannas at lower elevations to subalpine forest at high elevation in the Sierra Nevada Mountains.

3. Climate–fire interactions

In these primarily fuel limited systems, prior year precipitation is the primary determinant of the area burned by wildfire (Littell et al., 2009). Fire seasonality is largely attributed to annual precipitation patterns, with inter-annual fire activity influenced by the El Niño/Southern Oscillation (ENSO) (Swetnam and Betancourt, 1990). Historic fire seasonality, estimated from the relative location of fire injuries within the growing period of tree rings, tended to peak in spring and early-summer where the southwestern monsoon precipitation pattern is prevalent and in mid- to late-summer where the prevailing climate is Mediterranean (Baisan and Swetnam, 1990). The modern fire season also peaks during this time. Aside from seasonality, annually resolved fire records from over 400 years of tree-ring data also showed that climate synchronized fire occurrence across the Southwest, with up to half of study sites in Arizona and New Mexico burning in major fire years such as 1748 or 1851 (Swetnam and Brown, 2011). These fire events were significantly correlated with drought and the La Niña phase of the Southern Oscillation (Swetnam and Brown, 2011). Fires are also associated with La Niña conditions in the southern Sierra Nevada (Kitzberger et al., 2007; Trouet et al., 2010) and widespread and synchronous fire across vegetation types in California (Taylor et al., 2008; Westerling and Swetnam, 2003). While La Niña results in dry periods throughout most of the Southwest, this pattern is reversed in northern California, creating a weak association between El Niño and fire in the northern Sierra Nevada and southern Cascade Mountains of northern California (Taylor et al., 2008; Valliant and Stephens, 2009; Trouet et al., 2010).

Broad geographic patterns in fire activity determined by climate are refined across vegetation types. The high frequency of past fire events in the dry forest types of the Southwest, even during comparatively cool and moist periods (Beaty and Taylor, 2008), indicates that these systems occur in areas where climate and fuel conditions predispose them to being fire adapted. Forests dominated by ponderosa pine experience one of the most frequent fire return intervals.

Historic fire regimes of ponderosa pine forests of the Southwest are among the best studied in the world, due to the extensive record of fire-scarred trees and the pioneering application of dendrochronology to determine fire dates. Abundant charcoal has been associated with the arrival of ponderosa pine at the formation of modern southwestern vegetation in the Holocene, beginning around 13,000–11,000 years before present (Anderson et al., 2008; Weng and Jackson, 1999). Fires recurred at mean intervals between 2 and 16 years across the southwestern network of study sites examined by Swetnam and Baisan (1996). Using only "widespread" fires, those dates scarring 25% or more of the sampled trees, the mean intervals ranged from 4 to 36 years (Swetnam and Baisan, 1996). Frequent surface fires maintained relatively open forests of large trees with grassy understories by limiting regeneration establishment (Cooper, 1960; Covington and Moore, 1994), keeping the fuel complex resistant to severe burning. Traits of ponderosa pine, including thick bark, large buds, and the capability to recover from crown scorch, are highly adaptive for the frequent surface fire regime, reflecting the evolutionary history of this species in fire-prone, semi-arid environments (Keeley and Zedler, 1998).

Mixed conifer sites burned less frequently than ponderosa pine, but fires still recurred in mean intervals ranging from 3 to 25 or 10 to 26 years for widespread fires (Swetnam and Baisan, 1996). This decreasing fire frequency is in part driven by an increase in precipitation at higher elevations, influencing both fire type and frequency. Fire regimes in mixed conifer forest in Arizona and New Mexico historically reflected a transition from frequent surface fires at lower elevation to infrequent, stand-replacing fires at higher elevation. Fire regimes have been reconstructed over the entire elevational gradient of forest types in Grand Canyon National Park, Arizona (Fulé et al., 2003), and the Sangre de Cristo Mountains, New Mexico (Margolis and Balmat, 2009). At both sites the length of fire return intervals increased with increasing elevation, and fire events were synchronized across ponderosa pine and mixed conifer in years with drier climates. The infrequent fires in the spruce-fir forests found at higher elevations burned during years of exceptional drought at multi-century intervals (Margolis and Balmat, 2009; Margolis et al., 2011). Historic mixed conifer forest structure was more dense than that of contemporaneous ponderosa pine forests (Lang and Steward, 1910), but was significantly less susceptible to severe fire than modern mixed conifer forests (Fulé et al., 2003).

