Review

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Ecological effects of large fires on US landscapes: benefit or catastrophe?[A]

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Abstract. The perception is that today's large fires are an ecological catastrophe because they burn vast areas with high intensities and severities. However, little is known of the ecological impacts of large fires on both historical and contemporary landscapes. The present paper presents a review of the current knowledge of the effects of large fires in the United States by important ecosystems written by regional experts. The ecosystems are (1) ponderosa pine–Douglas-fir, (2) sagebrush–grasslands, (3) piñon–juniper, (4) chaparral, (5) mixed-conifer, and (6) spruce–fir. This review found that large fires were common on most historical western US landscapes and they will continue to be common today with exceptions. Sagebrush ecosystems are currently experiencing larger, more severe, and more frequent large fires compared to historical conditions due to exotic cheatgrass invasions. Historical large fires in south-west ponderosa pine forest created a mixed severity mosaic dominated by non-lethal surface fires while today's large fires are mostly high severity crown fires. While large fires play an important role in landscape ecology for most regions, their importance is much less in the dry piñon–juniper forests and sagebrush–grasslands. Fire management must address the role of large fires in maintaining the health of many US fire-dominated ecosystems.

Additional keywords: fire effects, fire regimes, megafires.

Introduction

Large wildland fires pose an interesting dilemma for fire management in North America. Many politicians, members of the public, and government agency land managers have come to believe that large wildfires (fires $>10\,000$ ha) are an ecological disaster because they are perceived to burn vast areas with high fire intensities and burn severities (Brown 1985; Mutch *et al*. 1993; GAO 2002; Daniel *et al*. 2007). However, these same fires can return fire to deteriorating ecosystems where fires have been excluded for over 70 years, thereby improving ecosystem health and reducing fire hazard (Agee 1998). Little is known of the ecological impacts, both short- and long-term, of large fires on historical and contemporary landscapes (Moreno 1998). This uncertainty fuels the debate that surrounds both the causes and ecological consequences of large fires, which confuses the fire management community and the general public

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(Daniel *et al*. 2007). Just because a fire is large doesn't necessarily mean that it is unnatural or undesirable, unless, of course, it burns homes and threatens human life and property. Large fires may provide a unique opportunity for ecosystem restoration and fuel management treatments (van Wagtendonk 1995).

The present paper presents a review and summary of the current knowledge of the ecological effects of large fires in some major ecosystems of the United States. It is organised by those important biomes that typically experience large fires, with each section written by local experts. Regions include the Pacific North-west, southern California, Northern Rockies, the south-western United States, Great Basin, and Midwest with emphasis on the following ecosystems: ponderosa pine, Douglas-fir, sagebrush–grasslands, piñon–juniper, chaparral, Great Lakes conifer–hardwood forests, and spruce–fir. We attempt to address many important issues associated with large fires including (1) the scale of fire heterogeneity; (2) climate, vegetation, and topographical factors that may precondition large fire events; (3) departure of current large fire effects from those that occurred historically; (4) responses of key biota to large fires; and (5) broad generalities between and across major biomes. We also contrast effects of small fires and large fires in a spatial domain.

Background

Conventional wisdom holds that 70 years of fire exclusion, coupled with timber harvesting practices and livestock grazing, have tended to increase hazardous fuels that are now capable of fostering larger and more severe fires (Brown 1985; Arno and Brown 1991; Mutch *et al*. 1993; Kolb *et al*. 1998; Keane *et al*. 2002; Pinol *et al*. 2005). In the absence of fire, subsequent vegetation development will generally increase canopy and surface fuels, and these fuels will be more connected and continuously distributed across landscapes (Baker 1992; Ferry *et al*. 1995). It follows that increased fuels may burn in intense, large fires that could kill most plants, propagules, and animals (Agee 1998), and alter many soil and biophysical site conditions (Ryan 2002). Regeneration of diverse post-fire plant communities on severely burned areas may be delayed or prevented because of adverse site conditions in the burned area, and this could increase the frequency and severity of erosional events (McNabb and Swanson 1990). However, large fires were common on historical landscapes where the majority of burned area occurred during large fire events (Strauss *et al*. 1989; Malamud 2005; Cui and Perera 2006). In fact, the annual area burned by large fires during the pre-Euro-American settlement period is much larger than the annual area burned by large fires today, in some places by an order of magnitude (Arno 1980; Barrett *et al*. 1997). Some studies suggest that the area of unburned patches within a large fire perimeter may actually increase with fire size (Eberhart and Woodward 1987), and that fire exclusion has had little effect on large fire dynamics and increased fuels won't promote large fires (Johnson *et al*. 2001; Schoenberg *et al*. 2003*a*; Bridge *et al*. 2005).

Many factors can precondition the regional landscape to experience large fires. Most large fires in the contiguous US occur in years of moderate to severe drought (Swetnam and Betancourt 1998; Heyerdahl *et al*. 2001; Baker 2003; Swetnam and Baisan 2003). This is especially true in topographically complex landscapes where, in normal climate years, subalpine to alpine ecosystems may stay moist throughout the year and thereby retard fire spread (Wadleigh and Jenkins 1996). Most area burned in large fires occurs during short-term wind events and under very hot and dry (low relative humidity) conditions (Cohen and Miller 1978; Schoennagel *et al*. 2004). Some evidence suggests that the amount and contagion of dead and live vegetation (fuels) on landscapes can also contribute to large fires (Gardner *et al*. 1997; Turner *et al*. 1998), but this may not be true in all ecosystems and geographical regions owing to the interactions of high winds, spotting, and large fire behaviour (Bessie and Johnson 1995; Schmoldt *et al*. 1999). Lightning starts most of today's large fires, especially in the western US, while human-ignited fires are also important in the eastern and south-eastern US, and California (Stephens 2005). Ignition dynamics (number, location, and source) are critically important in climate–fuel–fire interactions because without ignitions, large fires are impossible, even in extreme drought years (Ricotta *et al*. 1999).

The effects of large fires on ecosystems can be distinctly different from small fires. Large fires can create large burn patches (Agee 1998) that could slow wind and mammal dispersal of seed from unburned edges, thereby delaying common vegetation development processes. Large fires may be more severe because more fuel may be consumed and the additional heat generated can kill more aboveground vegetation and the deeper heat pulse will kill more belowground biota (Ryan 2002). Large fires are more difficult and costly to fight, and the subsequent effects of those fires may be economically more damaging than small fires (Butry 2001; Calkin *et al*. 2005; Daniel *et al*. 2007). Last, large fires will tend to affect the most people and destroy the most property because of their sheer size. However, severe large fires may be required on many landscapes to emulate historical fire dynamics and sustain healthy ecosystems (Fulé *et al*. 2004). As large fires were common on most historical US landscapes, it follows that there are many plant and animal species that depend on the severity pattern created by large burns (Habeck and Mutch 1973; Agee 1993; Hutto 1995; Debano *et al*. 1998).

