CHAPTER TEN

North Coast Bioregion

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In the early days of forestry we were altogether too dogmatic about fire and never inquired into the influence of fire on shaping the kind of forests we inherited.

FRITZ (1951)

Description of Bioregion

Physical Geography

The North Coast bioregion is classified as being within the California Coastal Steppe, Mixed Forest, and Redwood Forest Province of the Mediterranean Division of the Humid Temperate Domain (Bailey 1995). Specifically, it is composed of the Northern California Coast and the Northern California Coast Ranges Sections (Map 10.1) (Miles and Goudey 1997).

Mesozoic sedimentary bedrock from the Franciscan Formation is the dominant type in the bioregion. Sandstone, shale, and mudstone are most common with lesser amounts of chert, limestone, and ultramafic rocks. Basalt, andesite, rhyolite, and obsidian can be found in the volcanic fields of Sonoma, Napa, and Lake Counties, while granitic rocks similar to those found in the Sierra Nevada are located west of the San Andreas Fault near Point Reyes and Bodega Bay (Harden 1997). Soils in the northwestern California ecological units have been classified as Alfisols, Entisols, Inceptisols, Mollisols, Spodosols, Ultisols, and Vertisols (Miles and Goudey 1997).

The bioregion is topographically diverse. Elevations range from sea level to around 1,000 m (3,280 ft) in the Northern California Coast Section and from around 100 to 2,470 m (328–8,100 ft) in the Northern California Coast Ranges (Miles and Goudey 1997). Slope gradients vary from flat valley bottoms to steep mountain slopes that are commonly greater than 50%. Numerous mountain ranges (e.g., Kings Range, South Fork Mountain, Yolla Bolly Mountains, and the Mayacamas Mountains) and rivers (e.g., Smith, Klamath, Mad, Van Duzen, Mattole, Eel, Noyo, Navarro, Big, Russian) are located within the north coast region.

Climatic Patterns

Three predominant climatic gradients help determine the vegetation patterns in northwestern California: (1) a west–east gradient extending from a moist, cool coastal summer climate to a drier, warmer interior summer climate; (2) a north–south latitudinal gradient of decreasing winter precipitation and increasing summer temperatures; and (3) a montane elevational gradient of decreasing temperature and increasing precipitation. These gradients, while important individually, interact in a complex fashion, especially away from the coast.

The bioregion experiences a Mediterranean climate with cool, wet winters and cool to warm, dry summers. Over 90% of the annual precipitation falls between October and April (Elford and McDonough 1964). Annual precipitation varies from 500 to 3,000 mm (20–118 in) (Miles and Goudey 1997). The Pacific Ocean greatly moderates temperature, resulting in a sharp west to east temperature gradient. The mean maximum monthly temperature at Fort Ross, for example, varies from 13.8°C (56.4°F) in January to 20.2°C (68.4°F) in September, a difference of only 6.4°C (43.5°F). In contrast the mean maximum monthly temperature at Angwin, near the Napa Valley, varies from 11.2°C (52.2°F) in January to 30.5°C (86.9°F) in July (Western Region Climate Center 2001). Most coastal forests experience summer fog, an important water source that increases soil moisture and reduces plant moisture stress (Dawson 1998). Summer relative humidity and temperature is strongly influenced by proximity to the Pacific Ocean and the presence of summer fog. Whereas lightning does occur along the North Coast bioregion during the summer fire season, it is much less prevalent than on the higher ridges and mountains to the east (Keeley 1981). van Wagtendonk and Cayan (2008) found that lightning strike density ranged from 0.9 to 9.3 $yr^{-1}\ 100^{-1}\ km^{-2}$, with density increasing with distance from the Pacific Ocean and increasing elevation for the period between 1985 and 2000. Notwithstanding the lightning fire potential in northwestern California, ignitions by Native Americans likely accounted for most prehistoric fires (Fritz 1931, Lewis 1993, Stephens and Fry 2005).

Synoptic weather systems in northwestern California influence fire activity (Hull et al. 1966). Gripp (1976), in a study of critical fire weather in northwestern California, found that 37.5% of fires larger than 120 ha (300 ac) were associated with the Pacific High (Postfrontal) Type. The Great Basin High Type accounted for 29.7% of the fires, the Subtropical High Aloft Pattern was linked with 21.9%, and other miscellaneous

MAP 10.1 North Coast and North Coast Ranges Ecological Zones.

types were associated with 10.9%. The Pacific High (Postfrontal) and the Great Basin High Types produce warm, dry east winds (foehn winds) that displace the marine air mass off the coast (Hull et al. 1966).

Ecological Zones

The Northern Coastal Scrub and Prairie Zone is found in the fog belt along the California coast in a discontinuous band below 1,000 m (3,280 ft) elevation from Santa Cruz north to the Oregon border (Heady et al. 1988). Northern Coastal Scrub was described by Munz and Keck (1959) and is variously dominated by species such as coyote brush (*Baccharis pilularis*), yellow bush lupine (*Lupinus arboreus*), salal (*Gaultheria shallon*), and California huckleberry (*Vaccinium ovatum*). Scrubs dominated by salal, California huckleberry, ferns, and blackberry (*Rubus* spp.) are more common in the northern part of the bioregion. Coastal prairies are interspersed within the Northern Coastal Scrub and are often dominated by grass and herbaceous species, with one study recording an average of 23 species m^{-2} (Stromberg et al. 2001).

The Northern Chaparral Zone is typically made up of chamise (*Adenostoma fasciculatum*) dominated chaparral that occurs in the inland portion of the north coast (Fig. 10.2). Other common shrubs include manzanita (*Arctostaphylos* spp.), California-lilac (*Ceanothus* spp.), and oak (*Quercus* spp.). Large areas of chaparral are found near Clear Lake and on the lower elevations of the interior northern mountains.

The Northern Coastal Pine Forest Zone (Fig. 10.3) is made up of isolated stands along the north coast (Barbour 2007). Principal species include shore pine (*Pinus contorta* ssp. *contorta*), Bishop pine (*P. muricata*), Bolander pine (*P. contorta* ssp. *bolanderi*), knobcone pine (*P. attenuata*), and pygmy cypress (*Hesperocyparis pygmaea*).

The Sitka Spruce (*Picea sitchensis*) Forest Zone (Fig. 10.4) is generally found inland of the Northern Coastal Scrub and Prairie Zone in a narrow strip approximately 1–2 km (0.6–1.2 mi) wide (Zinke 1988) extending south from the Oregon border and terminating near Fort Bragg. Along rivers and in the Wildcat Hills south of Ferndale, Sitka spruce forests can extend inland as far as 25 km (15.5 mi) (Zinke 1988).

The Redwood (*Sequoia sempervirens*) Forest Zone (Fig. 10.5) is inland of the Sitka Spruce Forest Zone due to intolerance of salt spray and strong, desiccating winds (Olson et al. 1990). The distribution of redwood forests to the north, east, and south is mostly limited by inadequate soil moisture and excessive evapotranspiration (Mahony and Stuart 2001). Redwood forests occur in an irregular narrow strip, ranging in width from 8 to 56 km (5–35 mi) (Olson et al. 1990). Stands in Napa County are 68 km (42 mi) from the coast (Griffin and Critchfield 1972).

Increased evapotranspiration inland limits the coastal conifers allowing for complex mixtures of Douglas-fir (*Pseudotsuga menziesii* var. *menziesii*) and a variety of evergreen and deciduous broad-leaved trees defining the Douglas-fir–Tanoak (*Notholithocarpus densiflorus*) Forest Zone (Fig. 10.6). Notable among the tree species present are tanoak, Pacific madrone (*Arbutus menziesii*), Oregon oak (*Quercus garryana*), and California black oak (*Q. kelloggii*) (Stuart and Sawyer 2001). Douglas-fir and tanoak forests dominate inland lower montane forests. Montane forests characteristically have Douglas-fir mixed with ponderosa pine (*P. ponderosa*) and white fir (*Abies concolor*) at higher elevations.

Low-elevation riparian forests are interspersed throughout much of the Sitka spruce, Redwood, and Douglas-fir–Tanoak Forest zones. These forests are typically represented by the presence of red alder (*Alnus rubra*) and big-leaf maple (*Acer macrophyllum*) in addition to the common conifer species present in the respective zones. Further inland other species are associated with riparian forests including California bay (*Umbellularia californica*) and California buckeye (*Aesculus californica*). We do not consider these forests as a separate zone, but note that there are likely subtle but distinct differences in the fire ecology of these forests relative to their adjacent upland forest zones.

The Oregon Oak Woodland Zone (Fig. 10.7) occurs sporadically throughout the North Coast and North Coast Ranges. In the Redwood Forest Zone and the Douglas-fir–Tanoak Forest Zone, Oregon oak woodlands often occur in patches of a few to hundreds of hectares in size, usually near south and west facing ridges. In the warmer, drier parts of the North Coast and North Coast Ranges, Oregon oak can form open savannas or be interspersed with other tree species.

A few of the higher mountains support red fir forests and on the highest peaks, foxtail pine (*P. balfouriana*) and Jeffrey pine (*P. jeffreyi*) forests can be found. These are discussed in the Klamath Mountains chapter (chapter 11). Blue oak (*Q. douglasii*) woodlands and grasslands are found in the interior lowlands

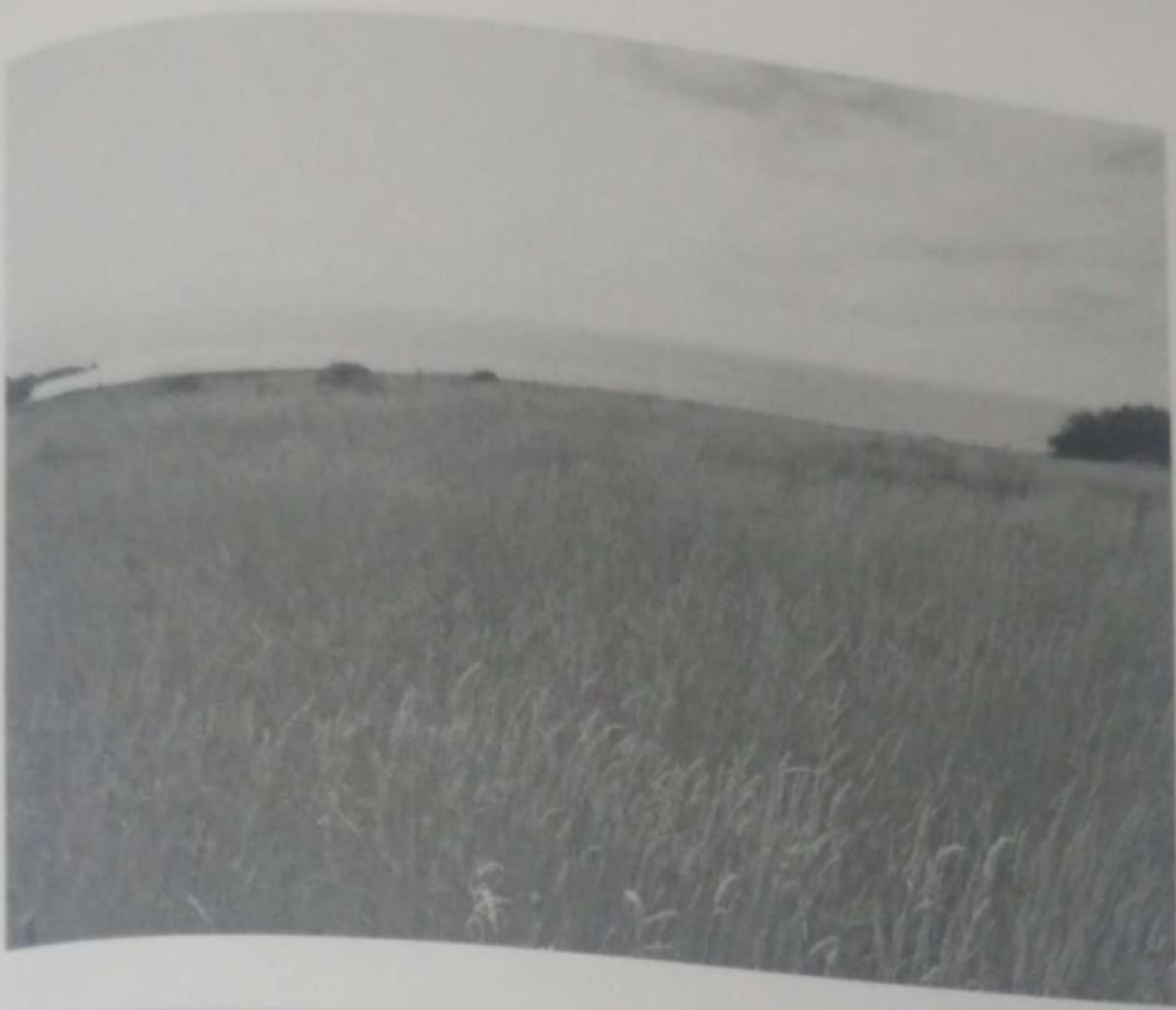

FIGURE 10.1 North Coastal Scrub and Prairie at Sea Ranch, Sonoma County (photograph by Rand Evett).