As in Arizona and New Mexico, historic fire frequency decreased with increasing elevation in California. Mixed conifer forests occupy a fire frequency transition zone, bounded by frequent-fire ponderosa pine-oak forest at lower elevations and infrequent-fire red fir (*Abies magnifica*) at higher elevations (Taylor, 2000; van Wagtendonk and Fites-Kaufman, 2006). In the mixed-conifer forest, low-moderate intensity surface fires were historically common; burning on average every 11–30 years (McKelvey and Busse, 1996; North et al., 2005; Taylor and Skinner, 2003; Van de Water and North, 2010). Regular fire resulted in the dominance of large, fire-tolerant pine species, including ponderosa pine, sugar pine (*Pinus lambertiana*), and Jeffrey pine (*Pinus. jeffreyi*) (North et al., 2005; Beaty and Taylor, 2007; Scholl and Taylor, 2010; Taylor et al., in press). Effects from regular fires also resulted in high structural heterogeneity in these forests at a range of spatial scales (Beaty and Taylor, 2008; North et al., 2007), further reinforcing a diversity of fire effects (Collins et al., 2011).

4. Fire suppression

By the end of the nineteenth century, fire regime disruption was nearly ubiquitous across the spectrum of southwestern forests, from ponderosa pine to mixed conifer forests (Swetnam and Brown, 2011). Surface fire regimes were first interrupted in portions of the Southwest nearly 200 years ago, when fuels were reduced by sheep when pastoralism was adopted by Navajo people in the Chuska Mountains (Savage and Swetnam, 1990). Initial fire exclusion from heavy grazing by livestock was accelerated with the completion of transcontinental railroads in the 1880s, which was accompanied by widespread logging across much of the region (Belsky and Blumenthal, 1997; Bahre, 1991). The railroad increased access to the west and precipitated land use changes that include more than a century of continued fire suppression, which reduced the extent and frequency of fire in most forest and range land ecosystems (Stephens et al., 2007). Although ecological problems associated with fire regime disruption were noted relatively early in the twentieth century by Leopold (1924) and Weaver (1951), organized fire suppression extended fire exclusion up to the present in many areas.

Southwestern forests have changed since the onset of fire exclusion in ways that make them more susceptible to severe burning as well as other disturbances. A legacy of past management actions, resulting in the absence of fire for a century or more, have altered the structure of frequent-fire forests. In the ponderosa pine forests of Arizona and New Mexico increased density, biomass, and canopy continuity have occurred at the expense of understory vegetation (Covington and Moore, 1994), despite substantial biomass removal through harvesting (Fulé et al., 1997). Mixed conifer forests in Arizona and New Mexico also increased in density, especially shade-tolerant species with long crowns that create fuel ladders, supporting the transition of fire from the surface to the canopy (Cocke et al., 2005; Fulé et al., 2003; Stephens, 1998; Taylor et al., in press). Mixed conifer forests in California have shifted from fire-tolerant pines to fire-intolerant species (e.g. white fir, incense-cedar (*Calocedrus decurrens*) comprising a majority of the basal area (Beaty and Taylor, 2007; North et al., 2005; Scholl and Taylor, 2010; Taylor and Skinner, 2003). The effects of fire suppression on forest structure have been further compounded by logging at the end of the 19th and beginning of the 20th century. Selective cutting of ponderosa pine trees, the most highly valued timber species, from the mixed conifer forests of Arizona and New Mexico has enhanced the shift toward mesic, shade-tolerant conifers (Cocke et al., 2005; Fulé et al., 2009). In the Lake Tahoe Basin on the Nevada–California border, selective cutting has increased lodgepole pine (*Pinus contorta)* at the expense of the historically dominant red fir and western white pine (*Pinus monticola*) (Taylor, 2004). While the trends of increasing fire severity and area burned are common in many forest types, there is some regional variation. For example, high-severity fire has not increased in the Klamath Mountains of northwestern California, despite a trend of increased fire size and annual area burned (Miller et al., 2012). Miller et al. (2012) attribute the dissected topography and associated temperature inversions in the Klamath region for moderating fire severity when compared to other parts of northern California.