Regional effects of large fires

Pacific North-west

Large and severe fires are historically characteristic of Pacific North-west forests. This section focusses on the forests west of the Cascade Mountains called the Douglas-fir region, which are influenced by a maritime climate that is wet and relatively warm compared with inland regions (Franklin and Dyrness 1973). Douglas-fir (*Pseudotsuga menziesii*) tends to be a dominant species, except along a narrow coastal strip dominated by Sitka spruce (*Picea sitchensis*) and western hemlock (*Tsuga heterophylla*), and at subalpine elevations dominated by mountain hemlock (*Tsuga mertensiana*) and subalpine fir (*Abies lasiocarpa*).

The fire regime

Although one might expect that moist forests (sometimes >2500 mm annual precipitation) would rarely burn, a summer dry season is characteristic of this region, and annual precipitation drops to less than 500 mm in the southern portion of the region. The fire regime transitions from one of high severity in

Table 1. Examples of fire return intervals for Pacific North-westforests
Sites are listed from north to south in the region

Forest type/location	Fire return Interval (years)	Source
Western Washington	230	Fahnestock and Agee (1983)
Mount Rainier, WA	434	Hemstrom and Franklin (1982)
Bull Run, OR	350	Agee and Krusemark (2001)
Central Oregon Cascades	95–145	Morrison and Swanson (1990)
Central Oregon Cascades	100	Teensma (1987)
Siskiyou Mountains	15–75	Agee (1991)
Siskiyou Mountains	12–19	Taylor and Skinner (1998)
Klamath Mountains	11.5–16.5	Taylor and Skinner (2003)

the northern Cascades to a mixed severity in the south as fire return intervals shorten. To the north, climate is the major driver of fire spread (Agee and Huff 1987), and large fires depend on previous winter drought, deep summer drought, ignition (lightning), and a dry, strong east wind (Agee 1991). The gradient of fire return intervals shown by field studies (Table 1) is also supported by a fire cycle model based on climate that shows approximately an order of magnitude increase in fire frequency from north to south in the region (Agee 1991). Regional synchronicity appears in these fire regimes, with more fire from 1400s to 1650, less fire from 1650 to 1800, and again more fire from 1801 to 1925 (Weisberg and Swanson 2003).

Large fires of 100 000–400 000 ha have occurred historically in the region (Fonda and Bliss 1969; Henderson *et al*. 1989; State of Oregon 1997). The most flammable conditions in these forests are in early successional stages (Agee and Huff 1987), so reburns are possible (Isaac 1940). Fonda and Bliss (1969) identified a series of large fires that covered the entire eastern Olympic Mountains of Washington, and Henderson *et al*. (1989) dated them to 1700 or 1701 and estimated their size at over 400 000 ha. Subsequently, a large subduction earthquake (9.0) on the Washington coast was dated to January 1700 (Atwater *et al*. 2005), which may have quake-thrown trees and created substantial dead fuel in a forest normally with few flashy dead fuels. Large fires in the Oregon Coast Range (100 000 ha+) occurred in 1848, 1853, and 1868 (State of Oregon 1997). The 175 000 ha Yacolt fire burned across south-western Washington in 1902. This was followed by a large fire in the northern coast range of Oregon in 1933, ignited by a logging crew. The 96 000 ha Tillamook fire had a spectacular blow-up, burning over 80 000 ha in just 20 h. Major portions of this fire reburned in 1939, 1945, and 1951 owing to snags catching fire and profuse cover of bracken fern, which is very flammable when cured (Isaac 1940). In 2002, the 200 000 ha Biscuit fire in south-western Oregon burned for months across many successional stages (Raymond and Peterson 2005).

Forest development patterns

In northern portions of the region with high-severity fire regimes, fires tend to kill all the trees, either because the fire was a crown fire or the fire was hot enough to scorch the foliage of tall, old-growth trees (>70 m) (Agee 1993). At least two patterns of succession have been documented: (1) tree regeneration is immediate; and (2) tree regeneration is delayed. These patterns may be due to the large scale of the event coupled with the availability of seed sources. In the Olympic Mountains, Huff (1995) found tree regeneration was rapid after fire burned 500-, 180-, 100-, 20-, and 1–3-year old stands, and that Douglas-fir establishment initially colonised the open landscapes created by the fire. After canopy closure, only western hemlock was able to establish itself, suggesting that the historical dominance of Douglas-fir over the region is testament to the importance of fire across the region. With most tree species in this region living 400–1000 years (Franklin and Dyrness 1973), only 1–2 fires per millennium are sufficient to maintain the dominance of Douglas-fir. Other evidence points to delays in forest regeneration, possibly due to lack of seed sources, shrub competition, or reburns (Franklin and Hemstrom 1981).

In the southern mixed-severity area of the Pacific Northwest, successional patterns are more complex (Weisberg 2004). Lower-severity fires historically created variability in tree sizes and a greater component of shade-tolerant tree species. Although Douglas-fir remains a dominant species, the diversity of other species is much higher than to the north. High burn severity patches in the landscape (often upper thirds of slopes and drier aspects; Taylor and Skinner 1998) typically regenerate to sprouting hardwood species or serotinous-coned conifers such as knobcone pine (*Pinus attenuata*). Lower-severity patches typically have several age classes of Douglas-fir and associated regeneration can include Douglas-fir, hardwoods such as Pacific madrone (*Arbutus menziesii*) and canyon live oak (*Quercus chrysolepis*), and shade-tolerant conifers such as white fir (*Abies concolor*) or grand fir (*Abies grandis*). Recent large-scale fires, such as the 1987 fires and Biscuit fire, have allowed the mixed-severity fire regime to again be emplaced on the landscape, thereby enhancing biodiversity (Martin and Sapsis 1991).

Managing landscapes

Even with Douglas-fir as the primary tree species, the landscape effects of fire were different between the mixed-severity and high-severity fire regimes. Patch edge was maximised in the mixed-severity fire regimes compared with the high-severity fire regime in the north or low-severity fire regimes like ponderosa pine to the east (Agee 1998), owing to relatively small patches of differing fire severity. The patchy forest landscape created by mixed-severity fire regimes has maintained high suitability for northern spotted owls (*Strix caurina* var. *occidentalis*) (Franklin

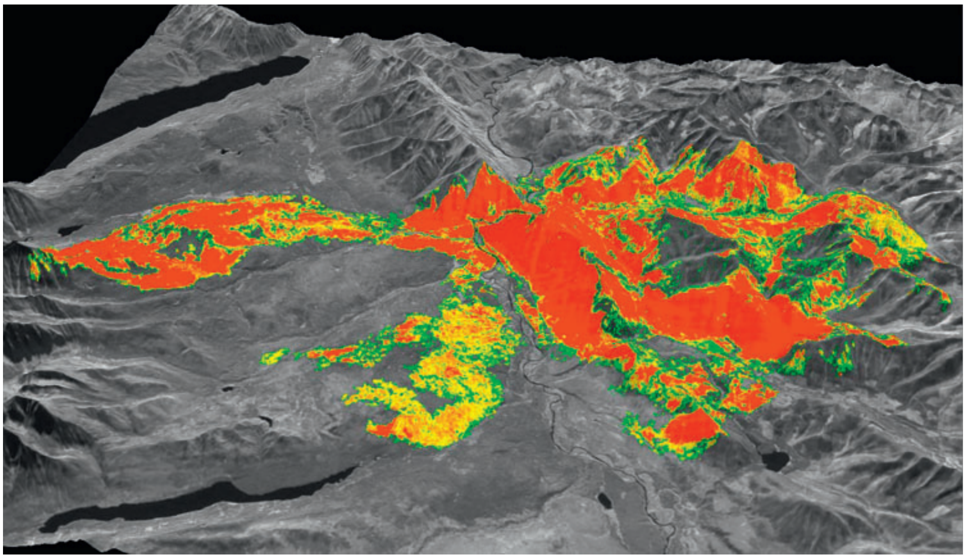

Fig. 1. A map of burn severity for the Moose fire near Glacier National Park, Montana, USA. Highest severities are in red while lowest are in greens and yellows.

et al. 2000, 2002), preferable to either more homogeneous old growth or large patches of other vegetation types. Maintaining this patchiness using prescribed fire and wildland fire use will create major challenges for land managers in coming decades, along with fragmentation from private forest land management (Spies *et al*. 1994).