FIGURE 10.2 North coastal chaparral zone (photograph by Scott Stephens).

FIGURE 10.3 North coastal pine forest zone. Bishop pine forest regeneration following the 1995 Mount Vision fire (photograph by Scott Stephens).

FIGURE 10.4 Sitka spruce forest zone. Young Sitka spruce forest at Patrick's Point State Park (photograph by John Stuart).

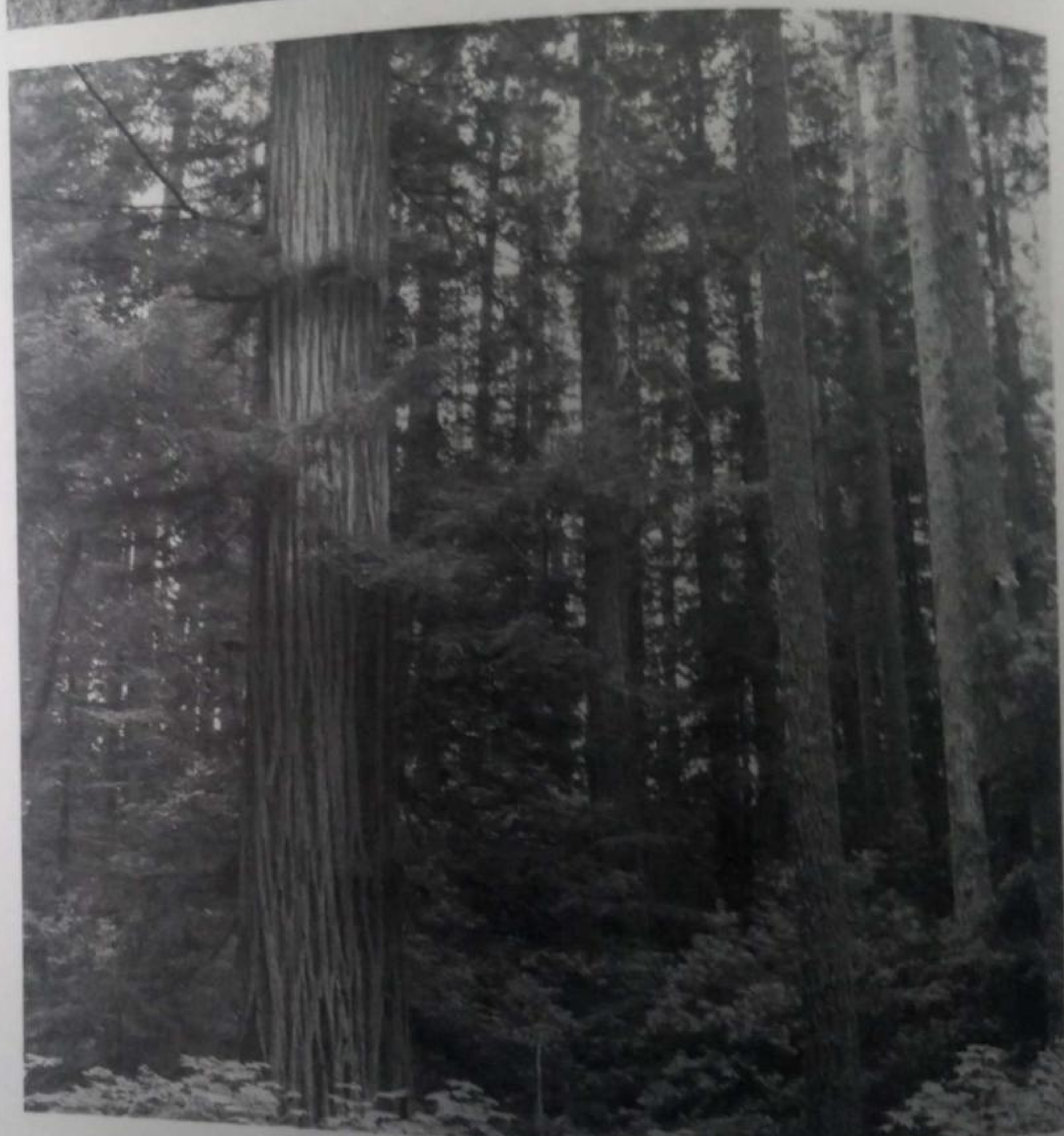

FIGURE 10.5 Redwood forest zone. Old growth redwood forest in Redwood National Park (photograph by John Stuart).

on the eastern border of the region (Stuart and Sawyer 2001) and are discussed in the Central Valley chapter (chapter 15).

Overview of Historic Fire Occurrence

Prehistoric Period

Holocene fire history reconstructions from lake sediments in western Oregon (Long et al. 1998, Long and Whitlock 2002) and the Klamath Mountains (Mohr et al. 2000) indicate relatively frequent fire during the warm, dry early to mid-Holocene and less frequent fire as the climate became cooler and wetter. Pollen analyses reveal increased levels of fire-adapted vegetation concomitant with thicker charcoal deposits and a warm, dry climate (Long and Whitlock 2002). Similar patterns were detected by Anderson et al. (2013) on the Point Reyes peninsula and presumably apply more generally to North Coast and North Coast Ranges.

The North Coast Ranges have experienced three major climatic periods since the end of the Pleistocene: a cool, somewhat continental climate from the early Holocene to about 8,500 years BP; a warmer period with presumably drier summers from 8,500 years BP to about 3,000 years BP; and a cool

FIGURE 10.6 Douglas-fir–tanoak forest zone. Young Douglas-fir–tanoak forest in Humboldt County (photograph by John Stuart).

FIGURE 10.7 Oregon oak woodland zone. Prescribed fire in Oregon oak woodland at Copper Creek (photograph by John McClelland).

moist climate since about 3,000 years BP (Keter 1995). Native Americans are known to have lived in the region since around 8,200 years BP to 8,600 years BP, and by the middle of the Holocene, humans lived throughout the North Coast (Sawyer et al. 2000). During the early Holocene pine forests with sparse shrubs and herbs dominated parts of the North Coast Ranges (West 1993). Redwood forests were present in relatively low abundance at this time but peaked in the mid-Holocene (~5,500 BP). As the climate warmed and dried, pine pollen counts remained high and oak counts increased while Douglas-fir pollen was reduced. The cool, moist climate in the late Holocene enabled Douglas-fir, tanoak, and true fir pollen counts to increase and oak counts to decrease (Keter 1995). Fire was presumably more frequent during the warm, dry period and less frequent during the cool, moist period.

Fire history studies from the last 1,000 years reveal a variable pattern of fire frequencies throughout northwestern California. The most frequently burned landscapes were ignited on a near annual basis by Native Americans (Lewis 1993) and were generally in close proximity to villages or in areas cultured for food and basketry materials such as in grasslands and oak woodlands. Vegetation adjacent to Native American use areas experienced more frequent fire than would be found in the same vegetation type further away (Whitlock and Knox 2002). Lightning fires were generally less frequent, but relatively more numerous at higher elevations in the North Coast Ranges than the coastal regions. In general, the

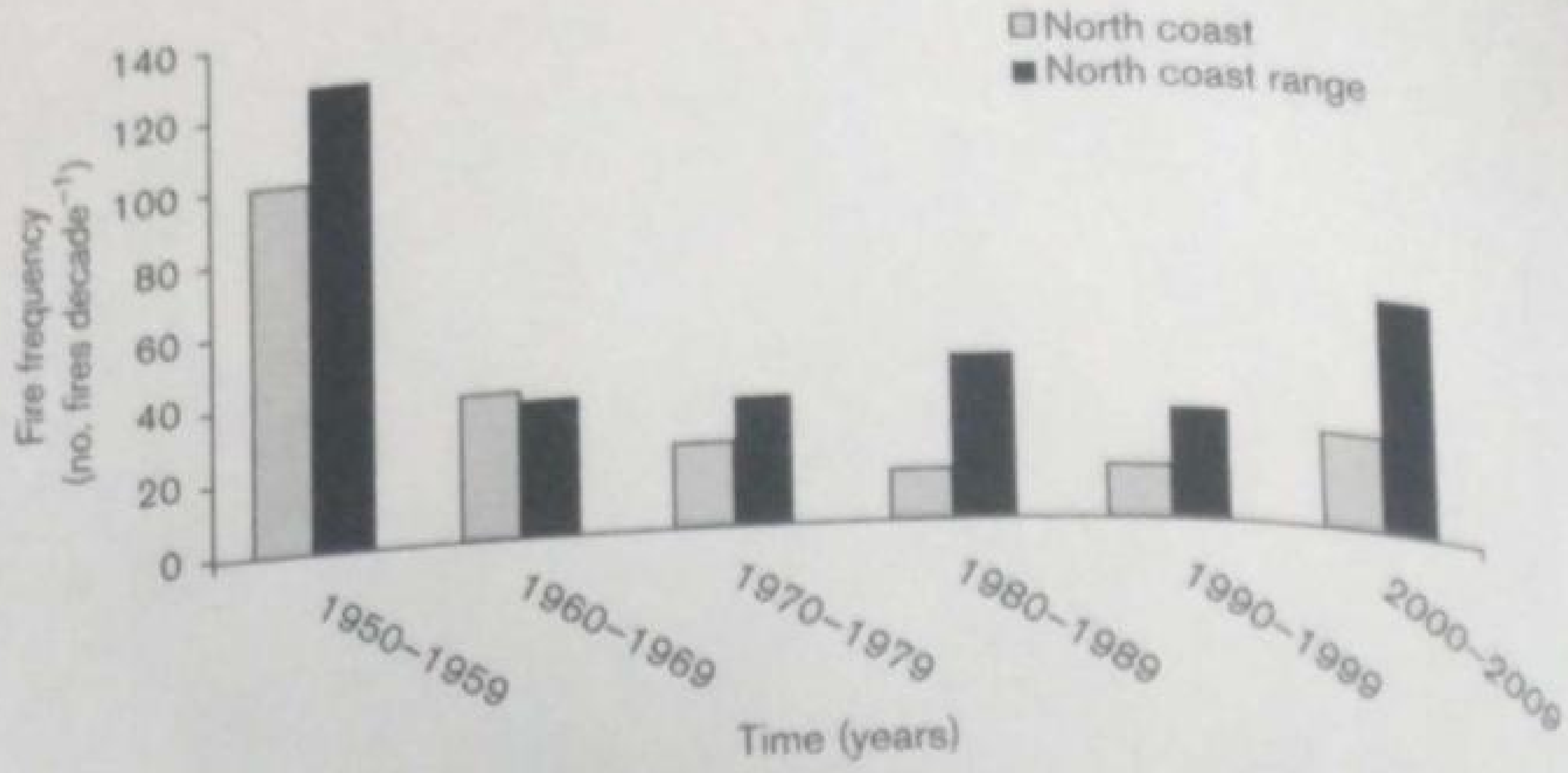

FIGURE 10.8 Fire frequency by decade for fires larger than 120 ha (300 ac) in the North Coast and North Coast Ranges Ecological Sections.

most frequent fire occurred in grasslands and oak woodlands, with less frequent fire in chaparral, mixed evergreen, and montane mixed conifer. The least frequent fire occurred in moist, coastal conifer forests (e.g., Sitka spruce forests).

