# 16 Longleaf Pine Restoration in Context

## *Comparisons of Frequent Fire Forests*

*Seth W. Bigelow, Michael C. Stambaugh, Joseph J. O'Brien, Andrew J. Larson, and Michael A. Battaglia*

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## INTRODUCTION

To see a frequent-fire forest burn for the first time is to experience a remarkable feat of nature. Most people are accustomed to the slow change of forests with the seasons, not the instantaneous conversion of green and brown plant mass to smoke and char. Yet to visit such a forest a week after it burns is to see bright green shoots emerging, highlighted against a background of charcoal. Frequent-fire forests, or forests that regularly experience low-intensity/low-severity fires, although surprisingly common, challenge commonly held notions about what forests are and how they function. They are found in North and Central America including the Caribbean basin and U.S. landscapes such as the upper Midwest, the central hardwoods area, the Rocky Mountains, the Intermountain West, the eastern Cascades range in the Pacific Northwest, and the southeastern Coastal Plain. Despite their drastic differences in range, ecology, anthropogenic alterations, and conservation challenges, these forests share many similarities.

The purpose of this chapter is to place the structures and processes of the frequent-fire longleaf pine (*Pinus palustris*) ecosystem in the broad context of other forest ecosystems that historically experienced frequent fire. We also compare the restoration challenges of longleaf pine with those in other frequent-fire forests. First, we address the ecological commonalities among frequently burned forests and the kinds of degradation that threaten them. We provide vignettes of five other frequent-fire forests; examine how their ecology, restoration goals, and restoration approaches differ from longleaf pine; and evaluate whether restoration trends and ideas in other frequent-fire areas might be relevant for the longleaf pine range (and vice-versa).

## COMMON FEATURES OF FREQUENT-FIRE FORESTS

At large spatial scales (such as 1000 km), climate is the most important influence on where frequent-fire forests occur (Hawbaker et al. 2013). Higher average annual temperatures are negatively correlated with fire-return interval (Guyette et al. 2012), but this relationship can be modified by soils (Murphy and Bowman 2012): for instance, droughty excessively well-drained soils can support frequent-fire forests and savannas in cooler areas such as the sandy glacial outwash plains of the upper Midwest. In mountainous areas, slope and aspect alter the interception of sunlight, thereby influencing forest type and fire-return interval over fine spatial scales (Cansler and McKenzie 2014). Because lightning is the sole source of ignitions that are not caused by humans, the incidence of lightning strikes is important; for example, the extremely high incidence of strikes in the Southeast contributes to the development and maintenance of its frequent-fire forest (Outcalt 2008). Annual precipitation has a U-shaped relationship with fire-return interval, in that forests with very short ($\leq$2 years) fire-return intervals only occur in places that experience either very low (Southwest) or very high (Southeast) annual precipitation (Guyette et al. 2012). The timing of precipitation delivery and warm temperature occurrence is also important; despite high precipitation, frequent-fire forests can develop in western U.S. areas that have high mountains and Mediterranean climates because most of the precipitation takes the form of snow and is lost in the spring snowmelt (Stephenson 1990). Although such relationships are recognized at local to national scales, a global synthesis on what determines fire regimes in world ecosystems has not yet been developed (Bond and Keeley 2005).

Frequent-fire forests possess convergent structural characteristics regardless of the climatic and soil factors that originally shaped their development. In savannas and woodlands, one of the most notable characteristics is the open canopy condition, which permits abundant sunlight to pass through to the forest floor (Battaglia et al. 2003; Bigelow et al. 2011). Open canopies are the result of complex feedbacks among fire, climate, and vegetation that can contribute to limitations on fire behavior (Collins et al. 2009; Mitchell, Hiers, et al. 2009; Scholl and Taylor 2010; Parks et al. 2015). Surface fires help to maintain an open canopy by scorching and killing lower branches and branch tips; they allow overhead winds to ventilate the stand and dry fine fuels, dissipating heat and smoke that would otherwise build up (Albini and Baughmann 1979). Equally important, frequent surface fires kill many of the small trees that would otherwise eventually occupy available canopy space (Grace and Platt 1995a). In comparison to the tree species that characterize crown-fire regimes, the species of frequent-fire forests tend to have thick bark at the base of their trunks, the potential to become tall, and self-pruning ability (Pausas et al. 2004).

The role of fire as an agent of mortality for young trees in frequent-fire forests has striking consequences for species composition and forest structure. Young trees (saplings) are disproportionately susceptible to mortality and the variation among species in sapling bark thickness and other heat-insulating qualities is considerable, which means that frequent fire favors fire-resistant species (van Mantgem and Schwartz 2003; Hammond et al. 2015; Pausas 2015). Such species tend to produce pyrogenic litter that creates a positive feedback to the fire regime (Nowacki and Abrams 2008; Platt et al. 2016). In frequent-fire forests that have active fire regimes, the tree populations often have a broad irregular distribution of sizes with several peaks representing age classes that have established in the canopy gaps formed by disturbances (Arno et al. 1995; Bailey and Covington 2002; Moser et al. 2002; North et al. 2005).

The cultural traditions involved in human use of fire vary among the frequent-fire forests across the North American continent, but one common theme is the universal practice among indigenous peoples of applying fire as a tool for modifying the environment (Ryan et al. 2013). This practice has persisted and developed in virtually all areas of the continent (Pyne 1982). Fire histories of western landscapes show an abrupt decrease in fire frequency following the period of rapid settlement that characterized the latter half of the 19th century (Arno et al. 1995; North et al. 2005; Sherriff and Veblen 2007). The indigenous practice of woods-burning was adopted by European settlers in the Southeast to a far greater extent than in other regions. These burning practices have served as a model for reintroduction of fire from the mid-20th century onward. The preeminent 20th century California fire ecologist Biswell (1989) described how the course of his career was changed in 1940 by a day spent with a Georgia timber company employee who was assigned to burn pinelands; the experience convinced him that prescribed fires could be used beneficially in forest management. Today, prescribed burning is still done on a much larger area in the Southeast than in any other region (Melvin 2015).