To the north, large stand-replacement patches of past centuries grew into old-growth patches that form the basic concept of the North-west Forest Plan with its late-successional reserves (FEMAT 1993). Mean old growth in the Oregon Coast Range province (2 200 000 ha) appears to have averaged ~47%, but as the scale became more local (40 000 ha), old growth percentages ranged from 0 to 100% (Wimberly *et al*. 2000). A policy of fire exclusion might work as an ecosystem management strategy in these forests as northern spotted owls prefer large blocks of old growth for nesting and roosting. However, when conditions are right, large fires are likely to occur and be relatively immune from firefighting efforts. Neither allowing all fires to burn nor suppressing all fires will be successful ecosystem management strategies across this complex region.

Northern Rockies

The topographic complexity of most northern Rocky Mountain landscapes, along with the convergence of maritime and continental climates, create diverse mosaics of vegetation communities and structures that are ultimately shaped by complex and dynamic fire regimes (Wellner 1970; Habeck and Mutch 1973; Arno 1980; Philpot 1990). The juxtaposition of grasslands and xeric forests (e.g. ponderosa pine) with montane forests (e.g. Douglas-fir–western larch (*Larix occidentalis*)) and nearby subalpine areas (lodgepole pine–subalpine fir) requires that the entire landscape must be sufficiently dry to support the large fires that occurred in the past, and this occurs primarily during years of prolonged drought (Arno 1980; Gruell 1983; Barrett *et al*. 1991). These large fires were mostly started from lightning or Native American burning (Barrett and Arno 1982) during times of widespread regional drought. Kitzberger *et al*. (2007) found warm, dry springs are often precursors to large fire years.

Large fires were common on the northern Rocky Mountain landscape before 1980. Barrett *et al*. (1997) found 35 large fire episodes occurred between 1540 and 1940 with most fire dates recorded across large regions of the interior Columbia River Basin. Most of these fire dates were recorded in xeric ecosystems with high fire frequency (e.g. ponderosa pine) and they burned 5–15% of the area. Barrett *et al*. (1997) also suggest that 'major fires prior to 1900 burned more area than any fire years since'. The fire years of 1910 and 1889 appear to be the largest in recent history (Koch 1942; Cohen and Miller 1978), but the fire years of 1869, 1856, 1846, 1833, and 1778 were also impressive in extent (Barrett *et al*. 1997). The main difference between historical and contemporary large fires may be that today's large fires may burn areas that have deep duff layers, heavy woody fuel loadings, and thick canopies due to decades of fire exclusion that will certainly result in severe fires (Kolb *et al*. 1998), but the extent of these conditions is unknown before the fire exclusion era. Large fires have been increasing in recent years with 1988 and most years since 2000 having at least one large fire (Schoennagel *et al*. 2004).

Many large Rocky Mountain fires leave severity patterns that are quite diverse in shape and size, which greatly affect subsequent post-fire ecological dynamics (Schoennagel *et al*. 2004; Baker *et al*. 2007; Lentile *et al*. 2007; Schoennagel *et al*. 2008) (Fig. 1). These large patches and complex mosaics facilitated the regeneration and survival of many plant and animal species. Large patches created from regional fires, for example, ensured the continued presence of western larch because the mature trees had thick bark and high crowns that increased survival after high-severity fires so they could disperse abundant seed in areas where all other trees were dead and seed sources were distant (Schmidt

Table 2. A comparison of burn severity and patch metrics for small fires (<3300 ha) and large fires (>10 000 ha)

The mean for burn severity and standard deviation for patch metrics are given in parentheses. *P*-values in bold indicate significance at 0.05 level using *t*-test statistics

Attribute	Small fires	Large fires	*P* value
Number of fires	25	11	–
Burn severity			
Unburned	0.21 (0.0139)	0.15 (0.0047)	0.10465
Low	0.25 (0.0086)	0.21 (0.0052)	0.27295
Moderate low	0.18 (0.0091)	0.18 (0.0013)	0.27295
Moderate high	0.19 (0.0091)	0.20 (0.0066)	0.70221
High	0.16 (0.0162)	0.25 (0.0113)	0.05251
Patch metrics			
Patch density (patches 100 ha^{-1})	91.2 (24.5)	66.8 (5.78)	**0.00272**
Largest patch index (dimensionless)	16.9 (9.25)	10.7 (7.48)	0.05812
Landscape shape index (dimensionless)	23.1 (9.59)	103.2 (45.44)	**0.00001**
Shape index (dimensionless)	5.08 (2.97)	8.85 (4.13)	**0.00387**
Fractal index (dimensionless)	1.06 (0.01)	1.06 (0.01)	0.35577
Edge density (m ha^{-1})	306.4 (49.8)	247.5 (13.3)	**0.00052**
Contagion (0–100 index)	34.0 (5.05)	35.4 (2.07)	0.38853
Interspersion-juxtaposition index (0–100)	64.1 (5.90)	61.8 (1.99)	0.18497

et al. 1976; Davis 1980; Barrett *et al.* 1991). In another example, it appears the intricate pattern of fire severity from the large Yellowstone fires in 1988 heavily influenced subsequent lodgepole pine regeneration (Turner *et al.* 2003), aspen and ungulate dynamics (Romme *et al.* 1995), and bird distributions dependent on stand-replacement fires (Hutto 1995). This unique pattern will also affect future fire dynamics and landscape structure as successional development advances at disparate rates depending on the availability of propagules (e.g. level of pine serotiny, survival of underground rhizomes, tubers, and seed, and sprouting potential), soil fertility, and post-fire climates (Turner *et al.* 1998, 1999).

Conventional wisdom holds that large fires tend to be more severe and therefore are undesirable (Laverty and Williams 2000; GAO 2002). To assess whether severity patterns from contemporary large fires were significantly different from small fires, we created digital burn severity layers for 11 large fires (>10 500 ha; see Fig. 1 for an example) and 25 small fires (<3300 ha) that occurred in the northern Rocky Mountains within the last 10 years (Table 2). These digital maps were generated from a classification of Landsat TM (Thematic Mapper) satellite imagery by deriving the differenced Normalized Burn Ratio, and then linking to the Composite Burn Index from ~1100 plots to categorise burn severity class (Key and Benson 2005). We then used *FRAGSTATS* (McGarigal and Marks 1995) to calculate important landscape metrics for each fire and summarised these metrics across large and small fires using statistical testing.