Historic Period

The removal and prevention of Native American ignited fires, the introduction of cattle and sheep by ranchers, and intensive logging altered fire regimes during the historic period. Changes in the fire regimes occurred as Euro-American settlement moved north from the San Francisco Bay area in the early nineteenth century to the northern counties in the mid- to late-nineteenth century. The earliest known Euro-American settlement in the region was established at Fort Ross in 1812 by Russian-American fur traders (Lightfoot et al. 1991). Burning by Native Americans of the North Coast was likely interrupted starting in the early to mid-1800s. After which, traditional burning practices were increasingly curtailed as indigenous populations were decimated by disease and warfare and as survivors were relocated to reservations (Keter 1995).

Early settler fires originated either from escaped campfires or were deliberately set to improve forage for livestock (Barrett 1935). Ranchers primarily grazed cattle during the early to mid-nineteenth century with sheep grazing increasing in the mid- to late-nineteenth century. Sheepherders were notorious as indiscriminate users of fire (Barrett 1935). Rather than burning for a single reason, Native Americans skillfully employed burning for multiple purposes at different times of the year (Lewis 1993).

During the mid- to late-nineteenth century fire frequency, intensity, and severity were generally high throughout this period. Loggers regularly burned recently cut lands to remove downed fuels and to facilitate log extraction by draft animals. The potential for fire to escape and burn into unlogged forests was high. By the late nineteenth century and through the 1920s, mechanical yarding systems and railroads enabled logging of whole watersheds in coastal and interior drainages. Following logging, some timberland owners attempted to convert forestland to grassland by repeatedly burning the logging slash and sowing grass seed (O'Dell 1996). Large fires were frequent in northwestern California during the historic period. For example, Gripp (1976) conducted an extensive review of northwestern California newspapers and various other documents and found that large fires in Humboldt and Del Norte counties from about 1880 to 1945 had an average three years between severe fires.

Fire suppression began on national forest lands in northwestern California in 1905 (Keter 1995). Fire suppression on private and state land, in the latter part of the nineteenth century through the early twentieth century, was largely the responsibility of the counties and various landowner associations. During the 1920s fire wardens used their power of conscription to recruit fire fighters and by the early 1930s, the California Division of Forestry assumed the role of fire suppression (Clar 1969). Effective fire suppression on private, state, and federal land did not begin until after 1945 when an increased number of returning soldiers that became firefighters had access to technologies used during World War II.

Current Period

Fire records dating to around 1915 exist for the bioregion, although reasonably complete records for large fires are only available since 1945 and complete records in digital format for fires of all sizes are available for only the past few decades. The records for fires larger than approximately 120 ha (300 ac) (CDF-FRAP 2015) reveal that two to three times as many fires occurred in the 1950s as in subsequent decades. In addition, there were consistently more fires in the North Coast Ranges than the North Coast (Fig. 10.8). The vast majority of fire records did not identify fire cause, although the large number of fires in the 1950s coincided with a period of increased logging. Fewer large fires in ensuing decades can be attributed to more effective fire prevention and suppression.

The North Coast has experienced a consistent decrease in cumulative area burned since the 1950s, and the North Coast Ranges similarly experienced progressively smaller areas burned from 1950 through the 1970s, but in the last few decades cumulative fire size is potentially trending upward (Fig. 10.9), possibly due to increased fuel load and continuity associated with logging and fire suppression (Stuart and Salazar 2000) and increasing global temperatures (Westerling et al. 2006). This trend of increased fire size in the bioregion appears to be continuing as demonstrated by the spate of destructive fires in the Northern Bay area during the fall of 2017.

Major Ecological Zones

Northern Coastal Scrub and Coastal Prairie Zone

Northern coastal scrub extends from Monterey County into Oregon in a narrow strip generally less than a few hundred

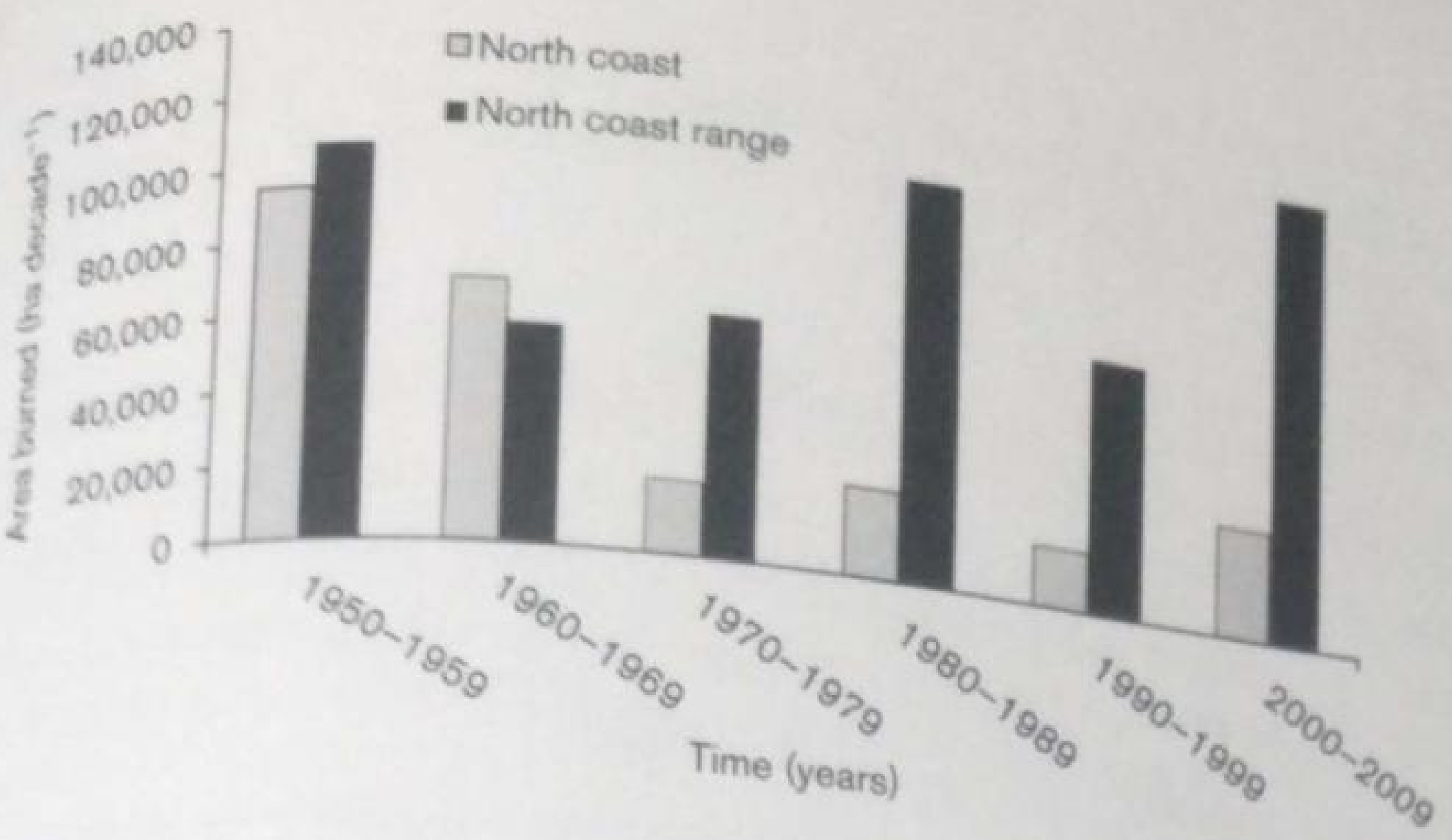

FIGURE 10.9 Area burned by decade for fires larger than 120 ha (300 ac) in the North Coast and North Coast Ranges Ecological Sections.

meters wide. Salal in combination with blackberry species and a rich mix of other shrubs, subshrubs, and herbaceous species often forms thickets and brambles. A transition zone between northern coastal scrub and coastal sage scrub lies in Marin and San Mateo Counties.

Prior to Euro-American settlement, the Northern Coastal Prairie was probably dominated by native perennial grasses, including California oat grass (*Danthonia californica*), purple needlegrass (*Stipa pulchra*), Idaho fescue (*Festuca idahoensis*), and tufted hair grass (*Deschampsia cespitosa*) (Heady et al. 1988). Highly susceptible to invasion, coastal prairie now includes many Mediterranean annual species as well as non-native perennial grasses, notably common velvet grass (*Holcus lanatus*) and sweet vernal grass (*Anthoxanthum odoratum*) (Hektner and Foin 1977).

FIRE RESPONSES OF IMPORTANT SPECIES

Salal, California huckleberry, coyote brush, thimbleberry (*Rubus parviflorus*), salmonberry (*R. spectabilis*), and California blackberry (*R. ursinus*) are all fire neutral, facultative sprouters (Table 10.1). While regeneration is not fire dependent, these species have the capability to aggressively recolonize burned landscapes through sprouting, seeding, or germination from buried seed.

Dominant grasses in the Northern Coastal Prairie are fire neutral, facultative sprouters. One study of coastal prairie native bunchgrass fire responses showed no significant changes in foliar cover or frequency for California oat grass, purple needle grass, and foothill needle grass (*Stipa lepida*) after burning (Hatch et al. 1999). Tufted hairgrass in other plant communities is resistant to all but the highest intensity fires and recovers to pre-burn levels in a few years (Walsh 1995). Idaho fescue is resistant to low-intensity burning but can be killed at higher intensities (Zouhar 2000). A small unpublished study at The Sea Ranch in Sonoma County showed temporary decreases in the cover of velvet grass and sweet vernal grass following prescribed burns with little effect on tufted hairgrass (Evett 2002).

FIRE REGIME–PLANT COMMUNITY INTERACTIONS

Fire is relatively uncommon in Northern Coastal Scrub but Native Americans would most likely have ignited any fires that did burn, and fire spread probably would have been dependent on warm, dry east winds. Pre-Euro-American fire intervals are unknown, but Greenlee (1983) estimated variable fire return intervals ranging from 1 to 100 years depending on the vegetation type and proximity to Native American villages. Northern Coastal Scrub is capable of self-perpetuating with or without fire. Ten years following the 1995 Vision fire in Point Reyes National Seashore where 5,000 ha (12,355 ac) burned, coastal scrub communities decreased by 27% and transitioned to either blue blossom scrub or Bishop pine communities (Forrestel et al. 2011).

The role of fire in maintaining coastal prairie prior to Euro-American settlement is poorly documented but likely included widespread burning (Blackburn and Anderson 1993). Soil and phytolith evidence suggests that many coastal prairie sites have been grassland for thousands of years (Bicknell et al. 1992). In the absence of burning or grazing, many of these sites with high grass phytolith content in the soil have been invaded by shrubs and trees, suggesting that regular aboriginal burning was required to maintain the coastal prairie (Bicknell et al. 1993, Evett 2000). Following displacement of the Native American populations, many ranchers practiced deliberate burning to maintain the prairie and promote understory forage (Bicknell et al. 1993). For the past 150 years, continuous livestock grazing has replaced frequent burning, which collectively resulted in the increase in dominance of non-native Mediterranean annuals. Whether the proliferation of non-native annuals was caused by the transition to heavy grazing, removal of burning, or a combination of the two is not well understood. Aboriginal fires were probably small and of low intensity due to the discontinuous nature of the coastal prairie and lack of fuel accumulation under a high-frequency fire regime. Seasonality of aboriginal fires is unknown; fires were probably more likely in the dry season of summer or early fall.

