Chapter 9
Fire Ecology of the North American Mediterranean-Climate Zone

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Abstract North America's Mediterranean climate zone (NAMCZ) includes most of California, southwestern Oregon, a slice of western Nevada, and northwestern Baja California, Mexico. Climatically, the world's Mediterranean climate regions are unique because the wet season is concurrent with the cold season, and the warm, dry season is akin to an annual drought of 3–7 months. Most of the NAMCZ receives sufficient precipitation in the winter and early spring to produce a crop of fuel just in time for the hot, dry summer. Vegetation in the NAMCZ is among the most fire-prone and fire-shaped on the continent. The NAMCZ supports all of the major fire

Ecoregions 4, South Cascades; 5, Sierra Nevada; 6, Central California Foothills and Coastal Mountains; 7, Central California Valley; 8, Southern California Mountains; 9, Eastern Cascade Slopes and Foothills; 78, Klamath Mountains and California High North Coast Range; 85, Southern California/Northern Baja Coast

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regime types represented in North America, but most of the modern landscape supports either the chaparral type (moderate frequency, high-severity), or the moderate frequency/“mixed” severity type including the extensive yellow pine and mixed conifer forest type, which before logging, reduced human ignitions and fire suppression supported a high frequency, low-severity fire regime. We compare historical (pre-Euro-American settlement) and modern fire regimes in the NAMCZ and discuss current pyrological and ecological trends, ecosystem management, conservation and restoration, and the future of fire and fuels management in a time of rapid global change.

Keywords California · Mediterranean-type climate · Fire regimes · Fire management

9.1 Introduction

9.1.1 Biophysical Environment

9.1.1.1 Geography

North America’s Mediterranean climate zone (NAMCZ, c. 295,000 km^2) includes most of California, southwestern Oregon, a small slice of western Nevada, and northwestern Baja California, Mexico (Fig. 9.1). The exact boundaries of what constitutes the NAMCZ are not universally agreed on, but most scientists subscribe to the California Floristic Province as mapped by Hickman (1993), which corresponds closely to the Mediterranean Division of Bailey (1995). The region is characterized by large NNW-SSE trending geographic features, such as the Coast Ranges, the San Andreas Fault, the Central Valley (which entrains the Sacramento and San Joaquin Rivers), the South Cascades, the Sierra Nevada, and the Peninsular Ranges. The Klamath Mountains, the Modoc volcanic plateau, and the Transverse Ranges, which stretch from Santa Barbara to the San Gorgonio Mountains, are important exceptions to this rule (Fig. 9.1). Aside from the Sacramento and San Joaquin, most major rivers in California flow from NE to SW, especially in the Sierra Nevada, which

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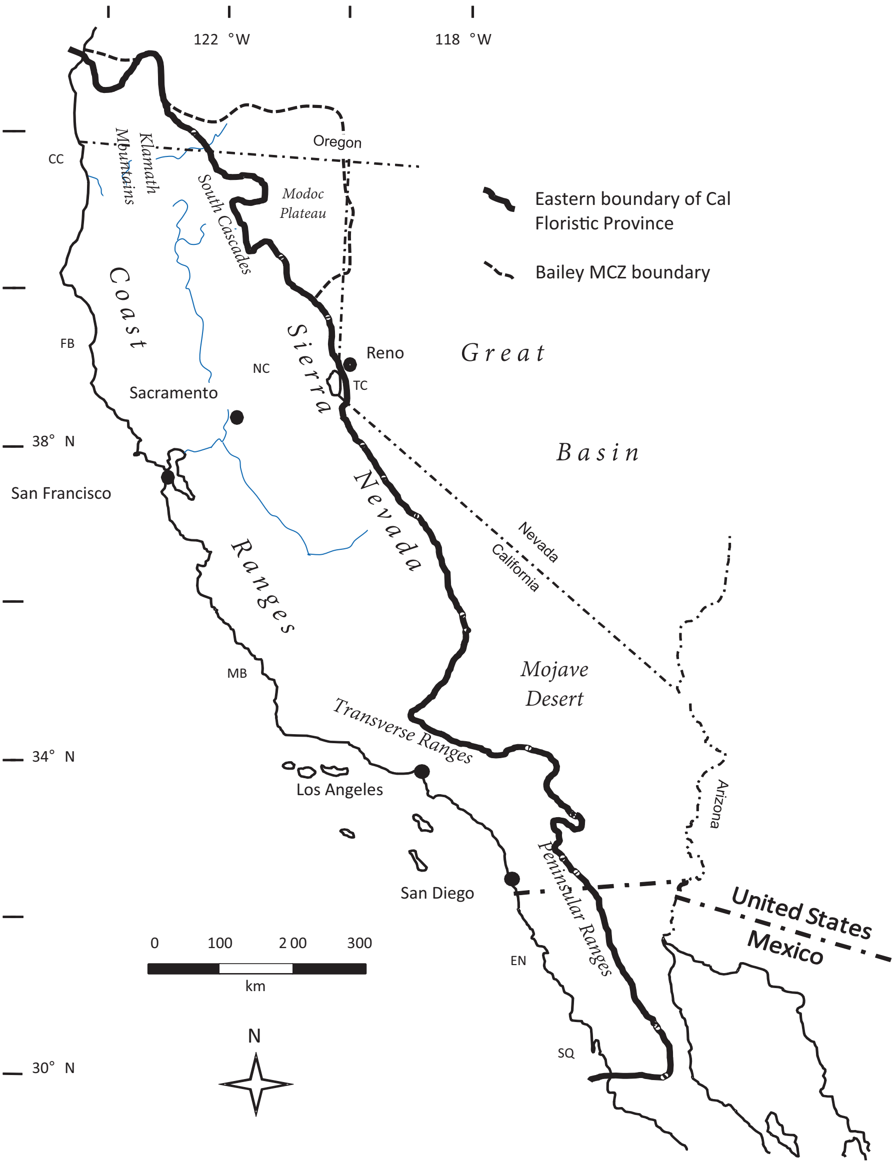

Fig. 9.1 North American Mediterranean Climate Zone (NAMCZ). Boundary approximated by the limits of the California Floristic Province (Hickman 1993); Bailey (1995) NAMCZ boundary coincides with the CFP except in the north, area of discrepancy indicated by dark dashed line. Two-letter codes represent locations featured in Figs. 9.2-9.4: CC = Crescent City, FB = Fort Bragg, NC = Nevada City, TC = Tahoe City, MB = Morro Bay, EN = Ensenada, SQ = San Quintín

supplies >60% of the state's developed water. In the northern half of the NAMCZ, elevations range from sea level to >2000 m in the Coast Ranges, to 2750 m in the Klamaths and >4300 m in the South Cascades. The Central Valley is very low elevation, with the Sierra Nevada rising gradually to the east to elevations mostly <3200 m north of 38°N, but rising south of that and reaching 4421 m at Mt. Whitney at 36.6°N. The eastern escarpment of the Sierra Nevada is very steep and geologically represents the western edge of the Great Basin.

"Southern California" typically refers to the lands south of and including the Transverse Ranges (Fig. 9.1). Elevations in the Transverse Ranges rise from sea level to >2600 m in the west and to over 3500 m in the east at San Gorgonio Mountain. The general NNW-SSE trend of the land is resumed south of the Los Angeles and San Gabriel Basins, where the Peninsular Ranges begin their southward march. The most important ranges here are the Santa Ana Mountains, the Laguna Mountains, and the granitic San Jacinto Mountains, which reach 3300 m elevation. South of the international border, low mountains along the coast are succeeded to the east by the Sierra Juarez and then the Sierra de San Pedro Mártir (up to 3096 m), whose eastern and southern reaches mark the end of the NAMCZ.

9.1.1.2 Climate

The geographic boundaries of the NAMCZ may be inexact, but the general gist is a region in which winters are cool and wet and summers are warm to hot and dry (Fig. 9.2). The climate of the NAMCZ is globally unique. Very few places on earth – the others being the world's other Mediterranean climate regions (the Mediterranean Basin, SW South Africa, S and SW Australia, and central, coastal Chile) – support a climate where the wet season is concurrent with the cold season, and the warm, dry season is an annual drought of 3–7+ months (Fig. 9.2). This pattern leads to the growing season being mostly out-of-phase with the wet season, which has fundamental effects on plant and animal communities. Importantly, most of the NAMCZ receives sufficient precipitation in the winter and early spring to produce a crop of fuel just in time for the hot, dry summer.

The major geographic gradients in NAMCZ climate align with latitude, distance from the ocean ("continentality"), and elevation. These climatic gradients interact with soils and disturbances, principally fire, to drive vegetation distribution. Figure 9.3 gives three precipitation-related measures of the latitudinal climatic gradients experienced in the NAMCZ along the west coast. At the southern end of the region (NW Baja California) precipitation is scarce (130–250 mm in the lowlands), the dry season is long (8 months with <25 mm precipitation), and interannual variability in precipitation (CVs >0.5) is extremely high. At the northern end of the region (NW California/SW Oregon), precipitation is copious (>1000 mm), the dry season is much shorter (2–3 months), and interannual variability in precipitation is lower (CVs <0.3) (Fig. 9.3). Very high interannual variability in precipitation is a distinguishing feature of the NAMCZ (Dettinger et al. 2011).

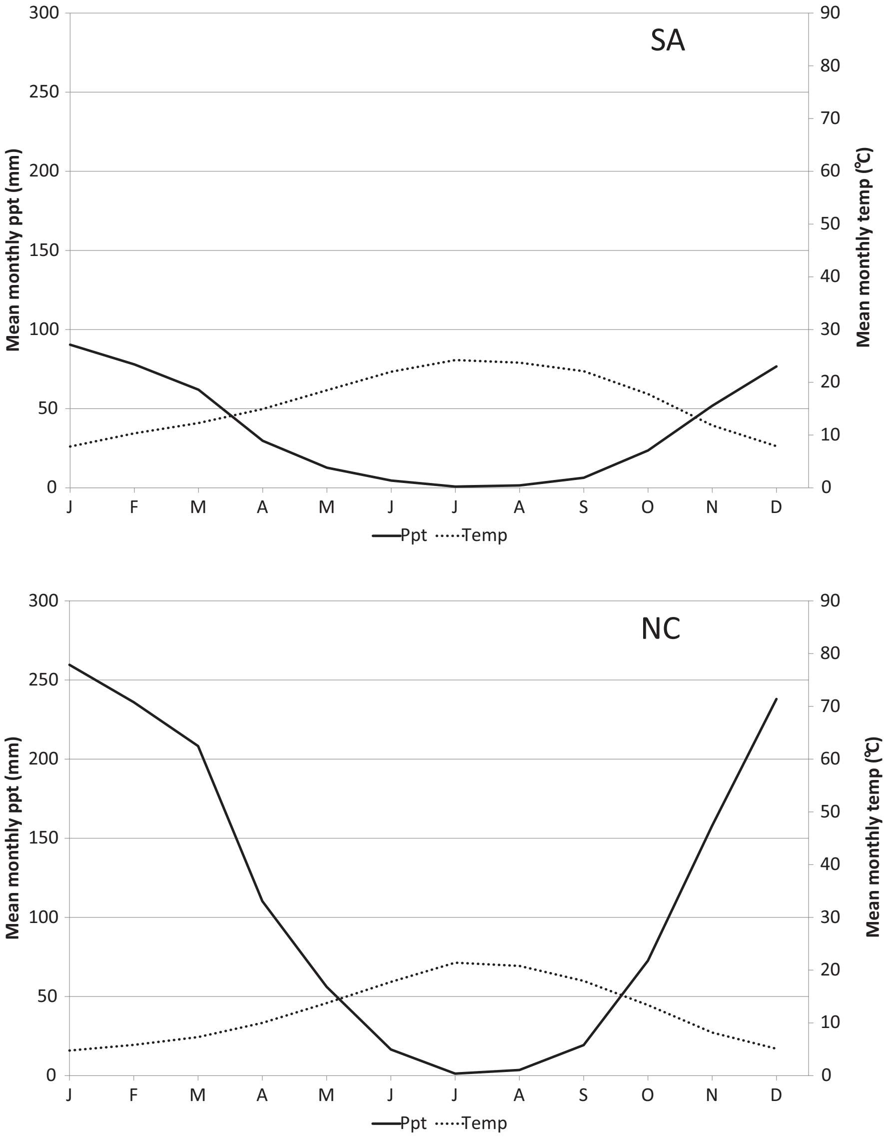

Fig. 9.2 Climate diagrams for Sacramento (SA, 6 m asl) and Nevada City (NC, 847 m asl), demonstrating typical MCZ patterns for precipitation and temperature for a lowland "hot Mediterranean" site (SA: Köppen class Csa) and an upland "warm/cool Mediterranean" site (NC: Köppen class Csb)

Natural vegetation varies dramatically as one travels from the coast to the interior of the NAMCZ to its eastern edge at the cordilleran crest. Patterns are driven principally by water balance, which is a function of growing season warmth and precipitation (as well as soil, occurrence of fog, etc.). Figure 9.4 depicts a set of

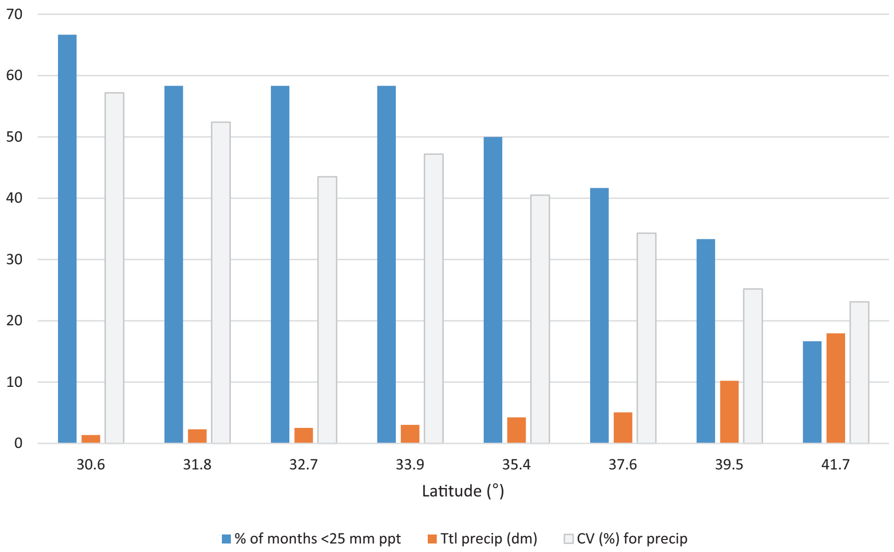

Fig. 9.3 Three precipitation-related measures from coastal NAMCZ stations, arrayed along a south-north latitudinal gradient. CV = interannual coefficient of variation. Stations are, from left: San Quintín, Mexico; Ensenada, Mexico; San Diego; Los Angeles; Morro Bay; San Francisco; Fort Bragg, and; Crescent City. See Fig. 9.1 for station locations. Data from: https://wrcc.dri.edu (US sites); https://smn.conagua.gob.mx (Mexico sites)

temperature- and moisture balance-related measures following an elevational transect approximately along Interstate Highway 80, from the coast at San Francisco (6 m asl) to Tahoe City (1904 m asl; see Fig. 9.1), about 300 km inland. Temperatures are moderate at the coast, and the difference between maxima and minima is small (only 17 °C). Mean minima drop inland, but summer highs reach their maximum in the Central Valley before dropping as the Sierra Nevada is ascended. Seasonal temperature amplitudes are progressively greater moving inland, rising to almost 33 °C at Tahoe City, which is near the eastern boundary of the NAMCZ. Climatic water deficit (CWD = potential evapotranspiration minus actual evapotranspiration) is highest in the Central Valley, moderate in San Francisco (the CWD portrayed is probably an overestimate because of the effect of coastal fog, which is difficult to measure) and Nevada City, and low in Tahoe City. Natural vegetation ranges from coastal mixed evergreen forest and prairie at San Francisco (with redwood groves in some watersheds), to grassland and oak woodland at Sacramento, to mixed conifer and interior mixed evergreen forests at Nevada City, to montane and upper montane conifer forests at Tahoe City.