We found that a larger proportion of the burned landscape was, indeed, in the highest-severity class for large fires compared with that for small fires (high severity averaged 25% of burned area for large fires and 19% for small fires), but this difference was not significant ($P > 0.05$, Table 2) and quite low (<25% of the area) (Lentile *et al.* 2007). In fact, there were few statistically significant differences in proportions of burned area for small and large fires across all burn severity classes. However, large fires were different from small fires in terms of landscape structure (Romme *et al.* 1998). Large fires tended to have larger patches (low patch density) that were more regular in shape (high landscape shape index) and had less edge than smaller fires (Table 2). The fractal dimension, contagion, and interspersion, however, were nearly identical across small and large fires, indicating that although the patches were larger, the patches tended to be adjacent to the lower-severity classes, creating diverse landscape mosaics.

There appears to be no indication that the frequency and severity of large fires in the northern Rocky Mountains have changed from historical fire regimes. This is partly because there has been insufficient time (only 70+ years) to evaluate any temporal changes in the long fire return interval ecosystems after the advent of fire exclusion policies. It is also because fighting fires during the weather and drought conditions that foster large fires tends to be difficult, so fire suppression probably has the least effect on reducing the size of large fires. Additionally, historical spatial distributions of fire severity in large fires are unknown because of the lack of spatially explicit legacy field data, so it is difficult to compare today's fire severity patterns with historical patterns. Presettlement large fires tended to leave fire-scarred trees on the landscape, indicating low fire severities, but it is difficult to quantify the historical patch distribution of burn severity because evidence of most high-severity burns is often lost. And last, large fire severity mosaics tend to be complex (Fig. 1, Table 2) so it would take measurements from many large fires to compute any statistically significant change in severity and frequency (Pratt *et al.* 2006).

Southern California

The Cedar Fire burned 110 600 ha of San Diego County shrublands in October 2003 and was the largest fire in the state since record-keeping began ~100 years ago (Fig. 2). Many hailed

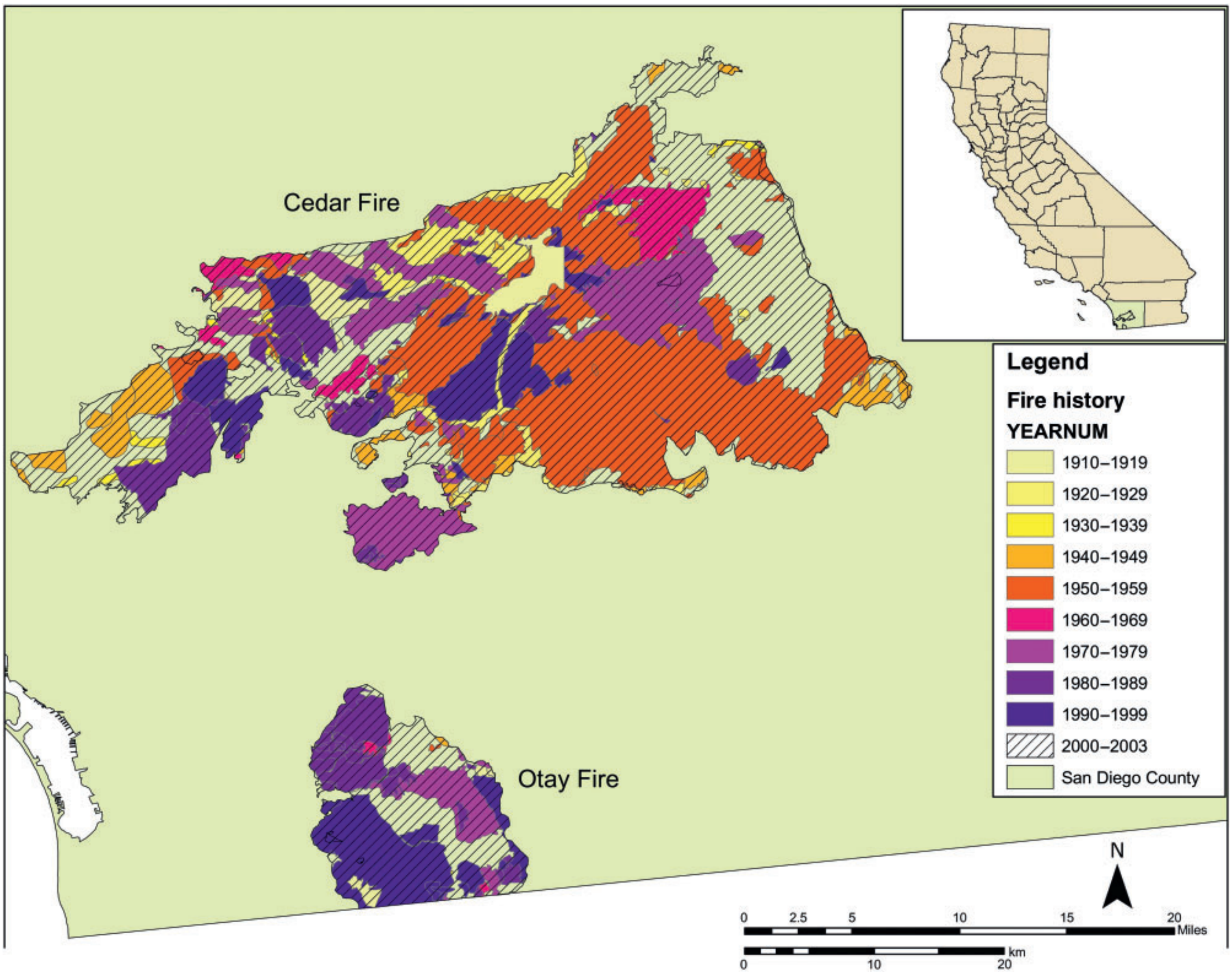

Fig. 2. The fire history of the area within the Cedar and Otay fires showing the great diversity of stand ages and sizes. YEARNUM, span of dates.

this as an anomalous event whose root cause was past land management practice coupled with an unusually long drought, and proclaimed it would have disastrous repercussions for the biota. However, close examination of historical records and scientific studies fails to support these contentions.

The primary cause of the final size for the Cedar Fire was the severe weather event that coincided with an intentional human ignition in a remote portion of the backcountry. Like most large wildfires in the western US, severe winds were a critical factor in the rapid spread of the fire. In southern California, large wind-driven fires are a common feature of the landscape because of the annual foehn winds known locally as Santa Ana winds (Keeley and Fotheringham 2006). These occur every year and there may be anywhere from 10 to 50 or more such days a year. The worst of these are the autumn winds that come at the end of the annual drought, which is typically 4–6 months. These winds commonly exceed 100 km h^{-1} with relative humidity less than 10%, and they are capable of spreading fires more than 10 000 ha in an hour.