The removal of fire and livestock grazing from the coastal prairie has profound effects on plant community composition. In most cases, the absence of fire or grazing results in conversion to Northern Coastal Scrub after 15 to 25 years (Ford and Hayes 2007). However, sites at The Sea Ranch in northern Sonoma County, where livestock grazing was removed in the late 1960s and fires were excluded, show a shift from mixed annual and perennial grasses to overwhelming dominance of non-native perennial grasses that reduced biodiversity and substantially increased fuel loads (Foin and Hektner 1986).

Small-scale experiments to reintroduce fire to coastal prairie sites to reduce nonnative species and restore native species dominance had mixed success. Burning at The Sea Ranch sites reduced cover of the nonnative grasses (velvet grass and sweet vernal grass) but increased cover of another nonnative grass,

TABLE 10.1
Fire response types for important species in the North Coast Bioregion

Lifeform	Type of fire response			Species
	Sprouting	Seeding	Individual	
Tree	None	Neutral	Killed	Shore pine, Sitka spruce, western hemlock, western red cedar
	None	Neutral	Survive	Grand fir, Port Orford cedar
	None	Stimulated (establishment)	Survive	Douglas-fir
	None	Stimulated (release)	Killed	Bishop pine, Bolander pine, pygmy cypress
	Stimulated	Neutral	Survive/top-killed	Big-leaf maple, California bay, canyon live oak, giant chinquapin, tanoak
	Stimulated	Stimulated (establishment)	Survive/top-killed	Redwood, California black oak, Oregon oak, Pacific madrone
Shrub	Stimulated	Neutral	Survive/top-killed	California huckleberry, salal
	Stimulated	Stimulated (establishment)	Survive/top-killed	Chamise, California blackberry, coyote brush, salmonberry, thimbleberry
	None	Stimulated (establishment)		Wedgeleaf ceanothus, blue blossom, common manzanita
Grass	Stimulated	Stimulated (establishment)	Survive/top-killed	California oat grass, Idaho fescue, purple needlegrass, tufted hairgrass

hairy oat grass (*Rytidosperma penicillatum*) (Evett 2002). Biological diversity increased because of increased cover of non-native forbs and annual grasses. Native grass species were mostly unaffected by burning and this was confirmed in a separate study in coastal San Mateo County (Hatch et al. 1999).

Northern Chaparral Zone

Chaparral constitutes 7% of the land area in California, but it hosts more than one-quarter of the state's endemic flora and fauna—nearly half of which are endemic to chaparral (Keeley and Davis 2007). Chaparral ecosystems have been extensively studied in the southern and central coasts of California (see chapters 16 and 17 of this text) but some work has also occurred in northern California, particularly in the coastal interior mountains (Potts and Stephens 2009, Potts et al. 2010). In many areas chaparral is located on steep slopes with thin soils and high solar radiation (Wilkin et al. 2017, Newman et al. 2018). Mature chaparral is commonly 1.5–2 m (4.9–6.6 ft) in height and nearly 100% shrub cover but cover varies with site productivity. Understory native herbaceous plants are uncommon in mature chaparral because of low light levels under the canopy and limited moisture. Chaparral was probably not as dense as it is today and Native Americans likely burned some areas to facilitate grasslands and oak woodlands that were higher valued resources.

Crown fires are the typical fire type in chaparral, characterized by high flame lengths and rapid rates of spread (Stephens et al. 2008). Chaparral has been burned for decades in northern California to increase range resources for livestock and to improve forage for black-tailed deer (*Odocoileus hemionus columbianus*) (Biswell 1989). Because of its high fire hazards chaparral has also been burned to reduce fuel loads. In southern California, the impacts of high velocity, dry foehn winds have been shown to be a dominant factor in chaparral fire regimes (Moritz et al. 2004).

FIRE RESPONSES OF IMPORTANT SPECIES

Chaparral plant species can be grouped into three functional groups, depending upon their post-disturbance regeneration response: obligate seeders, obligate sprouters, and facultative seeders. Obligate seeders (e.g., *Ceanothus cuneatus*, *Arctostaphylos manzanita*), primarily reproduce from long-lived fire stimulated soil-stored seed banks and lack any sprouting ability (Table 10.1). In contrast, obligate sprouting species regenerate well after fire primarily by sprouting vigorously from adventitious buds (e.g., Eastwood's manzanita [*A. glandulosa*], scrub oak [*Quercus berberidifolia*]). Obligate sprouters can also reproduce from seed, but most species only produce a few seeds that are intolerant of heat exposure (e.g., oaks), and a few species (e.g., Eastwood's manzanita) produce seeds that require fire cues (Keeley 1991). Facultative seeders reproduce both sexually by seed and vegetatively by sprouting (e.g., chamise). The seed bank of facultative seeders is often polymorphic, containing both fire-stimulated and fire sensitive seeds to ensure germination after and between fire events

(Zammit and Zedler 1988). Facultative seeder species should have an advantage in most chaparral ecosystems since they can successfully regenerate after diverse environmental cues.

FIRE REGIME–PLANT COMMUNITY INTERACTIONS

Chaparral, like most Mediterranean shrublands, is highly fire resilient and historically burned with high-severity, stand-replacing events every 30 to 100 years (Stephens et al. 2007). Though adapted to infrequent fires, chaparral plant communities can be exterminated by frequent fires ignited by humans (Keeley 2005). Historically, periodic summer or fall high-intensity wildfires were considered necessary to maintain a full suite of native chaparral plants because many species depend on fire cues for germination; however, many native chaparral plants can also germinate without fire (Potts et al. 2010, Wilken et al. 2017).

Northern Coastal Pine Forests Zone

Bishop, shore, Bolander, and knobcone pine forests are sporadically arranged along coastal bluffs and marine terraces from the Oregon border to the San Francisco Bay Area. Bishop pine occurs in disjunct populations in coastal California from Humboldt County to Santa Barbara County. Shore pine is widely distributed along the Pacific Coast from Yakutat Bay, Alaska, south to Mendocino County (Little 1971), primarily occurring along stabilized dunes and low lying deflation plains with poor site conditions and high water tables (Green 1999). Bolander pine and pygmy cypress are endemic to the pygmy forests of western Mendocino County. Knobcone pine forests primarily occur on dry steep slopes and ridgetops in the North Coast and Klamath ranges (Griffin and Critchfield 1972).

FIRE RESPONSES OF IMPORTANT SPECIES

Bishop pine is a fire dependent, obligate seeder, with postfire regeneration from crown-stored seed bank (Table 10.1). Cones can remain closed for years and primarily open after fire or on hot, dry days (Van Dersal 1938). Most Bishop pine stands are very dense and commonly have stand-replacing crown fires, although older trees have thick bark that enables survival following multiple surface fires (Stephens and Libby 2006). Cone serotiny is somewhat variable, with northern populations less serotinous than southern populations (Zedler 1986). Morphological differences in cones have been observed but were not associated with differences in cone opening temperatures (Ostoja and Klinger 1999).

Shore pine can be considered a fire neutral or fire-dependent obligate seeder, while Bolander pine and pygmy cypress are fire-dependent, obligate seeders (Table 10.1). Most shore pine have open cones (Lanner 1999), though serotiny has been observed (Griffin 1994). Bolander pine and pygmy cypresses are mostly serotinous and cones typically open after fire or by desiccation (Vogl et al. 1977, Lotan and Critchfield 1990). Detached or desiccated cones commonly open but deep litter beds and low light conditions limit seedling establishment.

Knobcone pine is a fire-dependent, obligate seeder (Table 10.1). Mature trees have thin bark and are often killed following fire. However, serotinous cones are retained in the canopy for up to 50 years and fires with temperatures of 160°C (320°F) or greater open cone scales and allow seed dispersal (Vogl 1973). Maturity in knobcone pine trees is reach by seven years, with 90% of individuals bearing an average of nine cones per tree (Keeley et al. 1999). Seed content of cones varies slightly with age but generally ranges from 60 to 100 seeds per cone with approximately 80% seed viability (Fry and Stephens 2013).

FIRE REGIME–PLANT COMMUNITY INTERACTIONS

Bishop pine is adapted to stand-replacing high-severity fires (Appendix 1). Prior to Euro-American settlement, the majority of these fires probably occurred in the late summer and fall when fuel moisture contents were low. High-severity fires were probably mostly associated with warm, dry east winds. Fire history data are not available to estimate the spatial extent of fires, but isolated Bishop pine groves surrounded by shrubland probably had moderate to large fires (> 100 ha [> 250 ac]).

The majority of Bishop pine forests have not burned in the last 40–90 years because of fire exclusion. However, in 1995 the Vision fire burned through 423 ha (1,045 ac) of Bishop pine forests in Point Reyes National Seashore. Following the fire, Bishop pine prolifically reseeded and expanded in area by 85% (Forrestel et al. 2011) (Fig. 10.3). Expansion occurred in areas that were previously Northern Coastal Scrub communities adjacent to preexisting Bishop pine stands (Harvey et al. 2011). Fourteen years following the Vision fire, two distinct Bishop pine community pathways were apparent, a low-density, open-canopy pathway with high shrub cover on gentle slopes and a high-density, closed-canopy pathway with low shrub cover on steep slopes (Harvey and Holzman 2014). The lack of fire could threaten the long-term existence of Bishop pine because it relies on high-severity crown fires that prepare seed beds, enable the release of large quantities of seed, and increase light availability to seedlings. Introduction of nonnative grasses into recently burned areas would increase the risk of extirpation because the grasses could produce a highly continuous fuelbed in 1 to 2 years. Little is known about the role of fire in Bolander pine and pygmy cypress forests. Fires are likely infrequent and of high severity when they occur.

Knobcone pine typically experiences a high-severity stand-replacing fire. In fires between 1930 and 1960 in the Mayacamas Mountains of the North Coast Range, 67% of stands examined had even-aged distributions while the remaining stands had multiage distributions (Fry et al. 2012). Multiage distributions were attributed to areas that experienced multiple fires or a low-intensity fire that promoted regeneration but retained some of the prefire cohort (Fry et al. 2012).

Sitka Spruce Forest Zone

Sitka spruce forests are variously dominated by Sitka spruce, western hemlock (*Tsuga heterophylla*), Douglas-fir, western red cedar (*Thuja plicata*), Port Orford cedar (*Chamaecyparis lawsoniana*), grand fir (*Abies grandis*), and red alder. Forests immediately along the coast are primarily composed of Sitka spruce.

As the distance from the coast increases the other co-occurring species become more common and Sitka spruce reduces in abundance.

FIRE RESPONSES OF IMPORTANT SPECIES

In general, Douglas-fir and red alder regenerate well following fire, whereas western hemlock, grand fir, western red cedar, and Port Orford cedar regenerate well in either undisturbed or disturbed forests. Douglas-fir is a fire-enhanced, obligate seeder and red alder is a fire neutral, facultative sprouter (Table 10.1). Both species require full sunlight and mineral soil to regenerate. Sitka spruce can establish well on organic seedbeds but require light gaps from small disturbances (Taylor 1990). Young red alder trees sprout vigorously, but older trees rarely sprout (Harrington 1990). Western hemlock, grand fir, western red cedar, and Port Orford cedar are all fire inhibited, obligate seeders and regenerate well on organic seedbeds in shade or partial shade (Burns and Honkala 1990).