## FOREST DEGRADATION: DISTURBANCE OUTSIDE OF HISTORICAL RANGE OF VARIATION

Fire maintains the structure and function of frequent-fire forests (Bond and Keeley 2005), and fire exclusion constitutes a damaging disturbance that has been all too common over the past 200 years (Stephens and Ruth 2005; Fill et al. 2015). Fire exclusion can take the form of active suppression of naturally ignited wildland fires, or neglect of prescribed fire as a management tool. Fire regimes have also been altered by more intensive harvesting than was typical during the development of frequent-fire forests and by landscape fragmentation, which prevents the spread of fire; such fragmentation can occur from roads, fences, housing, and wholesale conversion to agriculture (Duncan and Schmalzer 2004).

One common consequence of fire exclusion in frequent-fire forests is the establishment and growth of fire-intolerant shrub and tree species (Parsons and DeBenedetti 1979; Arno et al. 1995). This midstory development has the effect of decreasing transmittance of light to the forest floor, thus competitively eliminating many ground cover species because most plants of frequent-fire forests are adapted to abundant sunlight. Effects include increases in aboveground live biomass, total leaf area, and canopy-cover of trees that are less drought-tolerant and use more water per unit of leaf surface area than the trees they are replacing (Nowacki and Abrams 2008). Similarly, regeneration of dominant fire-adapted tree species may sharply decrease, either because their establishment depends on bare mineral soil, or because they are poor competitors and are sensitive to above- or belowground competition. This compositional change leads to increased whole-ecosystem water use, and can also result in decreased ecosystem resilience and increased drought vulnerability (Niinemets and Valladares 2006; Ganey and Vojta 2011; Dobrowski et al. 2015; van Mantgem et al. 2016).

Parsons and deBenedetti (1979) observed that fire exclusion in frequent-fire forest leads to a buildup of dead fuels and an altered canopy structure that forms "ladders" for fire to climb into tree crowns (the fire exclusion/fuel buildup perspective). Some of the evidence for this fire exclusion/fuel buildup hypotheses is that the proportion of a landscape burning at high intensity depends on the amount of time since the last fire or the amount of time since the departure from typical fire-return intervals (Harris and Taylor 2015; Steel et al. 2015). Others caution that a distinction must be made between fire exclusion in frequent-fire forests and fire exclusion in fire-dependent forests that have longer and more irregular fire-return intervals; for the latter, fire exclusion may not necessarily lead to larger or more intense wildfires (Johnson et al. 2001; Noss et al. 2006).

Expansion of shade-tolerant native trees from fire-free areas into fire-dependent forests is both a cause and a consequence of fire-regime interruptions. Under normal frequent-fire regimes, these trees are restricted to moist, shaded microsites (such as riparian areas or narrow ecotones between vegetation types). In part, these invading trees alter the fire cycle by shedding litter that holds moisture well or otherwise decreases flammability (Stephens et al. 2004). The shape and size of their litter, principally dead leaves, is a key determinant of their pyrogenic properties (Kane et al. 2008). Some nonnative plants can also disrupt the fire cycle. They carry fire exceedingly well and burn more intensely than the ground cover plants that they replace, predisposing canopy trees to increased mortality and thereby destabilizing the basic scaffolding of the forest (Brooks et al. 2004).

Isolated relict shade-intolerant trees of the historical forest are interspersed with dense, clumped stands of shade-tolerant trees in some fire-excluded forests (Gilliam and Platt 1999; Taylor 2004). In others, fire exclusion may simply result in higher density of shade-intolerant trees (Laughlin et al. 2011). These shade-tolerant trees tend to be Douglas-fir (*Pseudotsuga menziesii*) and the true firs (*Abies* spp.) in western landscapes, and mesic hardwoods in central and eastern landscapes (Larson and Churchill 2012; Hanberry et al. 2014). Dense stands of shade-tolerant trees result in a light-deprived ground cover that impedes the regeneration of shade-intolerant canopy trees (Veblen and Lorenz 1991; Gilliam and Platt 1999; Stambaugh and Muzika 2007; Bigelow et al. 2011).

Excessive harvesting of trees or other plants in frequent-fire forests also constitutes a disturbance that may be outside the historical range of variation. Commonly, frequent-fire forests are dominated by very large and old trees (legacy trees), which exert control on ecosystem properties and are therefore classified as keystone structures (Lindenmayer, Laurance, et al. 2012). This control can include suppressing competing trees, providing specialized habitats for mammalian or avian wildlife, and sequestering large amounts of carbon (Lutz et al. 2012). Large trees generally increase ecosystem heterogeneity both vertically (canopy height variability) and horizontally (within-stand patchiness). Common vegetative responses to the removal of large trees can include the release of younger age classes either of the same species or of more shade-tolerant species, setting the forest on a different successional trajectory. Such responses to large-tree removal usually result in stands that have more homogeneous structure in both their horizontal and vertical dimensions (Churchill et al. 2013). Once excessive harvesting of large old trees has occurred, restoring the characteristic ecosystem structure and function is difficult, especially in low productivity sites.