Many such fires have burned in the 20th century, although none reached the ultimate size of the Cedar Fire. Nonetheless, if one considers a longer-term view, we see that larger fires have occurred historically. For example, at the beginning of a 3-day Santa Ana wind event in the last week of September 1889, the Santiago Canyon Fire ignited in northern Orange County and burned into San Diego and Riverside counties, and likely exceeded 125 000 ha, making it the largest fire in California's history (J. E. Keeley and P. H. Zedler, unpubl. data). Thus, the Cedar Fire should be considered a 100-year event as opposed to an anomalous event. Other fire records such as marine sediments in the Santa Barbara Channel indicate a repeating cycle of large fire events for the region that extend the record back 500 years and indicate no change in the periodicity of such events (Mensing *et al*. 1999). The primary difference between the 1889 and 2003 events is the damage done to local communities. The 1889 Santiago Canyon Fire did not destroy any homes or kill anyone, whereas the 2003 Cedar Fire consumed 2232 homes and 14 lives were lost (Keeley and Fotheringham 2006). Thus, the recent event was far more catastrophic in human terms, owing primarily to the 100-fold increase in population density between 1889 and 2003 (http://www.census.gov/population/cencounts, accessed 12 June 2007).

Past fire management practices, in particular fire exclusion and lack of effective fuels management, have been suggested by scientists, managers and legislators in California as a key reason for the size of the Cedar Fire. Indeed, it has long been argued that large fire events such as this could be prevented by creation of landscape mosaics of fuels of different ages (Philpot 1974; Minnich and Dezzani 1991). However, when these hypotheses have been put to the test, they have not been supported. Schoenberg *et al*. (2003*b*), for example, found that after the first two decades of post-fire recovery, there was no change in the risk of burning for Los Angeles County shrublands. Moritz *et al*. (2004) also analysed the role of fuels throughout the southern and central coastal region and found that on 90% of the landscape, fuel age was of minimal importance to fire hazard.

The Cedar Fire is an excellent case study of the ineffectiveness of fuel mosaics in stopping these wind-driven fires (Fig. 2). Prior to that fire, this landscape was a complex mosaic of fuel ages created by both wildfires and prescription burning and many patches were 20 years of age or less, yet the fire rapidly burned through age classes as young as 3 years. Maintaining a landscape dominated by even younger age classes is logistically problematical. Although 20-year old chaparral will burn readily when pushed by strong Santa Ana winds, it is difficult to reburn chaparral younger than this under prescription burning conditions of wind speed and relative humidity (Keeley 2002). Perhaps the clearest demonstration that young age class fuels will not stop these fires is the 2007 fire event that followed closely on the 2003 fire event. In just San Diego County alone, the Witch and Harris fires consumed ∼100 000 ha and nearly one-third of that was reburning of 2003 Cedar and Otay fire scars (J. E. Keeley and P. H. Zedler, unpubl. data).

These observations call into question the value of landscape-scale prescription burning or other fuel modifications on these shrubland landscapes. One justification for continuing fuel modification work is that young fuels will sometimes act as barriers to fire spread under much more moderate weather conditions. However, such fires are seldom a threat to property or lives and thus a huge question mark remains as to whether they can be justified based on the cost/benefit ratio (Keeley 2005). Of course, regardless of weather conditions, fuel-modified zones do reduce fire intensity and thus increase the defensible space for fire suppression activities. In a region experiencing urban sprawl, where firefighting forces commonly are deployed in defensible postures during severe Santa Ana wind-driven fires, the most strategic place for fuel modifications is directly at the wildland–urban interface.

Last, media and other sources contended that the Cedar Fire was more intense than typical fires and, owing to the intensity and the massive size of this fire, the recovery of these ecosystems was threatened. Crown-fire ecosystems, such as chaparral, were not affected by the size of this fire. This is because these ecosystems have endogenous mechanisms for recovery that include resprouting from basal burls and long-lived dormant seed banks that are stimulated to germinate by fire. Colonisation is not important in their recovery and thus, unlike many forest ecosystems, the landscape pattern of burning is largely unimportant in post-fire recovery (Keeley *et al*. 2005). As for fire intensity, there is no evidence that fire intensity or fire severity was greater in this fire than in many other chaparral fires, now or historically. In addition, these ecosystems are highly resilient to a range of fire severities (Keeley *et al.*, in press).

South-western US

Large fires were historically common in the south-western USA (Arizona, New Mexico, and southern Utah and Colorado); a compilation of 63 fire history studies across the region showed that as many as two-thirds of the forests burned in synchrony in dry years such as 1748, with multiple ignitions in this lightning-prone region (Swetnam and Baisan 2003). However, the critical difference between historical and modern fires is that past fires burned largely on the surface, whereas large modern fires such as the Rodeo–Chediski fire of 2002 burn primarily as crown fires.

The predominant forest type of the South-west is ponderosa pine, generally as a monospecific forest type or together with Gambel oak. South-western ponderosa pine forests were historically characterised by frequent surface fire regimes (Swetnam and Baisan 2003). Disruption of the fire disturbance regime by livestock grazing, logging, and fire suppression triggered extensive tree regeneration (Cooper 1961), leading to dense forests that support canopy burning (Covington and Moore 1994; Fiedler *et al*. 2002). Recently, Westerling *et al*. (2006) argued that climate warming was directly associated with the increase in size and severity of western wildfire, but singled out the south-western ponderosa pine forest as an example of the destructive convergence of warmer climate with historically unprecedented fuel levels.

In elevation zones surrounding the ponderosa pine, some south-western forests were naturally characterised by severe fire. Across an elevational gradient from ponderosa pine through mixed conifer (ponderosa pine, Douglas-fir, white fir) to spruce–fir and aspen forest, historical fire regimes changed from frequent to infrequent surface fires, then to stand-replacing fires (Fulé *et al*. 2003). Since 1880, none of the high-elevation sites burned, increasing the continuity of dense fuels across the landscape (White and Vankat 1993). At low elevations in the South-west, woodlands of piñon pines and junipers included varying fire regimes (Romme *et al*. 2003), but at least some were historically characterised by stand-replacing fires (Floyd *et al*. 2004).

Large crown fires in south-western ponderosa pine forests have led to long-lasting changes in species dominance and habitats. A survey of severely burned landscapes found that only approximately a third recovered to support pine forests within the historical range of variability of forest structure (Savage and Mast 2005). Another third became 'hyper-dense' forests, susceptible to crown fire again, and the remainder appeared to have converted to alternative stable states dominated by grass or brush (Savage and Mast 2005). In mixed pine–oak forests, severe burning favoured regeneration by sprouting oaks at the expense of pines (Barton 2002). Invasive exotic species have capitalised on many large burns (Crawford *et al*. 2001) although invasives were notably limited following a severe fire in Grand Canyon National Park, perhaps because their seeds were not present in the relatively undisturbed, never-logged forest (Huisinga *et al*. 2005).

The Rodeo–Chediski fire complex in central Arizona in 2002, covering ∼189 000 ha, was by an order of magnitude the largest severe fire to date in the South-west US. This landscape-scale fire may represent the largest possible scales of fire in the region because it was essentially unaffected by almost all the attempted control actions and stopped only when it burned into adjacent dry grasslands. Yet even under the conditions of extraordinary drought and sustained strong winds that supported the Rodeo–Chediski fire, those portions of the landscape with recent (<11 year old) treatments of prescribed fire or cutting and burning displayed primarily surface-fire behaviour (Finney *et al*. 2005; Strom 2005). These patches of living forest in a matrix of fire-killed vegetation are somewhat unique to the south-west US and provide three useful lessons for fire ecologists: (1) treated sites not only survived the worst-ever fire conditions but also affected landscape-scale severe fire behaviour by protecting adjacent untreated vegetation on the downwind side (Finney *et al*. 2005); (2) untreated forests suffered high mortality and vegetation simulation modelling projected long-term (100+ year) dominance by oaks and other non-pine species, in contrast to continued pine dominance of treated sites (Strom and Fulé 2007); and (3) the survival of treated forests is strong empirical evidence of the effectiveness of ecological restoration and hazard fuel reduction efforts. The utility of such treatments in reducing fire severity had previously been shown in fire behaviour simulations (e.g. Fulé *et al*. 2001) or stand-level post-fire comparisons (e.g. Pollet and Omi 2002).