5. Projected climate–fire-vegetation interactions

The temperature increases of the past several decades are predicted to accelerate over the first half of the 21st century (Karl et al., 2009), and be reasonably consistent across the Southwest. Compared to 1971–2000, annual temperatures are anticipated to rise 1.4–1.7 °C by 2021–2050, 1.9–2.5 °C by 2041–2070, and 2.5–4.4 °C by 2071–2099 (Kunkel and Redmond, 2012). These increases are expected to be slightly more pronounced in the summer and fall than during winter and spring. Increasing temperatures will result in more frequent and prolonged heat waves, defined as approximately 20–40 more days per year in which maximum temperature exceeds 35 °C. Warmer temperatures will also lengthen the frost-free season by 25–40 days. Rising temperatures will increase potential evapotranspiration and decrease snowpack volume (Seager et al., 2007), especially at the lowest elevations currently accumulating snow (Das et al., 2009). Increased winter and spring warming in mountainous regions is likely to have substantial impacts on the timing and duration of snowmelt (Cayan et al., 2008). Changes in snowpack volume and the timing of melting is a significant concern because the region is in part dependent on winter snowpack for early summer water availability.

Compared to temperature, predictions of precipitation are both less certain and more variable across the Southwest. Precipitation is anticipated to decrease across most of the southwest, especially in southern Arizona, New Mexico and southeastern California, although the far northern part of the region may see modest precipitation increases. Region-wide, precipitation is expected to decrease 1–1.5% by 2021–2050, 3–4% by 2041–2070 and 2–5.5% by 2071–2099. The most dramatic decreases in precipitation are expected to occur in the spring and summer with only modest decreases in the fall and potentially small increases in winter precipitation. Regardless of changes in precipitation, the effects of increasing temperature are projected to cause regional drying (Seager et al., 2007).

Regional warming and the associated drying will likely alter future fire–climate relationships. This will either occur directly through drying of fuels or through climate-vegetation-fire interactions. Fire activity in montane evergreen conifer forests (cf. Lenihan et al., 2008a) is particularly sensitive to temperature variability because of its effect on snowpack duration and fire season length (Dettinger and Cayan, 1995; Westerling et al., 2006). Recent warming has influenced fire extent by lengthening the period fuels are dry enough to ignite and burn each year. Warming and early spring snowmelt have led to an exponential increase in lightning caused ignitions and an increase in both area burned and areal extent of high severity burns in recent decades (Lutz et al., 2009). Climate, land-use, and fire can also interact to influence the frequency of burning across the region. The spread of invasive grasses has increased fire frequency in semi-arid shrublands, and this trend is expected to continue because regional vegetation shifts toward increased annual grass lands are anticipated. This process is likely to be exacerbated by warming in coming decades (Stevens and Falk, 2009; Abatzoglou and Kolden, 2011).

A complex set of interactions between fire and fire suppression, climate, and the influence of both on forest growth will likely impact future forest distribution in the Southwest. Recent research attributes the majority of forest drought-stress in the region to growing season vapor-pressure deficit and winter precipitation (Williams et al., 2012). Changes in precipitation and temperature have already caused substantial mortality of piñon pine (*P. edulis*) in southwestern piñon-juniper woodlands (Breshears et al., 2005; Adams et al., 2009). Furthermore, competition for available water is greater in more densely forested fire-suppressed stands, making trees in dense stands more sensitive to drought conditions (Kerhoulas et al., in review). Given the projected increase in fire season length for the region (McKenzie et al., 2004) and the influence of fuel availability on area burned (Littell et al., 2009), climate-driven tree mortality may influence future fire behavior by literally adding fuel to the fire. Over multi-decadal time scales, however, reduced forest growth and higher mortality due to warming climate may result in less fuel on the landscape in the future in the increasingly arid portions of the Southwest (Diggens et al., 2010).

6. Future climate and fire management

Increasing large fire frequency in the Southwest correlates with the onset of earlier spring snow melt and the resultant increase in fire season length (Westerling et al., 2006); leading to an increased probability occurrence of conditions where fire escapes initial attack. Atmospheric greenhouse gas concentration is projected to increase the frequency of heat extremes and decrease precipitation across the Southwest (Diffenbaugh and Ashfaq, 2010), conditions that are conducive to wildfire events (Littell et al., 2009). Changes in climate under a business-as-usual emissions scenario are projected to increase the number of days with high fire danger by 2–3 weeks across the region (Brown et al., 2004). In California, regardless of the emission scenario and resultant projected climate, the frequency of large wildfires is projected to increase in forested systems (Westerling and Bryant, 2008). As more detailed information is available for potential fire activity with future climate change in California, we use it as an example for the Southwest. Although California provides a useful model, predictions for future climate–fire relationship across the Southwest represent an important knowledge gap.