Another major consequence of changes in the fire-vegetation cycle is the alteration of dead wood dynamics. Dead wood, both as standing snags and as coarse woody debris, is a major habitat element in frequent-fire ecosystems (Harmon et al. 1986). Even though altered frequent-fire ecosystems continue to produce dead wood, its quality and dimensions may preclude its use as specialized habitat by many wildlife species. Many woodpecker species will only use large-diameter snags for excavating their nests (Zarnowitz and Manuwal 1985). Large down wood is used for many purposes by wildlife, including as runways for small mammals and as subnivean runways and refuges in snowy areas by the American pine marten (*Martes americana*) and other mid-sized mammals (Haggstrom and Kelleyhouse 1996). Because down wood in frequent-fire forests changes constantly and is susceptible to consumption by surface fires, a frequent-fire forest that loses live large-diameter trees can require many decades to restore its large dead wood component (Knapp 2015).

## FIVE FREQUENT-FIRE FORESTS

The examples described above show some of the many disruptions threatening the characteristic fire-vegetation cycles that maintain structure and diversity of frequent-fire forests. To provide multiple points of reference for understanding the longleaf pine ecosystem, we present vignettes of five other U.S. and Caribbean basin frequent-fire forests (Figure 16.1). They represent unique ecosystems, each with its own conditions and restoration approaches. The selection, which provides a broad geographic representation of the frequent-fire forest, is in no way intended to be comprehensive but rather to show how the interplay between ecosystems and human concerns determines restoration needs and priorities.

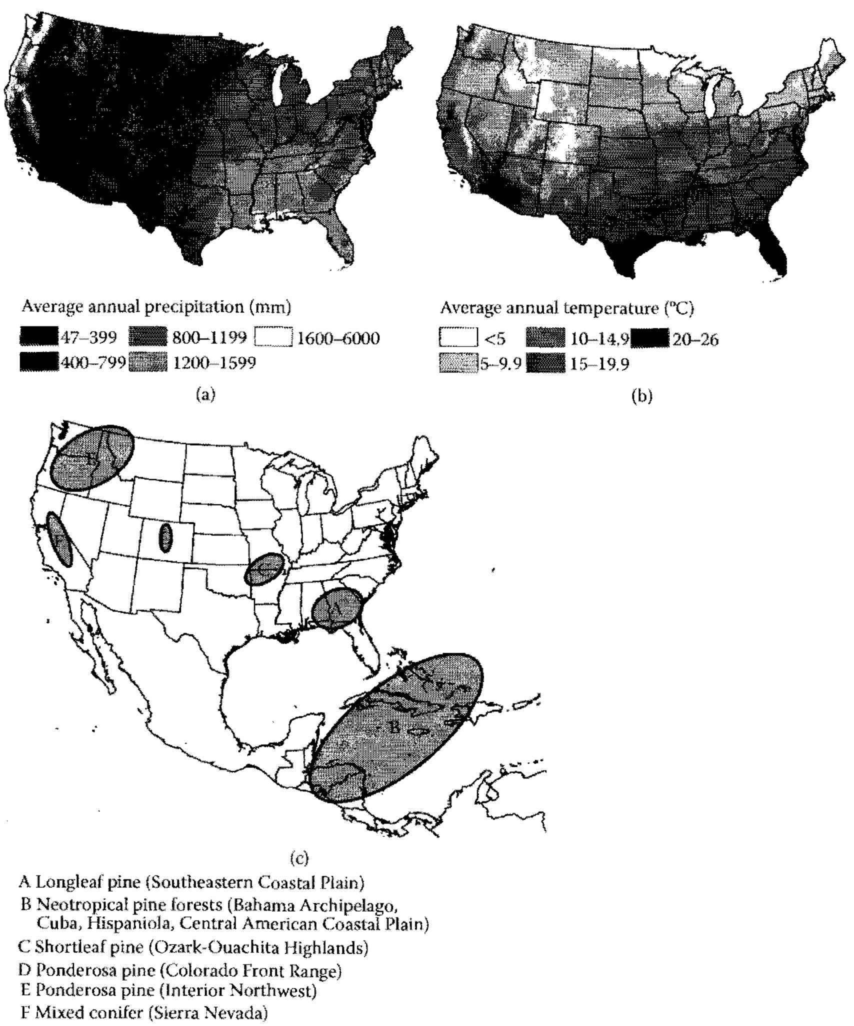



**FIGURE 16.1** Continental U.S. climate gradients, 1981–2011, and location of the frequent-fire forests: (a) Average annual precipitation (b) average annual temperature, and (c) location of five frequent-fire forest areas in the Caribbean and North America. (Data from Prism Climate Group, Oregon State University, http://prism.oregonstate.edu, created 15 December 2015.)

## Neotropical Pine Forests of the Caribbean Basin

The Caribbean Basin is home to several tropical pine ecosystems that depend on frequent fire to maintain structure and species composition, with fires typically recurring at 1–5-year intervals (Myers and Rodríguez-Trejo 2009). These forests occur in the Bahama Archipelago, Cuba, Hispaniola, and the Coastal Plain of Central America (Figure 16.2). The overstory is composed of one or two of several species—the Caribbean pine (*P. caribaea*), the West Indian pine (*P. occidentalis*), or the tropical pine (*P. tropicalis*)—all members of the southern pine group, subsection Australes; the midstory is sparse or absent; and the ground cover has a diverse assemblage of palms, shrubs, and herbs. In the Bahama Archipelago (Commonwealth of the Bahamas and Turks and Caicos Islands), the Bahamian variety of Caribbean pine (*P. caribaea* var. *bahamensis*) forms the canopy. On Hispaniola, the dominant pine is West Indian pine; in Central America, it is the Honduran variety of Caribbean pine (*P. caribaea* var. *hondurensis*). Cuban pine forests are dominated by the Caribbean variety of Caribbean pine, (*P. caribaea* var. *caribaea*), by tropical pine in sandy dry sites, or occasionally by the two species occurring together (de las Heras et al. 2005). The tropical pine forests are the ones most functionally similar to longleaf pine forests: they have a grass stage and their ground cover is dominated by graminoids.