With the exception of Douglas-fir and larger Port Orford cedar (Zobel et al. 1985), other potential canopy dominants and codominants in these forests are not fire resistant. Western hemlock, Sitka spruce, western red cedar, and red alder are shallow rooted and have thin bark (Burns and Honkala 1990). Grand fir is fire sensitive when small, but can develop bark thick enough to resist light surface fires (Howard and Aleksoff 2000).

FIRE REGIME–PLANT COMMUNITY INTERACTION

Fires were generally uncommon in Sitka spruce forests due to high fuel moisture and low lightning densities. Fuel moisture is almost always too high to support fire in these coastal forests with the exception of late summer and early fall after summer fog has dissipated. Even then, temperature and humidity are usually not conducive to burning so close to the Pacific Ocean. With the exception of ignitions by Native American in prairies or near villages, fire intervals were long to very long. Lower-intensity fires with long residence times and the thin bark of Sitka spruce resulted in a stand-replacing, high-severity fire in this ecosystem. Fires were small to moderate in size with moderate spatial complexity (Appendix 1).

Fire history in Sitka spruce forests is not well documented. Based on multiple sources, Van de Water and Safford (2011) estimated a range in fire return intervals for spruce-hemlock forests of California to range from 180 to 550 years. Agee (1993) reported fire intervals of around 200 years for the southern Oregon coast, 400 years from the northern Oregon coast, and 1,146 years in Sitka spruce forests of western Washington (Fahnestock and Agee 1983). Red alder is known as an early seral species that aggressively colonizes moist, mineral soils in full sunlight. Early growth is rapid and it usually dominates competitors for the first 25 years or so until it is overtopped by other conifers (Uchytil 1989). Port Orford cedar and western hemlock can regenerate in the shade of a canopy, while Sitka spruce and grand fir, though not as shade tolerant as Port Orford cedar or western hemlock, can self-perpetuate following windthrow or pockets of overstory mortality (Franklin and Dyrness 1973). There has been little change in fire regime from pre-European time to the current time period. The 70 to 90 years of effective fire suppression is much shorter than the pre-Euro-American settlement fire intervals.

Redwood Forest Zone

FIRE RESPONSES OF IMPORTANT SPECIES

Redwood is a fire-enhanced, facultative sprouter (Table 10.1). Seedling establishment is problematic in the absence of fire, windthrow, or flooding because of low seed viability (Olson et al. 1990) and unsuitable seedbeds. It is rare to find redwood seedlings that have established on their own litter because of the combination of damping-off fungi (Davidson 1971) and low light intensities commonly found beneath redwood canopies (Jacobs et al. 1985). However, exposure of mineral soil and canopy openings following fire, windthrow, or flooding often results in successful seedling establishment. Following recent wildfires in the Santa Cruz Mountains of the Central Coast, redwood seedling regeneration was highly variable and ranged from 0 to 450 seedlings ha^{-1} across three sites (Lazzeri-Aerts and Russell 2014).

Sprouting in redwood can occur from lignotubers at the root crown, induced lignotubers on layered branches, trunk burls, and from adventitious buds on tree trunks (Del Tredici 1998). The development of redwood lignotubers and axillary meristems are a normal part of seedling development (Del Tredici 1998) and may represent an evolutionary response to fire, windthrow, or flooding. Sufficiently intense surface fires can stimulate basal lignotubers to sprout, while crown fires often result in redwood "fire columns" (Jepson 1910, Fritz 1931) whose denuded trunks sprout new leaves and eventually develop new branches. In general, larger redwood trees (12 in [>30 cm] diameter at breast height) that experienced full crown scorching or consumption by wildfire were more likely to regenerate through epicormic sprouting than smaller redwood trees (Lazzeri-Aerts and Russell 2014). Post-fire basal sprouting (number, area, and height) per tree decreased with tree size (Ramage et al. 2010, Lazzeri-Aerts and Russell 2014). However, fire intensity (measured as bole char height) had a parabolic relationship with basal sprout area and height, where intermediate fire intensity resulted in the tallest and most numerous sprouts per tree (Ramage et al. 2010).

Redwood bark serves as either a resister or enabler of fire damage to the cambium layer depending on its thickness and the water content of its loosely packed, sponge like fibers. Bark 15–30 cm (6–12 in) thick (Fritz 1931) protects the cambium from heat damage, especially when moist. Small, thin barked trees, however, are susceptible to fire damage and are readily top-killed (Finney and Martin 1993). Dead cambium acts as an infection court for sapwood and heartwood rots. Surface fires burn into the rotten sapwood or heartwood forming basal hollows often referred to as "goose pens." Subsequent fires in goose pens continue to expand the cavity with the result that many trees are consumed standing (Fritz 1931) but cavity formation can also be assisted by the large radial growth rates of redwood (Finney 1996). Fire scars have been recorded to be as tall as 70 m (230 ft) (Sawyer et al. 2000). Redwood stands are among the most productive in the world (Fritz 1945) and consequently produce impressive fuel loads. Litter load of old-growth redwood forests ranged from 9.6 to 16.7 Mg ha^{-1} (26.2–45.4 ton ac^{-1}) with depth ranging from 5.0 to 9.6 cm (2–3.8 in) (Graham 2009). Rapid litter decomposition helps to keep the litter loads relatively low. Pillers and Stuart (1993) found that the time to decompose 95% of the weight of oven dried litter ranged from approximately 7 to 11 years on four sites in Humboldt County.

Redwood litter is highly flammable in comparison to many other western conifer species (Fonda et al. 1998) because leaves do not abscise individually but fall as sprays of leaves. The resultant fuelbed is made up of loosely compacted fuel enabling the litter to quickly respond to moisture changes and greater oxygen permeability during combustion (Pyne et al. 1996) but when fog is present redwood litter will not readily burn. Litter is the predominant surface fuel that carries fire, but understory plants can either inhibit or intensify fire behavior. Coastal redwood forests in Del Norte County, for example, include many plants (Mahony 1999) with a moisture content high enough to retard fire spread. Inland redwood forests, in contrast, have a higher proportion of sclerophyllous understory trees and shrubs that can exacerbate fire behavior.

FIRE REGIME–PLANT COMMUNITY INTERACTIONS

Redwood forests have a complex fire history, typically burning across a wide range of conditions over space and time (Stephens and Fry 2005). This variation mainly reflects differences and changes in ignition types and climatic gradients. Many fires were ignited by Native Americans with less common ignitions occurring from lightning. Thus, stands in closer proximity to Native American village sites or resource use areas usually burned more frequently (Brown and Swetnam 1994, Norman 2007). Three main climatic gradients in the redwood region contribute to variation in redwood forest fire regimes (Lorimer et al. 2009). Forests with more moisture and lower temperatures (coastal and northern forests) burned less frequently than forests with less moisture and higher temperatures (inland and southern forests). Forests with less fog or during greater fog-free periods burned more frequently than forests with more fog. Also, climate–fire relationships may be muted by the presence of Native American burning due to burning across wider climatic conditions.

Historical information on fire intensity and severity in redwood forests is not directly ascertainable, but redwood forests generally experienced low- to moderate-intensity surface fires (Appendix 1). Stuart (1986) found that during prescribed burns in Humboldt County, flame lengths on redwood litter and duff were 10–90 cm (4–35 in) long, yet beneath California huckleberry the flames were 300–500 cm (9–16 ft) long. In September 2003, lightning ignited the Canoe fire, which burned 5,554 ha (13,744 ac) of predominantly old-growth redwood forest in Humboldt Redwoods State Park. Fire behavior was generally characterized as a low-intensity surface fire with flame lengths ranging from 15 to 30 cm (6–12 in) and rates of spread of 7.9 km hr^{-1} (4.9 mi hr^{-1}) (Scanlon 2007). Occasional passive crown fires occurred, especially along the southern and eastern edges of the range. On average, fire severities were lowest in the coolest, wettest regions and highest in the warmer, drier areas.

Numerous fire history studies in redwood forests indicate that the fire return interval was quite variable. Fire return intervals from 125 to 500 years have been reported for coastal forests in Del Norte County where redwood grows in association with Sitka spruce, western hemlock, and western red cedar (Veirs 1982, Mahony and Stuart 2001). Norman (2007) estimated that the fire return interval for warmer redwood forests in eastern Del Norte County were relatively frequent (mean = 23 years, range 5 to 86 years); fire intervals in its southern range generally ranged between 6 and 44 years (Table 10.2). Some redwood stands had fire intervals of 1 or 2 years due to regular Native American burning in prairies surrounded by redwood forest, near villages, and along travel corridors (Fritz 1931, Lewis 1993). Redwood tree ring growth patterns are often complacent (i.e., uniform) with high numbers of missing and false rings (Brown and Swetnam 1994), which makes cross-dating and assigning years difficult. While numerous fire history studies have been conducted in redwood forests throughout much of its range, only two studies have successfully cross-dated chronologies (Brown and Swetnam 1994, Norman 2007). However, a recent study by Carroll et al. (2014) has provided the most comprehensive and extensive cross-dated chronologies in redwood to date, with the potential to develop more detailed fire histories in redwood forests.

Fire season in redwood forests was also historically variable across sites. Most studies indicate that the fire season occurred between the late summer or early fall, based on the fire scar position in the latewood or between annual rings. For instance, Brown and Swetnam (1994) and Brown and Baxter (2003) both determined that 91–95% of fires occurred in the late summer or early fall. In Del Norte Coast Redwoods State Park, however, Norman (2007) found greater occurrence (32% of fire scars) of early season fires.

Few studies have attempted to estimate fire size in redwood forests. Fire size is generally thought to be small, though occasionally larger fires would occur, especially during droughts or warm, dry east wind events. Stuart (1987) estimated fire sizes in old-growth redwood stands in Humboldt Redwoods State Park to be approximately 786 ha (1,940 ac) for the presettlement period, 1,097 ha (2,710 ac) for the settlement period, and 918 ha (2,270 ac) for the postsettlement period. Stuart and Fox (1993) examined fire records between 1940 and 1993 for Humboldt Redwoods State Park and nearby areas and found 30 fires larger than 120 ha (300 ac) burned (average of 505 ha [1,250 ac] and range from 100 to 1787 ha [250–4,400 ac]).

Succession and the climax status of redwood forests have been discussed since the early 1900s. Some authors contend that redwood is a seral species that, in the absence of fire, windthrow, or flooding, would eventually be succeeded by more shade-tolerant species as individual redwoods die (Cooper 1965, Osburn and Lowell 1972). Most ecologists, however, have argued that redwood is self-perpetuating and should be considered as a climax or fire subclimax species (Fisher et al. 1903, Weaver and Clements 1929, Veirs 1982, Olson et al. 1990, Agee 1993). While redwoods are generally highly resistant and resilient to numerous forest disturbances, climate-change induced warming and reduction in fog may have important effects on redwood growth and survival in the near future. A recent study determined that fog frequencies in the redwood region have declined by 33% since the 1900s (Johnstone and Dawson 2010). However, the growth response to this warming and reduced fog has been shown to differ across the redwood region. Carroll et al. (2014) found that redwood growth was negatively correlated to dry summer conditions in the southernmost portion of the range and positively with decreased summer cloudiness (i.e., fog) in the northern portion of the range.

