Forest Ecology and Management 396 (2017) 217–233

Contents lists available at ScienceDirect

Forest Ecology and Management

journal homepage: www.elsevier.com/locate/foreco

Tamm Review: Shifting global fire regimes: Lessons from reburns and research needs

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ARTICLE INFO

Article history:
Received 17 January 2017
Accepted 31 March 2017

Keywords:
Climate change
Fire-dependent ecosystems
Fire regimes
Grasslands
Semi-arid forests
Savannas
Wildland fire

ABSTRACT

Across the globe, rising temperatures and altered precipitation patterns have caused persistent regional droughts, lengthened fire seasons, and increased the number of weather-driven extreme fire events. Because wildfires currently impact an increasing proportion of the total area burned, land managers need to better understand reburns – in which previously burned areas can modify the patterns and severity of subsequent fires. For example, knowing how long past fire boundaries can function as barriers to fire spread may empower decision-makers to manage some wildfires as large-scale fuel treatments, or alternatively, determine where prescribed burning or strategic wildfire management are required. Additionally, a clear understanding of how prior burn mosaics influence future fire spread and burn severity is critical knowledge for landscape and fire-dependent wildlife habitat planning under a rapidly changing climate. Here, we review published studies on reburns in fire-adapted ecosystems of the world, including temperate forests of North America, semi-arid forests and rangelands, tropical and subtropical forests, grasslands and savannas, and Mediterranean ecosystems. To date, research on reburns is unevenly distributed across the world with a relative abundance of literature in Australia, Europe and North America and a scarcity of studies in Africa, Asia and South America. This review highlights the complex role of repeated fires in modifying vegetation and fuels, and patterns of subsequent wildfires. In fire-prone ecosystems, the return of fire is inevitable, and legacies of past fires, or their absence, often dictate the characteristics of subsequent fires.

Contents

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http://dx.doi.org/10.1016/j.foreco.2017.03.035
0378-1127/

1. Introduction

Over recent decades, many parts of the world have witnessed a dramatic rise in the incidence and area burned by wildfires (Westerling et al., 2006; Flannigan et al., 2009). Globally, rising temperatures are extending droughts and increasing the number of weather-driven fire events and lengthening fire seasons (Jolly et al., 2015). Large increases in the annual area burned have been documented for the western United States (Littell et al., 2009; Miller et al., 2012; Dennison et al., 2014), bush regions and eucalypt forests of Australia (Williams et al., 2009), boreal forests (Kasischke and Turetsky, 2006), subtropical pine forests (Mitchell et al., 2014), and tropical rainforests and savannas (Brando et al., 2014). Because wildfires generally impact a much greater proportion of the area managed than prescribed burns and other fuel reduction treatments, land managers need to better understand reburns – prior wildfires, which can modify the patterns and severity of subsequent fires. For example, knowing how long past fire boundaries can function as barriers to future fire spread can enable decision-makers to manage more wildfires as large-scale fuel treatments, or alternatively, determine when and where fire suppression or prescribed burning may be strategically needed. An understanding of how past burn mosaics can regulate future fire spread and burn severity is also critical for landscape and fire-dependent habitat planning, especially in the context of a changing climate. Here, we review studies of reburns in fire-adapted ecosystems that share a legacy of 20th-century fire exclusion (see Box 1).

Box 1 High severity fires can accelerate vegetation lifeform changes.

Given 21st-century climate change scenarios, widespread non-native species invasions, and past fire exclusion across most global wildfire ecosystems, this is a challenging time for fire managers. Many historically fire-prone ecosystems have experienced a chronic fire deficit and now exhibit highly altered fire regimes. In previously fuel-limited systems with frequent fire returns, fires now burn more intensely and contagiously over broader areas. For example, in ponderosa pine (*Pinus ponderosa*) dominated forests of the western United States (US), frequent surface fires historically maintained light and patchy surface fuels under low- or mixed-severity fire regimes (Agee and Skinner, 2005; Hessburg and Agee, 2003). After nearly a century of fire exclusion, surface and canopy fuels have accumulated, as has horizontal and vertical continuity of fuels at patch to regional landscape scales – predisposing some dry pine and mixed-conifer forests to more intense fire behavior and severe fire effects (Hessburg and Agee, 2003; Hessburg et al., 2005; Malleck et al., 2013; Stephens et al., 2012). Similar trajectories of increased forest cover, density, layering, fuel accumulation, and susceptibility to large, high-intensity fires have been documented in African dryland savannas (Bond and Archibald, 2003; Staver et al., 2011), South American cerrado (Geiger et al., 2011; Pivello, 2011), subtropical pine forests and savannas (Brockway et al., 2005; Stambaugh et al., 2011), and Australian dryland Eucalypt forests and woodlands (Burrows and McCaw, 2013). In a time of climate change, high-severity fires can accelerate vegetation lifeform changes via their frequency and size (Odion et al., 2010).

Table 1
Summary of major vegetation types and fire management issues.

Biome	Vegetation type/ region	Fire and fuels management issues
Semi-arid forest and rangelands (North America)	Mixed conifer forests and savannas	Fire exclusion and fuel accumulation; altered fire regimes, expanding wildland urban interface
	Sagebrush and other shrub steppe ecosystems	Invasive grass species and altered fire regimes
Tropical ecosystems	African grasslands and savannas	Maintenance of grasslands and savannas
	Amazon rainforest and Cerrado	Forest clearing and escaped wildfires in rainforests; fire exclusion and fuel accumulation in the Cerrado.
	Australian eucalypt forests and savannas	Altered fire regimes; invasive species
	Subtropical forests, southeastern United States	Fire exclusion and fuel accumulation; altered fire regimes
Mediterranean ecosystems	European Mediterranean	Decrease in rural burning practices and need for prescribed burning programs

Managers of fire-excluded ecosystems face a backlog of work to restore fire-regimes and promote resilience to future fires (Varner et al., 2005; Moritz et al., 2014; Hessburg et al., 2015). In many regions, only a small fraction of fire-dependent landscapes are treated each year (Quinn-Davidson and Varner, 2012; Ryan et al., 2013; Fernandes et al., 2013; McCaw, 2013). Simultaneously, the area burned by wildfires has increased sharply in recent decades, often with different ecological outcomes than historical fire regimes or prescribed burns (Russell-Smith et al., 2007; Cansler and McKenzie, 2014). Mechanical treatments to reduce existing surface and canopy fuels may be necessary in some landscapes before fire can be reintroduced and mitigate the risk of very large, high-severity fires (Stephens et al., 2012; Mitchell et al., 2014; Hessburg et al., 2016).

Depending on the severity and extent of prior fire events, past fires can act as temporary barriers to future fire spread (Collins et al., 2009; Moritz et al., 2011; Teske et al., 2012; Parks et al., 2015a). Because fuels are consumed and reduced for a time after fires, there is less available fuel to burn in subsequent wildfires. After vegetation recovers, previously burned landscapes can be readily reburned in subsequent fire events (Parks et al., 2015b).

In many regions of the world, wildfires are influencing a much greater proportion of the area than prescribed fire and other fuel reduction treatments, and land managers need a better understanding of reburns to inform wildland fire management. For example, knowing how long past fire boundaries remain barriers to fire spread may empower decision-makers to manage some wildfire ignitions as large-scale fuel treatments, or alternatively, determine when and where strategic wildfire management or prescribed burning are required (Agee and Skinner, 2005; Finney et al., 2007; Reinhardt et al., 2008). An understanding of how burn mosaics can regulate future fire spread and burn severity is also critical for landscape planning. Specifically, it is important to anticipate where past fires may or may not act as barriers to subsequent fires (e.g., Price et al., 2015), and where vegetation and fuels management are necessary to restore fire regimes that perpetuate fire-adapted vegetation and habitats. There also is a growing concern that large, high-severity fire events can synchronize changes in lifeform dominance across large areas (Lonsdale, 1994; Hessburg et al., 2007, 2016; LaRosa et al., 2007; Meyn et al., 2007; Moreira