9.1.1.3 Vegetation

The world's Mediterranean climate zones are well-known for supporting sclerophyllous ("hard-leaved", mostly evergreen) shrublands and savanna-like woodland vegetation (Cody and Mooney 2018; Underwood et al. 2018). In the NAMCZ, the

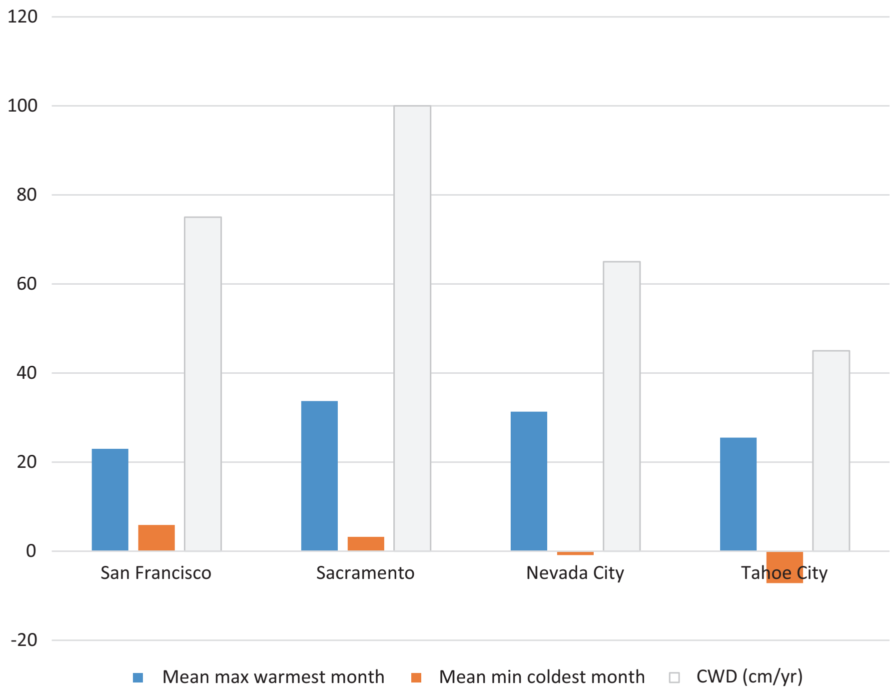

Fig. 9.4 Temperature- and water balance-related measures for an E-W transect from San Francisco to Lake Tahoe. Temperature data from https://wrcc.dri.edu. Climatic water deficit (CWD) data estimated from Fig. 3c in Flint et al. (2018)

sclerophyllous shrublands are known as chaparral, and lowland woodlands are dominated by oak (*Quercus*) species. There are at least 22 full species of oak in California ("at least" due to so many hybrids; Baldwin et al. 2012), plus relatives in the genera *Chrysolepis* and *Notholithocarpus*. Along the north and central coasts and in moister, low elevation locations in inland mountains, oaks and other hardwoods form dense stands often with Douglas-fir (*Pseudotsuga menziesii*). Drier versions of these "mixed evergreen" forests occur in southern California as well, where they are dominated by canyon live-oak (*Q. chrysolepis*) and big-cone Douglas-fir (*P. macrocarpa*).

At higher elevations (>300 m elevation in northern California, >1300 m in southern California) pines (*Pinus* spp.) were historically the dominant taxon in woodlands and open forests, especially yellow pines (*P. ponderosa, P. jeffreyi*) and sugar pine (*P. lambertiana*) (pines are also diverse in the NAMCZ, with 18 full species; Baldwin et al. 2012). Since Euro-American settlement, selective logging, loss of human ignitions, fire suppression, and pest and pathogen epidemics in these "yellow pine and mixed conifer" (YPMC) forests have greatly decreased populations of pines, and modern forests are often dominated by less fire-tolerant species that are better resource competitors in low light conditions, such as white fir (*Abies* *concolor*), incense cedar (*Calocedrus decurrens*), and Douglas-fir. Forests above the 50:50 snow/rain line (the elevation above which >50% of precipitation is snow) are known as "upper montane" in California, and are dominated by red fir (*A. magnifica*) in most of the state (red fir is absent south of 35° latitude), together with western white pine (*P. monticola*) and often lodgepole pine (*P. contorta* ssp. *murrayana*). At higher elevations, subalpine forests replace red fir, dominant species include mountain hemlock (*Tsuga mertensiana*), and a suite of cold-tolerant pines, including lodgepole, whitebark (*P. albicaulis*), limber (*P. flexilis*), and foxtail (*P. balfouriana*).

The driest forest types in the NAMCZ are dominated by pinyon pines (mostly *P. monophylla*) and juniper (mostly *Juniperus occidentalis* and *J. osteosperma*). In the NAMCZ, these types are primarily found in the Bailey (1995) extension to the California Floristic Province (Fig. 9.1), and along the eastern edge of the NAMCZ south of 36° latitude. Vegetation of the NAMCZ has been described in depth in numerous publications, including Barbour et al. (2007), Mooney and Zavaleta (2016), and van Wagtendonk et al. (2018a, b). We direct the reader to these and other sources for more detailed ecological information.

9.1.2 Evolutionary and Human History

The origin of the Mediterranean-type climate in the mid-Miocene led to high levels of diversification in sclerophyllous and other species, with fire being a major driver of speciation (Rundel et al. 2016). Fire adaptations in the NAMCZ flora are widespread, and range from traits such as seedbanking, serotiny, and fire-cued germination in high-severity fire regimes (fire severity is a measure of the effect of fire intensity – usually biomass loss – on a given ecosystem), to fire-resistant traits (e.g., thick bark, self-pruning of lower branches) in low-severity fire regimes (McCune 1988; Keeley at el. 2011). Fire-dependent recruitment varies greatly among ecosystems in the NAMCZ and is closely correlated with the fire regime. Because of their dense crowns, highly combustible foliage, horizontal fuel continuity, and very low fuel moistures in the late dry season, sclerophyllous shrublands in the NAMCZ support high-severity fire regimes. Obligate seeding and soil seed-banking are common in these shrublands (primarily chaparral and sage scrub) and serotiny is common in conifer taxa that coexist with the sclerophyll species. Low- and moderate-severity fire regimes, such as those that typified oak woodland and YPMC forest before Euro-American settlement, do not provide strong selection for niches adapted to heat-driven germination and the number of fire-dependent species is low (and almost entirely restricted to a few genera that are much more diverse in chaparral habitats) (Keeley and Safford 2016). Fire regimes where fire is an ecological novelty – such as those that characterize very wet forests, deserts, extremely low productivity sites, etc. – support essentially no fire-related traits, but such ecosystems are relatively rare in the NAMCZ.

Native Americans are thought to have first settled California around 12,000 ybp. By the time of Spanish arrival in the NAMCZ in the eighteenth century, the density of native populations was probably the highest in North American north of Mexico (Anderson 2006). Fire was an important management tool and was used for multiple purposes. Fires were set to promote oak growth for acorn harvesting, to stimulate hardwood sprouting for basketry materials, to enhance wildlife forage, for hunting purposes, and to reduce forest fuels that might lead to severe fires. Fire impacts were large in highly populated areas and in the vicinity of important cultural sites (hunting grounds, water sources, settlements) that were mostly at lower elevations. As a result, Native Americans had important heterogenizing effects on lowland vegetation, including reductions in density and cover of woody vegetation, expansion of oak savanna, reduction of chaparral, and general promotion of useful plants (Anderson 2005). In mid- and high-elevation forests (above c. 1400 m) native impacts were more geographically restricted, seasonal (or nonexistent), and much less severe, and background rates of lightning ignition were probably high enough to mask native additions to fire frequency (Barrett et al. 2005).

European settlement in California began with arrival of the Spanish during the late eighteenth century. Impacts on fire and vegetation were concentrated in southern coastal regions and near missions, where native people were forcibly relocated and herds of livestock reduced herbaceous fuels (Minnich 2007). Anglo-Americans began to settle the NAMCZ en masse after the discovery of gold in the Sierra Nevada in 1848. By the end of the nineteenth century, indigenous populations had been decimated along with their cultural burning practices, and early logging led to loss of the largest trees in many areas and wholesale clearcutting in others (especially redwood and YPMC forests). Fire suppression was formally initiated in the 1920s and 1930s to protect timber resources, despite scientific evidence that fire – as it burned in those days – was generally not a destructive force (Leiberg 1902; Show and Kotok 1924) and in the face of opposition by many forest landowners (Cermak 2005). In combination with industrial timber harvest after World War II, this led to major structural changes in NAMCZ forests, including further losses of large trees, increases in stand density, shifts toward even-aged structure, loss of stand heterogeneity, and increases in prevalence of shade-tolerant, fire-intolerant species (Safford and Stevens 2017) (note that these changes did not occur or were not as extreme in forests with longer fire return intervals (FRIs) or in most wilderness areas). Artificially low fire frequencies in many areas permitted expansion of housing into forested lands that support very high fire risk, with the risk rising over the time as fuels continue to accumulate due to the lack of fire. Over the last few decades, NAMCZ fires have destroyed roughly 500–1000 homes per year on average, but recent years have seen an astronomical leap in the level of destruction, with wildfires in 2017 and 2018 combining to kill more than 150 people and destroy 30,000 homes (Fig. 9.15). Logging pressure on public lands has eased considerably over the last 30 years, but this has had the perverse outcome of severely reducing wood product processing capacity in the NAMCZ, which has become a major economic barrier to forest thinning work. Current policy directions in the federal and state fire and forest management agencies are to increase the footprint of mechanical forest thinning and prescribed fire, and permit naturally ignited fires to burn on more of the landscape under moderate weather conditions, but the scale of the problem, the rapidity of climate warming, and the lack of economic resources are major barriers to success.

9.2 Major Vegetation Types and Fire Regimes of the NAMCZ

In this section we describe the ecological role of fire, fire adaptations in the biota, human impacts and management, and past, current, and projected future trends in fire regimes for the major forest and woodland types in the NAMCZ. We also include coverage of chaparral, as the ecology and management of fire in the chaparral ecosystem is a major concern in the NAMCZ, and a brief section is devoted to grasslands and meadows, due to their important dynamic relationship with woody vegetation.

The section is organized by historical fire regime – the fire regime believed to have characterized the ecosystem before the arrival of Euro-Americans in the NAMCZ – using the widely used classification developed by the LANDFIRE program (Table 9.1; Schmidt et al. 2002). Many terrestrial ecosystems have seen the fire regime under which they assembled change dramatically under recent human influence. Where this is the case, it is called out. Fire occurrence and behavior depend on the growth of vegetation, which results in a broadly inverse relationship between fire frequency and intensity (if ecosystem type and climate are kept constant; Safford and Van de Water 2014). Although fire intensity is rarely robustly measured, a surrogate – fire severity – can be accurately mapped using remotely sensed imagery, and fire frequency can be determined from fire perimeter atlases, which have been kept in the California part of the NAMCZ since the early twentieth century. The LANDFIRE frequency x severity fire regime classification is thus simple, measurable, and mappable, and departures from previous states can be relatively easily determined.