Forest soils include acidic quartz sands in Cuba and the Central American Coastal Plain and volcanic clays in Hispaniola. Pines also occur on exposed limestone outcrops in the Bahama Archipelago, and portions of Hispaniola and Cuba. Limestone is an unusual substrate that has contributed to high levels of endemism in these forests (Cano Carmona and Cano Ortiz 2012).

### Fire Regime

Fire-return intervals in the area are typically <10 years and can be as frequent as annually (Myers et al. 2006; Harley et al. 2013). Canopy-derived fuels are critical in maintaining fuelbed continuity in the many Caribbean pine forests growing on exposed limestone. For example, in pine rockland forests—such as those of the Bahamas and Hispaniola—fire frequency is largely determined by overstory fuel production because the exposed limestone interrupts the continuity of other fuels. Vegetation occupies pockets of soil separated by patches of bare rock. If mature trees are present, pine needles carpet the rock allowing fire to spread across the landscape by creating fuel continuity among the patches of vegetation. Overstory productivity is driven primarily by rainfall. In areas with Ultisols and other better developed soils, ground cover that carries fire more evenly and regularly can often develop (Kellman 1984). Such areas include Honduran coastal areas and the western highlands of Cuba (de las Heras et al. 2005).

### Changes Resulting from Fire Exclusion, Grazing, and Logging

When fire is excluded, shrub-form broadleaved species are released and can replace pines as the dominant overstory in as little as 25 years. Litter shed by these broadleaved species contributes to changes in the ground cover moisture, which make broadleaf-dominated areas much less likely to burn. In areas where fire is customarily applied, overly frequent or intense fire can produce undesired outcomes. For example, fires lit annually at the height of the dry season in the Miskito savannas of Honduras result in intense and complete burns that inhibit pine regeneration, causing the conversion of extensive pine forests into grasslands. Elsewhere, lower-intensity annual fires can result in regeneration failure and ultimately, conversion to grassland (Myers et al. 2006).

In pine rockland ecosystems, the loss of the pine overstory results in fuel discontinuity, interrupting the fire regime and allowing the release of fire-suppressed broadleaved species. Nevertheless, these frequent-fire forests can often recover from overstory-removing disturbances such as hurricanes if regeneration persists and reestablishes fuel continuity (O'Brien et al. 2008). In areas where annual fire is applied in combination with logging, conversion of woodland to prairies can occur rapidly, as has happened in some areas of Honduras (Myers et al. 2006). Another form of ecosystem degradation is the nearly range-wide replacement of tropical pine with plantations of the Caribbean variety of Caribbean pine, which does not have a grass stage (de las Heras et al. 2005).

(a)

(b)

**FIGURE 16.2** Frequent-fire pine forests of the Caribbean basin: (a) Ground cover fire in a Caribbean forest, Abaco National Park, Bahamas; (b) Caribbean pine, Rio Platano Biosphere Reserve, Honduras.

*(Continued)*

(c)

**FIGURE 16.2** ***(Continued)*** Frequent-fire pine forests of the Caribbean basin: (c) West Indian pine, Sierra de Bahoruco, Dominican Republic. (Photographs courtesy of Joseph O'Brien.)