Since climate influences the distribution of vegetation types, simulations that included changes in climate and vegetation project an increase in total area burned in California (Lenihan et al., 2008a). Projected area burned in virtually all years was greater than mean area burned during the historic reference period (1895–2003). However, annual area burned prior to the reference period used by Lenihan et al. (2008a,b) may have been higher in some locations (Stephens et al., 2007). By 2100, area burned increased by 9–15% and much of this increase was driven by the expansion of flammable grasslands. As the area burned increased, biomass consumption varied under different climate scenarios. The amount of biomass burned in the cooler and less dry PCM-A2 scenario was about 18% greater by 2100 compared to the reference period. In contrast, biomass burned was initially higher and then fell below historic values after several decades under warmer and dryer scenarios because of lower fuel production. A significant increase in area burned and large wildfire occurrence is also predicted by Westerling et al. (2011) using a wider range of climate scenarios and static vegetation patterns. By the mid 21st century most scenarios indicate a significant increase in large wildfire occurrence and burned area, particularly in forested areas at mid-elevations. These locations include federal land and private lands with low density development that will expose property owners to higher fire risk.

Fire interacts with other climate influenced disturbance factors in several ways. Extended drought lengthens fire seasons (Westerling et al., 2006), resulting in longer periods each year with high probability of burning. Severe, extended droughts in the first decade of the 21st century have been described as “global-change-type” drought (Breshears et al., 2005) and they resemble climate projections for the Southwest in the mid- to late 21st century (Seager et al., 2007). These droughts have been associated with landscape and even regional-scale die-off of woodland vegetation, especially piñon pine (Allen et al., 2010). Projections of future fire regimes in the Southwest, based on linear extensions of the current relationship between climate and fire, predict future temperatures will drive a multi-fold increase in area burned (McKenzie et al., 2004). Interactions among disturbance factors could alter future scenarios. For example, drought is likely to increase the probability of bark beetle outbreaks in many pine species (Waring et al., 2009) and decrease the productivity of vegetation/fuel (Diggens et al., 2010).

Tree mortality caused by mountain pine beetle tends to increase during drought, as stressed trees have reduced defenses (Parker et al., 2006). In the mixed conifer forests of California, higher tree density due to fire suppression intensifies drought effects and increases mortality from both water stress and beetle outbreaks (Guarin and Taylor, 2005). Recent mortality in southwestern ponderosa pine forests can increase the short-term risk of fire as canopy foliage dries and surface fuels accumulate under beetle-killed trees (Hoffman et al., 2012). While beetle outbreaks elevate the likelihood of near-term increases in large wildfire frequency, there is uncertainty in long-term projections of wildfire frequency because of the uncertainty associated with climatic influences on vegetation and the potential for fuel becoming a limiting factor (Danato et al., 2013). At longer time scales, however, beetle-caused mortality may reduce fine fuel loads and canopy fuel continuity by thinning stands, thereby potentially decreasing the size and severity of future fires (Hoffman et al., 2012). Uncertainly surrounding the impacts of bark beetles underscores the difficulty in predicting the impact of bark beetle attack on long term wildfire scenarios (Simard et al., 2011; Moran and Cochrane, 2012).

7. Climate, fire management, and carbon

Forest carbon sequestration has potential for mitigating rising atmospheric carbon dioxide concentration. While forests sequester carbon from the atmosphere, it is not stored permanently. Once a tree dies the woody biomass decomposes and the carbon is transferred to both the soil and atmosphere, and disturbances such as fire can alter the rate and apportioning of carbon between pools. The consumption of surface fuels and soil organic matter account for most of the emissions associated with fire, regardless of severity (Campbell et al., 2007; Meigs et al., 2009). However, in fire-adapted systems, these emissions are rapidly resequestered by forest growth. In frequent fire systems, such as mixed-conifer forests, the carbon emitted from a low-severity surface fire can be resequestered in fewer than 7 years (Hurteau and North, 2010). In the case of stand-replacing fires, such as those that occur in interior lodgepole pine, forest recovery will resequester the carbon emitted during the fire over the course of one fire rotation, which is usually longer than a century (Kashian et al., 2006).