Whether redwood is dependent on fire is an open question and should be answered on a case-by-case basis with each redwood community assigned a place on a fire dependency continuum. A redwood-western hemlock/salmonberry association (Mahony and Stuart 2001) in coastal Del Norte County, for example, is fire tolerant but not fire dependent (Veirs 1982). Low to moderate surface fires kill fire susceptible species such as western hemlock, western red cedar, and Sitka

TABLE 10.2
Fire intervals in redwood forests

Location	Mean fire interval (years)	Composite area (ha)	Source
Del Norte and northern Humboldt Counties	50–500; 12–26	1; variable	Veirs 1982; Norman 2007
East of Prairie Creek Redwoods State Park	8	0.25–3	Brown and Swetnam 1994
Humboldt Redwoods State Park	11–44	7	Stuart 1987
Southern Humboldt County	25	12	Fritz 1931
Jackson State Forest	6–20	4–20	Brown and Baxter 2003
Salt Point State Park	6–9	~200	Finney and Martin 1989
Annadel State Park	6–23	14 trees[a]	Finney and Martin 1992
Near Muir Woods National Monument	22–27	75	Jacobs et al. 1985
Western Marin County	8–13	5–10; 10 trees in one stand	Finney 1990; Brown et al. 1999
Jasper Ridge Preserve, San Mateo County[b]	9–16	1–3	Stephens and Fry 2005
Big Basin Redwoods State Park[b]	~50	Variable, dependent on estimated fire area	Greenlee 1983
Santa Cruz Mountains[b]	61	51 trees[a]	Jones and Russell 2015

NOTES:
a. Point data.
b. study sites located south of North Coast bioregion.

spruce. These shade-tolerant species quickly regenerate, grow, and have the capacity to develop into important ecosystem structural elements. A redwood-tanoak/round-fruited sedge (*Carex globosa*)–Douglas iris (*Iris douglasiana*) association (Borchert et al. 1988) near the drier southern limits of the range, in contrast, could be considered as fire dependent (Veirs 1982). In these areas, reproduction from seed may be limited due to drier soil conditions, but redwoods can persist through postfire resprouting (Noss 2000). The vast majority of redwood's range is found on sites intermediate to those described above with individual stands occupying intermediate positions on the fire dependency continuum. While the redwood species may not be dependent on fire to perpetuate in these forests, the associated ecosystem structures and functions could provide critical habitat for a variety of wildlife species. Additionally, fire can increase the availability and cycling of soil nutrients, increase understory plant diversity, and create natural edges.

Fire regime changes have not been consistent across the redwood range. Post-World War II (1950–2003) natural fire rotations for the northern, central, and southern redwood zones for large fires (>331 ac [134 ha]) were 1,083 years, 717 years, and 551 years, respectively (Oneal et al. 2006). The current fire intervals in the central and southern redwoods zones greatly exceed the 6- to 44-year presuppression intervals (Table 10.2). It is possible that a fire regime change in the northern range of redwood will eventually occur because of modifications in fuel loads and structure following logging and increasing global temperatures. Only about 10% of the redwood range is currently old-growth forest, with the remaining forest composed of young-growth (Fox 1988). Some of the young-growth forests are over 100 years old and dominated by large redwoods with fuel complexes similar to old growth, but most young forests are composed of small diameter, dense complexes of conifers intermixed with broad-leaved trees and shrubs. Other fuel complex alterations include: the presence of large, persistent redwood stumps and logging slash, as well as greater shrub and herbaceous plant cover. In general, fuel load and continuity have increased. Eventually, fire size, intensity, and severity may increase, while fire complexity may change and the fire type may have a greater occurrence of passive/active crown fire.

Douglas-Fir–Tanoak Forest Zone

Douglas-fir–tanoak forests are widely distributed in areas inland from the redwood belt. Both tanoak and Douglas-fir are major components of northern mixed evergreen forests and Douglas-fir–hardwood forests (Sawyer et al. 1988). Douglas-fir dominates the overstory, while tanoak dominates a lower, secondary canopy. Other important tree associates can include: Pacific madrone, giant chinquapin (*Chrysolepis chrysophylla*), big-leaf maple, California bay, canyon live oak (*Quercus chrysolepis*), ponderosa pine, sugar pine (*Pinus lambertiana*), incense-cedar (*Calocedrus decurrens*), California black oak, and Oregon oak.

FIRE RESPONSES OF IMPORTANT SPECIES

Douglas-fir regeneration is often episodic in part due to irregular seed crops (Strothmann and Roy 1984), and establishment is enhanced when good seed crop years are followed by low to moderate surface fire. If, however, there was a poor seed crop other species would become established and inhibit Douglas-fir seedlings in ensuing years. Seedlings are most likely to become established on moist, mineral soil, while relatively few seedlings are found on thick organic seedbeds (Hermann and Lavender 1990). Douglas-fir is considered moderately shade tolerant in northwestern California (Sawyer et al. 1988). Young seedlings are better able to tolerate shade than older seedlings, although Douglas-fir seedlings growing on dry sites need more shade. In northwestern California, optimum seedling survival occurs with about 50% shade, but optimum seedling growth occurs with 75% full sunlight (Sawyer et al. 1988). Mature Douglas-fir has thick corky bark and is fire resistant. Other fire adaptations include: great height with branches on tall trees over 30 m (100 ft) above the ground, rapid growth, longevity (up to 700 to 1,000 years old), and the ability to form adventitious roots (Hermann and Lavender 1990).

Douglas-fir and tanoak litter decomposes to 5% of original dry weight in 7 years and 9 years, respectively (Pillers 1989). Because of this, fuelbeds are typically thin. Douglas-fir leaves are short and fall individually resulting in compact litter (Engber et al. 2011). Tanoak litter, however, is deeper and less dense promoting good flammability when dry (Engber and Varner 2012a).

Tanoak, Pacific madrone, chinquapin, canyon live oak, bigleaf maple, and California bay are all fire neutral facultative sprouters (Table 10.1). Tanoak generally dominates its associated broad-leaved trees and reproduces from seed under most light conditions, but fares best in full sunlight (Sawyer et al. 1988). While tanoak is very tolerant of shade, regeneration from seed in dense shade is limited. Tanoak can be easily top-killed by fire but vigorously sprouts from dormant buds located on burls or lignotubers (Plumb and McDonald 1981). Stored carbohydrates and an extensive root system aid in a rapid and aggressive postfire recovery (McDonald and Tappeiner 1987). Resistance to low-intensity surface fires increases with size because of increased bark thickness (Plumb and McDonald 1981). Dried tanoak leaves are very flammable in comparison to most California oaks (Engber and Varner 2012a).

FIRE REGIME–PLANT COMMUNITY INTERACTIONS

Pre-Euro-American settlement fires were relatively frequent in Douglas-fir–tanoak forests due to their warmer, drier, inland locations and increased lightning activity at more interior and higher elevations (Keeley 1981). In the North Fork of the Eel River, for example, an average of about 25 lightning strikes occur per year (Keter 1995). Native Americans were the primary ignition source in the North Coast Ranges, as they regularly burned to promote food plants and basketry materials (Keter 1995). Similar to other regional forest types, fires most likely occurred during the months of July through September (Keeley 1981).

There is little literature describing pre-Euro-American settlement fire size, intensity, and severity in Douglas-fir–tanoak forests. Ethnographic data on indigenous populations and fire history data suggest that fires sizes, intensities, and severities were highly variable. In areas subject to frequent burning by Native Americans, fire intensities and severities were low (Lewis 1993). Other areas experienced fire intensities and severities that varied spatially and temporally across the landscape resulting in a complex mosaic of mostly multiaged stands of varying sizes (Rice 1985, Wills and Stuart 1994). Fires in interior sites spread more extensively than those closer to redwood forests. Surface fires were common and were intermixed with areas that supported passive/active crown fires.

The few fire history studies of Douglas-fir–tanoak forests indicate that average presuppression fire return intervals varied from 10 to 16 years (Rice 1985, Wills and Stuart 1994). Further inland in the North Coast Range, dry forest sites containing Douglas-fir and other species, without tanoak, had shorter fire return intervals between 4 and 6 years with slightly higher and more variable fire return intervals on more mesic sites (Skinner et al. 2009). These high-frequency fires likely promoted much more open forests with greater cover of understory plant species. Presuppression fire sizes were undoubtedly variable. In addition to environmental factors, such as soil type and topography, successional trajectories depend on fire severity, seed availability, and sprout density. The climax forest is characterized by an overstory of Douglas-fir with tanoak dominating the lower, secondary canopy. Fire suppression and past logging have increased the density of shade-tolerant tanoak in many Douglas-fir–tanoak forests (Hunter et al. 1997). However, Douglas-fir is often able to maintain its dominance because of its large size and longevity.

Sawyer et al. (1988) describe several successional pathways for Douglas-fir–tanoak forests. Following a severe, extensive stand-replacing fire, seed producing Douglas-fir are killed leaving the sprouting tanoak or other sprouting hardwoods to dominate during early succession. Salvage logging and broadcast burning can enhance sprouting hardwood dominance; while the absence of postfire treatment will promote the persistence of shrubs and sprouting hardwoods (Stuart et al. 1993). Eventually Douglas-fir will slowly reinvade as growing space is created by maturing, self-thinning hardwoods. A moderate- to low-severity surface fire, in contrast, would not kill many Douglas-fir but would kill Douglas-fir seedlings and saplings (Rice 1985), allowing Douglas-fir trees to dominate the overstory and tanoak to perpetuate in the understory and secondary canopy. A third scenario in more open mixed stands would allow both Douglas-fir and tanoak to persist.

Since Euro-American settlement, Douglas-fir–tanoak forests have been modified through the absence of fire. The most significant change in old growth is the greater density of understory shrubs and trees creating greater vertical fuel continuity, increasing the probability that a surface fire could burn into the crown. Many Douglas-fir–tanoak forests, however, have been logged or have experienced a stand altering wildfire. The current regime in these forests can be characterized as having longer fire return intervals due to effective and aggressive fire suppression (Skinner et al. 2009), greater intensity because of increased fuel loads from slash and increased densities of understory shrubs and trees, and greater severity because of the accumulated ladder fuels and increased dead surface fuel loads. Seasonality of fire occurrence is probably unchanged and fire size changes are uncertain.

Oregon Oak Woodland Zone

Oregon oak is distributed from southwestern British Columbia through western Washington and Oregon into California in the North Coast Ranges and Sierra Nevada (Little 1971). Oregon oak woodlands often occur in the margin between conifer forest and prairies in the North Coast Ranges. Oregon oak can also be found in open savannas, closed-canopy oak woodlands, or other broadleaved trees stands, mixed stands with conifers, or other broadleaved trees (Burns and Honkala 1990). Common associates of Oregon oak woodlands are Pacific madrone, California black oak, California bay, and Douglas-fir.

FIRE RESPONSES OF IMPORTANT SPECIES

Oregon oak and California black oak are fire-enhanced, facultative sprouters (Table 10.1). Seedling establishment for these oak species is relatively less common than sprouting, but may be enhanced by the removal of the litter layer (Arno and Hammerly 1977). Both oak species have moderately thick bark that can often withstand low- to moderate-intensity surface fires. During higher-intensity fires, Oregon oak and California black oak are frequently top-killed by fire but vigorously sprout from the bole, root crown, and roots (Sugihara et al. 1987). Sprouting has been reported to decrease with age (Sugihara and Reed 1987) and increase with higher severity fire (Cocking et al. 2012, 2014).

FIRE REGIME–PLANT COMMUNITY INTERACTIONS

Prior to Euro-American settlement, Oregon oak woodlands experienced frequent, low-intensity surface fires, many of which were ignited by Native Americans. Mean fire return intervals varied from 7 to 13 years in Oregon oak woodlands in Humboldt County (Reed and Sugihara 1987). Fires probably spread in cured herbaceous fuels and oak litter. The litter of Oregon oak and California black oak are among the most flammable of California oak species (Engber and Varner 2012a). There is no information on fire sizes in this vegetation type but they probably were diverse depending on annual variation in climate. Fires ignited by Native Americans under moister conditions probably would not spread far into adjoining conifer forests (Gilligan 1966), those ignited in the summer and fall could have been extensive because of continuous dry herbaceous fuels. Frequent surface fires produced open savannas in the Bald Hills in Humboldt County (Sugihara et al. 1987) (Appendix 1).