Figure 9.5 maps the LANDFIRE fire regimes as we hypothesize them to have been distributed before Anglo-American settlement beginning in the mid-nineteenth century. Figure 9.5 is broadly similar to the LANDFIRE historical fire regime map (2014 version, at: https://www.landfire.gov/viewer/) in southern California, the southern Sierra Nevada, and the Modoc Plateau, but differs notably in the northern

Table 9.1 LANDFIRE fire regime group characteristics

Fire regime group	Fire return interval	Dominant fire severity classes	Example ecosystems treated in the text
I	0–35	Low	Oak woodland, yellow pine, mixed conifer forest
II	0–35	High (mostly herbaceous vegetation)	Grasslands and meadows
III	35–200	Low and moderate (“mixed”)	Mixed evergreen forests, redwood forests under native American fire regime, upper montane forests, pinyon-juniper woodlands
IV	35–200	High and moderate	Chaparral, serotinous conifers, subalpine forests and woodlands
V	≥200	Any (usually high)	Wet coastal forests, some subalpine forests/ woodland

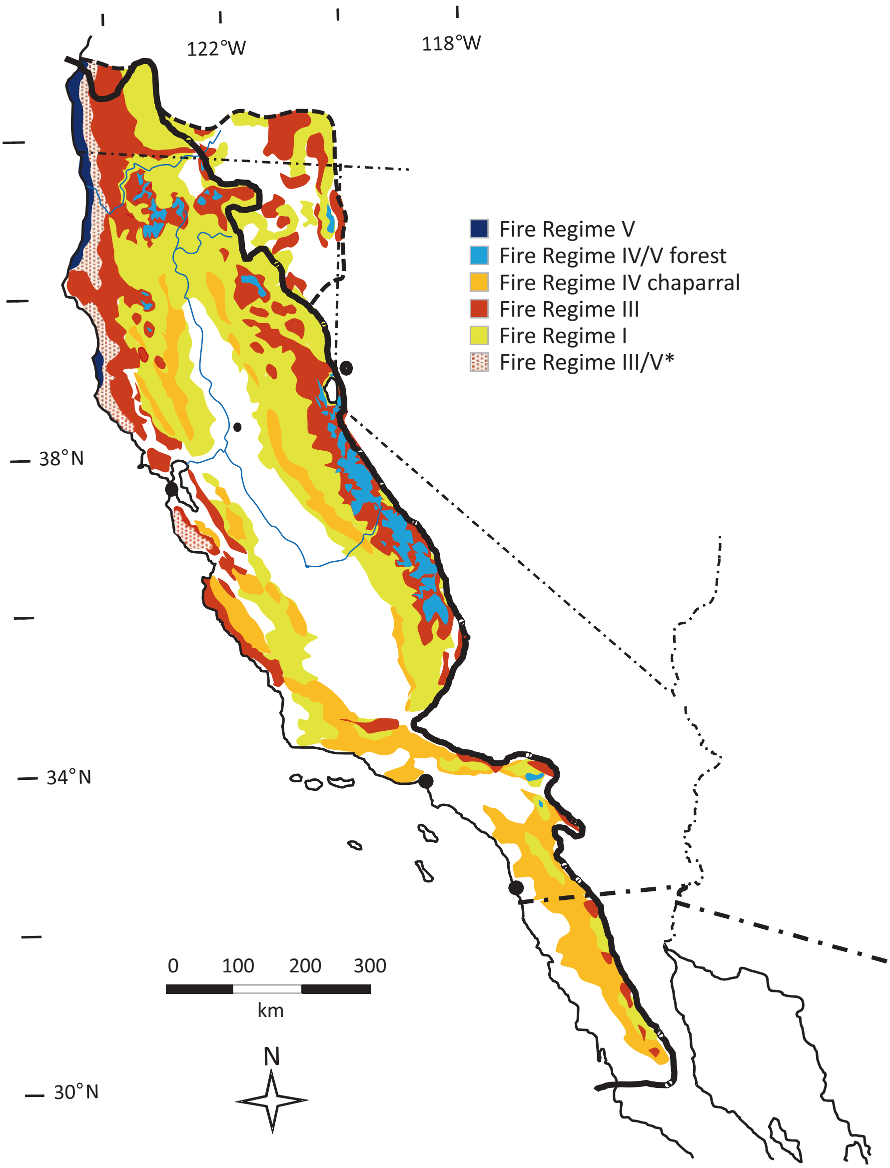

Fig. 9.5 Schematic map of hypothesized pre-Anglo-American settlement (before 1850) fire regimes of major forest types and chaparral in the North American Mediterranean Climate Zone (NAMCZ). Geographic boundaries as in Fig. 9.1. Fire regimes from Schmidt et al. (2002). Blank areas within the NAMCZ boundaries support urban and agricultural areas, grasslands, shrublands, or desert vegetation. *Stippled red pattern denotes moist redwood-dominated forests that rarely experience lightning ignitions but were burned frequently by Native Americans

Sierra Nevada, southern Cascades, Klamath Mountains, and the Coast Ranges. In these regions LANDFIRE maps a nearly uniform swath of fire regime (FR) I and misses important variability in topography, vegetation, climate, and fire ecology.

In particular, the LANDFIRE map (1) omits the coastal strip of wet forest dominated by fire avoiding species like sitka spruce (*Picea sitchensis*), western hemlock (*Tsuga heterophylla*), and western red cedar (*Thuja plicata*) (FR V); (2) does not recognize fire regime differences between mixed evergreen forest (FR III; Halofsky et al. 2011; Steel et al. 2015) and YPMC (FR I); and (3) severely undermaps the extent of red fir forest (FR III).

9.2.1 Fire Regime I: Frequent, Mostly Low-Severity Fire

9.2.1.1 Oak Woodlands

Oak woodlands are (mostly) open canopied formations dominated by a deciduous or mixed deciduous-evergreen tree canopy over an understory of (mostly) grass, forbs, and shrubs. These ecosystems are generally found in lower elevation valleys and foothills in areas of relatively high climatic water deficit. Dominant species vary based on elevational and water availability gradients, and include the deciduous species blue oak (*Quercus douglasii*), valley oak (*Q. lobata*), Engelmann oak (*Q. engelmannii*, only southern California), and Oregon white oak (*Q. garryana*, mostly somewhat higher elevations in the NW part of the NAMCZ), and the evergreen species coast live oak (*Q. agrifolia*) and interior live oak (*Q. wislizenii*). Two other oaks are very important components of higher elevation forests: the evergreen canyon live oak (*Q. chrysolepis*) is the dominant oak in mixed evergreen forests (and is also common in drier YPMC forests), and California black oak (*Q. kelloggii*) is the dominant oak in YPMC forests. Allen-Diaz et al. (2007) provides an excellent overview of oak woodlands in California including information on species autecology and geographic distribution.

Oaks and other hardwoods have a number of adaptations that allow them to tolerate fire, most importantly the ability to resprout after disturbances that kill aboveground stems (McDonald and Tappeiner 1996). Sprouting can be vigorous, and growth rates of sprouts can be high, allowing oak species to quickly reoccupy sites after disturbance. That said, oaks differ notably in their response to fire, principally as a result of differences in bark characteristics and resprouting capacity. Most California oaks can survive low- to moderate-intensity fire, but canyon live oak and interior live oak have relatively thin bark that makes them somewhat more susceptible to topkill (Plumb and McDonald 1981; Allen-Diaz et al. 2007). Valley oaks and blue oaks will crown sprout after even 100% crown scorching, but crown sprouting is uncommon in California black oak (Fry 2008). Seedlings and saplings of all hardwood species are more susceptible to mortality from fire than mature trees (Holmes et al. 2008).

Lower elevation hardwood forests and woodlands have been dramatically reduced in extent since the discovery of gold in California in 1848 (Bolsinger 1988; Thorne et al. 2008). Widespread clearing for agriculture, rangeland expansion, and urban development have eliminated oak woodlands from most of their former ranges; most California oaks have little or no commercial value as timber. About three-quarters of all oak woodlands are privately owned in the NAMCZ (Allen-Diaz et al. 2007). At higher elevations oaks have done better, and comparisons between plot data and maps from the 1930s and modern-day plots (FIA or direct resamples) show increased density and extent of oaks in mixed evergreen, yellow pine, and mixed conifer forests in most of the Sierra Nevada national forests (McIntyre et al. 2015). This is likely due to recovery from early degradation and the relative success of oaks in areas cleared of conifer competitors by harvest, fire, pests and pathogens.

Most dendrochronology records indicate that historical FRIs in oak woodlands were among the shortest of all vegetation types found in the NAMCZ (FR I) (Chap. 1, Fig. 1.5, Table 1.2), and may have been as frequent as every few years prior to European settlement (Allen-Diaz et al. 2007; Fites-Kaufman et al. 2007). Lightning strike densities are very low in areas supporting oak woodlands in the NAMCZ so most fires were and continue to be anthropogenic (Chap. 1, Fig. 1.4, Table 1.1). Extensive ethnographic evidence documents widespread use of fire by indigenous people in the NAMCZ to manipulate hardwood forests for food and basketry, to enhance habitat for game species, and to improve access for hunting (Anderson 2006; Mensing 2006). Fires set by Native Americans were likely frequent and of low-intensity, promoting oak growth and reducing fuel loads and the probability of high-severity fire that can damage large, mature oaks (Nowacki et al. 2012). The most frequently burned landscapes were generally near villages and were ignited on a nearly annual basis (Anderson 2006).

Estimates of departure from presettlement fire regimes indicate that most hardwood vegetation types in the NAMCZ are now burning much less frequently than prior to European settlement. Current FRIs for most types are now 55–70 years, compared with reference FRIs of 7–35 years (Safford and Van de Water 2014). In Oregon white oak forests an estimated pre-European fire rotation of 19 years increased to 238 years after 1905 (Taylor and Skinner 2003). Standiford and others (2012) found mean FRIs of 12.8 years in a southern Sierra Nevada blue oak woodland between 1850 and 1965, with no fires recorded since that time. In the central Sierra Nevada, Stephens (1997) calculated a mean FRI of 7.8 years between 1850 and 1947, followed by a complete absence of fires in a mixed oak-pine forest, including blue oak, gray pine, interior live and canyon live oak.

Fire exclusion has had a dramatic effect on some oak woodlands, increasing stand density and cover as well as shifting species composition to dominance by less fire-tolerant conifers such as gray pine (*P. sabiniana*) in dry locations (typically blue oak woodlands) and Douglas-fir in more productive sites (typically Oregon white oak woodlands). In the absence of disturbance, conifers can overtop, suppress, and ultimately kill most hardwood species (this dynamic is rare with gray pine, which has a shorter life span than the oaks) (Fig. 9.6). Today, prescribed fire is a favorite tool to maintain open canopies and oak dominance in stands that would

Fig. 9.6 Winter photo of an Oregon white oak woodland on a patch of shallow soil embedded in a matrix of Douglas-fir forest, Klamath Mountains, Six Rivers National Forest. The patch formerly extended onto deeper soils surrounding the site, but with fire exclusion Douglas-fir invasion has reduced the patch size; note active conifer invasion in the rear of the photo (photo Hugh Safford)

otherwise succeed to conifer forest (Fry 2008). Fire exclusion has also changed fire behavior in oak woodlands, and even more so in higher elevation hardwood-dominated forest types like mixed evergreen forest. Although fires are now less frequent, they are generally larger and of higher severity when they occur (Mallek et al. 2013). For example, increased surface and ladder fuels and dense stand conditions in California black oak forests have led to a higher proportion of high-severity fire (Agee 1993; Taylor and Skinner 2003) that can sometimes be fatal to black oak (Kauffman and Martin 1987). High-intensity fire fueled by dense adjacent conifer stands in some areas has also led to a significant shift in stand structure. Although oaks have the ability to resprout and regain dominance in conifer encroached stands, their structure is often modified from one of large, widely spaced mature oaks to small, dense, multi-stemmed individuals (Cocking et al. 2014). On the other hand, fuel loading may now be lower in foothill oak woodlands as a result of grazing, leading to lower fire frequency and severity.

A major issue in warmer, lower elevation oak woodlands is the general lack of recruitment in deciduous oak stands, attributed to various causes ranging from herbivory, to invasive plant competition, to climate change (Allen-Diaz et al. 2007; Reynier et al. 2016). Projected changes in climate will also likely increase the frequency of fire and area burned across the NAMCZ and most models also project increases in fire severity (Restaino and Safford 2018). Increased fire activity generally favors hardwood species (Allen-Diaz et al. 2007; Thorne et al. 2008), and modeled projections expect a general expansion of hardwood-dominated forest (e.g. Lenihan et al. 2008). However, many oak species are vulnerable to higher-severity fire, particularly in the seedling and sapling stages (Holmes et al. 2008), suggesting that altered fire behavior could negatively affect hardwood recruitment in the future (Swiecki and Bernhardt 2001). Increased fire occurrence can also promote annual grasses and other nonnative species that compete with oak seedlings and promote frequent fire (Bradley 2009), and ultimately lead to a loss of woody vegetation, including oaks and other hardwoods. Overall however, by far the most serious threat to oak woodland sustainability is continued habitat loss to agricultural and urban conversion (Allen-Diaz et al. 2007).

9.2.1.2 Yellow Pine and Mixed Conifer Forests

The main tree species represented in yellow pine and mixed conifer (YPMC) forests in the NAMCZ include the yellow pines (*Pinus ponderosa, P. jeffreyi*), sugar pine (*P. lambertiana*), incense cedar, white fir, and Douglas-fir, and the hardwoods black oak, canyon live oak, tanoak, madrone (*Arbutus menziesii*) and bigleaf maple (*Acer macrophyllum*). In the central and southern Sierra Nevada, giant Sequoia (*Sequoiadendron giganteum*) occurs in groves associated with moist mixed conifer stands. The mix of hardwood and conifer taxa in a given YPMC stand is driven largely by ecological differences in tolerance to shade, drought, and fire. Species that are highly tolerant of fire and drought but intolerant of shade (black oak and yellow pines) are more abundant in sites with lower water availability such as south aspects, upper slope positions and ridgetops, and lower elevations. Species that are less tolerant of fire and drought, but more tolerant of shade (white fir, incense cedar, and Douglas-fir) are more abundant in sites with higher water availability such as north aspects, lower slope positions, and higher elevations (Barbour et al. 2007; Safford and Stevens 2017; Bohlman et al. 2021).

Before Euro-American settlement, YPMC forests supported fire regimes characterized by frequent, low- to moderate-severity fires (Agee 1993; Van Wagtendonk et al. 2018a, b). The YPMC fire regime was primarily fuel-limited (as opposed to climate-limited), with the high frequency of fire and the heterogeneous nature at which fires burned limiting fuel accumulation and reinforcing structural patterns that supported primarily low- to moderate-severity fire, with only small proportions (5–15%) burning at high-severity (Safford and Stevens 2017). Prior to Euro-American settlement (pre-1850s), the mean FRI for YPMC forests was 5–20 years, with the higher end representing the wetter extreme found in moist mixed conifer forests (Van de Water and Safford 2011).

Species adaptations specific to fire in YPMC trees include thick bark, self-pruning of lower branches, and high litter flammability. These traits are associated with survival of low- to moderate-intensity surface fire and a reduced likelihood of fire entering the tree canopy (Keeley and Zedler 1998; Stevens et al. 2020). Although frequent fire favors pines (especially yellow pines), in the absence of fire species that are less fire-tolerant when young (e.g., white fir, Douglas-fir, incense cedar) develop an increased fire tolerance as trees mature (Agee 1993; Skinner et al. 2018). Overstory and understory species mostly lack specialized reproductive traits tied directly to fire, as the intense fires that provide the selective pressure for fire-cued germination or serotiny were historically rare (Keeley and Safford 2016). Where such traits exist, they are primarily found in a few members of the shrub genera *Ceanothus* and *Arctostaphylos*, which are widespread in the NAMCZ and whose centers of diversity are in lowland chaparral.

YPMC forest structure and composition before Euro-American settlement was driven by interactions between topography, climate, and frequent low- to moderate-severity fires. Fire was a key functional process that helped to maintain relatively open canopies, limit fuel accumulation, decompose biomass, promote the dominance of shade-intolerant species, and create structural heterogeneity on the landscape. YPMC forests were historically variable in structure and composition, but were generally characterized by relatively low tree densities, large tree sizes, high seedling mortality as a result of recurrent fire, and highly heterogeneous understory structure (Fig. 9.7; Safford and Stevens 2017; Bohlman et al. 2021).