Mortality and regeneration are key regulating components of forest carbon dynamics. Under historic conditions with a functioning fire regime, little net change in forest carbon stocks is expected over one fire rotation because post-fire regrowth sequesters the carbon emitted through combustion and decomposition. However, climate can also negatively affect system recovery following disturbances such as fire, and thus limit long-term carbon stocks. Regeneration following severe fire in ponderosa pine forests can be slow, patchy, and dependent on distance to seed sources (Haire and McGarigal, 2010). Severe fires eliminated overstory ponderosa pine at half of the sites examined by Roccafort et al. (2012) across Arizona, and pine regeneration was absent from 57% of sites. Sites lacking ponderosa pine regeneration may recover as aspen or oak forests depending on elevation, but Roccafort et al. (2012) suggest a few sites are likely to convert to grasslands. Likewise, Savage and Mast (2005) found that at half of the high-severity burn areas in Arizona and New Mexico they surveyed, regeneration was not sufficient to support forest recovery. They suggest that prolonged drought may be a causal factor in this lack of recovery. Severe fire can also alter competitive interactions, allowing shrubs to exclude forest regeneration and establish a self-reinforcing feedback with fire (Collins and Stephens, 2010; Nagel and Taylor, 2005). Increased mortality, reduced regeneration, and forest conversion have implications for forest carbon dynamics in the context of fire and changes in forest structure and fuel loads resulting from fire-exclusion (Hurteau and Brooks, 2011). These two factors are also areas of uncertainty that need further investigation to improve efforts to model forest carbon dynamics following fire events.

In the Southwest, forest carbon stocks have been altered by fire exclusion. Stocks have increased in some areas because of ingrowth (Collins et al., 2011; Hurteau et al., 2011) and decreased in others because of loss of large trees which contain a disproportionate amount of the carbon (Fellows and Goulden, 2008; Lutz et al., 2012; North et al., 2009a). Regardless of the effects of fire-exclusion on current carbon stocks relative to the carbon stock size with frequent fire, forest structural changes and increased fuel loads necessitate the removal of carbon from systems to restore fire as a process and reduce the risk of severe wildfire. Reductions in the carbon stock and emissions associated with treatment can be counterbalanced by avoided wildfire emissions in some systems, but in others thinning and burning treatments release more carbon than wildfire burning in untreated forest (Hudiburg et al., 2011; Hurteau and North, 2010; Mitchell et al., 2009; Stephens et al., 2009). As indicated by Hudiburg et al. (2011), the net carbon balance between treatment emissions and wildfire changes as a function of the productivity of the system, with the carbon balance in favor of treatment in less productive systems. The authors also suggest that with changing climate, decreases in productivity may alter the carbon balance in forest types that are currently better left untreated from a carbon perspective. Given the limited geographic scope of work in this area, more research is needed to evaluate these trade-offs in forest types across the region.

Direct emissions resulting from high-severity wildfire are greater than emissions from low- and mixed-severity fire on a per unit area basis (Meigs et al., 2009; Wiedinmyer and Hurteau, 2010). Further, the emissions from severe wildfire may only have a small impact on post-fire carbon dynamics relative to post-fire reductions in net primary productivity (NPP) due to tree mortality. Tree mortality increases with increasing fire severity, thus, fire-induced mortality influences both the size of the live tree carbon stock and NPP. Treatments designed to reduce high-severity fire risk reduce mortality and the fraction of live tree carbon that is at risk of being killed by fire (Stephens et al., 2009). In a study examining the effects of 19 different wildfires that burned through treated and untreated forest, North and Hurteau (2011) report that in untreated forest, 70% of the post-fire carbon stocks were dead material; whereas the treated forest only had 19% in dead material. Between carbon loss from decomposition and foregone NPP, fire-induced mortality can have substantial effects on forest carbon dynamics. Several years following low- and moderate-severity fire, Meigs et al. (2009) report a small carbon sink in ponderosa pine and mixed-conifer forest in Oregon. Whereas, high-severity wildfire can transition a forest from sink to source for periods ranging from years to decades (Campbell et al., 2004; Dore et al., 2008; Meigs et al., 2009).

Fire effects on soil carbon pools are multifaceted. Bormann et al. (2008) report that post-fire erosion on the Biscuit Fire resulted in approximately 60% of the mineral soil carbon being lost and Ross et al. (2012) report substantially less soil carbon in burned sites even decades after the burn. While there is some indication that fire effects on soil carbon are limited, fire severity and soil drainage can influence soil carbon loss following wildfire (Harden et al., 2000; Kashian et al., 2006; Wirth et al., 2002). Conversely, fire can add to the soil carbon pool by creation of black carbon. Black carbon includes charcoal and other products of incomplete biomass combustion (Preston, 2009). DeLuca and Aplet (2008) estimate that between one and ten percent of biomass consumed by fire is converted to black carbon. Because of its recalcitrant nature, black carbon can remain in the soil for a centuries.