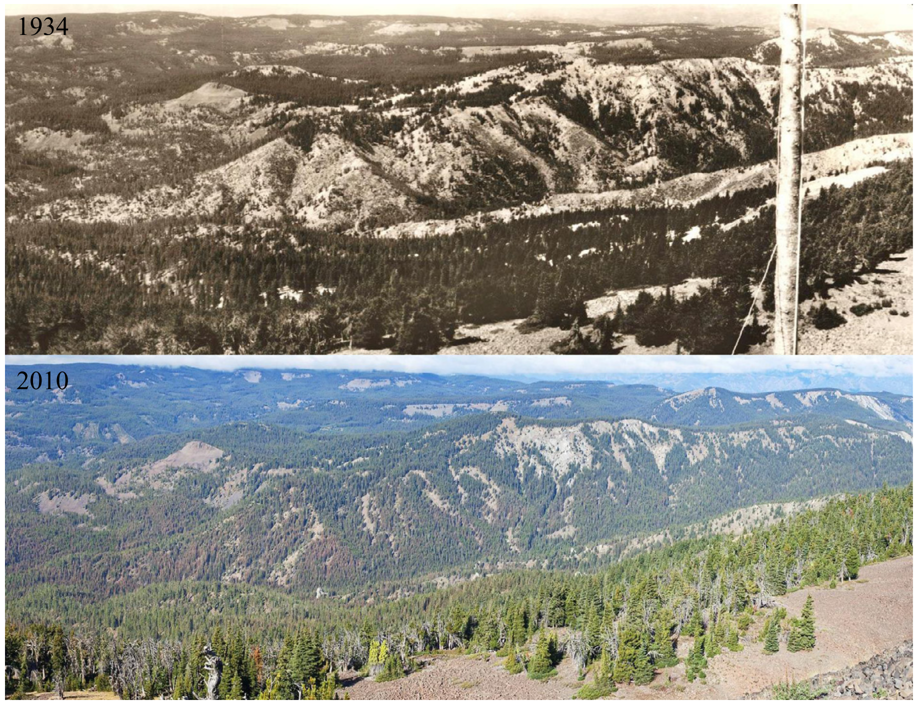

Fig. 1. Repeat panoramic photographs of dry ponderosa pine and dry ponderosa pine-Douglas-fir mixed conifer forests in eastern Washington. In the top photo, the effects of topography and frequent fire are evident; south-facing slopes and ridges support open canopy forest, sparse woodlands, or grassy meadows. Fires occurred less frequently on north facing slopes and valley bottoms; these environments typically supported denser forests and more complexly layered structures. In the lower photo, a century of fire exclusion has all but eliminated the once complex patchwork maintained by frequent fires. Both photos are of the Mission Peak area, facing west. The top photo is from the William Osborne collection, 1930, National Archives, Seattle, WA. The bottom is a John Marshall photograph from 2010.

et al., 2011), creating more chronic, large-scale, grass-fire or shrub-fire cycles (Odion et al., 2010).

In this paper, we provide a review of empirical studies of reburns in fire-adapted ecosystems across the world that have been impacted by fire exclusion. The term "reburn" is used to encompass fire-on-fire interactions even though in some instances past wildfires can act as barriers to fire spread. We confine the review to published field and remote sensing studies, and intentionally exclude simulation studies because validation is generally lacking. Our review focuses on the semi-arid forests and rangelands of North America, tropical and subtropical forests, grasslands and savannas, and the European Mediterranean (Table 1). This review was motivated by two central questions: (1) how did prior wildfires impact the spread and severity of future wildfires, and (2) what are the management implications of fire-on-fire interactions? We found that research, to date, on reburns is unevenly distributed across the world with a relative abundance of literature in Australia, Europe and North America and a scarcity of studies in Africa, Asia and South America. Although fire is a dominant process in boreal and chaparral ecosystems, they were omitted from this review because fires in these ecosystems are generally high severity and driven by extreme weather events. In a larger review, it would have been appropriate to include chaparral and boreal systems because even though they are generally characterized as being more climate than fuel limited, there is recent evidence of past fires influencing subsequent fire occurrence and severity in these ecosystems (Fernandes et al., 2012; Héon et al., 2014).

2. Semi-arid forests of western North America

Of fire-prone ecosystems, semi-arid temperate forests of western North America are among the most studied (Fig. 1). Research has largely been motivated by concerns over departures from historical fire regimes and the increased incidence of large fires over recent decades (Hessburg et al., 2005; Cansler and McKenzie, 2014; Wimberley and Liu, 2014). Across dry mixed-conifer forests, historical fire frequency varied from frequent, low-severity fires to moderately frequent mixed-severity fires, depending on location and site conditions (Heyerdahl et al., 2001; Taylor and Skinner, 2003; Hessburg et al., 2007).

The reduction in 20th-century wildfire across western US forests has been termed a "fire deficit" by some (Marlon et al., 2012; Parks et al., 2015b). Numerous factors have contributed to widespread fire exclusion in dry western forests, including cessation of most intentional aboriginal burning, livestock grazing that reduced grass cover, and active fire suppression (Agee and Skinner, 2005; Hessburg et al., 2005). After decades without fire, many sites that were once characterized by virtue of their fire regime as open forests, savannas, or sparse woodlands, have increased forested area, horizontal and vertical continuity of surface and canopy fuels, and are now predisposed to fires of increased severity (Veblen et al., 2000; Miller et al., 2009; Savage et al., 2013; van Mantgem et al., 2013). Landscape mosaics of high elevation forests also have become increasingly homogenous, likely due to effective suppression of small to mid-sized fires, and ineffective suppression of large fire events during extreme fire weather events (Malleck et al., 2013; Hessburg et al., 2015).

Currently, research from the western US provides the most specific evidence of how long fires remain barriers to subsequent fires, and how past fires mitigate subsequent burn severity (summarized in Table 2). The effectiveness of prescribed burning on mitigating wildfire severity, in particular, has been extensively studied. Across a wide range of studies, prescribed burning alone, or in combination with other fuel reduction treatments, has been shown to be effective at mitigating burn severity in ponderosa pine and semi-arid mixed conifer forests by reducing the intensity and often the spread of subsequent wildfires except under the most extreme fire weather conditions (Agee and Skinner, 2005;

Table 2
Fire-on-fire interaction metrics and findings from semi-arid forests in western North America.

Metric	Region, forest type	Duration or effect	References
Evidence of past fires as effective barriers to subsequent fire spread	Southwest mixed conifer	Up to 6 yr	Parks et al. (2015a,b)
	Rocky Mountains mixed conifer	14–18 yr	Parks et al. (2015a,b)
		Rarely effective as absolute barriers	Teske et al. (2012)
	Interior PNW mixed conifer	Up to 35 yr	Prichard and Kennedy (2014)
Reduced burn severity in previously burned areas	Southwest mixed conifer	Up to 9–10 yr	Finney et al. (2005) Cram et al. (2006) Strom and Fulé (2007) Parks et al. (2014) Kennedy and Johnson (2014)
	Rocky Mountains mixed conifer	10–20 yr	Battaglia et al. (2008)
		Up to 22 yr	Parks et al. (2014) Stevens-Rumann et al. (2016)
	Interior PNW mixed conifer	15–30 yr	Thompson et al. (2007) Lyons-Tinsley and Peterson (2012) Prichard and Kennedy (2014)
	Sierra Nevada mixed conifer	7–20 yr	Safford et al. (2009)
		11–17 yr	Collins et al. (2009) Van Wagtendonk et al. (2012)
Increase in fire-resistant vegetation	Southwest mixed conifer	Increase in large-diameter trees following repeat fires	Holden et al. (2007)
	Rocky Mountains mixed conifer	Repeat fires favor ponderosa pine over lodgepole pine	Larson et al. (2013) Stevens-Rumann and Morgan (2016)
Past fires or fuel breaks assist in suppression	Interior PNW mixed conifer	Fuel breaks were effective 46% of time (coordinated with suppression activities)	Syphard et al. (2011a,b)

Reinhardt et al., 2008; Fulé et al., 2012; Jain et al., 2012; Stephens et al., 2012). Numerous studies have established that strategically-placed prescribed burns can reduce subsequent wildfire severity (Schoennagel et al., 2004; Finney et al., 2005; Strom and Fulé, 2007; Safford et al., 2009; Lyons-Tinsley and Peterson, 2012; Kennedy and Johnson, 2014; Prichard and Kennedy, 2014; Waltz et al., 2014).

Prescribed burns in semi-arid mixed conifer forests exhibit a range of durations as effective barriers to fire spread; retrospective studies following wildfires have been documented that prescribed burns were effective at mitigating burn severity after only 2 yr to as many as 30 yr post treatment (Battaglia et al., 2008; Prichard et al., 2010; Hudak et al., 2011; Lyons-Tinsley and Peterson, 2012; Waltz et al., 2014). Fire weather, location, and site productivity may influence the length of fuel treatment effectiveness and overall resistance to large high severity fire events (Turner and Romme, 1994; Graham, 2003; Schoennagel et al., 2008; Lydersen et al., 2014). The duration of fuel treatment effectiveness is dependent on several factors such as topo-edaphic setting (Hessburg et al., 2015), treatment unit size (Graham, 2003), rates of surface fuel accumulations and post-fire tree regeneration (Battaglia et al., 2008; Shive et al., 2013; Stevens-Rumann et al., 2013), and fire weather (Graham, 2003; Hudak et al., 2011). Several studies found that strategically-placed fuel treatments aided fire suppression operations during subsequent wildfire events (Graham, 2003; Cram et al., 2006; Safford et al., 2009; Hudak et al., 2011).