Fire exclusion has resulted in an estimated 29–44% loss in the spatial extent of prairies and oak woodlands in the Bald Hills region of Redwood National Park (Sugihara and Reed 1987, Fritschle 2008). Frequent surface fires once inhibited seedling establishment and reduced the density of Douglas-fir and other competing conifers (Barnhardt et al. 1996). In the absence of fire conifers can overtop, suppress, and eventually kill the shade-intolerant Oregon oak (Hunter and Barbour 2001). In a study of Oregon oak and California black oak stands in Humboldt and Mendocino counties, most oaks (>80%) established before 1905, while the majority (73%) of Douglas-fir established after 1950 (Schriver and Sherriff 2015).

Future Direction and Climate Change

Climate Change

The combination of fire exclusion, logging, and increased global temperatures has altered the fire characteristics of the North Coast bioregion. Fire size and frequency has increased over the past few decades due in part to increasing temperatures and earlier springs (Westerling et al. 2006, Miller and Safford 2012). This trend of increased burned area is predicted to continue over the next century (Westerling et al. 2011); however, some models indicate fire size may decrease for some areas along the coast (Lenihan et al. 2008). While the amount of area burned each year continues to increase, the rates of burning are still considered much lower than presettlement levels (Safford and Van de Water 2014). Research assessing changes in fire severity for the North Coast bioregion has not been conducted and findings from other bioregions have been mixed (Miller et al. 2009, Miller et al. 2012, Mallek et al. 2013). Return intervals for some ecosystems closer to the coast are generally thought to be longer and may not have deviated substantially from presettlement fire severity. However, more inland forest types may experience increases in fire severity.

Management Issues

Native American burning was interrupted in the mid-1860s largely because of the impacts of introduced diseases. Managers in this region are increasingly incorporating the influence of past Native American ignitions into restoration and management objectives because Native Americans were an integral component of this region for thousands of years (Underwood et al. 2003; Crawford et al. 2015) and continue to live in the region today.

Fire has been an essential part of coastal prairie ecology and fire exclusion has led to undesirable consequences. Livestock grazing can mitigate some of these consequences (Hayes and Holl 2003) but effective management should include prescribed burning. The sparse published data on burning of coastal prairie in California combined with anecdotal evidence and evidence from outside the region suggest regular burning on a 3- to 5-year rotation will slow the invasion of some nonnative species without reducing cover of existing native species. If the seed bank contains viable seeds of displaced native species, burning may stimulate increased cover of these species. To achieve restoration objectives on most sites, extensive reseeding of desirable native species is required, followed by a regular prescribed burning and/or grazing program (Evett 2002). See sidebar 10.1 for information on chaparral management in the north coast.

The majority of closed-cone pine forests have not burned in the last 40 to 80 years because of fire exclusion (except for areas burned in the 1995 Mt. Vision and the 2017 Northern Bay Area fires). The lack of fire could threaten the long-term existence of closed-cone pines (e.g., Bishop, knobcone) because they rely on high- and mixed-severity fires to prepare seed beds, enable the release of large quantities of seed, and remove the canopy thereby increasing the light reaching the forest floor. Where possible, wildfire or use of higher intensity prescribed fires could perpetuate these forests types.

The majority of the remaining redwood forests are relatively young and more homogeneous than their old-growth forest counterparts, due to a legacy of logging that occurred throughout the region. As such, many land managers are interested in restoring young-growth redwood forests to promote old-growth forest structures and improve wildlife habitat and greater connectivity (Porter et al. 2007). Much of this restoration has focused on the use of mechanical thinning treatments to reduce stand density, increase growth, and increase structural complexity (O'Hara et al. 2010, Teraoka and Keyes 2011, Plummer et al. 2012). One potential drawback of this approach is the increased accumulation of surface fuels (activity fuels) following thinning treatments (Agee and Skinner 2005) and may require subsequent fire or other fuel reduction treatments to reduce potential risks to wildfire.

Redwood forests once experienced relatively frequent fire at varying intervals depending on their geographic location. Redwood sites should use fall-ignited prescribed fires to

SIDEBAR 10.1 NORTHERN CALIFORNIA CHAPARRAL

Kate M. Wilkin

Chaparral typically burns in high-severity stand-replacing events every 30 to 100 years (Quinn and Keeley 2006). These fires are quite dangerous for the people living in the wildland-urban interface and fire hazard reduction treatments, such as prescribed fire and mastication, are often implemented to proactively protect people and infrastructure (Keeley 2002, Dicus and Scott 2006). These treatments help reduce wildfire hazard, yet can have drawbacks for ecosystems adapted to infrequent crown fire (Keeley 2002, Schoennagel et al. 2004). In California chaparral, non-native plants are known to invade after fuel treatments, but mainly persist in areas with the most shrub cover reduction (Merriam et al. 2006, Potts and Stephens 2009).

Many are concerned that fuel reduction treatments altered chaparral ecosystems through changes to the historic fire regime (Parker 1987, Keeley 2002, Wilkin et al. 2017). Though adapted to lower frequency fires, increased fire frequency can convert chaparral to a non-native annual grassland and reduce species diversity, especially under global-change-type drought (Syphard et al. 2007, Pratt et al. 2013). Despite wide spread application of fuel treatments in chaparral, there have been few large-scale, long-term experiments to determine how fuel hazards, non-native species invasion, and shrub composition change through time.

A recent study addressed these research gaps by examining the effects of fuel treatment type (fire or mastication) and season (fall, winter, spring) over a 13-year timespan in chamise-dominated chaparral of northern California (Fig. 10.1.1) (Wilkin et al. 2015). Both treatment type and season of application had distinct influences on plant communities and fire hazards (Fig. 10.1.2). In contrast to prescribed fire treatments, masticated areas had consistently lower shrub cover, higher nonnative plant abundance, and more non-native grasses. However, mastication surprisingly increased buckbrush (*Ceanothus cuneatus*) cover, an obligate seeder and a preferred deer browse, compared to some fire treatments (Fig. 10.1.2). Fall treatments had consistently lower shrub cover, greater non-native plant abundance, non-native annual grasses, and greater buckbrush cover than spring or winter treatments. Ten years after treatment, fall mastication had the lowest shrub fuel load, but highest annual grass cover. These results highlight ecological and fire hazard reduction trade-offs among treatment types and season of treatment in chaparral shrublands.

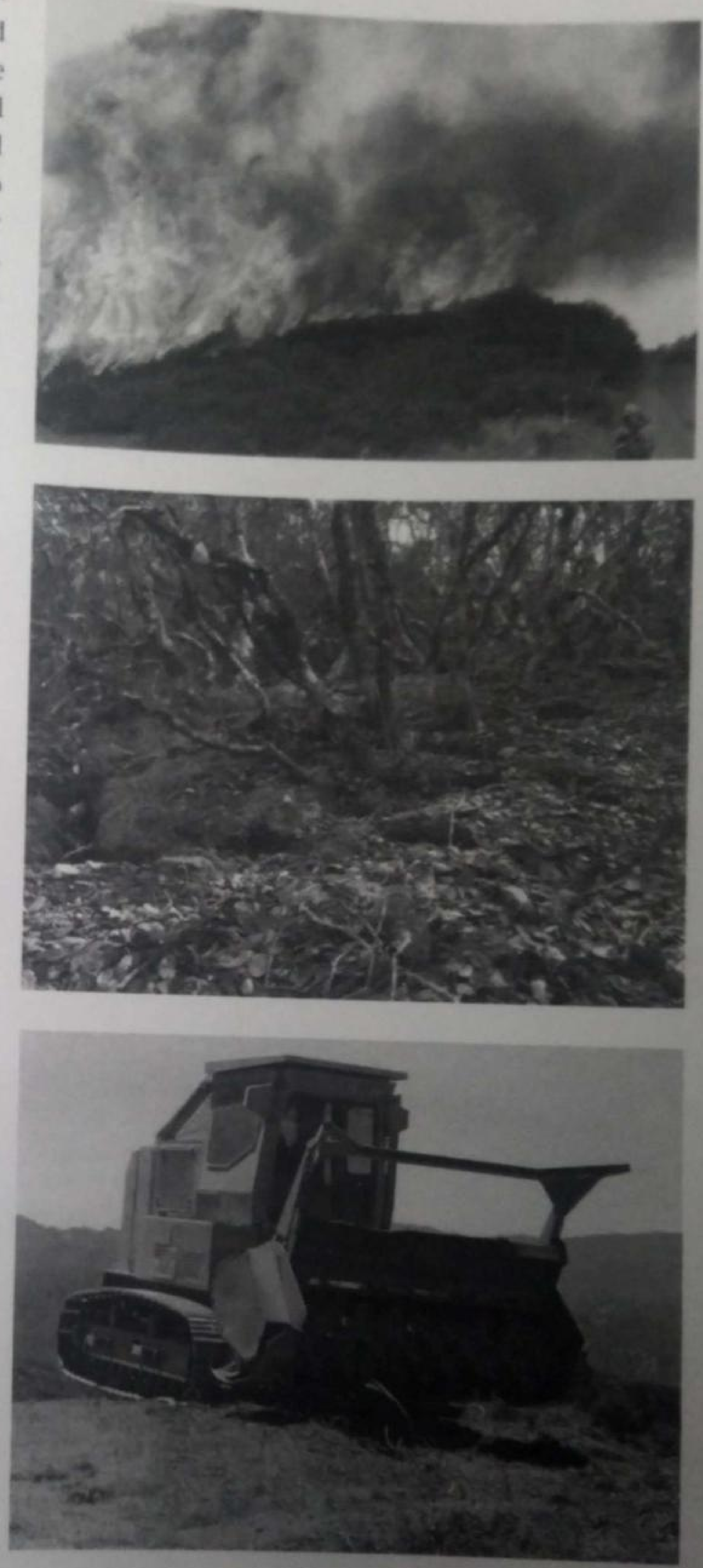

FIGURE 10.1.1 Fuel reduction treatments included prescribed fire (top), a control (center), and mastication (bottom) (photo credits: S. Stephens, K. Wilkin, and D. Fry).

(continued)

(continued)