8. Climate, fire and biodiversity

Climate and disturbance are key factors that determine the distribution of plant species and communities. In the Southwest, both are changing at unprecedented rates (MacDonald, 2010; Seager and Vecchi, 2010; Westerling et al., 2006). The combination of changing climatic conditions, altered fire regimes and fire management practices has the potential to impact species composition and biodiversity patterns by influencing vegetation distribution and abundance. Comparison with historical plant surveys indicates that plant species distributions are shifting with changing climate (Kelly and Goulden, 2008) and further shifts are predicted in response to climate change and disturbance, but details remain unclear (Lawler et al., 2009). Uncertainties about ecosystem responses are especially high in arid and semi-arid regions because of tight linkages between vegetation dynamics and hydrologic cycling (Jackson et al., 2009a).

Nevertheless, there is strong evidence that the distribution of vegetation types is responsive to climate (e.g. Harrison and Prentice, 2003) and disturbance regime (Bond et al., 2005; Staver et al., 2011). Predicted changes in distributions inferred from both climate envelop models and dynamic vegetation models point to northward and upslope movement of all vegetation types, spread of species and ecosystems adapted to arid conditions, and potential contraction of forest ecosystem distributions (Lenihan et al., 2008b; Schlaepfer et al., 2012). Increases in western conifer mortality rates in response to recent climate change highlight the risks to tree populations (Allen et al., 2010; van Mantgem and Stephenson, 2007; van Mantgem et al., 2009), and evidence is mounting that increased mortality is linked to increasing temperature and drought conditions (Adams et al., 2009; Anderegg et al., 2012; McDowell et al., 2011). Forest persistence will rely both on adult tree survival as well as successful tree regeneration and establishment (Grubb, 1977; Jackson et al., 2009b). Increasing fire frequency and severity would likely help facilitate and/or accelerate these transitions (Bond et al., 2005; Lenihan et al., 2008b). In addition, land use practices and biological invasions (both intentional and unintentional) are impacting ecosystems over very large areas (Stevens and Falk, 2009; Bradley, 2010; Balch et al., 2013), potentially providing a catalyst for widespread and rapid changes in plant distributions.

Climate change projections and the associated effects on vegetation and fire for the southwest are exemplified by simulations conducted for California using future climate scenarios from the IPCC Fourth Assessment Report (Cayan et al., 2008). The impact of future climate on vegetation and fire was assessed by driving a dynamic vegetation model (MC1) with climate model output for different emission scenarios to compare the extent and location of seven life form vegetation types simulated for the period 1961–1990 and 2070–2999 (Lenihan et al., 2008a) (Fig. 1). The climate data were monthly time series from two general circulation models (GCMs), GFDL and PCM1, that project a 1.8–3 °C temperature increase with a doubling of CO_2. Vegetation and fire simulations were reported for high (A2) and low (B1) emission scenarios. Projections ranged from moderately dry to no change in precipitation with intermediate temperature increases (<3 °C) to larger temperature increases (>4 °C) and substantial drying for California compared to IPCC AR4 model simulations (Cayan et al., 2006).

Simulations project significant changes in vegetation types in California under the different climate change scenarios. The extent of alpine/subalpine forest and evergreen conifer forest declined under all scenarios. Mixed evergreen forest, in contrast, increased in extent converting evergreen conifer forest to mixed evergreen forest. Evergreen conifer forest was also replaced by shrubland, woodland, and grassland with large reductions in moisture or increases in fire. The expansion of mixed evergreen forest was most evident under the PCM-A2 which simulates higher levels of effective moisture which promotes forest expansion. PCM-A2, overall, projects a 23% increase in forest cover, while the warmer drier scenarios project a 3–25% decline. Mixed evergreen woodland and shrublands are also projected to decline under all scenarios, and the vegetation types were replaced mainly by grassland under the driest conditions (Fig. 1). Under the cooler moister PCM-A2 scenario, the woodland and shrubland decline was due to both grassland expansion and forest encroachment. Grasslands expanded under all scenarios due to reductions in effective moisture and fire. Grasslands expanded into desert under the cooler-moist scenario but desert expanded into grassland under the drier scenarios (Lenihan et al., 2008a). Area burned in California is projected to increase under all scenarios with the greatest increase under the driest scenario (Lenihan et al., 2008b). An identical trend of increased area burned under climate change in California was identified by Westerling et al. (2011) who used static vegetation cover for wildfire projections.