In contrast to the use of fire in dry mixed-conifer forests and in mountain big sagebrush (*Artemesia tridentata* ssp. *vaseyana*) steppe ecosystems, the use of prescribed and wildfires in Wyoming big sagebrush (*A. tridentata* ssp. *wyomingensis*) may not reduce subsequent fire effects. With the expansion of cheatgrass (*Bromus tectorum*) and other invasive grasses, continuous grass cover that cures early in the fire season can contribute to a grass-fire cycle that favors non-native grass expansion and dominance (Bunting et al., 1987; Balch et al., 2013; Miller et al., 2013). Prescribed burning in shrub-steppe ecosystems can exacerbate the expansion of cheatgrass, especially where it is already present (Keeley, 2006). In addition, certain sagebrush species, including big sagebrush (*Artemisia tridentata* ssp. *tridentata*), may be locally extirpated following repeated wildfires (Pickford, 1932), while other species may be slow to regenerate due to limiting seed dispersal distances and seed rain (Bunting et al., 1987). However, careful management of natural ignitions or prescribed fires during the peak growth of cheatgrass (early spring), followed by additional restoration efforts, such as herbicides and planting of native species, may help to re-establish native shrubs (Baker, 2006).

Wilderness areas and national parks in the western US provide a unique laboratory for studying the effect of past wildfires, following a change in federal policy in 1970 that allowed for management of natural ignitions for resource benefit. For example, wildfires over the past 40 yr have been managed to restore the historical role of fire within the Gila Aldo Leopold Wilderness of New Mexico (van Wagtendonk, 2007; Holden et al., 2010), many wilderness areas in Montana and Idaho, and Yosemite National Park in California. Research has examined the impact of repeated wildfires on burn severity (Collins et al., 2009; Holden et al., 2010; Parks et al., 2014,), fire effects (Holden et al., 2007) and how past wildfires often act as temporary barriers to subsequent wildfires (Teske et al., 2012; Haire et al., 2013; Prichard and Kennedy, 2014; Parks et al., 2015a; Holsinger et al. 2016).

Much of the research on past wildfire perimeters serving as barriers for subsequent fires has demonstrated that past perimeters can confine or at least slow subsequent fire spread. Haire et al. (2013) examined wilderness wildfires in the US Southwest, Northern Rockies, and Sierra Nevada and found that previous wildfires negatively influenced subsequent fire sizes. In a study of fire atlas data in the Northern Rockies, Teske et al. (2012) demonstrated that previous fire perimeters were breached by subsequent fires along 80% of their shared borders, but the size of the reburned areas was usually small (ca. 40–400 ha). The size of the reburned area at the shared fire borders was larger where time since fire was longest, likely due to accumulated surface fuel loads and fuel ladders. Similarly in north-central Washington State, Prichard and Kennedy (2014) documented instances where past wildfires within 3–34 yr were effective barriers to subsequent wildfire spread in high elevation mixed-conifer forests. In a cross-regional comparison of fire atlas data in the northern Rockies and southwestern

Table 3
Recommended metrics for future studies on fire-on-fire interactions.

Topic	Metric	Description and application	Sample references
Evidence of past fires as effective barriers to subsequent fire spread	Duration (yr)	Evidence of treatment longevity of past fires would assist long-term fire management planning and strategic firefighting operations	Teske et al. (2012) Parks et al. (2015a,b)
	Percent of perimeter breached	In many cases, past fires are not absolute barriers but are only partially breached and reburned	Collins et al. (2007) Teske et al. (2012)
	Interactions with fire weather, topography, and vegetation type	Fire weather, topographic position and vegetation can influence the duration and penetrability of past fires	Parks et al. (2015a,b)
Burn severity within previously burned areas	Burn severity	Subsequent burn severity provides evidence that past fires may effectively mitigate subsequent burn severity	See Table 2
	Residual patch mosaic of burn severities	Landscape metrics of residual patch mosaic (percent area in high, moderate, low and unburned), patch size distributions and interactions with topography and vegetation have important implications for wildlife habitat mosaics and evidence for self-regulation of burn severity at larger spatial scales	Hessburg et al. (2005, 2007) van Wagtendonk (2007) Boer et al. (2009) Collins et al. (2009) Burrows and McCaw (2013) Haire et al. (2013)
	Interactions with fire weather	Several studies demonstrate lack of treatment effectiveness under extreme fire weather. A better understanding of these potential thresholds is needed	Graham (2003) Hudak et al. (2011) Lydersen et al. (2014)
	Interactions with topography	Evidence of effectiveness on steeper slopes. Analysis of topographic effects may assist strategic placement of fuel treatments and managed wildfires	Heyerdahl et al. (2001) Taylor and Skinner (2003) Kellogg et al. (2008) Safford et al. (2009)
	Burn severity and vegetation	Vegetation type and age can either reduce or increase burn severity	Collins et al. (2007) Holden et al. (2007) Thompson and Spies (2009) Miller et al. (2012)
Increase in fire-resilient vegetation	Vegetation type change Resilient stand characteristics	Species or vegetation type to more fire resistant assemblages Mean tree size, canopy base height	Brown and Johnstone (2012) Larson et al. (2013) Stevens-Rumann et al. (2016) Holden et al. (2007) Bennett et al. (2013)
Use of past fires in wildland fire operations	Barriers to fire spread Anchor points for prescribed burns or backfires	Evidence of effectiveness of barriers when combined with fire suppression Use of past fires in firefighting	Syphard et al. (2011a,b)

United States, Parks et al. (2015a, 2016) found that past wildfires diminished both the ignition potential spread of subsequent wildfires, and that time-since-fire was an important factor. In related studies Parks et al. (2014) and Stevens-Rumann et al. (2016) reported that burn severity was uniformly lower in reburned areas than in previously unburned areas up to 22 yr between fires, and the reduction in burn severity diminished with time since fire in forested ecosystems. In a recent study of factors contributing to wildland fire spread in the western US, Holsinger et al. (2016) reported that recent burns (<5 years) generally regulated fire spread but that topography often interacts with fuels. For example, in the Gila-Aldo Leopold Wilderness area, herbaceous fuels and fine-grained topography both limited fire spread. In contrast, weather and fuels, including recently burned areas, were primary drivers of fire spread in the northern Rockies.

Extremely hot, dry weather coupled with high winds have been shown to reduce burning thresholds in young regenerating post-fire lodgepole pine (*Pinus contorta* var. *latifolia*) forests (Turner and Romme, 1994) and past fuel treatments (Lydersen et al., 2014). Graham (2003) found that some past wildfires and prescribed fires were effective barriers and reduced severity within portions of the 2002 Hayman Fire, but extreme fire weather days overwhelmed other past wildfire perimeters. Haire et al. (2013) also reported higher elevation moist, mixed-conifer forests experienced higher severity reburns especially with longer time-since-fire.

Several studies have demonstrated that burn severity of repeated wildfires is not uniformly lower in reburned areas – instead, subsequent burn severities generally reflected those of the previous wildfire. For example, in California and southwestern Oregon, three studies found areas that burned at low- to moderate-severity tended to burn at the same or lower severity in a second fire (Thompson et al., 2007; Collins et al., 2009; van Wagtendonk et al., 2012). However, if an area burned previously in a high-severity fire, a higher proportion of the area burned at high-severity in the subsequent fire (Collins et al., 2009; van Wagtendonk et al., 2012). Thompson et al. (2007) similarly found that areas that experienced severe crown fire damage in an initial wildfire had a high prevalence of damage in a subsequent fire 15 yr later. In a study of repeat wildfires in Yosemite National Park, Collins et al. (2009) reported lower reburn severity when time since fire was less than 11–17 yr. Further, regardless of time since fire, forests dominated by lodgepole pine had significantly higher burn severity than red fir (*Abies magnifica*) forests (Collins et al., 2007). In Douglas-fir (*Pseudotsuga menziesii* var. *menziesii*) dominated stands of northern California, Miller et al. (2012) reported that single wildfires and reburns had the same percentage of high burn severity (9–10%), where wildfires occurred within 30 yr of one another. However, when wildfires were greater than 30 yr apart, high burn severity in reburned areas decreased to 5% (see Box 2).

Box 2 Explaining regional and site differences in fire effects.