Sagebrush ecosystems

Sagebrush ecosystems comprise the most widespread shrublands in western North America (McArthur and Stevens 2004); consequently, the variation and change in associated fire regimes merit consideration. Sagebrush ecosystems include all landscapes dominated by one or more sagebrush taxa (most commonly big sagebrush: *Artemisia tridentata*) and cover a wide range in elevation (600–3000 m) sharing ecotones with a variety of grassland, shrubland, woodland, and forest types (Wright and Bailey 1982; McArthur and Stevens 2004). Abundance and diversity of associated plant species increase with increasing productivity potential and are correlated with soils and climate variables (Beetle 1960; Wright and Bailey 1982; McArthur and Stevens 2004).

Fire is the dominant disturbance associated with historical sagebrush ecosystems (Wright and Bailey 1982). Post-fire regeneration of sagebrush is from seed with some exceptions (e.g. silver sagebrush sprouts from roots; Wright *et al*. 1979). Because sagebrush seeds lack long-distance dispersal (Young and Evans 1989; Chambers 2000) and do not persist in soil seed banks (McDonough and Harniss 1974; Meyer and Monsen 1992), recovery rates are long when fire intensity and size result in large uniform burns depleted of viable seed (Welch 2005). Without fire, many sagebrush landscapes are subject to tree encroachment and dominance (Tausch 1999*a*; Miller *et al*. 2000; Heyerdahl *et al*. 2006) whereas others, particularly those found on drier sites or at greater distances from trees, persist as treeless shrublands in the absence of fire (Baker and Shinneman 2004).

Historical fire regimes

Favourable years for large fires in sagebrush ecosystems occur when hot dry summers follow wet spring conditions, indicating the importance of fine fuel (i.e. grass) accumulation and conditioning (Wright *et al*. 1979). Historical fire-free intervals are believed to have been shorter on more productive *v.* less productive landscapes owing to greater fuel production. Proximity to fire-prone forest types (i.e. ponderosa pine and dry mixed-conifer) likely contributed to relatively short fire-free intervals on some landscapes.

Historical fire frequency has been estimated for a few sagebrush-dominated landscapes using evidence collected from fire-scarred trees located near the forest–shrubland ecotone. This method has produced estimates of historical mean fire interval that range from 12 (Miller and Rose 1999) to 40 years (Houston 1973; Arno and Gruell 1983; Heyerdahl *et al*. 2006). Studies of post-fire succession suggest that mean fire-free periods of 30+ years are often needed for sagebrush recovery on productive sites (Harniss and Murray 1973; Humphrey 1984; Watts and Wambolt 1996; Wambolt *et al*. 1999, 2001) and much longer on less productive sites (Wright and Bailey 1982; West and Yorks 2002; Welch 2005). Existing evidence supports historical mean fire intervals of 35 to 80 years for productive sagebrush-dominated landscapes and 100 to 200+ years for less productive sites. By applying various correction factors, Baker (2006) argues for historic fire intervals (fire rotations) that are approximately two to three times longer than these estimates.

Historical fire spatial patterns in sagebrush ecosystems must be inferred owing to difficulty in obtaining direct perimeter measures. Habitat requirements for big sagebrush-dependent fauna, such as greater sage grouse (*Centrocercus urophasianus*) and the non-migratory pigmy rabbit (*Sylvilagus idahoensis*), suggest that large fires were rare (Crawford *et al*. 2004; Welch 2005). Conversely, precursor climate–fuel cycles favourable for extensive fire spread are common today and presumably were at least as common historically. The potential for periodic large fires was greatest on productive, contiguous expanses of sagebrush grasslands such as those that occurred from northern Nevada and eastern Oregon to western Wyoming. Short-term sagebrush recovery after large fires would have depended on seedling establishment from unburned, viable seed left in the seed bank or short-range seed dispersal from unburned islands and the burn perimeter. Both processes operate after modern fires and must have been important historically.

Modern fire regimes

Fire frequency on sagebrush landscapes has changed relative to that of pre-Euro-American settlement owing to the combined impacts of livestock grazing, fire suppression, exotic species introductions, and altered anthropogenic ignition patterns (Wright and Bailey 1982). Fire on many landscapes has been absent for 80–140 years. Encroachment by conifers is widespread and has resulted in partial to complete conversion to woodlands on favourable sites (Tausch 1999*b*; Miller *et al*. 2000). Where encroachment is advanced, fuel loads are greatly increased and fuel type and structure are significantly altered. Woodland expansion also reduces landscape-scale structural diversity (Tausch and Nowak 2000). Ultimately, woodland expansion preconditions landscapes for more extreme fire behaviour and larger fires. Large fires during the past 20 years

Fig. 3. A large stand replacement fire in sagebrush grasslands facilitated by abundant fine fuels due to cheatgrass invasions (photo by Stan Kitchen).

in piñon–juniper woodland–sagebrush shrubland mosaics of Nevada and Utah confirm these expectations.

Fire intervals for many sagebrush ecosystems of low to moderate productivity are perhaps 10 to 20 times shorter than what is estimated for presettlement conditions (Whisenant 1990; Peters and Bunting 1994). This increase in fire is due primarily to the spread and dominance of the Eurasian annual cheatgrass (*Bromus tectorum*), in sagebrush understoreys during the last century. This cool-season grass cures earlier than perennial grasses, effectively lengthening the fire season. The dense, somewhat continuous fine fuel matrix facilitates rapid fire spread and larger fires (Fig. 3). Cheatgrass competitively excludes perennial seedlings, effectively arresting post-fire succession. One effect of this cheatgrass-truncated succession is that multiyear series of adjacent smaller fires have similar ecological impacts to single large fires. Consequently, large tracts of sagebrush ecosystem have been converted to annual grasslands prone to short fire-free intervals and large fires (Whisenant 1990; Peters and Bunting 1994). Unburned sagebrush shrublands adjacent to these annual grasslands remain at high risk.

Although fire size in historic sagebrush landscapes is poorly understood, it is generally accepted that recent large fires fuelled by woodland-induced homogenisation of landscapes and cheatgrass-dominated understoreys are outside the historical range of variation. These changes in fire regime and vegetation–fuel structure impact vast areas in the semiarid western United States and are felt at all trophic levels. Effects are particularly harmful on landscapes where post-fire recovery is slowest. We can expect the trend for larger, more damaging fires in sagebrush ecosystems to continue until aberrations in fuel conditions that drive fire are corrected.