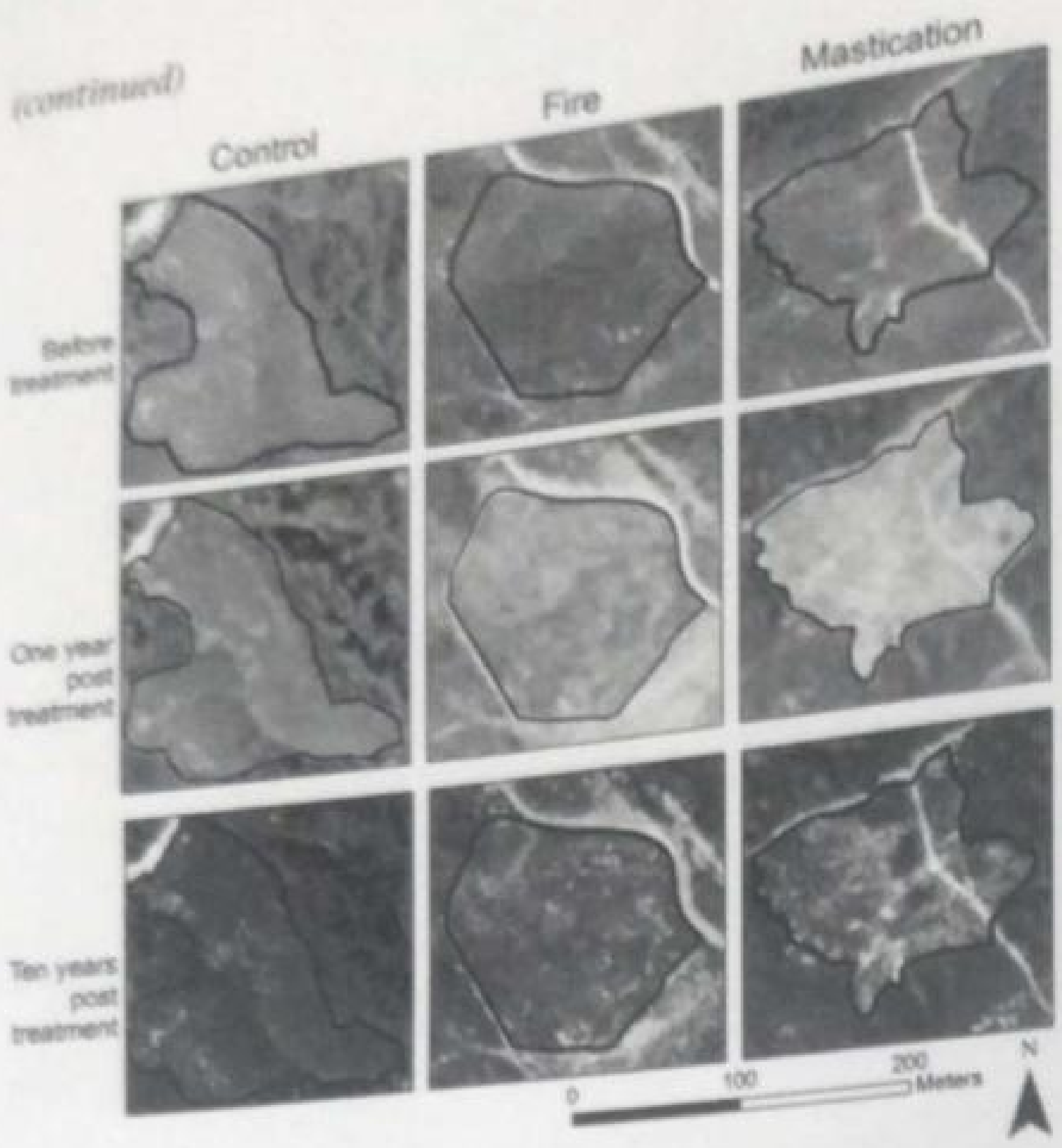

FIGURE 10.1.2 Repeated imagery of northern California chaparral fire hazard reduction study, demonstrating different long-term recovery between fire and mastication treatments (Wilkin et al. 2017). Shrub cover was persistent and continuous through time in the control unit. One year after treatment, the masticated unit had fewer shrubs, whereas fire treatment had rapid shrub recovery. A decade after treatment, differences in shrub recovery were maintained, where the fire unit had a nearly continuous shrub canopy and the masticated unit had a shrub-grass mosaic. The before treatment photos are black and white aerial photos from a USGS Digital Ortho Quarter Quad. The posttreatment photos are aerial images from National Agricultural Imagery Products for Mendocino County California.

Management Implications

Prescribed fire

- Generally fosters long-term native plant diversity and community structure
- Reduces fire hazard for a shorter time than mastication
- Decreases certain native shrubs

Mastication

- Fosters some native shrub species, but nonnative understory species invade and persist
- Reduces fire hazard more than fire, but may also increase fire frequency due to an increase amount of highly flammable annual grasses
- Increases certain native shrubs, such as the obligate seeder buckbrush

Season of treatment

- Fall treatments slow shrub recovery more than winter and, even more so, spring treatments
- Fall treatments promote greater nonnative plants and nonnative annual grass density than other seasons
- Fall fire treatments increase the preferred dear browse buckbrush. Within mastication treatments, season did not influence buckbrush outcomes

reintroduce fire as an ecosystem process. Some prescribed fire is presently occurring in parks in this area but this should be expanded. While redwood trees may not be solely dependent on fire to perpetuate in these forests, many ecosystem structures and functions can be enhanced or maintained.

Pre-Euro-American settlement fires were relatively frequent in Douglas-fir–tanoak forests due to their warmer, drier, inland locations and increased lightning activity at higher elevations; fire use by Native Americans was also important in areas with tanoak since their acorns were an important food. Fire suppression has reduced fire frequency, particularly in young-growth forests. The most significant change in old growth is the greater density of understory shrubs and trees that has increased vertical fuel continuity, which can increase the probability of high-severity crown fires.

A major concern for forests containing tanoak in the North Coast bioregion is sudden oak death, a disease caused by the pathogen, *Phytophthora ramorum* (Rizzo and Garbelotto 2003). The pathogen has reached epidemic levels in the north and central coast of California and is projected to continue to spread over the coming decades (Meentemeyer et al. 2011). In its lethal form the pathogen can kill native tree species, including tanoak, leading to rapid ecosystem changes.

Sudden oak death alters the distribution, availability, and abundance of fuels. In some locations more than 95% of the trees in a stand can be killed (USDA Forest Service 2009). Once dead, the tanoak leaves are often retained in the canopy for two or more years. Kuljian and Varner (2010) found that infected tanoak leaves had 5–10% lower foliar moisture content than leaves of uninfected tanoak, while the foliar moisture content of dead leaves in the canopy of sudden oak death killed tanoak ranged between 5.9% and 26.4% over the fire season. Increased availability and abundance of canopy fuels

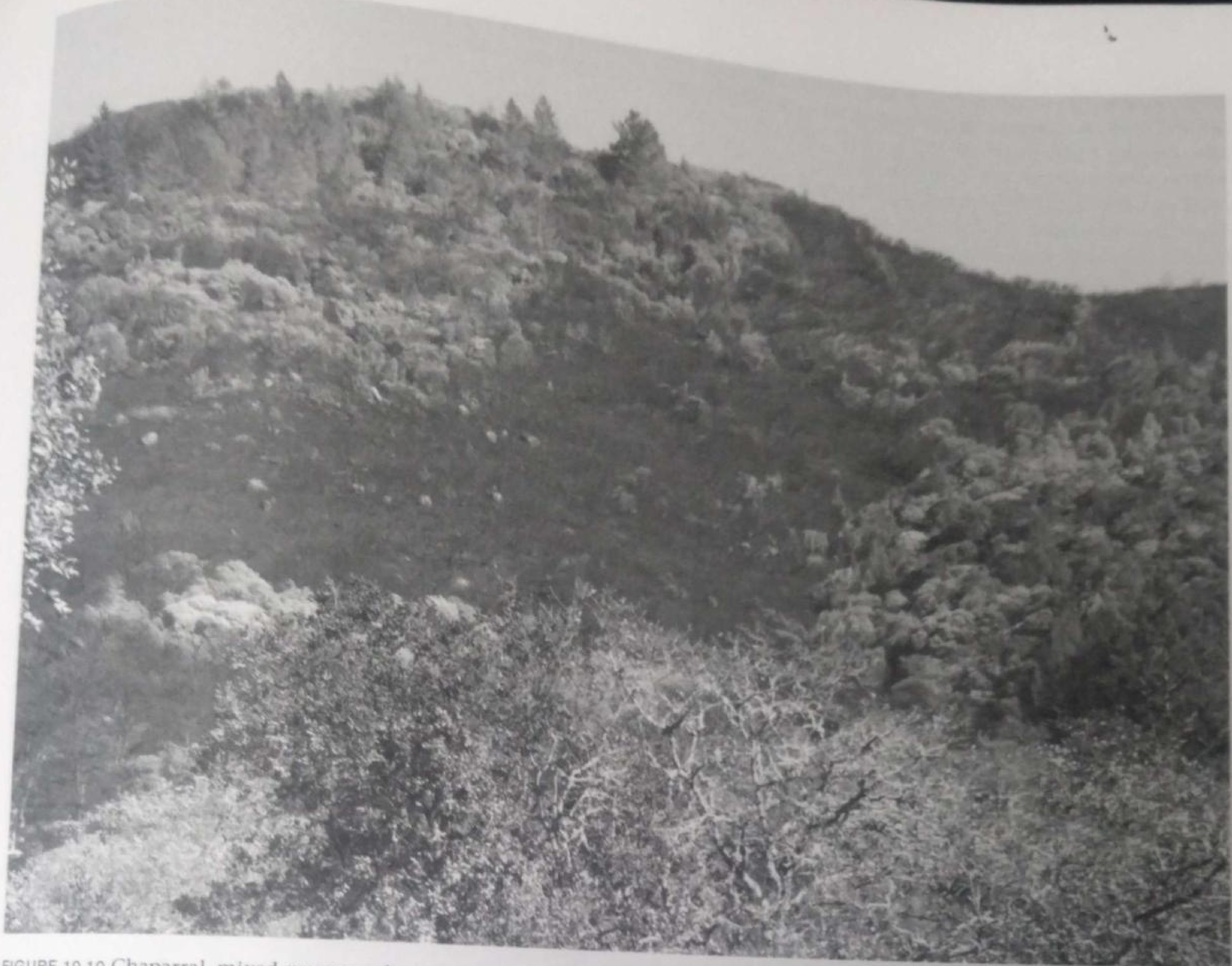

FIGURE 10.10 Chaparral, mixed evergreen forest, and oak woodlands burned in the Bouverie Preserve near Glen Ellen by the 2017 Nuns Fire (photo by Scott Stephens).

increases the potential for crown ignition and torching in these forests, which have been substantiated by laboratory experiments (Kuljian and Varner 2013) and modeling studies (Valachovic et al. 2011, Forrestel et al. 2015). Increases in the surface and crown fire behavior in sudden oak death infected stands can increase fire severity. In a recent study, researchers found that the density of standing dead tanoak was positively related to fire severity, leading to increased redwood mortality (Metz et al. 2011, 2013) following the 2008 wildfires in the Big Sur area. The possible impact of sudden oak death on the 2017 Northern Bay Area fires is an important question.

Removal of fire from coastal grasslands and Oregon oak woodlands has dramatically changed these ecosystems. The establishment of conifers and loss of Oregon oak have had cascading effects in Oregon oak woodlands. The reduction of Oregon oak populations reduces the amount of food and habitat important to wildlife in the region. Douglas-fir canopies reduce light penetration, which also reduces native herbaceous species resulting in a net loss of biodiversity in invaded areas (Livingston et al. 2016). The increased presence of Douglas-fir and the reduction of herbaceous plants also reduce the flammability of these ecosystems by increasing fuelbed bulk density and increasing fuel moisture (Engber et al. 2011). Lessened flammability results in reduced surface temperatures that are essential in killing young Douglas-fir trees (Engber and Varner 2012b) and contributes to a positive feedback that further promotes Douglas-fir invasion and continued loss of Oregon oak woodlands and prairies.

Frequent surface fires historically maintained these open oak woodlands and prairies in the Bald Hills in Humboldt County. The use of fall ignited prescribed fires should be used to enhance and maintain these ecosystems. Management ignited prescribed fires have been ineffective in reducing the density of competing conifers when they are over 10 ft (3 m) in height (Sugihara and Reed 1987). Hand removal of small Douglas-fir has been used in some areas and larger trees can be girdled to increase the dominance of the oak (Hastings et al. 1997). To maintain oak dominance, a minimum fire frequency of 3 to 5 years has been recommended (Sugihara and Reed 1987).

Fire research on the ecosystems of the North Coast bioregion should be expanded through adaptive management and experimentation including in riparian and wetland habitats. This area includes significant amounts of federal and state land along with the highest proportion of industrial forests in California. Site specific questions on fire's role in North Coast ecosystems need to be addressed on the full spectrum of private and public lands. Most fire-related research has been on public land and relatively little has been done on the extensive private lands found in this bioregion. Fire has been an essential ecological process throughout most of the North Coast and has greatly contributed to the biological diversity of the region. Fire management in this area could focus on improving the resilience of these ecosystems to maintain and enhance the biodiversity of the region. The recent 2017 Northern Bay Area fires were tragic in terms of lives and property lost but initial burn patterns may have coarsely followed vegetation types with high-severity fire in chaparral, mixed severity in mixed evergreen forests (coast live oak [*Quercus agrifolia*], California bay, Pacific madrone, Douglas-fir), and low severity in oak woodlands (Fig. 10.10). Much more information on the drivers and effects of these fires is needed to better understand their impacts.

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