## Restoration, Management, and Conservation

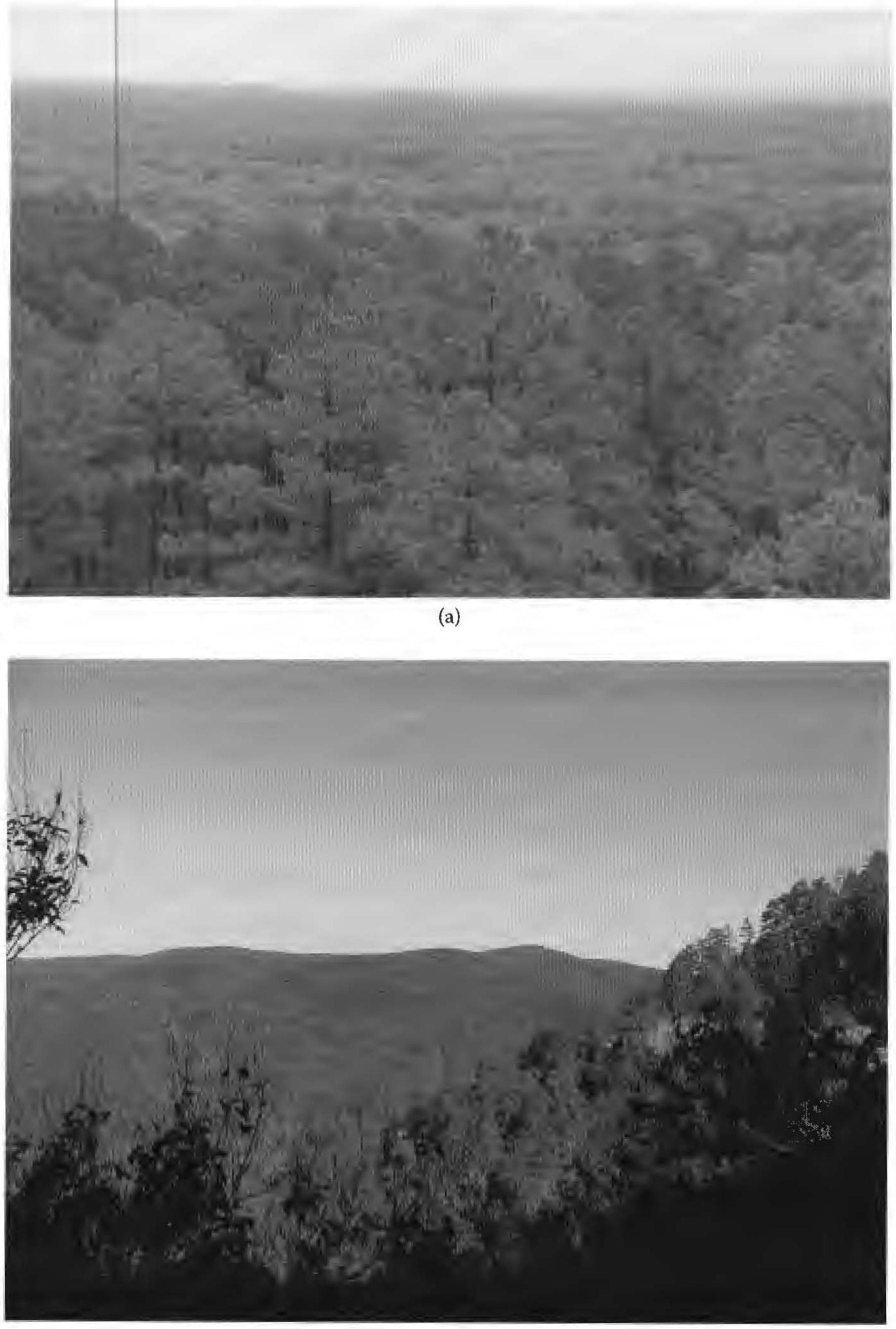
Few conservation measures are in place for tropical pine ecosystems. However, an even more critical issue for management of Caribbean basin pine forests is the maintenance of appropriate fire regimes. Wildfire is a major concern, and countries such as the Dominican Republic and Honduras have forbidden the intentional burning of forests. The exception is the Commonwealth of the Bahamas, which has recognized the value of fire as a management tool and has recently instituted legal changes to allow prescribed burning.

Sea level rise resulting from climate change is placing stress on these ecosystems and sometimes hindering recovery from disturbance. Inundation of islands by salt water following hurricane storm surges exacerbated by high sea levels can result in the complete loss of pines (Ross et al. 2009; Maschinski et al. 2011). In the Turks and Caicos, an infestation of an invasive scale insect resulted in the near complete loss of both the pine overstory and the regeneration in the ground cover (Malumphy et al. 2012). The sea level rise likely decreased the availability of fresh groundwater, which in turn would have increased the vulnerability of the pines to the invasive insect. In both situations, formerly pine-dominated areas are likely to transition to tropical broadleaved forests in the absence of intensive management intervention. Ultimately, the shift to dominance by broadleaved, fire-sensitive trees will lead to the loss of endemic fire-dependent plants and animals. The interaction of sea level rise and the ecology of fire-dependent ecosystems in low islands is a major conservation concern.

### Shortleaf Pine Forests in the Ozark Highlands

Shortleaf pine (*P. echinata*) is the northernmost member of the southern pine group, extending beyond the Piedmont Plateau and into the central hardwoods forest. Its broad range from

Connecticut to West Texas is indicative of its ability to grow in a multitude of climates, given favorable site conditions. The Ozark-Ouachita Highlands area of Missouri, Arkansas, and Oklahoma represents the northwestern range margin (Figure 16.3). Its climate is transitional between continental and subtropical. The northern margin of the shortleaf pine range aligns



(a)

(b)

**FIGURE 16.3** Shortleaf pine forests: (a) Ozark-Ouachita Highlands in Missouri, and (b) Boston Mountains in Arkansas. (Photographs courtesy of Michael Stambaugh.)

with the 43 cm winter precipitation isohyet; tree growth is limited by extreme winter temperatures (Fletcher and McDermott 1957; Stambaugh and Guyette 2004). Historically, shortleaf pine was a dominant forest type in this area, occurring in pure stands even near the outer margin of its range (within about 100 km).

Shortleaf pine is highly tolerant of stressful conditions, which provides an advantage over its oak (*Quercus* spp.) and hickory (*Carya* spp.) competitors on steep slopes and poor soils (Fletcher and McDermott 1957). Like eastern redcedar (*Juniperus virginiana*) it can occupy cliff edges and rock fissures. Southerly slope aspects with high solar radiation provide warm microsites where shortleaf pine thrives in northern parts of its range. It can occur naturally on gentle slopes, but usually only those with impoverished soils. Many shortleaf pine sites are sandy, or have a fragipan (a hardened, brittle soil horizon) that restricts growth of competing species (Graney and Ferguson 1972).

Shortleaf pine is an early successional shade-intolerant species, with little capacity to replace itself in closed-canopy forests, and only limited potential for recruitment through gap-phase succession in mixed oak-pine stands (Stambaugh and Muzika 2007). Historical sources—such as explorer notes, surveyor records, tree-rings, and photographs—consistently portray shortleaf pine as occurring in open, fire-maintained plant communities (Schoolcraft 1821; Guyette et al. 2006). Pine-bluestem (*Andropogon* spp.) savannas or pine-oak/oak-pine woodlands were common historical shortleaf pine communities, a sharp contrast to the closed-canopy shortleaf pine forests of today (Hanberry et al. 2014).