Piñon and juniper ecosystems

Piñon and juniper woodlands occupy ~19 million ha in the Intermountain West (Miller and Tausch 2001). Throughout this region, these trees are associated with sagebrush-steppe, where they frequently form distinct woodlands adjacent to sagebrush communities or grow at varying densities within shrub–steppe communities (West 1999; Miller and Tausch 2001; Weisberg *et al*. 2007). Numerous studies have documented expansion of these woodlands into shrub–steppe communities since the late 1800s (Cottam and Stewart 1940; Tausch *et al*. 1981; Miller and Rose 1995; Knapp and Soulé 1996; Gedney *et al*. 1999; Weisberg *et al*. 2007; Miller *et al*. 2008). Presently, many of these shrub–steppe communities are in various states of woodland succession (Miller *et al*. 2008). In past literature, woodlands that have recently encroached into shrub–steppe are often not distinguished from historic (persistent) piñon and juniper woodlands. In the central and northern Great Basin in Utah, Nevada, and Oregon, less than 10% of trees greater than 1 m tall were over 150 years old (Johnson and Miller 2006; Miller *et al*. 2008). However, relatively extensive old woodlands occupy portions of the Colorado Plateau (Floyd *et al*. 2000, 2008; Eisenhart 2004), although the proportion has not been defined, and the Mazama Ecological Province in central Oregon (Miller *et al*. 2005). In the current section, we are addressing what we define as persistent woodlands (Romme *et al*. 2007) as 'those found where site conditions (soils and climate) and disturbance regimes are inherently favorable for piñon and juniper'. These are areas that have supported piñon and juniper during the past several 100 years and either support stands of trees that are hundreds of years old or stands of young trees re-establishing following a stand-replacement event. Piñon and juniper encroachment into sagebrush–steppe communities is addressed in another section of the current paper.

Woodlands where trees have been able to mature and reach ages of hundreds of years are commonly found on steep rocky slopes, shallow soils high in clay content, sedimentary soils, and soils high in sand (Burkhardt and Tisdale 1969; Tausch *et al*. 1981; Holmes *et al*. 1986; Miller and Rose 1995, 1999; Burwell 1998; Floyd *et al*. 2008). Little evidence suggests stand structure in closed persistent piñon and juniper woodlands has changed in the past 150 years resulting from altered fire regimes (Waichler *et al*. 2001; Eisenhart 2004; Miller *et al*. 2005; Floyd *et al*. 2008). In Mesa Verde, Floyd *et al*. (2000) concluded fire frequency during the past 50 years has probably not varied outside the range of historic variability for closed persistent woodlands. Widespread fires in persistent woodlands are typically infrequent (>200 years) high-severity stand-replacement crown fires (Floyd *et al*. 2000, 2004, 2008; Waichler *et al*. 2001; Baker and Shinneman 2004; Miller *et al*. 2005). Accumulations of old wood and standing snags that have persisted for hundreds of years are supportive evidence that fire has been absent in these woodlands (Betancourt *et al*. 1993; Waichler *et al*. 2001; Floyd *et al*. 2003; Eisenhart 2004). Floyd *et al*. (2008) estimated a 400–600-year fire rotation for piñon–juniper in the Glenn Canyon Recreation Area and Waichler *et al*. (2001) could only find evidence of small single to several-tree fires in western juniper woodland with trees exceeding 1000 years old.

Although total fuel loads can be abundant in persistent woodlands, surface fuels are often sparse and tree canopies can be discontinuous within the stand, requiring high winds and dry conditions to support a widespread high-severity fire (Miller *et al*. 2000; Floyd *et al*. 2008). Increasing amounts of dead canopy foliage as a result of drought, insects, and disease also increased the potential for large high-severity fires. During the late 1500s, severe drought and large stand-replacement fires are suspected to have caused extensive mortality of piñon and juniper woodlands across the South-west (Swetnam and Betancourt 1998). The lack of abundant and continuous surface fuels limits the occurrence of widespread low-intensity fires, which, when they do occur, are typically small (Wangler and Minnich 1996; Waichler *et al*. 2001).

However, the relationship between fire and persistent woodlands well may be in a state of change. Elevated CO_2 levels appear to have resulted in longer fire seasons and higher summer temperatures during the past decades (Westerling *et al*. 2006), which have increased canopy cover and foliage biomass (Knapp and Soulé 1996; Knapp *et al*. 2001; Soulé *et al*. 2004) and increased abundance of introduced annuals (Tausch 1999*b*; Floyd *et al*. 2008). This will likely increase the probability that these historic woodlands will burn and possibly change successional trajectories to new steady-states. A large proportion of persistent woodlands present at the time of Euro-American settlement was established during the Little Ice Age (~1300–1850). In the past, woodland succession was largely dependent on the understorey composition before the fire and severity of the fire (Erdman 1970; Barney and Frischknecht 1974; Everett and Ward 1984; Koniak 1985; Wangler and Minnich 1996). In the absence of weed encroachment, the response of native biota and successional trajectories appear to have changed little across this vegetation group. However, exotic plants, such as cheatgrass (*Bromus tectorum*), can dramatically change successional trajectories following woodland fires where native understorey species are decreased, resulting from high fire severity or past overgrazing. The contiguous cover of exotic annuals can result in repeated fires, which limit the establishment of native species, creating a new steady-state of vegetation (Tausch 1999*b*).

Piñon–juniper savannas are defined as stands having a well-developed grass understorey plus a low to moderate density of trees (Romme *et al*. 2007). In a review of these ecosystems, Romme *et al*. (2007) concluded that although fire, climate, and herbivory influenced their structure and composition, the relative importance or interactions have been poorly documented. Savannas in parts of Arizona and New Mexico are less extensive today than they were in the past (Leopold 1924; Sallach 1986; Miller 1999; Fuchs 2002). The decline has been the result of infill by piñon and juniper. It seems obvious that fire intensity and probably severity would be greater in these newly developed woodlands when compared with their historic structure.

Great Lakes conifer–hardwood forests

Fire regimes of mixed conifer–hardwood forests within the northern Great Lakes Region are also complex and heterogeneous, and depend on local interactions between climate, physiography, vegetation, landscape context, and human factors (Heinselman 1973; Cardille *et al*. 2001; Cleland *et al*. 2004; Schulte *et al*. 2005). Dry conditions conducive to fire occur on an annual basis in the upper Great Lakes region, and are usually correlated with the prevalence of warm, dry south-westerly winds; extremely dry conditions occur on a decadal basis (Lorimer and Gough 1988). The fire season spans April to October and, generally, peak fire incidence is before leaf-out in the spring or in autumn (Haines *et al*. 1975; Lorimer and Gough 1988). The fire season can span the whole summer, however, where climate, physiography, and vegetation synergistically interact to form extremely fire-prone systems such as the Boundary Waters area of north-eastern Minnesota and the Mio Outwash Plains of the northern Lower Peninsula of Michigan (Haines *et al*. 1975). These areas tend to experience the largest fires in the region (Heinselman 1973; Simard and Blank 1982).