The removal of fire from YPMC forests, through the suppression of wildfires and the near complete elimination of cultural burning, has led to widespread changes in structure and composition and major alterations to numerous ecological processes and functions (Safford and Stevens 2017; Bohlman et al. 2021). Other human-caused disturbances have also had important interactions with fire exclusion. With the discovery of gold and the expansion of the railroad system, YPMC forests were intensively logged in the nineteenth century, and US Forest Service timber harvest policies between the Second World War and the 1980s led to a second wave of heavy cutting. During the first period large trees were selectively removed, especially from the most valuable timber species (i.e., yellow pines, sugar pine), indirectly promoting the dominance of smaller-diameter, fire-intolerant trees, and leading to major changes in species composition and structure (Leiberg 1902; Hasel 1932; Laudenslayer and Darr 1990; Safford and Stevens 2017). Intensive grazing from domesticated livestock also began during this period and, although it is difficult to ascertain the long-term impacts that grazing has had on YPMC forests, early settler accounts and modern fire history studies indicate that grazing reduced understory cover and may have also reduced fire frequency in some areas by reducing fine fuel loading (Skinner and Chang 1996; Fry and Stephens 2006). For the past several decades, air pollution has also had a significant impact on mixed conifer forests, particularly in the mountains of southern California (Miller et al. 2012c). Increased ozone levels and nitrogen deposition have been shown to reduce growth and vigor in trees, especially yellow pines, ultimately leading to increases in tree mortality associated with drought and bark beetles (subfamily Scolytinae; Jones et al. 2004; Temple et al. 2005). This increase in mortality has led to shifts in species composition as well as an increased susceptibility to wildfires (Miller et al. 2012c).

In general, modern YPMC forest stands have much higher tree densities and are dominated by small-diameter individuals of shade-tolerant species, fuel loads are higher, canopy cover has increased with fewer canopy gaps present, and understory plant cover is lower (Fig. 9.7; Safford and Stevens 2017; Bohlman et al. 2021).

Fig. 9.7 Upper: Mixed conifer forest, Sierra de San Pedro Mártir (SSPM), Baja California, Mexico. The site was not logged, effective fire suppression and exclusion has been practiced here only since the 1980s. SSPM is a well-known reference site for restoration work on dry YPMC sites in California, USA. Tree species include Jeffrey pine, sugar pine, white fir, lodgepole pine, and canyon live oak. Lower: Typical mixed conifer stand after 19th and twentieth century logging and a century of fire exclusion, Plumas National Forest, northeastern Sierra Nevada. Species include ponderosa pine, sugar pine, white fir, canyon live oak, and incense cedar (photos Hugh Safford)

Much of the erstwhile fine-grained heterogeneity has been lost and in most areas YPMC stands have become more homogenous and contiguous across the landscape. These structural and compositional changes have led to a major shift in the dominant fire regime, from FR I to FR III or IV, which are characterized by infrequent moderate- to high-severity fires (Fig. 9.8; see Table 9.1). The primary role of fire has changed from one of forest maintenance (of relatively open-canopy, fuel-limited conditions with dominance primarily by fire-tolerant species) to one of forest transformation, where dense stands of largely fire-intolerant species and heavy, continuous fuel accumulations are more likely to burn at high-severity, resulting in major ecosystem changes (Steel et al. 2015; Coppoletta et al. 2016; Stephens et al. 2018a). These hot fires often lead to massive recruitment of shrubs, which can be a major impediment to forest recruitment, especially if such sites burn again (Coppoletta et al. 2016; Tepley et al. 2017). Recent work has also shown that high-severity burning can have major, long-term impacts to soil biogeochemical processes (Dove et al. 2020).

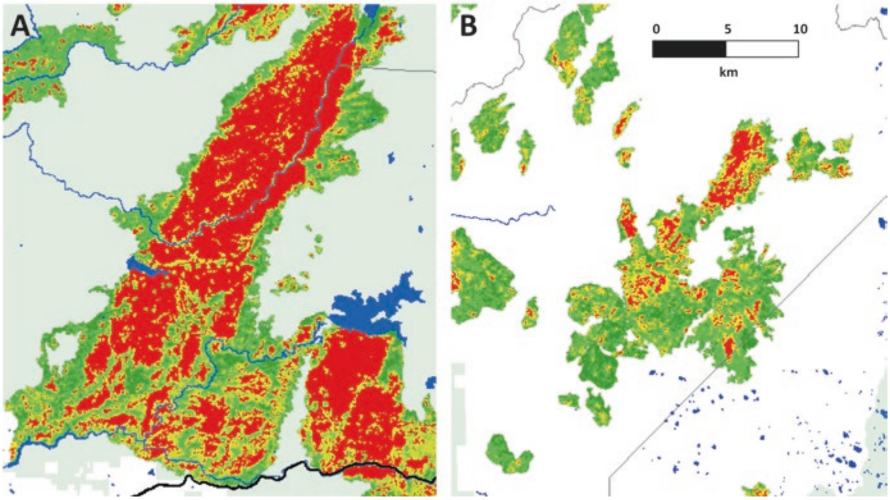

Fig. 9.8 Fire history and fire severity between 1984 and 2016 in two contrasting management landscapes in Sierra Nevada YPMC forest. Subfigures are ~780 km^2 each. Green shading indicates National Forest lands, dark blue indicates water bodies. Within fire perimeters, red = high-severity burning, yellow = moderate-severity, light green = low-severity, dark green = low-severity or unburned: (**a**) Eldorado National Forest (ENF): number of fires = 9, mean size = 8234 ha, overall % high-severity = 43.8; (**b**) Yosemite National Park (YNP): number of fires = 40, mean size = 382.6 ha, overall % high-severity = 9.4. ENF was extensively logged and has been managed under a fire suppression policy for >100 years; YPMC forests here now burn with FR III or IV characteristics (Table 9.1). YNP was not logged and was managed under fire suppression until 1972, when a fire management policy was adopted permitting many naturally ignited fires to burn depending on weather conditions. Most fires in YPMC forest types in YNP burn with FR I characteristics. Note that YNP area is higher elevation on average and includes some upper montane and subalpine forest; some of the blank area is rock

Changes in the YPMC fire regime are having major impacts on biota in the NAMCZ. Bird communities are heavily influenced by fire, and many species are uncommon in areas that remain long-unburned (White et al. 2015). Trends of increasing fire severity benefit some animal taxa (e.g., certain woodpecker and bat species) but put at serious risk populations of species that depend on older forest conditions, including lichens, the spotted owl (*Strix occidentalis*), the northern goshawk (*Accipiter gentilis*), and the Pacific fisher (*Pekania pennanti*) (Jones et al. 2016; Miller et al. 2018). The lack of fire has negatively influenced local plant diversity in YPMC forests, as has the increase in the area of high-severity burn patches (Stevens et al. 2015; Richter et al. 2019). Aquatic invertebrates can be strongly affected by physical changes to streams and decreases in water quality caused by high-severity burning (Oliver et al. 2012). Terrestrial invertebrates and small mammals don't seem to be greatly affected by variation in fire severity (Fontaine and Kennedy 2012), but some species are strongly tied to conditions of open canopy and low surface fuels (e.g. Dalrymple and Safford 2019). Overall, a diversity of burning conditions is important to community diversity in nearly every taxonomic group that has been studied thus far.

Seasonal drought is an important characteristic of the NAMCZ. However, from 2012 to 2016 California experienced a multiannual drought that had heavy impacts on YPMC forests, especially in the southern Sierra Nevada. Paleodata suggest it was the most intense drought event in over 1000 years (Griffin and Anchukaitis 2014). Tree mortality was primarily due to an explosion in bark beetle populations after water-stressed trees – due to drought compounded by high stand densities – could no longer fend them off, thus creating a huge increase in the available food source (Young et al. 2017; Fettig et al. 2019; Restaino et al. 2019). The long-term effects of this massive mortality event on fire behavior are yet to be seen, and may include an increase in fire intensity and size as snags decompose and fall to the ground, creating a vast network of contiguous large fuels (Stephens et al. 2018a; Young et al. 2020). Future projections suggest that multiannual droughts will become more common as the climate warms, creating a major ecological and management conundrum for arid and semi-arid forest types that are already near their "climatic edge" (Dettinger et al. 2018; Williams et al. 2020).

In response to the deteriorating condition of California YPMC forests there has been heightened interest in restoration of more resilient forest conditions, focused often on pre-Euro-American settlement compositions, structures, spatial patterns and the natural disturbance regimes that maintained them (North et al. 2009, 2012; Safford and Stevens 2017), although not necessarily with those conditions as the ultimate end state (Safford et al. 2012a). Monitoring shows that forest management focused on reduction in tree densities and overall fuel loading via mechanical thinning and/or prescribed burning has been successful in reducing tree mortality due to drought, beetle outbreaks, and wildfire (Fettig et al. 2007; Safford et al. 2012b; Restaino et al. 2019; Tubbesing et al. 2019). Reintroduction of fire as a natural ecological process, as in wildland fire use/managed wildfire programs implemented in the Sierra Nevada National Parks and some US Forest Service wilderness areas, has also shown great promise (Fig. 9.8; North et al. 2012; van Wagtendonk et al. 2012; Boisramé et al. 2017a, b).

Current trends in YPMC fire activity in the NAMCZ show increases in burned area, fire frequency, fire size, and – in most of the region – fire severity (Miller et al. 2012b; Mallek et al. 2013; Steel et al. 2015; Nigro and Molinari 2019). High-severity fire patch sizes have also increased (Stevens et al. 2017; Steel et al. 2018), resulting in changed successional trajectories associated with distance to seed source, harsh postfire conditions, and dense shrub cover (Coppoletta et al. 2016; Welch et al. 2016; Tepley et al. 2017; Shive et al. 2018). Tellingly, in locations in the NAMCZ where timber harvest was less intense and fire exclusion less complete – e.g., national parks in the Sierra Nevada and northern Baja California – these patterns have been much less accentuated or absent altogether (Fig. 9.8; Miller et al. 2012a; Rivera-Huerta et al. 2016). Future projections are for continuation and acceleration of the same fire trends and increased influence of multiyear droughts (Restaino and Safford 2018; Dettinger et al. 2018). Perhaps the major challenges to YPMC sustainability are the loss of forested habitats and the ecosystem services they provide to altered disturbance regimes and the rapidly warming climate (McKenzie et al. 2008; Kershner 2014; Keeley and Safford 2016). To date, forest managers have chased relatively narrow restoration goals, mostly on a site-by-site basis, and achievement of those goals does increase resilience to fire and drought (Allen et al. 2015; Hanberry et al. 2015). However, with the magnitudes of change and values at risk growing by the day, forest and fire managers will have to work together on much larger scales of space and time if YPMC forests are to be conserved in a state that any of us will recognize in 50 years (North et al. 2009, 2012).

9.2.2 Fire Regime II: Frequent, Mostly High-Severity Fire

9.2.2.1 Meadows and Grasslands

For the most part, FR II is represented by meadow and grassland vegetation types, where woody vegetation may be present but is not dominant. Meadows and grassland comprise much of the white area in the NAMCZ in Fig. 9.5, especially in the Central Valley between the Sierra Nevada and Coast Ranges, but also scattered in smaller patches in forested landscapes. FR II is basically inimical to forest growth, but herbaceous ecosystems characterized by this fire regime are often successionally related to the forest types that surround them and they often carry fire to and from forests. In general, meadows tend to remain green throughout the growing season because they are groundwater-dependent and rely on the persistence of a shallow water table, whereas grasslands are not dependent on shallow groundwater and often turn brown and senesce before the end of the growing season (Ratliff 1985; Lowry et al. 2011). Meadows and grasslands are of great ecological importance. They store high amounts of carbon, provide important refugia for many terrestrial and aquatic biota, reduce peak water flow after storms and during runoff, recharge groundwater, protect stream banks and lake shores, filter sediments, reduce sedimentation, supply forage, and provide numerous recreational opportunities (Ratliff 1985; Kattelmann and Embury 1996; Weixelman et al. 2011; Dass et al. 2018).

Fire is a natural occurrence in most meadow and grassland vegetation types. These ecosystems can act as a fire break or as a corridor for fire spread, depending on site conditions such as fuel moisture, topography, microclimate, vegetation, and surrounding forest fuels (Pettit and Naiman 2007). Fires tend to burn more frequently in dry grasslands (assuming sufficient productivity) than in wet meadows, which may support fire frequencies >35 years. Drier grasslands also tend to burn more homogeneously than moister sites or sites with a higher component of forbs, where fire may burn in a patchy mosaic. Regardless of how much of the overstory is consumed, most species in meadow and grassland ecosystems can rapidly recover following fires and other disturbances, due to traits such as high belowground biomass, strong resprouting capacity, clonal growth, and investment in underground storage organs (Veldman et al. 2015).

Meadow and grassland ecosystems often depend on disturbance, such as frequent fire or herbivory, to limit the abundance of trees and shrubs, promote herbaceous productivity, consume dead plant material, return nutrients to the soil, stimulate reproduction, and maintain plant diversity (Veldman et al. 2015). When fire is removed from these systems, meadow and grassland ecosystems can rapidly transition to forest or shrubland (Fig. 9.9), with an associated loss of plant diversity

Fig. 9.9 Prescribed fire being applied to a meadow fringe in the Lake Tahoe Basin to reduce the density of invading lodgepole pine (photo Hugh Safford)

and wildlife habitat (Norman and Taylor 2005). However, in areas with shallow soil, low soil moisture availability, or high concentrations of toxic metals, grasslands tend to persist even in fire excluded landscapes (Veldman et al. 2015).

Dynamic vegetation models suggest that increased fire activity in the NAMCZ will promote the expansion of grasslands (Restaino and Safford 2018) and such a trend is already apparent in parts of lowland southern California, where weedy annual grasses are expanding (Safford et al. 2018). Diverse grasslands are projected to be more resilient to changes in climate than forests, and it is thought that they will also provide a more reliable carbon sink because of higher resilience to drought and wildfire (Dass et al. 2018). However, excessively frequent fire can shift species composition in meadow and grassland ecosystems to dominance by non-native, annual grasses. These grasses can in turn promote frequent, high-severity fire, eliminating native, perennial species (D'Antonio and Vitousek 1992; Chap. 12).