Likely explanations for regional and site-specific differences in fire effects within previously burned areas are: (1) variability in tree species adaptations and their susceptibility to fire, (2) prevalence of flammable shrub fields following moderate to high-severity fire events, and (3) time sensitive changes in the susceptibility of forest structural conditions as a result of stand dynamics processes (Oliver and Larson, 1996). For example, young regenerating Douglas-fir and ponderosa pine trees are more susceptible to fire damage than older trees, and species such as lodgepole pine, Engelmann spruce (*Picea engelmannii*), and subalpine fir (*Abies lasiocarpa*) have lower resistance to burning than thick-barked conifers such as mature ponderosa pine, western larch (*Larix occidentalis*), and Douglas-fir. van Wagtendonk et al. (2012) and Coop et al. (2016) suggested that the pattern of burn severity in a second wildfire was likely due to a vegetation shift from forests to shrublands. In a ponderosa pine dominated ecosystem of the southwestern US, several studies found areas that previously burned in high severity events shifted vegetation to grass or shrubland-dominated ecosystems, while areas that burned as low-intensity surface fires perpetuated forest cover (Savage et al., 2013; Coop et al., 2016). In California, Coppeletta et al. (2016) reported that although burn severity in reburned areas was largely driven by weather, rapid shrub regrowth after the initial fire influenced higher burn severity. Thompson and Spies (2009) also documented that crown fire was associated with understory shrub dominance, which provided abundant fuel ladders and an easy transition from surface to crown fires.

Several studies have demonstrated that repeated fires alter forest structure and composition, increasing resilience to subsequent fires. In the southwestern US, for example, in a 50-year retrospective analysis of long-term fire effects, Holden et al. (2007) found a higher proportion of larger diameter ponderosa pine in areas that experienced several fires in comparison with those that had experienced a single fire. They attributed the difference to tree thinning effects of lower severity fires, which removed smaller thin-barked trees, reduced inter-tree competition for moisture and nutrients, and favored survival of larger diameter trees. In Montana and Idaho, Larson et al. (2013) and Stevens-Rumann and Morgan (2016) demonstrated that an initial low-severity burn in ponderosa pine dominated forests decreased stand density and fuels, but also initiated a dense cohort of regenerating lodgepole pine. However, a reburn within 20 yr decreased lodgepole pine establishment, surface fuels, and stand density. Thus, in some mixed-conifer patches, where fire-prone species such as lodgepole pine might proliferate after a fire, high frequency, low-severity burns can maintain open canopy ponderosa pine forests and perpetuate low-severity fire regimes. Conversely, repeated high-severity wildfires over short intervals can result in minimal forest regeneration and lagged tree establishment with no remaining overstory; an outcome recognized as early as 1921 (Larsen, 1921), which often favors fire tolerant vegetation communities and/or lifeform shifts (Savage and Mast, 2005; Larson et al., 2013; Stevens-Rumann and Morgan, 2016).

In summary, increased fire frequency via prescribed burning and managed wildfires in frequent fire forests can increase resilience to subsequent wildfires by reducing surface and canopy fuels, and altering overstory and understory forest structure (Donato et al., 2009; Perry et al., 2011; Hessburg et al., 2015, 2016). Depending on their frequency, severity, and vegetation adaptations to fire, repeat wildfires may shift species composition (Larson et al., 2013) or life form (e.g., forest to shrubland or grassland, Turner et al., 1993; van Wagtendonk et al., 2012). Fuel reduction measures that are maintained either by managed wildfire or prescribed burning may be a cost-effective way to maintain open-canopy structures, lower surface fuel loads, and reduce likelihood of future high severity fire events (North et al., 2012).

Previous wildfires in fire-prone forests of the western US can serve as temporary fuel breaks to subsequent wildfires (Graham, 2003; Collins et al., 2007; Teske et al., 2012); effectiveness decreases with time since fire, varies with burn severity and forest type, and thresholds to burning may be greatly reduced during extreme fire weather conditions. Western US wildfire seasons have already increased in length, and the likelihood of extreme fire events is increasing (Flannigan et al., 2009; Littell et al., 2009). These changes will test the effectiveness of fuel reductions from past burns. Fire weather, prior fire severity, species composition, regeneration, succession, and stand dynamics processes all play a role in the likelihood and longevity of reduced burn severity of subsequent wildfires in the semi-arid forests of western North America.

3. Tropical and subtropical ecosystems

With the exception of tropical rainforests, fire is a frequent visitor of many grasslands, savannas, and dry tropical and subtropical forests. Here, we review what is known about reburns and management of repeat fires in Australian forests and savannas, subtropical forests and grasslands of the southeastern US, African grasslands and savannas, and the Amazon rainforests and cerrado of Brazil. Fire exclusion via interruption of aboriginal burning practices, fire suppression, land conversion, and cattle grazing has in many cases resulted in similar changes in fire regimes to those of semi-arid temperate forests and rangelands of the United States (Russell-Smith et al., 2004; Welch et al., 2013; Bennett et al., 2014). However, in some savanna ecosystems, frequent, wide-ranging fires are presenting new challenges.

3.1. Australian eucalypt forests and savannas

Historically, aboriginal burning practices were an integral part of fire regimes in fire-prone ecosystems of Australia (Dyer et al., 2002; Whitehead et al., 2003). Research on traditional fire knowledge has intensified in recent decades in an effort to guide more proactive fire management that supports restoration of fire regimes, and promotes native biodiversity (Russell-Smith et al., 2003; Price et al., 2005). For example, in northern Australia, within the Arnhem Land Plateau, traditional fire knowledge is being used to inform restoration of early-season fire regimes and landscape burn mosaics. Frequent, wide-ranging grassland and savanna fires are reducing biodiversity of northern Australian savannas by adversely impacting rainforest inclusions and assemblages of fire sensitive or obligate seeder species (Russell-Smith et al., 2003; Williams et al., 2009). Climate there is strongly monsoonal, and over 75% of fires in northern Australian savannas burn during an extended (August–November) late dry season. Fires at this time of year tend to burn at much higher intensity than in other seasons (Williams et al., 1998; Gill et al., 2009). Area burned is dependent on grass continuity and fuel availability, with low area burned in extremely dry or wet regions, and highest area burned in areas of moderate rainfall (Spessa et al., 2005). Contemporary fire management in northern Australia is focused on restoring spatial patchiness of burn mosaics across landscapes, burning outside of the late dry season, and increasing the chances of longer fire-free intervals within some areas of heterogeneous burn mosaics (Gill et al., 2003; Price et al., 2005). A recent concern for Australian savannas is the expansion of exotic grasses such as Gamba grass (*Andropogon gahyanus*), Para grass (*Urochloa mutica*), and mission grass (*Pennisetum polystachyon*), that attain high biomass, and like cheatgrass in semi-arid shrublands of the western US, are shifting fire regimes to a high frequency, high severity, grass-fire systems (Lonsdale, 1994).

In areas of high rainfall, such as forest-savanna ecotones of northeastern Australia, recent expansion of rainforest species into woodland eucalypt savannas was attributed to cessation of aboriginal burning and reduction in fine fuels with cattle grazing (Hopkins et al., 1996; Russell-Smith et al., 2004). Long fire-free intervals can support the expansion of rainforest species into savannas, but this trend has only been noted in areas of sufficient rainfall and on soils with adequate moisture holding capacity (Russell-Smith et al., 2004). In contrast, dry eucalypt woodlands that were historically dependent on variable-intensity surface fires experienced an increase in frequent wide-spread, high-intensity fires in recent decades (Russell-Smith et al., 2007). These more frequent fires (now about every 3 yr) tend to lower surface fuel loads and associated carbon stocks. Alternatively, increasing the fire return interval to ~10 yr, or burning under moist conditions may mitigate future fire hazard while maintaining some surface fuels critical for eucalypt establishment (Aponte et al., 2014). Bennett et al. (2014) found a similar trade-off, and advocated for prescribed burning under high moisture conditions or slightly longer fire return intervals (~10 yr). More frequent and/or higher intensity prescribed burns can greatly decrease survivorship of small diameter eucalypt trees (Bennett et al., 2014) (see Box 3).

Box 3 Changes afoot in Australian eucalypt forest prescribed burning.

> Over the past decade, following several severe fire seasons and increased focus on risk mitigation, constraints on prescribed burning have greatly reduced the area burned (Russell-Smith et al., 2007). Because prescribed burning targets are not being achieved, a new system of fire management zoning has been proposed to include community protection zones, in which fuels are to be burned at least every 4 yr in a 5-km radius of communities, a bushfire modification zone, in which fuels are reduced at 5–7 yr intervals to mitigate wildfire behavior and increase suppression opportunities within a 20-km radius of communities, and a broader biodiversity management zone >20 km from communities in which mosaic burning is managed to maintain and promote biodiversity (Burrows and McCaw, 2013).