Shifts in ecosystem boundaries in response to climate change and fire regime may dramatically change biodiversity and ecosystem function (Chapin et al., 2008; Morin et al., 2008). For example, loss of forest at lower tree line and replacement with shrublands has clear impacts on plant species composition that in turn impact habitat quality and quantity as well as carbon and water cycles (Barger et al., 2011; Huxman et al., 2005; Jackson et al., 2005; Staus et al., 2002). Although the effects of changing temperature and precipitation on ecosystem structure and function and subsequently on patterns of biodiversity have been explored (e.g. Cramer et al., 2001; Gordon and Famiglietti, 2004; Loarie et al., 2008; Porporato et al., 2004; Stephenson, 1990), relatively little is known about the regional-scale biodiversity impacts of interactions between changing climate and altered fire regimes (D'Odorico et al., 2010; Turnbull et al., 2011).

In the fire-prone forests of the Southwest, the increase in forest density and build-up of surface fuels resulting from fire suppression have substantially impacted forest understory plant diversity. Understory diversity can be restored though forest structural manipulations and the reintroduction of fire (McGlone et al., 2009; Wayman and North, 2007; Webster and Halpern, 2010). The effects of fire reintroduction on understory plant diversity and cover can be rapid and dramatic if there is a local source of native propagules (Moore et al., 2006). Plant community recovery

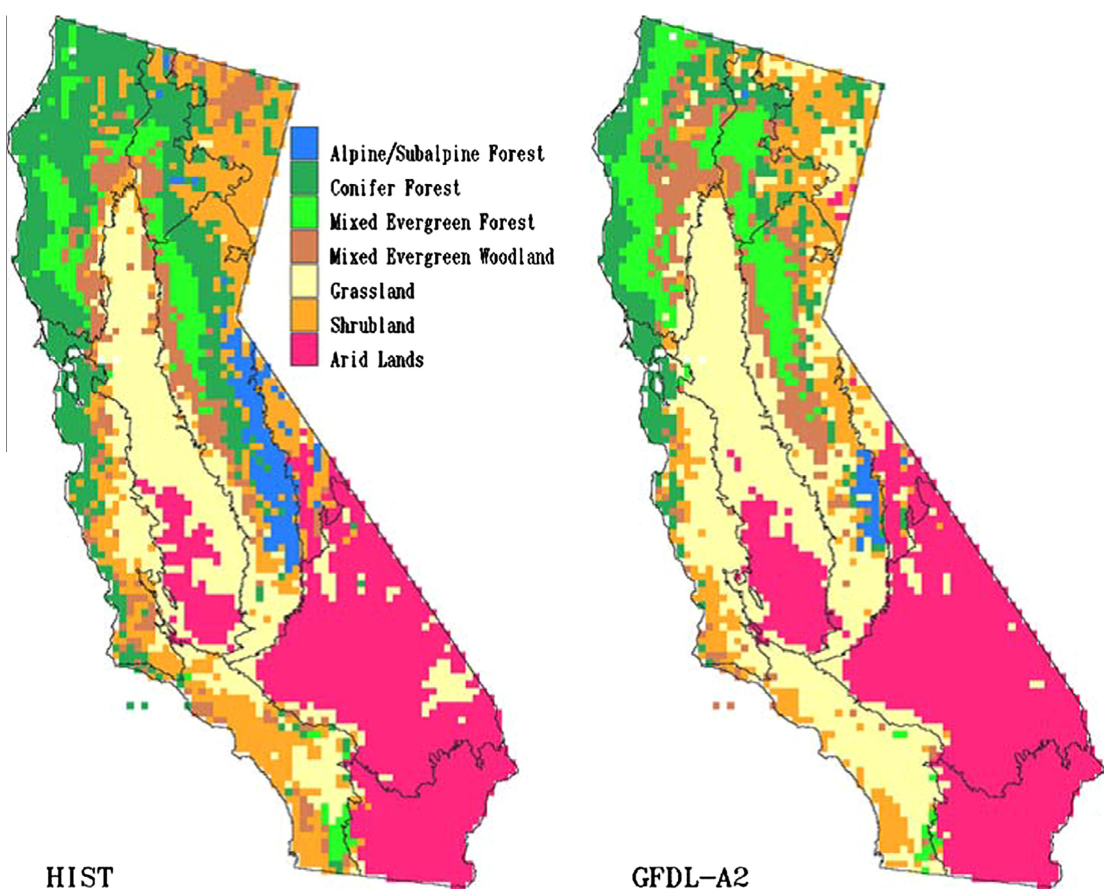

Fig. 1. The distribution of vegetation classes simulated for the historical and end-of-century (2070–2099) periods using PCM1 projections under an A2 emission scenario. Figure from Lenihan et al. (2008b).