### Fire Regime

Frequent surface fire regimes (1–15-year intervals) characterized historical shortleaf pine communities throughout the Ozark-Ouachita Highlands (Stambaugh et al. 2013). Historical fire events were sometimes extensive, particularly during droughts and times of increased human population size and economic activity (Guyette et al. 2002, 2006). Shortleaf pine fire tolerance is based in part on a thick bark and the ability to resprout when small, for example, < 20 cm d.b.h. (Garren 1943). Other indicators of fire tolerance and dependence include its self-pruning ability, the presence of axillary buds, and the inability of seedlings to establish on deep litter (Grano 1949). Frequent burning promotes shortleaf pine establishment by reducing litter, killing fire-sensitive hardwoods, and increasing transmission of light to the ground cover thereby minimizing damping-off by reducing moisture near the ground surface. Other poorly quantified effects of repeated frequent burning include decreases in coarse woody debris, changes in organic matter, and fuel type changes from leaf litter to forbs and grasses. Burning to restore shortleaf pine provides ancillary benefits of increased soil nutrient availability, wildlife forage, and plant diversity (Masters et al. 1993; Liechty et al. 2005).

The history of fire in shortleaf pine forests since the 17th century provides insights into the conditions necessary to promote the species. Research on historical fires in shortleaf pine stands indicates that long-term, very frequent fire (1–4-year intervals for decades) can be detrimental to the survival of shortleaf pine seedlings (Dey and Hartman 2005; Stambaugh and Muzika 2007). The current understanding is that shortleaf pine requires intermediate and variable fire frequency to allow for regeneration and recruitment while controlling hardwood competition.

### Changes Resulting from Fire Exclusion, Grazing, and Logging

Challenges with sustaining shortleaf pine often arise from deep shade conditions, inadequate seed sources, and hardwood competition. Much of the present hardwood-dominated forest in the Ozark-Ouachita Highlands originated as shortleaf or mixed shortleaf-hardwood forests and underwent succession to hardwoods as a result of fire exclusion.

When hardwoods are clearcut on sites formerly dominated by shortleaf pine, their sprouts and small residual stems nearly always outcompete planted shortleaf pine even when the pines were established in advance of harvesting. This hardwood-shortleaf pine competitive asymmetry is the primary reason for the decline of shortleaf pine in the absence of fire. Furthermore, current prescribed fires may be less effective at killing or top-killing competing plants than fires of historical regimes because they typically occur under different weather conditions and burn through altered fuel types and thus are less intense and cover smaller areas.

### Restoration, Management, and Conservation

Fire suppression policies of the 20th century have caused widespread failure in shortleaf pine regeneration and recruitment. Site preparation techniques used for restoration from bare-ground include the ineffective (and possibly counterproductive) practice of mechanically ripping subsoil layers and burning to enhance growth and survival (Gwayze et al. 2007). Restoration techniques for fire-suppressed, hardwood-choked shortleaf pine forests include silvicultural thinning to 13 $m^2/ha$, removing midstory hardwoods but retaining some oak and hickory clumps, and prescribed burning every 3–4 years (Hedrick et al. 2007). Shortleaf pine regeneration and growth is difficult to promote on sites that are dominated by hardwoods.

Perhaps the greatest challenge for shortleaf pine management is to increase public acceptance of prescribed fire and other means of controlling hardwood competition (Hedrick et al. 2007). Its history of successful regeneration on highly disturbed sites indicates that shortleaf pine can tolerate extreme disturbances and suggests that areas being restored from hardwoods to shortleaf pine may require more frequent or more severe disturbances than prescribed fire alone. This is particularly true during the initial restoration phases of forests that have high hardwood density. In such situations, higher severity effects can be achieved by felling, girdling, or applying herbicides. Hardwood competition arising from the 20th century propensity for fire suppression may be unprecedented, demanding flexible and innovative management strategies.

## Ponderosa Pine and Mixed-Conifer Forests of the Interior Northwest

Dry ponderosa pine (*P. ponderosa*) and mixed-conifer forests make up the major frequent-fire conifer ecosystems of the Interior Northwest and northern Rocky Mountains (Figure 16.4), defined as the area east of the Cascade crest and west of the Continental Divide within the Columbia River watershed (Weaver 1943; Habeck and Mutch 1973). On the hottest and driest forest sites, ponderosa pine forms pure or nearly pure forests and savannas. On more mesic sites, which is the dominant condition for the area, ponderosa pine forms mixed-species stands with Douglas-fir, grand fir (*Abies grandis*), and western larch (*Larix occidentalis*). Douglas-fir and grand fir are both shade-tolerant short-needled species that become dominant in the absence of fire. Grand fir prevails on moister, milder sites compared to Douglas-fir, which is more tolerant of lower temperatures and larger moisture deficits. The thick bark of mature Douglas-fir makes the species moderately fire tolerant (Arno et al. 1995; Clyatt et al. 2016). Western larch is a highly shade-intolerant early-seral species that is exceptionally tolerant of fire (Harrington 2012; Hopkins et al. 2014). It forms extensive even-aged or multiaged stands originating from stand replacement or mixed-severity wildfires (Marcoux et al. 2015), in addition to being a component of mixed-species, frequent fire-maintained stands on drier sites.