In southwestern Australia, active prescribed burning was implemented in the 1960s to mitigate future wildfire hazard (Fig. 2). Boer et al. (2009) examined the effects of long-term prescribed burning in eucalypt forests, evaluating fires dating back to the 1960s. They demonstrated that prescribed burn mosaics alter fuel structure and significantly decrease the incidence, severity, and extent of subsequent wildfires. Past prescribed fires were effective in mitigating subsequent wildfire severity for 6–9 yr. Prescribed fires in this region promote diverse seral stages and may not only decrease future fire hazard, but also maximize biodiversity (Wittkuhn et al., 2011; Burrows and McCaw, 2013).

3.2. Subtropical forests of the southeastern United States

Wildland fire management, wildlife habitat, and forest restoration have been extensively studied in subtropical southern pine and mixed-wood forests of the southeastern US (reviews by Brockway et al. (2005), Van Lear (2005) and Mitchell et al. (2009)). Across the southeastern US, herbaceous and litter surface fuels dominate fire behavior. Historical fire regimes were characterized by frequent, low-intensity surface fuels that perpetuated light, flashy surface fuels that rapidly became available for repeated surface fires (Stambaugh et al., 2011). Climate is humid with high annual rainfall, punctuated by seasonal droughts in the growing season and winter dormant seasons. Most wildland fires occur during the winter dormant season and are carried by dry grass, litter and shrubs. The sandy soils in the Coastal Plain support mixed forests of longleaf pine, slash pine and loblolly pine with grassy understories. Excessively drained soils can contribute to rapid drying of fine surface fuels during dry seasons, and historic

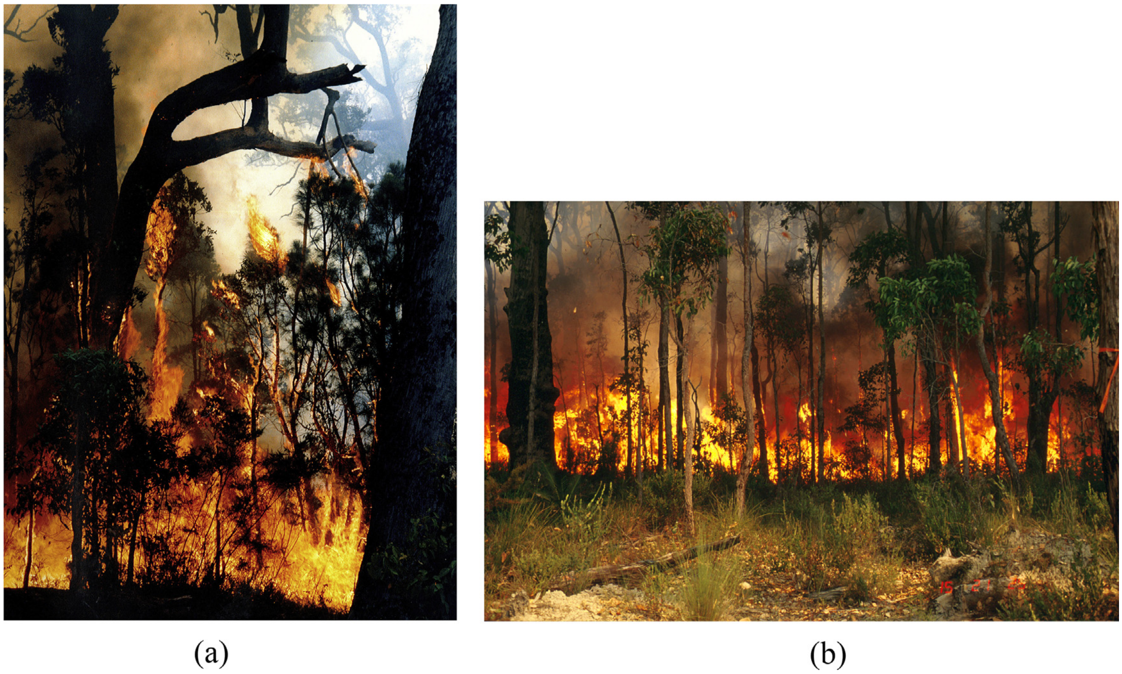

Fig. 2. Prescribed burn in eucalypt forests of southwestern Australia (photo credit Miguel Cruz, CSIRO).

fire return intervals were frequent at 1–3 years (Stambaugh et al., 2011). Riparian areas and low depressions support hammocks and swamps that have much longer fire return intervals, associated with periods of prolonged drought. The mesic pine flatwoods of the interior Piedmont region have palmetto-gallberry understories and burn somewhat less frequently (3–5 years) and with greater intensity than the grass-dominated Coastal Plain savannas and forests (Stambaugh et al., 2011). The mountainous forests of the southern Appalachian have the greatest component of mixed broadleaf deciduous forests and typically burn at longer time intervals (10–40 years) (Brose et al., 2001; Brose and Wade, 2002). Humans have been contributing to frequent burning in southeastern pine forests for millennia. Native Americans used fire to manage fire risk around settlements and across landscapes to maintain open pine forests and savannas (Van Lear, 2005). There is some evidence that when EuroAmerican settlers colonized the southeast, they increased the frequency of burning in some areas (Stambaugh et al., 2011) (see Box 4).

Box 4 Fire-dependent longleaf pine.

Longleaf pine (*Pinus palustris*) forests were once the dominant forest type throughout the southeastern US and are now one of the most threatened ecosystems in North America (Stambaugh et al., 2011; Mitchell et al., 2014). Longleaf pine is highly adapted to fire with the ability to recruit in canopy gaps and a fire-resistant seedling "grass stage" in which needles sprout in a bunch grass form during which time the seedling root systems develop and can readily resprout after light surface fires. Once seedlings have sufficient root systems and reserves, they shoot up into a sapling form with a terminal bud out of reach of typical surface fires. Native longleaf pine and bunchgrass assemblages generally grew in open park-like forests and supported many fire-adapted understory plants and obligate wildlife species including the endangered red-cockaded woodpecker. Through agricultural clearing, development, and fire suppression, the extent of longleaf pine has been reduced from 38 to 1.2 million ha, and most remnant patches occur in scattered fragments (Brockway et al., 2005).

Today, prescribed burning is commonly practiced throughout the southeastern US and is the source of available information on the effects of reburns. Frequent intentional burning (every 3–4 years) in the region is driven by the need to reduce rapidly accumulating surface and ladder fuels, and for maintaining longleaf pine and other fire-dependent species (Brose and Wade, 2002; Brockway et al., 2005; Melvin, 2012, Fig. 3). Davis and Cooper (1963) published a survey of wildfire occurrence, size and intensity of wildfires following regional fire years in 1955 and 1956 in north Florida and south Georgia. Wildfire occurrence generally increased with time since fire, and height of bark char, which was measured as a proxy for fire intensity, was higher in >3 year old roughs (1.8–10.7 m) than sites that had burned within the recent 2–3 years (0.9–3.6 m). They concluded that fire return intervals of 3–4 years were generally recommended to reduce the chance of catastrophic wildfires. Brose and Wade (2002) provide a summary of surface fuel characteristics in sites that had been burned ranging from 1 to 5 years (termed age of rough). They demonstrated major differences in shrub cover and height, which contributed to wide differences in potential fire behavior and recommended a prescribed burning interval of 3–5 years to maintain a low severity fire regime. In a study of time-since-fire and burn severity in southern pine flatwoods of Florida, Malone et al. (2011) found that burn severity is low in fires with return intervals of 1–4 years but spikes to a high burn severity at 5–6 years during peak litter and woody fuel accumulations. Burn severity actually decreases after 7 years post fire, associated with an encroachment of midstory hardwoods which are less flammable (Kane et al., 2008). In a study of invasion

(a)

(b)

Fig. 3. Longleaf pine forest during and following an understory prescribed burn (photo credit Roger Ottmar, US Forest Service Pacific Northwest Research Station).

of tropical pine savannas in southern Florida, Stevens and Beckage (2009) found that long-term fire suppression was leading to the development of relatively inflammable midstories of hardwoods, including the invasive shrub Brazilian pepper (*Schinus terebinthifolius*).

With their arable soils, pine forests were historically cleared for agriculture and are today increasingly fragmented from land use. Many old fields have reverted back to slash and loblolly pine with mixed hardwood understories. These sites have had infrequent fire and support high-intensity fires only during unusually dry periods (Mitchell et al., 2014). Frequent burning may also be necessary in some landscapes to control the spread of invasive species and perpetuate a frequent low-severity fire regime. Long unburned ecosystems present challenges for reintroduction of fire in areas with deep organic soils because smoldering ground fires can contribute to soil heating that damages roots and tree cambial tissue (Varner et al., 2005).