9.2.3 Fire Regime III: Moderately Frequent, Mostly Low- and Moderate-Severity Fire

9.2.3.1 Mixed Evergreen Forests

Mixed evergreen forests in the NAMCZ are the dominant vegetation in the dark red areas in Fig. 9.5. These forests are dominated by Douglas-fir (*P. menziesii* in the northern three-quarters of California, and *P. macrocarpa* ["big-cone Douglas-fir"] in the southern quarter) and a mix of evergreen and deciduous broad-leaved species. Commonly occurring species include canyon live oak, tanoak, Pacific madrone, Oregon oak, California black oak, giant chinquapin (*Chrysolepis chrysophylla*), big leaf maple, California bay (*Umbellularia californica*), ponderosa pine, sugar pine, and incense-cedar (Stuart and Sawyer 2001; North et al. 2016). Mixed evergreen forest occurs from near sea level to c. 1200 m in northern California, higher in inland canyons, the Sierra Nevada, and in southern California, where stands of canyon live oak and big-cone Douglas-fir can be found as high as 2200 m. Mixed evergreen forests are most widespread in the more maritime climates of the coast ranges, with high precipitation and humidity, abundant fog, limited temperature ranges, and low temperatures for their latitudes (Whittaker 1960). The mountains paralleling the coast limit the eastward extent of the cool, wet maritime influence, but mixed evergreen forests follow river canyons inland for 100 km at least. In the Sierra Nevada, mixed evergreen forests are common along the lower west slope in the northern two-thirds of the range, where moisture is sufficient. As in coastal California, the type also winds eastward up river canyons (Fig. 9.10). In southern California and the southern Sierra Nevada mixed evergreen forests are drier and more dominated by canyon live oak, and big-cone Douglas-fir is a common associate in the former (Minnich 2007).

Before Euro-American settlement, fires in mixed evergreen forests tended to be relatively frequent with high variability in fire sizes, intensities, and severities that

Fig. 9.10 Mixed evergreen forest in the canyon of the Middle Fork of the Yuba River, northern Sierra Nevada, showing typical change in forest structure and composition due to aspect. North-facing slope (far side of river) is dense forest of hardwoods and Douglas-fir, south-facing slope (near side of river) is open mixture of canyon live oak, interior live oak, gray pine (*P. sabiniana*), and chaparral (photo Hugh Safford)

resulted in a complex mosaic of multi-aged stands of varying sizes (Skinner et al. 2018; Stephens et al. 2018b; Wills and Stuart 1994). As in the oak woodland belt, lightning strike density in NAMCZ mixed evergreen forests is very low (van Wagtendonk and Cayan 2008), so most fires were set by Native Americans, who used fire to promote food plants, basketry materials, and other forest products (Keter 1995). Fire intensities and severities in areas subject to frequent burning by Native Americas were low (Lewis 1973).

Mean pre-Euro-American settlement FRIs in Pacific Douglas-fir dominated mixed evergreen forests ranged 5–80 years with an average FRI of about 30 years (Skinner et al. 2009; Van de Water and Safford 2011). Longer FRIs occurred in more mesic sites and at higher elevations. More frequent fires on drier, low elevation sites likely promoted more open forests with a greater cover of understory plant species. In northern California and the Sierra Nevada, fire suppression and past logging have increased the density of shade-tolerant species (especially Pacific Douglas-fir) in mixed evergreen forests and have resulted in a lengthening of the FRI to 2–3 times longer than expected under the pre-Euro-American settlement regime (Safford and Van de Water 2014). However, in southern California mixed evergreen forests' current fire frequencies may be higher than under the pre-Euro-American regime.

Fires in northern California and Sierra Nevada mixed evergreen forests were dominated historically by low- to moderate-severity/mixed-severity burning and fire season was primarily late summer to early fall (Leiberg 1900; Agee 1993; Skinner et al. 2009, 2018). In southern California the fire season is much longer and fires in mixed evergreen can occur in dry springs, summer, fall, and even early winter. The close proximity of mixed evergreen and chaparral stands in southern California also results in a much higher potential for high-severity fire (Minnich 1988). Fire season length varies depending on fuel moisture patterns and trends, and lower elevations and south- and west-facing slopes tend to burn earlier in the season (Show and Kotok 1929; Frost and Sweeney 2000). Fire size, fire severity, and the total area burned each year are driven to a great extent by topography and landscape conditions. Moderate and high-severity fire effects increase in importance with moisture, elevation, stand density, and the presence of fire-intolerant species. Fires tend to burn at lower severities on lower slopes and north- and east-facing aspects and higher severities on mid and upper slope positions and south- and west-facing aspects (Taylor and Skinner 2003; Alexander et al. 2006). Areas of high-severity fire tend to be more prevalent in denser mixed evergreen forests and in stands with high shrub and hardwood cover (Minnich 1988; Thompson and Spies 2009; Miller et al. 2012b; Estes et al. 2017).

Many of the hardwood tree species in mixed evergreen forest are thin-barked and easily damaged or killed by fire, but often prodigious postfire resprouting leads to relatively swift recovery (Plumb and McDonald 1981). Mixed evergreen conifers other than big-cone Douglas-fir do not resprout and fire intense enough to kill conifers shifts dominance to hardwoods for decades at least. Warming temperatures, rising winter precipitation, and increasing nutrient inputs from air pollution also enhance hardwood competitiveness with conifers, and all of these are projected under most climate models for the NAMCZ (Bedsworth et al. 2018). A trend analysis between the 1930s and 2000s demonstrated strong increases in hardwood stem densities in mixed evergreen forest and related forest types (McIntyre et al. 2015). Lenihan et al. (2008) used a dynamic vegetation-fire-climate model to project a large increase in the area of mixed evergreen and related forest types by the end of the twenty-first century. In contrast, Sudden Oak Death (SOD), caused by the introduced water mold *Phytopthora ramorum*, is wreaking havoc in coastal mixed evergreen forests between 35° and 40° latitude. Adult tanoaks are being extirpated from many stands and live oaks and black oak have also been heavily affected. Expansion of SOD has recently slowed, but warmer, wetter future climatic conditions projected for coastal forests indicate that spread of the pathogen could increase by up to tenfold by 2030 (Meentemeyer et al. 2011). Important interactions exist between SOD-related tree mortality and severity of subsequent fire (Metz et al. 2011), suggesting that increasing fire in mixed evergreen forest and adjoining redwood stands (Metz et al. 2013) could have unforeseen consequences.

9.2.3.2 Redwood Forests under the Native American Fire Regime

Redwood (*Sequoia sempervirens*) is found in groves of varying size along the northwestern and central coasts of the NAMCZ. Distribution of redwood is closely connected to the occurrence of summer fog. “Maritime” redwood groves coexist with a suite of fire-intolerant conifers that stretch down the coast from the Pacific Northwest (redwood is salt intolerant and does not grow along the coast itself. These wet coastal forests are treated briefly below (Sect. 9.2.5.1; Chap. 10). Inland of these wet coastal forests, late summer and early fall conditions are dry enough in redwood groves to allow fire (red stippled polygons in Fig. 9.5). Lightning strike densities are very low in these areas (van Wagtendonk and Cayan 2008) so the rich fire history uncovered by fire scar studies in these groves is almost entirely due to Native American fire use. In addition, the complex topography of the region leads to a complicated landscape mosaic that embeds redwood groves in many other forest types and fire regimes (Varner and Jules 2017). Redwood is an extraordinary tree. Adults are the tallest trees in the world, they can live thousands of years, and the species has no known mortal pathogens or pests. In addition, the species is very fire-tolerant (thick, spongy bark), and when aboveground parts are damaged, killed, or buried (by flood sediments), redwood resprouts readily from epicormic locations and the root crown. The species also furnishes a beautiful, rot-resistant wood and heavy logging over the last 150 years has left very few old-growth stands.

Historical FRIs in inland redwood stands averaged ~15–40 years (Van de Water and Safford 2011), but there are stands near Native American cultural sites that supported astonishingly frequent fire (nearly annually; Stephens et al. 2018b). Redwood litter is surprisingly flammable and can easily carry fire under the right conditions. Although these sites are extremely productive, fuel depths tend to equilibrate at <4 cm depth due to high decomposition rates. Based on the rarity of lightning-ignited fire in the redwood belt it seems clear that redwood is not “fire-dependent”, but its dominance on the ground certainly benefits from the fact that its seedlings survive best on open mineral soil (Griffith 1992) and most of its late-seral competitors are highly sensitive to fire. As a result, prescribed fire is increasingly employed in redwood forest management (Engber et al. 2017). Major ecological changes in redwood forests include a major reduction in fire due to the general loss of Native American ignitions, a decline in both fog occurrence and late summer humidity and fuel moistures due to climate warming, and the introduction of SOD (Stephens et al. 2018b). SOD causes high mortality of oaks and tanoak and can elevate dead fuel loads and fire intensity such that adult redwoods are killed by fire, an otherwise rare occurrence (Metz et al. 2013). Redwoods are highly resilient trees and it is uncertain how their populations will respond to ongoing global change.

9.2.3.3 Upper Montane Forests

Upper montane forests occupy the transition zone between lower montane frequent-fire forest types like YPMC and higher elevation subalpine forests and woodlands. Throughout most of the NAMCZ, upper montane forests are distinguished by the presence of red fir, which co-occurs with white fir and lodgepole pine at lower elevations, and mountain hemlock, lodgepole pine, and western white pine at higher elevations (Fig. 9.11). So defined, upper montane forests occur in most of the dominant mountain ranges of the region except southern California, Baja California, and the White, Inyo, and Warner Mountains where red fir is absent. Climatically, upper montane forests grow mostly above the "50:50 line", i.e., the elevation above which the proportion of snow to rain exceeds 1:1 (and also the elevation of the average freezing line in winter storms) (Safford and Van de Water 2014). Winter snowpack in these forests is typically very deep.

The fire regime in upper montane forests before Euro-American settlement was highly variable, with fires exhibiting considerable variation in size, frequency, and severity. Fires were predominantly slow-moving surface fires due to the presence of natural terrain breaks, heavy snowpack, cool and moist growing conditions and compact fuel beds (Skinner 2003; van Waterdown et al. 2018a, b). Passive crown fires would occur on occasion, however they were often associated with dry and windy conditions, pockets of fuel created by tree mortality, and patches of montane chaparral in the understory (Skinner 2003).

Prior to fire suppression in the early twentieth century, fires were moderately frequent in upper montane forests with mean FRIs ranging 30–80 years across the NAMCZ (Van de Water and Safford 2011; Meyer and North 2019; Coppoletta et al. 2021). FRIs generally increased with elevation, with longer return intervals in more mesic, high-elevation forest types (e.g., red fir and mountain hemlock), due to slower fuel accumulation and weather conditions that were less conducive to ignition and fire spread (Skinner 2003; Meyer and North 2019). FRIs also increased with latitude, with mean estimates ranging from around 33 years in the southern and central Sierra Nevada (Meyer and North 2019) to 43 years in the southern Cascades and Klamath mountains of northern California (Coppoletta et al. 2021). The absence of fire over the past 50–100 years in many present-day upper montane forests has resulted in a lengthening of FRIs, with the majority of stands currently characterized by intervals that are 1–2 times longer than expected under the pre-Euro-American settlement fire regime (Meyer and North 2019).

Most fires in upper montane forests occur during the short dry season, in late summer or early fall when lightning strikes are more common (van Wagtendonk et al. 2018a, b). Fires were historically small (typically <4 ha) and patchy, with more extensive fires limited to years with below average regional precipitation and/or coincidence with severe fire weather (Taylor 2000; Meyer and North 2019). Examination of fires over the past 100 years suggests that fire size and seasonality are similar between historical and contemporary upper montane forests in the NAMCZ (Meyer and North 2019; Coppoletta et al. 2021). However, climate projections also suggest that future fires may be larger and occur both later and earlier in

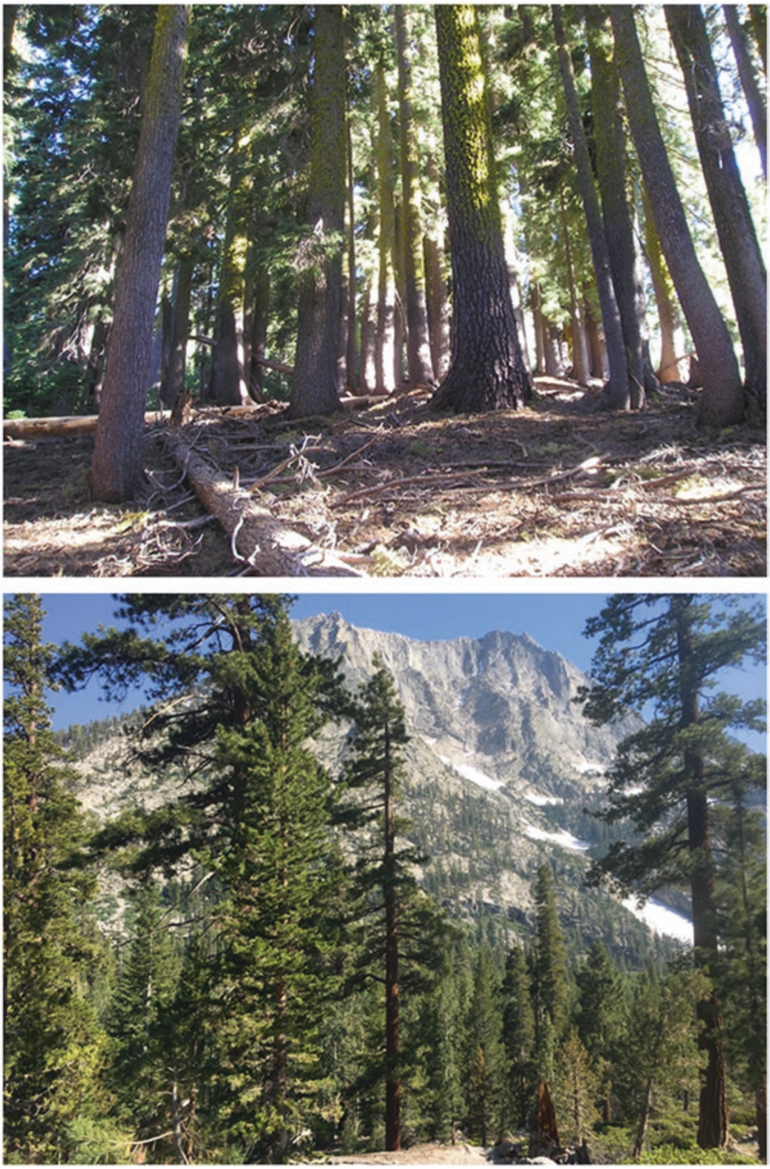

Fig. 9.11 Upper: Long-unburned Shasta red fir (*A. magnifica* var. *shastensis*) stand, Siskiyou Mountains, Klamath National Forest. This site receives high precipitation and fuel accumulations in such forests are high in the years between fires; Lower: Upper montane forest, southern Sierra Nevada. Main species are lodgepole pine, western white pine, red fir, and Jeffrey pine. These rocky and dry-summer sites are notably patchier than forests in the Klamath/Siskiyou Mountains or northern Sierra Nevada and fire effects are less severe. (photos Hugh Safford)

the season due to decreasing snowpack, warming temperatures, and decreases in the snow to rain ratio, which result in earlier drying of fuels and extension of the fire season (Westerling et al. 2006).