The characteristic structure of Interior Northwest frequent-fire forests is a fine-grained mosaic of individual trees, tree clumps, and open areas (Larson and Churchill 2012). Tree densities are typically low, ranging from about 10 to 300 per hectare; higher stand densities tend to occur on moister sites, north-facing aspects, and valley bottoms (Hopkins et al. 2014; Clyatt et al. 2016). This structure, which is mediated by topography, emerges from a cycle of patchy tree mortality and regeneration that is driven by native bark beetles and frequent fires (Weaver 1943; Larson and Churchill 2012). Forests of north-facing aspects and canyon bottoms burn less frequently but at higher severity than forests of adjacent south-facing aspects (Figure 16.4). Large, widely distributed old trees of fire-tolerant species dominate the canopies of open dry sites and dense multistory patches (Lutz et al. 2012; Hagmann et al. 2014; Hessburg et al. 2015).

## Fire Regime

Tree-ring based estimates for historical point fire-return intervals were as low as 1 year and as high as >50 years in the dry forests of the Interior Northwest (Weaver 1959; Heyerdahl et al. 2001; Wright and Agee 2004), typically ranging from 5 to 20 years for ponderosa pine forests and 10 to 50 years for mixed-conifer forests (Arno 1980; Agee 1993; Arno et al. 1995). Virtually all of these tree-ring based fire histories show a cessation of frequent fire from about 1880 to 1905, although a few sites with continuing or restored frequent-fire regimes exist in large wilderness areas (Larson, Belote, Cansler, et al. 2013; Clyatt et al. 2016).

Most frequent-fire forests in the Interior Northwest occur in heterogeneous, topographically complex landscapes (Figure 16.4) that also contain mixed- and high-severity fire regime forests (Hessburg et al. 2015). Historical reconstructions show that the fire regime is strongly controlled by climate; years with widespread fires had warm springs and warm-dry summers, and years with fewer or smaller fires had cool springs and cool-wet summers (Wright and Agee 2004; Heyerdahl et al. 2008). The synchronizing effect of climate on fire regimes persists in the modern record, with warm-dry springs and summers being strongly associated with burning on larger acreages (Morgan et al. 2008; Littell et al. 2009), larger areas burned at high severity, and greater spatial aggregation of high-severity areas within fires (Cansler and McKenzie 2014).

(a)

**FIGURE 16.4** Ponderosa and mixed-conifer forests of the Interior Northwest: (a) Ponderosa pine and mixed-conifer forest on steep topography 15 years after the 1994 Butte Creek fire, North Cascades National Park Complex, Washington. *(Continued)*

(b)

(c)

**FIGURE 16.4 (*Continued*)** Ponderosa and mixed-conifer forests of the Interior Northwest: (b) Old-growth ponderosa pine forest with pinegrass ground cover on gentle topography, Dugout Research Natural Area, Oregon; (c) ponderosa pine and Douglas-fir woodlands and forests, Imnaha River Canyon, Oregon. (*Continued*)

(d)

**FIGURE 16.4** ***(Continued)*** Ponderosa and mixed-conifer forests of the Interior Northwest: (d) Fire-excluded ponderosa pine/mixed-conifer forest—with an overstory dominated by ponderosa pine with occasional Douglas-fir and ground cover dominated by grand fir and Douglas-fir seedlings, Minam River Valley, Oregon. (Photographs, in order of appearance, courtesy of Alina Cansler, Derek Churchill, Andrew Larson, and Andrew Larson.)

### Changes Resulting from Fire Exclusion, Grazing, and Logging

Successional changes to forest structure and composition caused by disruption of the historical frequent-fire regime were already apparent by the 1930s (Weaver 1943). In the decades-long absence of fire, heavy fuel loads accumulated while forests became denser and more dominated by shade-tolerant trees (Lunan and Habeck 1973; Arno et al. 1995). These changes were exacerbated by selective harvesting of the largest and most fire-resistant mature pines (Hessburg et al. 2005; Naficy et al. 2010; Merschel et al. 2014). The combined effects of past management includes increased surface fuel loads (Agee and Lolley 2006), increased late-successional multistory stand structures (Figure 16.4), and fewer large trees across the landscape (Hessburg et al. 2015).

### Restoration, Management, and Conservation

Five principles for restoration apply broadly to frequent-fire pine and mixed-conifer forests of western mountain landscapes. First, planning and management needs to be conducted at appropriate scales to restore multilevel landscape patterns and processes. Restoration of these frequent-fire forests cannot be achieved through a stand-based approach; the complex topography and landscape heterogeneity of arid western mountainous forests require planning at larger scales (Hessburg et al. 2015). Second, topography is the best guide for restoration of successional and habitat patchworks. Patch sizes and stand structures need to be tailored to ridge, valley, and aspect topographies because these topographic settings give rise to contrasting forest communities and fire regimes (Heyerdahl et al. 2001). Third, spatial patterns of trees need to reflect the expected fine-scale heterogeneity that is appropriate for the natural disturbance regimes and biophysical setting. Historical reconstructions of fine-scale tree spatial patterns are useful for achieving this effect (Larson and Churchill 2012; Churchill et al. 2013; Clyatt et al. 2016). Fourth, successful reintroduction of frequent fire requires close coordination of fire and vegetation management specialists for the management of fuel amount and configuration so as to avoid undesired fire behavior. Methods for reducing risk of crown fire include increasing canopy base height and reducing both crown bulk density and surface fuels (Agee and Skinner 2005). Finally, large old trees—the structural backbone of dry, frequent-fire forests—need to be retained and recruited. Often the most fire-resistant individuals in the population (Wyant et al. 1986; Regelbrugge and Conard 1993), they provide ecosystem services such as habitat for vertebrates and long-term carbon sequestration.