Under climatic change projections, the future of fire-on-fire interactions in the southeastern US is likely to become more complicated. Increased annual temperatures will likely be associated with increased hurricane activity and other extreme weather events including prolonged drought. Although many uncertainties exist, fire seasons could be longer and be punctuated by extreme weather events that could lead to large wildfire seasons (Liu et al., 2012). Combined with constraints on prescribed burning periods, more forests in the southeastern US may burn under high severity wildfire events associated with extreme fire weather. Although windows of opportunity for prescribed burning could be shortened in the future, there is a critical need for frequent burning to promote ecosystem resilience in a warming climate (Brockway et al., 2005). Adaptive management with a focus on monitoring ecosystem responses will be an essential part of future fire management (Mitchell et al., 2014).

3.3. African grasslands and savannas

Tropical savannas represent some of the most extensive grasslands in the world (Kucera, 1981). In Africa, humans have occupied and managed savannas for millennia, and they represent a major ignition source (Boko et al., 2007). Lightning ignitions are also common, but they tend to coincide with the rainy season in tropical savannas, whereas human ignitions are responsible for much of dry season burning and the greatest area burned (Archibald et al., 2009). Historically, African savannas burned every 1–5 yr, and fires maintained grassy understories and reduced competing shrub and tree vegetation (Kucera, 1981).

Wildland fires in tropical grasslands and savannas are predominantly fueled by grass and litter. After an area is burned, fuel accumulation and curing are highly dependent on weather. With adequate rainfall, grasses quickly grow back and can support fire spread after a few weeks of dry weather (Stott, 2000). Because human ignitions are so common, area burned is mostly dictated by fuel continuity. Fires are less common in developed areas due to fuel fragmentation from agricultural fields and roads (Archibald et al., 2009).

Frequent fire plays a central role in maintaining African grasslands and savannas. In its absence, shrubs and trees encroach (Higgins et al., 2000). In an examination of fire management policies in Kruger National Park in South Africa, Bond and Archibald (2003) concluded that a mix of prescribed and lightning-ignited fires are necessary to maintain mesic savannas that would otherwise succeed to closed-canopy forest. They also recommended prescribed and lightning-ignited fires of variable intensities and sizes to maintain biologically diverse grasslands in more arid landscapes. Some African parks have attempted a lightning-only policy but found that high intensity fires during the dry season did not result in the desired burn mosaic necessary to support the highest plant and habitat diversity (van Wilgen et al., 2004).

3.4. Amazon rainforest and cerrado

Fire is an infrequent visitor in some ecosystems, including moist tropical Amazonian rainforests. In these ecosystems, repeated fire can lead to long-term habitat degradation and conversion to non-forest vegetation. Fire return intervals in Amazonian forests range from 500 to 1000 yr (Cochrane et al., 2009). Lightning commonly occurs in the Amazon, but ignitions are typically extinguished by humid understories (Pivello, 2011) (see Box 5).

Box 5 Forest-cerrado ecotone shifts over the centuries.

Over multi-centenary and longer time scales, paleoecological records show dynamic shifts in the borders of cerrado and tropical forests that are closely tied to climatic and fire frequency changes (Pivello, 2011). Rainforest retreat was recorded during the early Holocene

(11 and 10,000 yr BP) and again between 8000 and 4000 yr BP (Pessenda et al., 2004). Burning practices of pre-Columbian tribes likely slowed the advance of rainforest during moister periods and maintained greater expanses of savanna. Savanna vegetation, with its ability to resprout and need for open growing conditions, is favored by frequent fire. Depending on site productivity and longer fire return intervals, forest vegetation can establish and once closed canopy conditions are reached, can perpetuate longer fire free intervals by shading out grass-dominated understories and creating cooler, moister microsites (Hoffman et al., 2012).

Bordering the rainforests is the Great Plateau of Brazil, the vast Cerrado (tropical broadleaf woodlands and scrublands), representing the second largest biome in South America (Fig. 4). Climate is strongly seasonal with a distinctive wet season from November to April, and a dry season from May to October (Balch et al., 2008). Although lightning ignitions are common, indigenous people have a long history of intentionally igniting grasslands during the dry season (Welch et al., 2013). Fire return intervals are frequent, with fires every 1–2 yr in grasslands, and every 5–10 yr in savannas and woodlands (Fig. 5).

The cerrado supports a high diversity of fire-adapted plants and communities. Savanna vegetation is adapted to frequent fires, fueled by C4 grass understories and support small shrubs and trees that have bark and the ability to resprout after fire (Hoffman et al., 2012). Recent fire exclusion has led to heavy fuel accumulations and the potential for high-severity events (Pivello, 2011). In addition, contemporary timber harvest of some Amazonian rainforests and clearing for agriculture has led to accumulations of dry, flammable fuels nearby intact tropical rainforest and cerrado with high surface fuel loads. During prolonged droughts, fires in adjacent

Fig. 5. Burning cerrado vegetation. Many plants are deeply rooted and adapted to sprout or reseed after fires. (photo credit William Hoffman, North Carolina State University).

Fig. 4. Map of the neighboring Amazon Basin and Cerrado, South America.

cleared lands creep into intact tropical forests and burn their dry leaf litter. Because most rainforest tree species have thin bark and few adaptations to fire, tree mortality following these low-severity surface fires can be 100 percent (Cochrane et al., 2009). Following stand-replacing, low-severity events, dead tree and shrubs contribute to high downed wood accumulations and these surface fuel additions often lead to more severe future events (Brando et al., 2014). Repeated fires every 5–10 yr have contributed to tree regeneration failure and type conversion from forest to dry woodland and grassland, especially on rainforest margins (Cochrane et al., 2009). As the incidence of prolonged drought increases, the fate of the remaining Amazonian rainforest, particularly at its margins, is uncertain. Reducing the incidence of slash and burn agricultural practices and rainforest fragmentation appears to be the best defense against forest conversion by repeated burning (Laurance and Williamson, 2001; Fearnside, 2005).

In sharp contrast, fire-dependent cerrado ecosystems are among the most endangered in the world; their maintenance and restoration depend on active fire management. Within the cerrado, agricultural expansion has sharply reduced fire frequency and extent, and impacted fire-dependent habitats and species diversity. Land preservation and restoration of frequent surface fires through support of indigenous burning practices and prescribed burning programs are key to restoring native cerrado ecosystems (Geiger et al., 2011; Pivello, 2011; Welch et al., 2013).

4. European mediterranean

The Mediterranean Basin of Europe is prone to frequent fire, in part due to warm, dry summers, flammable vegetation, and steep terrain (Pausas et al., 2008), but also due to a long history of anthropogenic burning (Pyne, 2009; Fernandes et al., 2013). Paleoecological studies demonstrate that closed forest was the predominant physiognomic type prior to human settlement, and that climatic forcing alone cannot explain the significant shift to non-forest over much of the region. As fire frequency increased through forest clearing and intentional burning, forest area declined, they were replaced by heath- and shrublands, open woodlands, and grasslands (Colombaroli et al., 2007; Tinner et al., 2009; Morales-Molino et al., 2013). Decline in forested area and increased use of fire varied by era of human occupation. For example, at sites near human settlements, forested area declined as early as 6000 yr BP, whereas in more remote areas, declines occurred as late as 2000 yr BP (Kaplan et al., 2009). Even today, the rich floral diversity of the region is attributed to small-scale rural burning practices that created a diverse mosaic of vegetation types, favoring shrublands and grasslands over closed-canopy forests (Pyne, 2009; Moreira et al., 2011; Pausas and Paula, 2012).

Since the 1940s, rural populations have declined in many areas of the European Mediterranean, as people have moved to urban centers. Historical burning practices on silvo-pastoral landscapes of human design have correspondingly declined (Pausas and Fernández-Muñoz, 2012; Seijo and Gray, 2012; Huffman, 2013). In some countries, such as Portugal and Spain, agroforestry operations have expanded and tend to favor dense pine plantations (Fernandes et al., 2014). Both factors – the cessation of rural burning practices and afforestation – have contributed to more continuous flammable shrublands and forest cover, predisposing some landscapes to large, high-severity fire events (Pausas and Fernández-Muñoz, 2012). In other more mesic landscapes, a shift to broadleaf forest and short-needled conifer assemblages has actually reduced the likelihood of fire spread through shadier, mesic forest understories with lower herbaceous cover and higher live and dead fuel moistures of litter and understory plant species than open-grown vegetation (Azevedo et al., 2013; Fernandes et al., 2015). In some locations, expansion of forestland through cessation of pastoral burning has contributed to a sharp decline in the floristic diversity of previously fire-maintained heathlands (Ascoli et al., 2013) (Fig. 6). However, promotion of mature, continuous oak and short-needled conifer forests may provide fuel breaks and limit the extent of future large fire events (Fernandes et al., 2015).