The pre-Euro-American fire regime included both large, intense stand-replacing fires with variable frequencies and high-frequency, low-intensity surface fires (Heinselman 1973; Cleland *et al*. 2004; Schulte and Mladenoff 2005). Large conflagrations primarily occurred on locations with shallow soils and on glacial outwash plains, where they could be common. Drought-prone landforms also influenced fire frequencies in surrounding landscapes; fires ignited on droughty landforms spread to neighbouring, more mesic landforms, affecting vegetation composition and structure (Cleland *et al*. 2004; Schulte and Mladenoff 2005). Light to moderate surface fires occurred at frequencies of 1–2 times per decade on a variety of site types, where they maintained savannas, barrens, and pine forests (Heinselman 1973; Cleland *et al*. 2004; Schulte and Mladenoff 2005).

The most infamous large fires in the northern Great Lakes region occurred during the Euro-American settlement period. Some of these infamous fires include the 1871 Peshtigo Fire (518 000 ha), the 1881 Thumb Fire (41 000 ha), the 1894 Hinckley Fire (65 000 ha), the 1894 Phillips Fire (39 000 ha), the 1910 Baudette Fire (120 000 ha), and the 1918 Cloquet Fire (101 000 ha) (Mitchell and Robson 1950; Haines and Sando 1969; Lorimer and Gough 1988), many of which claimed many human lives in addition to consuming forest. The substantial buildup of logging slash and the careless treatment of fire at the time primed the environment for forest fire (Pyne 1997); conflagrations resulted when these conditions paired with extreme drought (Lorimer and Gough 1988). Many settlement-period fires burned several times more area than the largest previously recorded fires in the region. The Peshtigo fire, for example, covered an area almost three times greater than the largest known natural fire – a 180 000-ha fire in north-eastern Minnesota in 1864 (Heinselman 1973).

The number and size of fires dramatically dropped as fire control became effective in the early part of the 20th century and has been low ever since (Heinselman 1973; Cleland *et al*. 2004). For example, 232 000 ha burned annually in Michigan between 1910 and 1925; this average declined to just 11 000 ha between 1939 and 1948 (Mitchell and Robson 1950). An increase in fire activity is not expected at present because much of the regional forest has 'switched' to a new, less fire-prone state (Frelich and Reich 1995; Nowacki and Abrams 2008), and a novel fire regime given the known presettlement fire history of the region (Nowacki and Abrams 2008). This switch occurred as broad-leaved deciduous species such as sugar maple (*Acer saccharum*) – a shade-tolerant, competitively dominant tree in the region – expanded into historically pine-dominated (*Pinus banksiana*, *P. resinosa*, *P. strobus*) and oak-dominated (*Quercus macrocarpa*, *Q. rubra*, *Q. ellipsoidalis*) systems. Once established, shade-tolerant, deciduous species tend to inhibit fire ignition and spread (Nowacki and Abrams 2008). The interaction between fire suppression, fuel loads, and wildfire in the northern Great Lakes region is, thus, quite different from the western USA. Although fire is still a part of northern Great Lakes forests today, present landscapes are so fragmented by human land use that it is difficult to tease apart the effect of the natural dynamic from the human one; rather, human factors strongly interact with abiotic and biotic conditions to determine present fire dynamics (Cardille *et al*. 2001; Cleland *et al*. 2004).

Large fires are presently scattered across the region, but are more prevalent within north-western Minnesota, central Minnesota, and the north-central portion of Michigan's Lower Peninsula (Cardille *et al*. 2001). A few very large contemporary fires include the 1971 Little Sioux Fire in Minnesota (6000 ha), 1976 Seney Fire in Michigan (24 000 ha), 1980 Mack Lake Fire in Michigan (16 000 ha), and 2007 Ham Lake Fire in Minnesota (15 000 ha). Although the broad-scale distribution of fire is similar to the historical one, most fires and most large fires today occur in the aspen–birch forest type (Cardille and Ventura 2001) – historically, most fires occurred in conifer, mixed conifer–hardwood, or oak systems (Heinselman 1973; Cleland *et al*. 2004; Schulte *et al*. 2005). At a finer scale, the occurrence of large fires is highest in locations proximal to non-forest land cover, distant from cities, with low stream densities, and with low road densities (Cardille *et al*. 2001).

Are large fires a benefit or catastrophe in the northern Great Lakes region today? Given the sweeping differences between presettlement and present fire regimes, a major ecological concern is how any type of fire and its benefits might be reintroduced to the current, more fully humanised landscape (Schulte *et al*. 2007; Nowacki and Abrams 2008). Restoring pine and oak systems in the region will require fire. Historically dominant pine and oak species generally need high light and bare mineral soil conditions for successful germination (Burns and Honkala 1990). Fire allows them to maintain their competitive edge against more shade-tolerant, broad-leaved deciduous species (e.g. *Acer saccharum*, *A. rubrum*, *Tilia americana*), especially on higher-quality sites (Nowacki and Abrams 2008). Systems requiring the most frequent fire (1–2 times per decade), including pine barrens, pine savanna, and oak savanna systems, are among those most threatened habitats globally today, and are home to threatened and endangered species, including the Kirtland's warbler (*Dendroica kirtlandii*), sharp-tailed grouse (*Tympanuchus phasianellus*), the Karner blue butterfly (*Lycaeides melissa samuelis*), and several other butterfly and moth populations (Pregitzer and Saunders 1999; Will-Wolf and Stearns 1999). Restoring and maintaining these systems and species into the future will require active and concerted effort towards fire reintroduction and management.

Summary

In general, it appears that large fires were historically common on many landscapes of the US except for sagebrush ecosystems, and they will continue to occur on contemporary landscapes (Table 3). In many US ecosystems, large fires were an important ecosystem disturbance and many native plant and animal species have adapted to survive and thrive after these extensive events. However, large fire characteristics and subsequent effects appear to differ across most ecosystems presented here. Sagebrush ecosystems are currently experiencing larger, more severe, and more frequent fires compared with historical conditions owing to exotic cheatgrass invasions. Similarly, large fires in south-west ponderosa pine forest historically created a mixed-severity mosaic dominated by non-lethal surface fires whereas

Table 3. A summary of large fire characteristics as described in the present paper (Y, yes; N, no; M, more; L, less; S, same)

Ecosystem	Historically common?	Currently more severe?	Currently more frequent?	Important to ecosystem?	Pre-conditioning factors
Pacific North-west	Y	S	S	Y	Drought
Southern California	Y	S	S	Y	Santa Ana winds
Northern Rockies	Y	S	S	Y	Drought, wind
South-western US	Y	M	S	Y	Drought
Sagebrush–grasslands	N	M	M	N	Fuel contagion
Piñon–juniper	Y	S	S	N	Wind, hot weather
Great Lakes mixed forests	Y	N	N	Y	Drought, ignition

today's large fires are mostly high-severity crown fires. This is quite different from the other presented ecosystems where historical and current large fire effects are comparable (Table 3). Although large fires play an important role in landscape fire ecology for most regions, their importance is much less in the dry piñon juniper forests and sagebrush grasslands. We emphasise that there are limited data to critically evaluate any changes in historical large fire effects because fires have removed most of the evidence of past severity patterns and large fire return intervals tend to be somewhat long (Turner *et al*. 1998), making it difficult to determine the effects of land management (i.e. fire exclusion, resource extraction) that occurred over a relatively shorter time. Future fire management should recognise the importance of large fires to regional ecology as both a possible tool for the efficient restoration of fire-dominated ecosystems and an effective treatment for reducing fuel hazards.

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