Fire effects in upper montane forests are often characterized as moderate or "mixed" severity (although there is ambiguity associated with the concept of "mixed-severity" [Collins et al. 2017]). Fire severity patterns can be highly variable, both among years and within individual fires. Estimates based on historical data and contemporary reference sites in the NAMCZ indicate that the fire regime before Euro-American settlement was typically dominated by low- to moderate-severity fire effects, with high-severity fire relatively uncommon and limited in size (Laacke and Tappeiner 1996; Meyer and North 2019; Coppoletta et al. 2021). Variation in fire effects in upper montane forests can be related to fire weather conditions, fuel accumulation rates, and vegetation structure and composition (Fig. 9.11). Mature red fir develops thick bark making it more fire-tolerant than many of its associated species (Chappell and Agee 1996). Fire severity patterns in recent fires have slightly higher proportions and patch sizes of high-severity fire than presettlement fires and are approaching the upper limit of estimates from historical upper montane forests in the NAMCZ (Meyer and North 2019; Coppoletta et al. 2021).

Lengthening FRIs over the past century, in combination with past management practices (e.g., timber harvest), has led to noticeable changes in the structure of upper montane forests in the NAMCZ, although these are less accentuated than in YPMC forests at lower elevations. This includes increased tree densities, particularly in the smaller size classes, and concurrent decreases in the density of large diameter trees. Fire exclusion has also resulted in the simplification of forest structure, at both the stand and landscape scale, with shifts away from a heterogeneous, partially open canopy structure to one characterized by more continuous closed canopy conditions (Meyer and North 2019; Coppoletta et al. 2021). Late successional conditions, such as those that characterize much of the upper montane forest in protected areas, are important to a number of management focus species such as the American marten (*Martes americana*) (Kirk and Zielinski 2009), but such habitats are increasingly at risk under current trends in management, fire, and climate.

Upper montane forests of the NAMCZ are likely to experience substantial changes in fire regimes and forest composition and structure in coming decades. Modeling projects that future fires in these high elevation forests will be larger and more frequent due to decreasing snowpack, warming temperatures, and decreases in the snow to rain ratio, which result in earlier drying of fuels and extension of the fire season (Restaino and Safford 2018; Dettinger et al. 2018). Fire severity may also increase due to combined effects of increased moisture stress, greater fuel continuity, and forest densification associated with climate change and fire exclusion (Meyer and North 2019). Increased moisture stress, bark beetle outbreaks, and pathogens associated with climate change are already amplifying tree mortality in high elevation forests (Meyer et al. 2019). In response, there is increasing interest in the potential to employ forest and fire management strategies and tactics from the YPMC belt in an effort to buffer upper montane forests from drastic ecological changes (Meyer and North 2019; Coppoletta et al. 2021).

9.2.3.4 Pinyon-Juniper Woodlands

Pinyon and juniper woodlands in and adjacent to the NAMCZ (Chap. 11) are found in the high elevations of the Mojave Desert, in the southern California mountains and east of the Sierra Nevada crest. Species composition ranges from stands of pure pinyon pine (mostly *Pinus monophylla* but also *P. edulis* or *P. quadrifolia*) to pinyon mixed with juniper *(Juniperus occidentalis, J. osteosperma* or *J. californica*), oaks (*Quercus turbinella, Q. berberidifolia* or *Q. chrysolepis*), or Mojave yucca (*Yucca schidigera*; Bolsinger 1989). On average, the juniper species are somewhat more drought-tolerant than the pinyon species, with the driest and hottest sites tending to support pure juniper stands (Wigand 2017). Woodland structure varies with disturbance history, topography, hydrology, substrate and time since tree establishment (Miller et al. 2019). Pinyon-juniper woodlands provide densely-nutritious food sources for wildlife and indigenous peoples in the form of pine nuts and juniper berries, often surrounded by landscapes where high-value food resources are scarce.

Climatic variability and management have both induced marked geographic shifts in pinyon-juniper woodland distribution (Cole et al. 2013). From a core of temporally persistent woodlands – centered on low productivity areas with shallow, rocky soils and low fire frequencies (Miller et al. 2019) – pinyon-juniper expanded into deeper-soiled sites in the 19th and early 20th centuries. It is thought that a wetter climate and decreased fire frequency, related to livestock grazing reducing potential fuels, created favorable conditions for cone production and seedling establishment (Barger et al. 2009). Livestock grazing also provided tree seedlings with a competitive advantage after understory plants were selectively removed (Miller and Rose 1999; Touchan et al. 1995). Recently, pinyon-juniper woodlands have experienced widespread mortality and range contraction due to multiannual droughts in the western US (Gaylord et al. 2013). Management activities (cutting, chaining, burning) have also reduced pinyon and juniper populations where their expansion has impacted livestock forage or encroached on important greater sage grouse (*Centrocercus urophasianus*) habitat. Such activities have been shown to have marked and long-term effects on non-target wildlife communities (Gallo et al. 2016).

Pinyon and juniper species are sensitive to fire, with relatively thin bark and low canopies. High mortality is characteristic of most fires in all but the sparsest woodlands. Postfire succession is slow. After fire, tree densities remain low until c. 40–50 years postfire, when tree cover begins to recover (assuming presence of a seed source and nurse shrubs; Wangler and Minnich 1996; Miller and Tausch 2001). In the early stages of postfire recovery, juniper species typically precede pinyon species in establishing but with enough precipitation pinyon may overtake juniper in dominance over time (Wangler and Minnich 1996; Abella et al. 2012). Understory shrubs proliferate in the years following the fire (peaking between 30–50 years) and then decrease in cover as trees establish (Roundy et al. 2014; Williams et al. 2017). Herbaceous understory cover peaks within the first 10 years following fire and declines with succession of woody species (Miller and Heyerdahl 2008; Bates et al. 2017).

Fire regimes in pinyon-juniper systems vary tremendously, depending on geographic and topographic position, climate, soil substrate, understory composition and structure, tree canopy closure, and ignition frequency. Historically, fires in persistent woodlands, typified by sparse understory, tended to occur every few centuries and were stand-replacing, patchy, small (<10 ha), and driven by extreme weather (Baker and Shinneman 2004; Floyd et al. 2017). Given the heterogeneous nature of the typically small fires that occurred historically in woodlands, they can be described as having a high-severity or mixed-severity fire regime depending on the scale analyzed (Slaton and Stone 2013). In Nevada and Utah woodlands, which are comparable to the eastern California populations, fires in persistent woodlands occurred every 187–502 years (Slaton and Stone 2013). In southern California, the fire rotation period was estimated at 480 years (Wangler and Minnich 1996). Given the variation in fire severity (mixed or replacement) and length of FRIs, pinyon-juniper woodlands in and adjacent to the NAMCZ can be classified into FR III, IV, or V depending on site characteristics.

Large fire occurrence declined beginning in the late nineteenth century, likely driven by livestock-impacts on fuels and cessation of Native American cultural burning (Marlon et al. 2012). In the last 30 years the trend has reversed, with increasingly large high-severity fires affecting most of the pinyon-juniper range (Floyd et al. 2017; Board et al. 2018). Increases in fire occurrence are positively correlated with tree density, warmer and drier conditions, and the presence of non-native annual grasses, especially cheatgrass (*Bromus tectorum*) (Brooks et al. 2004; Westerling et al. 2006). Such grasses can create a monoculture that is more continuous and cures earlier in the year than the native understory, creating an easily ignitable fuel source that carries fire over large areas. As these grasses quickly re-occupy a site following fire, they perpetuate a fire cycle that inhibits the re-establishment of trees (Fig. 9.12; Nowak et al. 1994). In eastern California, >20% of pinyon-juniper woodlands show a current FRI of ≤55 years (compared to the historical FRI of 187–502 years; Slaton and Stone 2013). In southern California populations, 20% have a FRI of ≤55 years (compared with the historical FRI of ~50–250 years; Van De Water and Safford 2011). Increasing fire frequencies in these woodlands and widespread drought-related pinyon mortality (Mueller et al. 2005; Shaw 2006) forebode a future range contraction in pinyon-juniper woodlands.

9.2.4 *Fire Regime IV: Moderately Frequent Fire, Mostly Moderate- and High-Severity*

9.2.4.1 Chaparral and Serotinous Conifers

Chaparral is widespread throughout the NAMCZ, extending from southern Oregon, through the foothills of the Sierra Nevada and Coast Range, to northern Baja California, Mexico. Chaparral is the dominant vegetation in the orange areas in Fig. 9.5. Chaparral reaches its peak abundance in southern California in the

Fig. 9.12 Former single-leaf pinyon woodland south of Reno, just east of the NAMCZ. Site burned under severe fire weather conditions and was invaded by exotic and native grasses after fire. The stand in the center of the hillside was skipped over by the wind-driven fire. Since the photo was taken, the site has burned again through grassy fuels. Ignition densities are high in this area due to a nearby highway and expanding human habitation (photo Hugh Safford)

foothills (below 1400–1600 m) of the Transverse and Peninsular mountain ranges (Rundel 2018). Chaparral is dominated by evergreen sclerophyllous shrubs with the occasional presence of – often serotinous – conifers, including knobcone pine (*Pinus attenuata*), Coulter pine (*P. coulteri*), big-cone Douglas-fir, and various species of cypress (including *Hesperocyparis forbesii*, *H. stephensonii*, *H. macnabiana*, *H. sargentii*, etc.); and hardwoods (especially oaks, *Quercus* spp.), California bay, and California ash [*Fraxinus dipetala*]) in more mesic sites. Fire in chaparral and associated serotinous conifers is characterized by stand-replacing fire at relatively moderate frequencies (mean pre-Euro-American settlement FRI of 55 years; range 30–90 years; van de Water and Safford 2011), allowing for formation of a closed canopy between events. Fire occurrence in coastal chaparral stands has been driven largely by human ignitions since the arrival of Native Americans, as lightning strike densities are very low in these areas (Chap. 1, Fig. 1.2, Table 1.1; van Wagtendonk and Cayan 2008). Inland stands have somewhat more direct exposure to lightning ignitions, but large fires may have often been driven by high elevation fires or smoldering snags or logs that were reignited and driven downhill to the west by fall and winter desert winds (the so-called "Santa Anas"; Minnich 1988). Similar, but less prevalent winds occur in northern California ("north winds" or "diablo winds") and have driven huge fire-caused losses of life and property in the last few years (notably 2015, 2017, and 2018). Most fires in chaparral and serotinous conifers occur under standard warm and dry summer and early fall conditions, but most large and economically damaging fires occur during late season wind events (Safford et al. 2018).

Postfire vegetation recovery of shrubs and serotinous conifers is typically endogenous and requires little recruitment from neighboring unburned areas. Chaparral species can be classified into three postfire regeneration strategies: obligate seeding, obligate resprouting and facultative seeding. Obligate seeding species are typically killed by fire and rely on a dormant seedbank that is stimulated to germinate by smoke, heat or chemical byproducts of fire (Keeley and Fotheringham 2000). Shrubs belonging to this group include many species within the genera *Ceanothus* and *Arctostaphylos*. Serotinous conifers and herbaceous annuals also respond to fire by engaging a dormant seedbank. Various serotinous (or partly serotinous) conifers intermix with chaparral shrubs and are subject to the same stand-replacing burning conditions. Obligate seeding species recruit heavily the first year following fire, however they experience little recruitment during fire-free periods (Keeley et al. 2006).

Aboveground portions and seeds of obligate resprouters are typically killed by fire, therefore resprouting is the only mechanism for postfire recovery. Resprouting is initiated from the root crown or specialized underground structures (e.g., lignotubers). Obligate resprouting shrubs in chaparral include scrub oak (*Quercus berberidifolia*), toyon (*Heteromeles arbutifolia*), some *Arctostaphylos* species, and various species in Rhamnaceae. Big-cone Douglas-fir can resprout from epicormic buds in the branches and upper bole (Gause 1966) when subjected to low or moderate-intensity fire, but may be killed by crown fire (Fig. 9.13; Minnich 1977). During fire free intervals, obligate resprouters can recruit into the shade of established chaparral shrubs. Facultative seeding species like chamise and red shanks (*Adenostoma fasciculatum* and *A. sparsifolium*) are capable of regenerating from both fire triggered seeds and crown resprouting, allowing these species to successfully respond to a variety of postfire conditions.

Stand-replacing fire in chaparral has many important functions, most notably it serves to reset succession and return vegetation to an early-seral stage. There is a postfire pulse of short-lived species that reach peak abundance within the first few years following fire, including fire followers (aka pyrophytes, species whose germination requires fire) and other annuals that respond to the sudden availability of light, as well as geophytes that have not flowered since the last fire. In these first postfire years species richness can be very high (Keeley et al. 1981, 2005). The explosion of postfire species is more muted in low productivity sites where light was not as limiting before fire (Safford and Harrison 2004). Postfire recruiters can exhibit impressive flower displays that result in the deposition of seeds that remain dormant in the soil until the next fire. These species give way in a few years to woody obligate resprouting, obligate seeding and facultative seeding species and become largely restricted to the soil seed pool once the shrub and tree overstory begins to develop (Keeley et al. 2005). Full recovery of the shrub canopy tends to occur 8–15 years after fire (Keeley et al. 1981; Rundel 2018).