Recent increases in the incidence of large and severe wildfires in parts of the Mediterranean region have been linked to a warming climate (Fernandes et al., 2013). This trend is expected to continue under predictions of warmer and drier summers (Seidl et al., 2011). Additionally, there is evidence that a shift to large, high-severity fire is contributing to homogenization of post-fire landscapes, which may perpetuate a cycle of large fires in the future (Loepfe et al., 2010; Moreira et al., 2011). Multiple studies within the Mediterranean basin have demonstrated that patterns of vegetation and fuels and connectivity of flammability of vegetation are critical drivers of large fire spread and severity. In a study of changing fire regimes in the Catalonia region of northeastern Spain, Lloret et al. (2002) examined land cover trends in 1956, 1978 and 1993 and relationships between landscape patterns and fire occurrence. They found an overall trend in the homogenization as marginal agricultural and pasturelands were abandoned, and flammable shrubland and forest cover increased with a concomitant increase in wildfire area. Within a larger study of changing fire regimes in Catalonia, Loepfe et al. (2010) use a combination of land cover maps, satellite images and fire records to evaluate landscape homogenization associated with the abandonment of marginal agricultural lands. They demonstrated that increased continuity of forests and shrublands have contributed to a positive feedback with fire size. Further, recent large fires have actually perpetuated landscape homogeneity because they tend to be high severity and perpetuate flammable shrublands over forest vegetation. Fernandes et al. (2014) studied trends in fire hazard and area burned from 1943 to 2011 in Portuguese public forestlands. Following widespread agricultural abandonment in the 1950s and *Pinus pinaster* plantation projects, flammable shrublands and plantations dominated mountain landscapes in Portugal and contributed to increased burned area and a shift to a more severe, weather-dependent fire regime. Similarly, Curt et al. (2013) evaluated fire return intervals of wildland fires in southeastern France and found that although there was little evidence of fuel age influencing fire frequency in flammable shrublands, the size and continuity of vegetation patches was a strong predictor of fire

Fig. 6. Prescribed burn in Mediterranean heathlands of Portugal (photo credit Luís Mário Ribeiro, ADAI-CEIF).

probability. They concluded that management for varied fuel types across fire-prone landscapes, including mature oak forests and young shrublands with low biomass, can disrupt fuel connectivity and potentially limit future fire size and probability of spread. Fernandes et al. (2016a) also found that fuel connectivity was an important predictor of large fire spread and that areas with lower edge density had greater fire spread. Fernandes et al. (2016b) also report that large fires also tend to burn in areas with higher accumulated fuels and that shrublands <6 years since fire have lower loads and may contribute resistance to fire spread across landscapes.

Prescribed burning in the Mediterranean region is not a widespread practice, but it is being proposed to decrease the risk of wildfires near communities, make pine plantations more resilient to fire, and to restore more heterogeneous plant assemblages and wildlife habitats (Moreira et al., 2011; Ascoli et al., 2013; Fernandes et al., 2015). Managing unplanned wildfires for resource benefit and also to mitigate future wildfire severity has also been proposed (Regos et al., 2014). Resumption of traditional burning practices for increased forage production is also being considered, as is the use of unplanned wildfires for resource benefit during mild fire weather (Fernandes et al., 2013; Regos et al., 2014). Many constraints currently limit the use of prescribed burning, including negative public perceptions, a shortage of professionals to conduct burn operations, high population densities, the large extent of the rural-urban interface, and explicit legislation in some countries that ban or limit its use (Fernandes et al., 2013) (see Box 6).

Box 6 Growing awareness of the importance of fire.

Even though barriers to prescribed burning are numerous, there is growing awareness of the importance of repeat fires to maintaining the floristic and habitat diversity of the region (Fig. 6). Despite substantial investments in fire suppression, the incidence of large, high-severity fires has increased in recent decades (Regos et al., 2014). In a warming climate, maintaining a fire-regulated mosaic of vegetation may be critical to mitigating the effects of future wildfires (Moreira et al., 2011).

5. Conclusions

5.1. Lessons learned

Our review of fire interactions within fire-adapted ecosystems highlights the complex roles fire plays in modifying vegetation patterns, available fuels, and the size and severity of subsequent wildfires. Repeated short-interval burning typically limits the distribution and biomass of future vegetation – generally favoring maintenance of grassland, shrubland, savanna, and open canopy forest mosaics over closed forests (Bond and Keeley, 2005; Bond et al., 2005). However, over the past century, fire regimes have shifted from frequent to infrequent fires in many woodlands, savannas, and dry forests, with higher burn severity due to fuel accumulations (e.g., Savage et al., 2013; van Mantgem et al., 2013).

Because of feedbacks to vegetation, shifting fire regimes can trigger changes in vegetation that alter long-term fire regimes. Non-native grasses have found a foothold in many fire adapted ecosystems and are predisposing affected landscapes to frequent fires, while perpetuating non-native grass cover and a grass-fire cycle (Brooks et al., 2004; LaRosa et al., 2007). Native woodlands and some interior tropical forests are currently threatened due to frequent fires fueled by grass and logging slash (Tunison et al., 2001; D'Antonio et al., 2011). During prolonged drought or dry weather events, grass and slash fires can also carry into rainforests with little adaptation to fire, and contribute to conversions of these forests to non-native grasslands.

Restoring frequent fire to long unburned landscapes can be expensive and risky. However, in all fire-prone ecosystems, fire *will* return, and landscape legacies of past fires, or their prolonged absence, will often dictate the characteristics of future fires (e.g., light underburn, patchy mixed-severity fire, or high-severity crown fire Peterson, 2002; North et al., 2015). To address the backlog of fuel accumulation in ecosystems that once had frequent fires, repeated low-intensity fires can be used during benign fire weather to reduce accumulated fuels or restore heterogeneous burn mosaics (Ryan et al., 2013). In dense forested areas, particularly with multiple canopy layers and ladder fuels, prior forest thinning may be required to reduce uncertainty and make the reintroduction of fire feasible (Mitchell et al., 2014; Hessburg et al., 2015, 2016).

How long past fires influence subsequent fire behavior and effects varies widely by biophysical and climatic setting. For example, grassland and many savanna ecosystems are dominated by fine herbaceous fuels that rapidly recover following fire; productive sites may be fully revegetated in a matter of weeks (Stott, 2000). Feedbacks to subsequent fires are therefore generally short compared to forests in which surface fuels are dominated by fine woody fuels that may take much longer to accumulate following wildfires. Unplanned wildfires are responsible for the majority of burning in many fire-prone landscapes. Thus it is important for managers to understand how they might be harnessed to restore native fire regimes, promote landscape heterogeneity, and create burn mosaics that can, except during the most extreme fire weather, mitigate subsequent fire spread and effects (see Box 7).

Box 7 Recommendations for fire prone ecosystems.

Recommendations for managing fire are similar across fire-prone ecosystems of the world. Active management of wildland fires, including prescribed fires and allowing unplanned wildfires to spread under favorable burn windows, can all restore ecological processes, increase landscape heterogeneity and habitat complexity, while reducing subsequent wildfire suppression costs, fire fighter risk, and carbon fluxes. However, frequent fires, particularly in areas where invasive grasses now dominate understory fuels, may jeopardize native vegetation and result in type conversions and/or reduced biodiversity and habitat. For fire managers, this is a ''*goldilocks dilemma*'' in which overly frequent fire in some ecosystems can alter native vegetation assemblages and habitat patterns, while long intervals between fires also can have negative consequences.

As we conducted our review, we were struck by the similar stories of 20th-century fire exclusion across the world's fire-prone ecosystems. Lack of fire from active fire suppression, cattle grazing, human settlement and other factors has led to widespread encroachment of open forests, shrublands and grasslands by closed canopied forests, and associated with this, an increased potential for more severe wildfires. In each case – in semiarid forests and subtropical forests of North America, Australian eucalypt forests and savannas, African savannas and grasslands, Brazilian cerrado, and throughout the European Mediterranean – fire regimes were historically influenced by aboriginal burning practices in addition to lightning-caused ignitions. Human societies in many parts of the world were, and in many places still are, reliant on landscape burning (Pyne, 2009; Butz, 2009; Pivello, 2011). Aboriginal peoples intentionally applied fire to maintain savannas, shrublands, and grasslands to enlarge forage areas for livestock and wild game, to drive game while hunting, to create safer human habitats and seasonal encampment areas, to promote production of edible herbs, roots and tubers, among other uses (Huffman, 2013). As societies interact with fire in the 21st-century and beyond, a new relationship to fire is clearly needed. It is therefore helpful to recognize that humans throughout the millennia have successfully coexisted with fire, continue to do so in many parts of the world, and that modern societies also can learn to coexist with fire (Fernandes et al., 2013; Moritz et al., 2014).