was delayed at an Arizona treatment site where invasive non-native seed sources predominated, but native species were superior competitors by 6 years post-treatment (Stoddard et al., 2011). Webster and Halpern (2010) report a threefold increase in species richness 20 years following a first-entry prescribed fire in the mixed-conifer forest of the southern Sierra Nevada. In a study of the effects of overstory thinning and prescribed burning, Wayman and North (2007) found that the combination of these treatments resulted in the largest increase in herb richness and abundance. They attributed this to the increase in light and soil moisture associated with thinning and the reduction in surface fuels with prescribed burning. Treatments do not always lead to native plant community recovery, however. A drought-prone Arizona site showed persistent colonization by cheatgrass (*Bromus tectorum*) after tree thinning and prescribed burning treatments (McGlone et al., 2009).

While we cannot ensure the persistence of a given set of species at a given location with changing climate (Stephens et al., 2010), the use of portfolio theory has been proposed as a means of dealing with the uncertainty of future climate when selecting tree species genotypes for reforestation and restoration (Crowe and Parker, 2008). The concept behind portfolio theory is to minimize the covariance of risk within a portfolio of investments. Like a diversified investment portfolio, the increased understory plant diversity promoted by the reintroduction of burning in fire-adapted forests provides a larger suite of species within a given geographic location. Thus, such management improves the chances that the understory community will contain individuals that will be capable of responding to or tolerating climate-driven changes.

9. Management implications

Substantial increases in the area burned in the western US are projected with changing climate (Littell et al., 2009; National Research Council, 2011). This projected trend, coupled with the current trend of increasing large fire frequency (Westerling et al., 2006), suggests that the relatively fire-free period of the 20th Century is unlikely to continue (Marlon et al., 2012). Thus, it is imperative that natural system management include fire as an ecological process. There have been significant research efforts to determine ways to restore fire as an ecologically beneficial process. In areas where tree density and fuels have increased substantially, research indicates that forest thinning coupled with prescribed burning is the most effective means of reducing high-severity fire risk (Fulé et al., 2012; Safford et al., 2009; Stephens et al., 2012). Structural manipulations alone are much less effective in reducing the risk of severe fire. However, regular prescribed burning can be beneficial for reducing fuels and altering forest structure. Research in mixed-conifer forests in Sequoia National Park suggests that multiple prescribed fire treatments can reduce forest density, with preferential survivorship of large trees, and retention of heterogeneous overstory spatial patterns (van Mantgem et al., 2011). Reintroduction of fire also provides numerous ecological benefits including increased understory species diversity, nutrient cycling, and improved regeneration of shade-intolerant species (North et al., 2009b; Zald et al., 2008). Furthermore, reducing the risk of high severity wildfire through restoration of forest structure and regular prescribed fire provides a level of stability to forest carbon stocks (Hurteau and Brooks, 2011). Restoration of fire regimes is also expected to build system level resistance and resilience to climate change (Fulé, 2008; Millar et al., 2007; Stephens et al., 2010). Reducing forest density adds a level of drought tolerance, allowing for sustained tree growth and carbon sequestration during periods of reduced precipitation (Dore et al., 2012; Kerhoulas et al., in review). However, given the uncertainty associated with tree species-specific responses to changing climate (Hurteau et al., in review), thinning efforts in mixed-species forest types should consider a strategy that restores evenness where appropriate. Significant challenges remain for fire restoration and North et al. (2012) highlight the disparity between the spatial extent of forests requiring fuels treatments and the scale of treatment implementation. They suggest an approach to move landscape "firesheds" from restoration to maintenance that includes wildland fire use, where naturally ignited fires are allowed to burn in treated areas.

In historically frequent-fire forests, stand replacing fire can result in conversion to alternative vegetation types through regeneration failure (Savage and Mast, 2005) or by creating conditions that prevent tree re-establishment because of increased fire frequency or competitive exclusion (McGinnis et al., 2010; Nagel and Taylor, 2005). While re-vegetation issues can be overcome with human intervention (e.g. planting, herbicides), the rapid pace at which the climate is changing (Loarie et al., 2009) suggests that restoring fire as a process to increase resistance to high-severity wildfire could provide a viable option for slowing vegetation change, and associated impacts to carbon cycling and biodiversity, over larger areas.

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