Fig. 9.13 Chaparral and embedded conifers in the Santa Ana Mountains, Cleveland National Forest, southern California. Left: Knobcone pine, a serotinous species, on ultramafic rock ("serpentine"), which reduces vegetation productivity and thus fire frequency. Note the smog, a common occurrence on windless days (photo Hugh Safford). Right: Burned chaparral and big-cone Douglas-fir in the Holy Fire (2018). Some of the Douglas-fir are crown sprouting but most were killed by the fire due to high shrub cover in the understory. This area has been severely degraded by air pollution, invasive species, and extraordinarily frequent fire. The watershed surrounding the photo site was burned in 1925, 1954, 1956, 1959, 1968, 1980, 1987 and 2018 (although not all fires affected the photo site itself). As a result, chaparral recovery will be limited (photo N. Molinari)

When elements of the fire regime are greatly altered, chaparral and serotinous conifer stands may be sent down different successional pathways. Depending on conditions that support tree growth and the specific sequence of fire events, long-unburned ("old-growth") chaparral can succeed to woodland or forest, for example after oak invasion and overtopping of shrubs after very long fire-free intervals (Callaway and Davis 1993). Today a much more common successional path converts areas dominated by chaparral and serotinous conifers via highly frequent fire into more disturbance-tolerant vegetation dominated by ruderal, usually non-native herbs. Today, the extensive wildland-urban interface and high human population density in southern California has greatly enhanced ignitions, resulting in excessive fire (Figs. 9.13) (Keeley and Fotheringham 2003; Syphard et al. 2007; Safford and van de Water 2014) and heightened wildfire risk to surrounding human communities (Jin et al. 2015). In highly fire-prone areas, obligate seeding shrubs and serotinous conifers are vulnerable to immaturity risk, where fire occurs before they have reached maturity or established a sufficient seedbank to recolonize (Zedler 1995). Overly frequent fire is a major threat to the survival of multiple serotinous or partly serotinous trees in the NAMCZ – such as *Pinus torreyana*, *P. muricata*, *Hesperocyparis forbesii*, *H. stephensonii*, and outlying populations of various other species – with the problem focused in southern California and northern Baja California. Overly frequent fire is also a threat to big-cone Douglas-fir, which is not serotinous but often grows in a chaparral matrix. Longer fire-free periods allow for canopy development and reduction of understory shrubs by shading, but high frequency chaparral fires are burning big-cone Douglas-fir stands so severely that they cannot resprout (Fig. 9.13). As a result of these trends, full fire suppression is a necessity from both management and ecological viewpoints in most chaparral-dominated landscapes (Safford et al. 2018).

Non-native annual grasses from other Mediterranean-climate regions are ubiquitous in California and often become abundant in lowland areas where woody species cover has been degraded by frequent fire (Zedler et al. 1983), nitrogen deposition (Valliere et al. 2017) or grazing (Tyler et al. 2007). Non-native annual grasses senesce and cure in the early summer and remain highly flammable until rain resumes in late fall. During this time, they increase the likelihood of ignitions and further contribute to the problem of excessive fire and ecosystem degradation (D'Antonio and Vitousek 1992; Fusco et al. 2019).

Chaparral and serotinous conifers in the NAMCZ grow mostly at warmer lower elevations that experience a 5–7+ month long dry season and are subject to great interannual fluctuations in precipitation. As such they are tolerant of climatic extremes, but postfire recovery can be impeded by exceptional droughts – as occurred from 2012–2016 – that leads to depletion of carbon stores and affects even the mostly deeply rooted resprouters (Pratt et al. 2014). Frequent droughts are also thought to be a major driver of the recent trend toward larger fires in the chaparral belt, the reasoning being that higher proportions of dead fuels in the canopy are driving more intense fire and farther-reaching windcast of embers (Keeley and Zedler 2009). Climate change models project enhanced precipitation variability, rising temperatures, increased occurrence of multi-year droughts, and an expanded fire season. These trends seem likely to accelerate loss of chaparral in areas with a surfeit of human ignitions (Molinari et al. 2018).

9.2.4.2 Subalpine Forests and Woodlands

Subalpine forests and woodlands (hereafter "subalpine forests") are distributed throughout high mountain regions of the NAMCZ, although they are absent in Baja California (Millar and Rundel 2016). Subalpine forests are the dominant vegetation in light blue areas in Fig. 9.5. These forests occur above the upper montane zone and are dominated by one or more high elevation conifers including whitebark pine, foxtail pine, limber pine, lodgepole pine, and mountain hemlock, sometimes in combination with red fir, western white pine, and Sierra juniper (*Juniperus grandis*). The Klamath Ranges contain isolated groves of subalpine fir (*Abies lasiocarpa*), Pacific silver fir (*A. amabilis*), Brewer spruce (*Picea breweriana*), Engelmann spruce (*P. engelmannii*), and Alaska yellow cedar (*Cupressus* *nootkatensis*) (Sawyer et al. 2009). The southern range limit of nearly all of these subalpine tree species occurs in the NAMCZ. Subalpine forests experience a prolonged winter snowpack, short growing season, cool summer and cold winter temperatures, and precipitation that mainly occurs as winter snow (Agee 1993; Millar and Rundel 2016). These forests are frequently interspersed with exposed bedrock, snowfields, scrub vegetation, meadows, and riparian ecosystems and can vary structurally from patches of widely-spaced trees to denser stands in more productive sites (Fites-Kaufman et al. 2007) (Fig. 9.14).

Pre-Euro-American settlement fire regimes in the NAMCZ subalpine forests were characterized by long FRIs (generally 80–200 years or longer), with relatively shorter FRIs (generally 40–70 years) recorded in lodgepole pine stands (Meyer and North 2019; Coppoletta et al. 2021). The long FRI in subalpine forests is likely due to slower rates of fuel accumulation, discontinuous fuel loading, shortened fire seasons, and weather conditions that limit fire spread (Agee 1993; Skinner 2003; van Wagtendonk et al. 2018a, b). Fire rotation estimates in these forests typically range from 450–2100 years (Meyer and North 2019; Coppoletta et al. 2021), although longer fire rotations up to ~21,000 years have been recorded near tree line (van Wagtendonk 1995). Shorter fire rotations of ~50 to 150 years were observed in lodgepole pine and mountain hemlock-red fir stands that occurred at lower elevations in the subalpine zone. Most subalpine forests have missed one fire cycle at most and have a low FRI Departure (FRID) (Safford and van de Water 2014). In comparison, lower elevation subalpine stands (e.g., lodgepole pine, western white pine, mountain hemlock-red fir forests) have generally missed two fire cycles and have a moderate FRID more characteristic of upper montane forests. In recent decades the frequency and extent of wildfires has increased in high elevation forests of the region (Schwartz et al. 2015), leading to reductions in FRID within these subalpine landscapes.

Both pre-Euro-American and contemporary subalpine forests are classified as climate-limited fire regimes, whereby fire occurrence depends primarily on climate or weather limitations rather than fuel conditions (Agee 1993; van Wagtendonk et al. 2018a, b). In the Late Holocene (last 4000 years), fire activity in California's subalpine forests was primarily driven by changes in climate, including dynamics of the El Niño–Southern Oscillation (Hallett and Anderson 2010). Recent warming climate trends may also be responsible for changes in fire patterns within the region's subalpine forests (Schwartz et al. 2015). Most fires in the subalpine zone were historically very small (<4 ha) but variable and occurred during late summer or fall, a pattern that is mostly conserved in the contemporary period (Meyer and North 2019; Coppoletta et al. 2021).

Pre-Euro-American settlement fire severity patterns were variable in NAMCZ subalpine forests and are classified primarily as low or mixed-severity (the latter characterized as more even proportions of low-, moderate-, and high-severity fire effects), depending on the forest type and climate and weather conditions (Meyer and North 2019; Coppoletta et al. 2021). High-elevation white pine (e.g., whitebark, foxtail) forests typically experience low- or mixed-severity fire, and mesic lodgepole pine or mountain hemlock forests are generally characterized as

Fig. 9.14 Upper: Mountain hemlock recruiting into former perennial snowfield, Eldorado National Forest, central Sierra Nevada (photo Hugh Safford); Lower: Meadow Fire (2014) in Yosemite National Park, in upper montane and subalpine forest. Dominant species are Jeffrey pine and red fir on the slope, lodgepole pine, red fir, and mountain hemlock on the plateau. The event occurred under high winds from the southwest (to the photo's right). Most of the photo foreground and middle-ground burned, indicating the high heterogeneity typical of fires in high elevation forest in the Sierra Nevada (photo E. Brodie)

mixed-severity (Agee 1993; van Wagtendonk et al. 2018a, b). In most subalpine stands dominated by white pines, sparse surface and canopy fuels, natural terrain breaks, and low tree densities and biomass facilitate slow-moving surface fires with occasional torching of individual trees or tree clusters. In contrast, denser and more productive stands of mesic lodgepole pine, mountain hemlock, and subalpine fir forests support more frequent tree torching or localized crown fires (Meyer and North 2019). More severe fire effects and larger patches of high-severity fire occur in subalpine forests during exceptional drought and high wind events that facilitate sustained crown fires, particularly in dense stands with high fuel loading (Meyer and North 2019). Similar to fire frequency and seasonality trends, contemporary fire severity patterns have not changed noticeably from pre-Euro-American settlement patterns in NAMCZ subalpine forests (Meyer and North 2019; Coppoletta et al. 2021). This, combined with the short fire season, denser, short-needled fuels, low fuel continuity, and high lightning incidence has led to widespread implementation of wildland fire use/fire management for resource benefit in subalpine forests, especially in national park and national forest wilderness areas in the southern Sierra Nevada (Meyer 2015).

Although fire exclusion has had minor impact on forest structure or composition in the NAMCZ subalpine zone, forest densification *is* occurring in these forests, but it is driven by warming temperatures, decreasing snowpack, and an expanding growing season (Fig. 9.14). It is also nearly entirely confined to the young age classes, and older trees are actually dying at an increasing rate (Dolanc et al. 2013). Over the last few decades annual burned area has increased markedly in high elevation forests (Schwartz et al. 2015), outbreaks of mountain pine beetle have grown (Meyer et al. 2016), and white pine blister rust (*Cronartium ribicola*) has continued its southward expansion. As more seedlings and saplings survive in these progressively less harsh landscapes, as more of the older trees die, and as the climate warms, these trends suggest that significant change is in store for subalpine forests in the NAMCZ over the coming decades.

9.2.5 Fire Regime V: Very Infrequent, Mostly High-Severity Fire

9.2.5.1 Wet Coastal Forests

Wet coastal forests in the NAMCZ (dark blue areas in Fig. 9.5) are a southern extension of much more extensive forests in the Pacific Northwest (Chap. 10). These forests occur in a narrow coastal strip – mostly only a few km wide – from southwestern Oregon to near Ft. Bragg, about 200 km north of San Francisco. Dominant species include Sitka spruce, Douglas-fir, western hemlock, western red cedar, grand fir (*Abies grandis*) and redwood. Other than redwood and Douglas-fir, these species are all very fire sensitive, and the fire scar record indicates that pre-Euro-American FRIs were long, averaging 180–550 years in California (Van de Water and Safford 2011). FRIs today are probably similar, although some areas with expanding human settlements along the coast could see increased fire frequencies in the future, especially as the climate warms and fog becomes less dependable.

9.3 The Future of Fire and Fuels Management in the NAMCZ

Table 9.2 summarizes the major disturbances and stressors affecting terrestrial ecosystems in the NAMCZ; nearly all are either anthropogenic in origin or magnified by human actions. For example, even among factors that are at their root "natural" – native pine beetle outbreaks, periodic droughts, secular changes in climate – human contributions to greenhouse gases and human management practices (e.g., certain logging and planting practices, fire suppression policies) have greatly increased the severity of the threat and/or reduced the resilience of affected ecosystems. All of these threats also interact with fire at some level, hence management decisions made about fire use, fire suppression, fuel management, and other fire-related issues will impact nearly every box in Table 9.2.

The climate in the NAMCZ is warming rapidly, summers are drying, and interannual precipitation variability is rising (Safford et al. 2012a; Bedsworth et al. 2018). Fire trends in most ecosystems in the region are strongly positive – more burned area, more severe fires, bigger fires, more economic damage (Fig. 9.15) – and modeling suggests more of the same for the next 50+ years (Dettinger et al. 2018; Restaino and Safford 2018). Taken on their own, not all of these trends are injurious in the ecological sense. For example, in FR I forest types the lack of fire over the last century is perhaps the major challenge to sustainability. In this sense, more fire in these ecosystems should be welcomed, but the problem is that it is too often the wrong kind of fire (Pyne 2019). In other ecosystems, for example chaparral and serotinous conifers, more fire is decidedly not the answer (Safford and Van de Water 2014; Keeley and Safford 2016).

Although federal resource management agencies technically transitioned from full fire exclusion to fire management in the late 1960s and early 1970s, almost all wildfires today continue to be suppressed as rapidly as possible. In some ecosystems, especially wet coastal forests, chaparral and serotinous conifers, and many areas of pinyon-juniper woodland, this policy is ecologically appropriate (Safford et al. 2018). However, on much of the forested landbase in the NAMCZ fire suppression has been an ecological disaster, and the 'fire paradox' (putting out every fire in the short-term leads to bigger and more destructive fires in the long-term; Moreira et al. 2020) has put us in a tricky management place, where current forest structure, composition, and fuel loading interact with other growing threats like drought, pests and pathogens, and air pollution to create a classic wicked problem (Balint et al. 2011).