5.2. Future research needs

When considering fire management options under shifting fire and climatic regimes, it is important to consider the process of fire within the context of available fuels, fire weather, and overarching climate (Fig. 7). Knowledge of the relative importance of climate and fuels on fire dynamics for specific ecosystems is critical to understanding how fire regimes may shift under climatic change and how we might use or encourage wildland fire to increase ecosystem resilience in a warmer world. The rate of fuel accumulation following past wildfires is a key spatial determinant of whether or not sites have available fuel biomass and continuity to carry fire, while the timing of subsequent fires is generally dependent on antecedent drought and fire weather. In moist productive ecosystems, fuels are typically sufficient to carry fires, but frequently benign fire weather conditions and high fuel moistures are not conducive to burning. In contrast, arid ecosystems commonly display dry conditions that are favorable to burning, but fuel biomass and continuity are lacking to support fire growth (Krawchuk and Moritz, 2011; Pausas and Paula, 2012). Under a warming climate, some regions of the world will experience a higher drought incidence and longer fire seasons, which will lead to more wildfire area burned (Jolly et al., 2015). In ecosystems with historically long intervals between fires, prolonged drought and extreme fire weather, including strong winds, can effectively reduce burning thresholds and make more fuels available to burn across landscapes (Turner and Romme, 1994; Moritz et al., 2011). However, increasing aridity in some ecosystems (e.g., dry regions of the European Mediterranean and portions of western Australia) is projected to actually limit fire occurrence rather than increase it due to lower productivity and concomitant decreases in available fuels (Bradstock, 2010; Krawchuk and Moritz, 2011) (see Box 8).

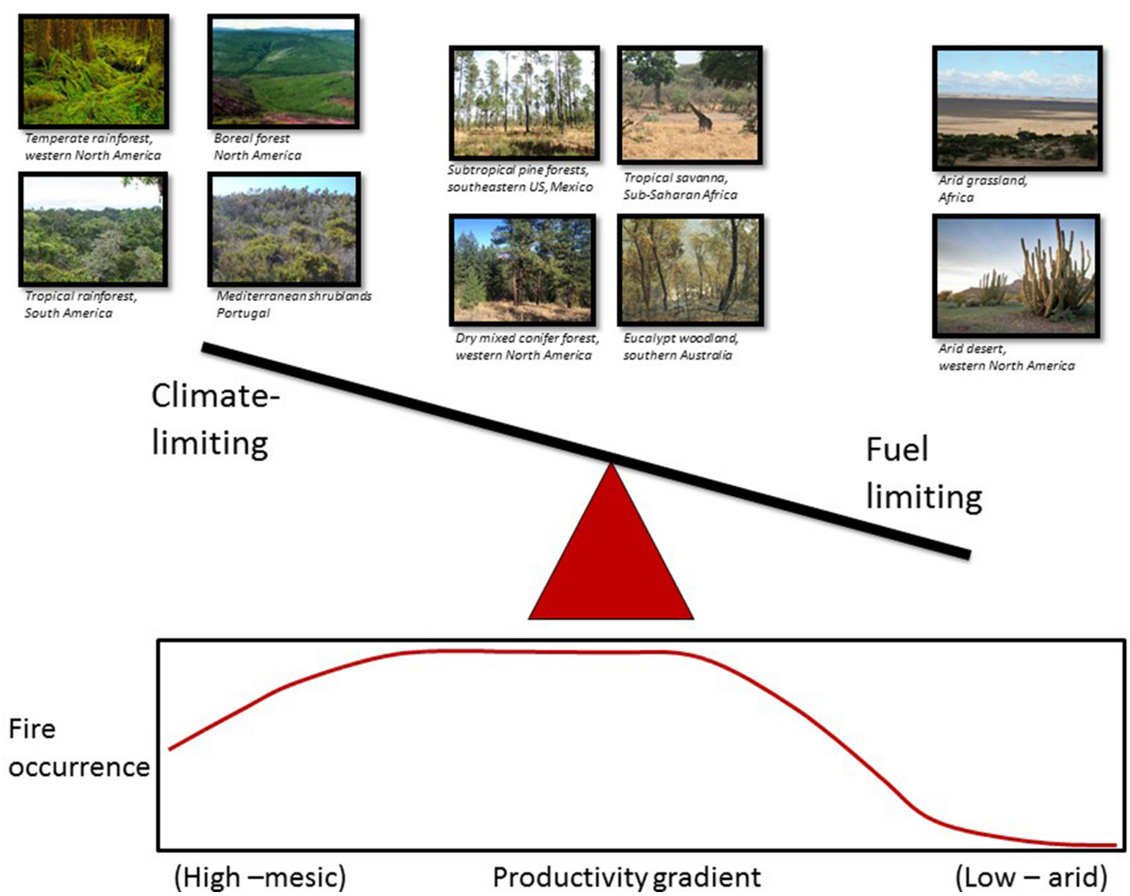

Fig. 7. Conceptual diagram of climate- and fuel-limits on fire regimes (after Krawchuk and Moritz (2011)). At the moist end of the productivity gradient, mesic climates support high biomass accumulations, and fires occur during fire weather events associated with prolonged drought and/or fire weather. At the dry end of the productivity gradient, fuels are almost always available to burn but fires are constrained by lack of fuel connectivity. Spanning the middle of the gradient are ecosystem types that are strongly influenced by fuels and climate; fires are generally frequent in these systems and strongly regulate spatial patterns and types of vegetation. (Temperate rainforest – Olympic National Park, boreal forest - Phil Higuera, University of Montana, Subtropical pine forest– Roger Ottmar, US Forest Service, tropical savanna, arid grassland, and tropical rainforest – Kim Romaine Bondi, Mediterranean shrubland - Luís Mário Ribeiro, ADAI-CEIF, Mixed conifer forest – Susan Prichard, Eucalypt woodland, Bushfire & Natural Hazards CRC http://www.bnhcrc.com.au, arid desert - https://www.nps.gov/orpi/index.htm.)

Box 8 Key areas of future research and development.

As more countries and regions consider managing unplanned wildfires for resource benefit, standardized metrics and study methods are needed to assess the duration and impact of past fires on subsequent fires. Managers and scientists need to better understand:

(1) the effects of repeated fires on burn severity, vegetation recovery, and post-fire vegetation and fuel succession;
(2) the period and conditions under which past wildfires and fuel reduction treatments served as effective barriers to subsequent fires;
(3) weather, fuel, and moisture conditions under which unplanned wildfires can be used (rather than suppressed) for resource benefit; and.
(4) how future climatic changes will alter fire-fire relationships, and the duration of those effects (see also Table 3).

We had originally aimed to provide equivalent information regarding fire regimes and reburns across the world's major bioregions. However, we discovered that there was a relative abundance of literature on the topic in some bioregions (Australian eucalypt forests and savannas, semi-arid forests and rangelands of western North America, and subtropical forests in the southeastern United States), and a relative scarcity in others (e.g., southeast Asia, African savannas, China, and Eurasian boreal forests). Given that wildfires may provide a positive feedback to global warming by releasing particulate black carbon and greenhouse gases to the atmosphere, and by increasing in severity and extent due to warming, it is vital that we better understand fire regime changes that are underway, and how fire management can create more resilient conditions. In a study of the relative contribution of regions to global fire emissions, van der Werf et al. (2010) found that tropical savannas and grasslands contributed 52% of emissions, followed by South America (15%), equatorial Asia (10%), boreal forests (9%) and Australia (7%). In a study of changing fire regimes in Siberian forests, Wallenius et al. (2011) stated that "[c]onsidering that Siberia contains the majority of the boreal forests of the world, it is paradoxical that its fire history is so little studied." A similar statement could be made of our knowledge of fire regimes and fire interactions in the world's greatest carbon stores. Knowledge gained from well-studied regions will hopefully serve as basis for future work in regions that have received less attention.

Acknowledgements

We thank Paulo Fernandes, Bob Gray, Phil Higuera, Bob Keane, Penny Morgan, Sean Parks, Morgan Varner and an anonymous reviewer for constructive comments on this manuscript. Discussions with James Dickinson, Richy Harrod, Nick Povak, and Brion Salter helped to inspire and frame this review. Rob Norheim assisted with figure creation. We acknowledge funding from the Joint Fire Science Program under project 14-1-02-30.

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