Table 9.2 Major disturbances and stressors affecting terrestrial ecosystems in the NAMCZ

Ecosystem	Fire	Invasive species	Pathogens	Insects	Warming climate	Drought	Other
Oak woodland	Lack of low-severity fire from fire exclusion increases stand densities and fuel, more severe fire	Woodland understory almost entirely converted to exotic species	Some SOD-driven mortality in wetter sites	Gold spotted oak borer (GSOB) killing large areas of oak in S. California, moving north	Dropping groundwater table is killing some stands	High mortality in some oak woodlands due to profound drought	Most oak woodland lost to agriculture and urban expansion. Low regen in deciduous spp. due to grazing, seed predation, invasive plant competition, & warming climate
Yellow pine and mixed conifer	Lack of low & moderate severity fire due to fire exclusion has increased stand densities and fuels, changed spp. composition, increased severe fire risk	Invasive annual grasses increase fire hazard & decrease seedling survival; invasion by European shrubs in wetter areas	White pine blister rust is killing sugar pine; root rot in firs interacts with beetles to kill trees; many other fungal pathogens	Pine beetle and fir engraver eruptions have killed hundreds of millions of trees in 2000s; GSOB is a developing problem	Increased water stress increases susceptibility to other mortality agents; fire season has expanded by up to two months	Deep droughts have interacted with other factors to drive high local mortality; major impacts on seedling survival	Ozone pollution is a major stressor for pine species, especially in southern California; interacts with other mortality factors. Tree harvest in 19th and 20th centuries removed most large, disturbance-tolerant trees
Meadows and grasslands	Fire suppression in montane meadows has promoted conifer invasion	Lower elevation grasslands are dominated in most areas by exotic species			Groundwater dropping, causing transition to more xeric conditions, soil drying and compaction	Droughts in interaction with warming summers can kill perennial grasses and forbs	Most grasslands lost due to agriculture and urban expansion. Heavy grazing compacts soils, leads to stream down-cutting, changed hydrology, changed species comp

(continued)

Table 9.2 (continued)

Ecosystem	Fire	Invasive species	Pathogens	Insects	Warming climate	Drought	Other
Mixed evergreen forests	Moderate effects of reduced fire frequency from fire suppression, increased fire severity where SOD has killed trees	Invasion of various exotic shrubs driven by disturbance and proximity to human infrastructure	SOD killing high numbers of oaks and tanoak in wetter areas	Gold spotted oak borer (GSOB) killing large areas of oak in S. California, moving north	Increased water stress increases susceptibility to other mortality agents	Deep droughts interact with other factors to drive some local mortality	Near major urban centers much mixed evergreen has been cleared for homes
Redwood forests	Increased fire severity where sudden oak death (SOD) has killed oaks and tanoak	Exotic shrub invasion linked to disturbance and proximity to human infrastructure	SOD killing oaks and tanoak, increasing fire severity		Increasing water stress		Reduced occurrence of coastal fog, increasing water stress and fire risk; logging removed most large trees
Upper montane forests	Lower fire frequency under fire suppression, some increases in fuels and stand densification		White pine blister rust killing western white pine	Mountain pine beetle outbreaks in lodgepole pine, fir engraver mortality in red fir	Increased water stress increases susceptibility to other mortality agents	Deep droughts interact with other factors to drive local mortality	Ozone pollution is a major stressor for pine species, interacts with other mortality factors
Pinyon-juniper woodland	Increase in fire frequency due to invasion of annual herbs, warmer and drier conditions, and increase in tree densities	Invasion of exotic annual grasses has increased fire hazard	Black stain root disease, pinyon blister rust, cubicle rot, armillaria are a few pathogens that impact pinyon	Beetle outbreaks have killed large areas of woodland; Mediterranean pine engraver is a new threat	Increased water stress increases susceptibility to other mortality agents, warmer/drier conditions increase fire frequency	Deep droughts have interacted with other factors to drive some stand-scale mortality	Grazing has reduced fuel and fire frequencies in productive sites, allowing for woodland expansion

Chaparral and serotinous conifers	Surplus of fire near urban areas, local extinction risk for obligate seeders & serotinous conifers, type conversion to grassland	Invasion of exotic annual grasses and forbs has increased fire hazard			Fire season has expanded to include almost entire calendar year	Many drought-tolerant shrubs killed by recent drought, increasing dead fuel loading and fire spread	Atmospheric N promotes exotic annuals in postfire landscapes. Much loss of chaparral to urban expansion. Grazing favors exotic annuals over native shrubland species
Subalpine forests	Increasing fire area linked to warming climate and increasing stand densities		White pine blister rust killing whitebark pine in wetter parts of the region	Beetle outbreaks have killed areas of whitebark and lodgepole pine	Increasing growing season length leads to more seedling survival, stand densification		
Wet coastal forests	Slight trend to more fire in some stands near frequent ignition sources	Exotic shrub invasion driv-en by disturbance, proximity to human infrastructure	*Phytophthora lateralis* is killing port-Orford cedar		Increasing water stress		Occurrence of coastal fog is dropping, increasing water stress and fire risk

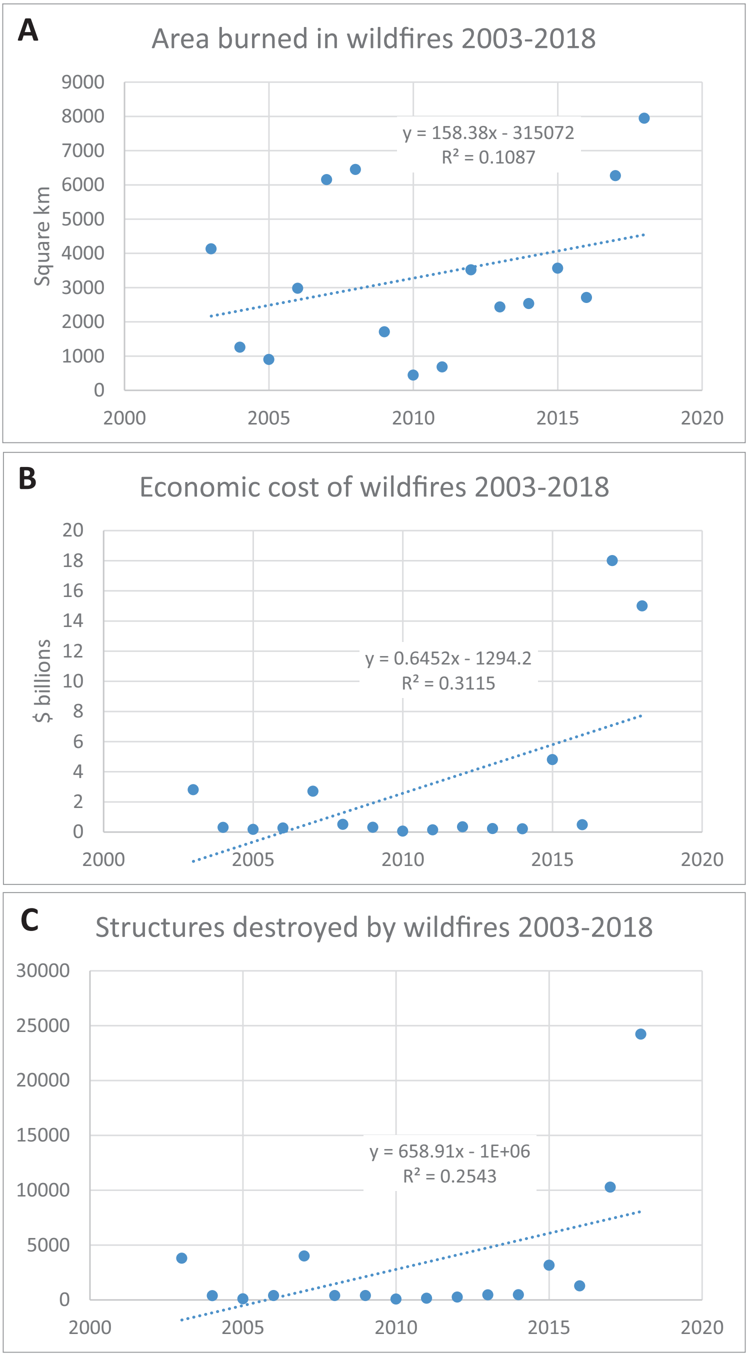

Fig. 9.15 (**a**) Area burned; (**b**) insured economic loss, and; (**c**) structures destroyed by wildfire in California (including areas outside NAMCZ), 2003–2018. Linear regression lines and formulas included. Sources: California fire perimeters (https://gis.data.ca.gov/search?source=california%20department%20of%20forestry%20and%20fire%20protection); Insurance Information Institute (https://www.iii.org/fact-statistic/facts-statistics-wildfires); Cal Fire annual incident summaries, for incidents since 2013 (https://www.fire.ca.gov/incidents/)

Other components of the wicked problem include political and cultural resistance to change, and the centrality of economics to any potential resolution. To this point, most progress has been made where political and social support for active management has been high; i.e., in high visibility locations and/or where fire risk to human life and infrastructure is high. These conditions are met primarily in the wildland urban interface (WUI), where the focus has been on mechanical and hand treatment with some implementation of pile burning and prescribed fire. Studies show that the work is largely successful in ameliorating fire behavior and reducing home losses and property damage, as long as weather conditions are reasonable and homeowners have taken at least minimal steps to protect their own property (e.g., Safford et al. 2009; Kennedy and Johnson 2014). Depending on the ecosystem and the nature of the management undertaken, some of this work also qualifies as ecological restoration (Winford et al. 2015). However, very little of the non-WUI forest landscape in the NAMCZ can be treated in this fashion because of a suite of interacting factors, including the lack of sufficient wood processing capacity in the region (especially for small diameter trees and woody biomass), the high cost of planning and implementing treatment, public environmental concerns and litigation, jurisdictional prohibitions (e.g., wilderness areas), the physiography of the landscape (e.g., steep slopes), and the geography of the road network. Taking these limitations into account, North et al. (2012) estimated that US Forest Service and US National Park Service fuel reduction – even when including escaped wildfires as a fuel reduction activity – accounts for less than 20% of the annual area burned in the Sierra Nevada before Euro-American settlement.

North et al. (2012) concluded that scaling up forest fuel reduction and restoration work in the Sierra Nevada (and the NAMCZ by extension) would require a major expansion in the use of managed wildfire. Yosemite and Sequoia-Kings Canyon National Parks have been national leaders in that realm (van Wagtendonk and Lutz 2007; Boisramé 2017a, b), and their success has been a beacon for similar efforts in neighboring national forests. Until recently however, such efforts were restricted to a few, large and remote wilderness areas. Promulgation of the 2012 Forest Planning Rule (36 CFR 219), which focuses heavily on ecological integrity and ecological and socio-economic sustainability, has provided the stimulus for more progressive, science-based land and resource management plans (LRMPs) on the national forests. To date, one new LRMP has been completed under the new rule in the NAMCZ, and two are on the verge of completion, all three are in the southern Sierra Nevada. A major advance in the new plans was the use of a fire risk assessment process (Thompson et al. 2016) to designate more than half of the landscape as fire maintenance or fire restoration zones, where naturally ignited fires will be permitted to burn under appropriate weather conditions (USDA 2019). Five national forests in the northwestern NAMCZ are poised to embark on the LRMP process in the next year, and a fire risk assessment for that region is just getting underway.

Another step in the right direction has been the notable uptick in prescribed fire use in the NAMCZ in recent years. The US Forest Service in California tripled its prescribed burn acreage between 2018 and 2019, the California Department of Forestry and Fire Protection (Cal Fire) hired 10 prescribed fire implementation crews and developed a prescribed burn monitoring program, The Nature Conservancy and other NGOs have implemented prescribed burning on thousands of ha, and the number of private citizens using prescribed fire as a management tool has skyrocketed. Formal training sessions for fire practitioners have multiplied (Bailey and Quinn-Davidson 2018), and there are now prescribed fire councils for most of California's forested areas. A major resurgence of Native American tribal interest in cultural fire use in the NAMCZ is also apparent (Lake et al. 2017), In Baja California, the Mexican forest (CONAFOR) and park services (CONANP) recently collaborated on their first formal prescribed burn in the Sierra Juarez, and plans have been made and sites selected to carry out a similar trial burn in the Sierra de San Pedro Mártir National Park.

Reasons for these positive developments are manifold. One is the emerging, science-driven understanding that fuel and carbon management in frequent-fire forests is inextricably tied to climate change, both in cause and effect (Hurteau and North 2009; State of California 2018; McLauchlan et al. 2020). Another is a developing social comfort with fire as a management tool and increasing interest in fire's role as a foundational part of traditional cultures in the NAMCZ (Lake et al. 2017; Crowder 2019). Yet another is a 2015 Memorandum of Understanding signed by the US Forest Service, Cal Fire, and a number of other state agencies and NGOs in California which committed land and fire management agencies "to increase the use of fire in meeting ecological and other management objectives". Cal Fire's Vegetation Treatment Program, approved at the end of 2019, sets an annual target of 200,000 ha of vegetation management and forest restoration on state and private lands, matching a long-term (but as of yet unattained) US Forest Service goal of 200,000 ha to keep up with California wildfire trends.

As noted above, economic questions remain at the center of the wicked problem. In California, the carbon cap-and-trade market, which allocates and sells emissions allowances for major greenhouse gas producers, has raised over $700 million dollars for investment in carbon-smart ecosystem and fuel management, and made available hundreds of millions of dollars in grants for fuel reduction, fire prevention and education by entities including federal agencies, fire safe councils, fire protection districts, resource conservation districts, and city and county governments (https://www.fire.ca.gov/grants/fire-prevention-grants/). These grants are administered by Cal Fire, which has also seen its annual operating budget balloon to $1 billion. In addition, the California Renewables Portfolio Standard stipulates that 60% of electricity sales must come from renewable sources by 2030. As of spring 2020, biomass energy contributes about 17% of the total energy delivered under RPS contracts (https://www.cpuc.ca.gov/rps), and forest residues contribute about 35% of the overall biomass fuel supply (Cal-SAF 2019).

Even with carbon management-related funding and progressive energy policies that incentivize biomass conversion to energy, there are many missing links in the supply and demand chains for forest products. For example, the number of California sawmills is now less than 30, down from 47 in 2000 and many more in the 1970s and 80s, and unlike mills in other parts of the USA these mills largely lack infrastructure to utilize production residues like chips, sawdust, and bark (Beck Group

2015). The lack of sawmills is especially acute in the southern Sierra Nevada and southern California, where one mill serves an area the size of Arkansas. In addition, the number of commercial-scale biomass energy facilities in California has dropped by >50% since the mid-1990s (Cal-SAF 2019). The cost per area of reducing forest fuels in the NAMCZ is among the highest in North America, which raises the costs of forest by-product commodification. Many innovative uses of forest management residues have been proposed – e.g., cross-laminated timber, new types of plyboard, small-scale and mobile biomass energy production (Beck Group 2015) – but to this point national and regional political and economic focus has been elsewhere and without subsidies few of these innovations will see the light of day.

9.4 Conclusion

The southern NAMCZ is experiencing the most rapid climate warming in the lower 48 states, and interannual variability in precipitation is increasing in the region, which already supports the least predictable rainfall in the USA (Safford et al. 2012a; Bedsworth et al. 2018). More than one in every eight Americans lives in the NAMCZ, and the Baja California portion supports the third most populous municipality in Mexico. The NAMCZ leads the USA in wildfire-caused human fatalities, home destruction, and economic losses, and the number of properties at extreme wildfire risk in California alone is almost three times higher than the next state in the list (Safford 2007; III 2020). The NAMCZ also leads the USA in wildfire suppression and management expenditures (Cook and Becker 2017). The level of fire adaptation in NAMCZ biota is unmatched in North America, and extreme modern-day departures from long reigning fire regimes threaten the resilience and sustainability of biota and ecosystems in one of the world's biodiversity hotspots. As global change accelerates, policy and management decisions made regarding fire and fuels will have broad ecological, social, and economic repercussions in the NAMCZ and beyond.

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