## Ponderosa Pine-Dominated Forests of the Colorado Front Range

The Front Range is the easternmost range of the southern Rocky Mountains, rising from the Great Plains, in a series of ridges and valleys that become the foothills of the Rocky Mountains and eventually reach 4300 m at the Continental Divide. This highly dissected landscape provides a range of elevations and aspects that influence the distribution of forest types.

The ponderosa pine-dominated forests of the Colorado Front Range are characterized by rugged, dry, and hot conditions. Annual precipitation is only 500 mm and average annual temperature is 6°C; most precipitation falls as snow but there is a monsoonal influence in the south. As shown in Figure 16.5, the vegetation is characterized by tall conifers with ground cover of herbs or shrubs (Peet 1981). Productivity of forests is fairly low: heights of typical 100-year-old ponderosa pines range from 8 to 24 m. At lower elevations (1700–2000 m) where the foothills meet the Great Plains, scattered Rocky Mountain ponderosa pines (*P. ponderosa* var. *scopulorum*) mix with shrubs or graminoids, depending on the soil type (Figure 16.5). Graminoids dominate the ground cover on finer textured soils and Rocky Mountain juniper (*Juniperus scopulorum*) is often present. An increase in elevation (2000–2200 m) or a northern aspect (or both) provides some additional moisture, allowing Rocky Mountain Douglas-fir (*Pseudotsuga menziesii* var. *glauca*) to become codominant with ponderosa pine. At elevations from 2200 to 2350 m and on southern aspects,

(a)

(b)

**FIGURE 16.5** Forests of the Colorado Front Range: (a) Low-elevation ponderosa pine stands on a rocky site at 1900–2200 m elevation; (b) ponderosa pine and Douglas-fir growing on coarse-textured soils. *(Continued)*

(c)

(d)

**FIGURE 16.5** ***(Continued)*** Forests of the Colorado Front Range: (c) Fire-excluded ponderosa pine/Douglas-fir forest with a dense forest floor of Douglas-fir seedlings; and (d) restored ponderosa pine forest showing individuals, groups of trees, and openings. (Photographs courtesy of Michael Battaglia.)

the xeric foothill woodland consists of sparse ponderosa pine with some Rocky Mountain juniper (Peet 1981). At 2450–2850 m, ponderosa pine woodlands occur on fine-textured soils with a ground cover of mountain muhly (*Muhlenbergia montana*). On coarse-textured soils, the tree density is higher and ponderosa pine and Douglas-fir grow together (Figure 16.5).

### Fire Regime

The historical (before 1860) fire regime for the Colorado Front Range was influenced by latitude, elevation, and aspect. Average fire-return intervals ranged from 12 to 59 years with more frequent fires at lower elevations (Veblen et al. 2000; Hunter et al. 2007). Fires occurred throughout the growing season but slightly earlier in the south (Brown et al. 1999; Brown and Shepperd 2001). A low-severity, frequent surface fire regime prevailed at lower elevations (Sherriff and Veblen 2006; Sherriff et al. 2014; Brown et al. 2015). At higher, steeper elevations, with mixtures of Douglas-fir and ponderosa pine, fire return-intervals were longer (>35 years) and historical fire regimes were likely of mixed severity.

Forests under mixed-severity fire regimes contain areas of low, moderate, and high severity burns. Low severity fires might kill seedlings and sometimes saplings, but the high severity fires would kill trees of all sizes, leaving patches of high mortality. Despite ongoing debate about the scale of high severity patches in these forest types and how high severity is defined (Fulé et al. 2014; Odion et al. 2014; Sherriff et al. 2014), the steep complex topography, variable weather conditions, and mixture of forest types clearly had a strong effect on the fire regime.

### Changes Resulting from Fire Exclusion, Grazing, and Logging

Ponderosa pine and mixed ponderosa-pine/Douglas-fir forests of the Colorado Front Range have increased in density since settlers began arriving from the East after gold was discovered in the mountains in 1859 (Kaufmann et al. 2000; Ehle and Baker 2003; Sherriff and Veblen 2006). In the early 20th century, fire frequency was substantially reduced with increases in timber harvesting, livestock grazing, and mining (Sherriff and Veblen 2007; Brown et al. 2015). These activities promoted new tree establishment of Douglas-fir in particular, contributing to denser forests and increased canopy continuity across the landscape (Figure 16.5). The prevalence of openings, especially small ones (<50 m in diameter) has decreased (Dickinson 2014). Present-day low-elevation ponderosa pine forests have more regular, homogeneous spacing and age structure than historical forests (Brown et al. 2015). As a result of these changes, contemporary forested landscapes have become vulnerable to high-severity crown fire; over recent decades, several wildfires have burned at high intensity across large, contiguous areas (Graham 2003) resulting in complete overstory mortality at spatial scales that limit ponderosa pine regeneration, especially at lower elevations (Chambers et al. 2016).

### Restoration, Management, and Conservation

The Colorado Front Range urban corridor—an area mostly on the Great Plains but also adjacent to forests—is already densely populated and development into the wildland-urban interface is projected to increase (Theobald and Romme 2007). By 2010, the increase in negative ecological, social, and economic impacts from large, high-severity fires led to funding under the U.S. Forest Service Collaborative Forest Landscape Restoration Program, whose goal is to reduce the threat of uncharacteristic fire while increasing forest resilience to fire, insects, disease, drought, and climate change (Haas et al. 2015). The Front Range program uses collaborative, science-based ecosystem restoration and has developed desired conditions for ponderosa-pine dominated forests that follow the principles articulated by Hessburg et al. (2015). The intent is to establish forests that are spatially heterogeneous across plots, stands, watersheds, and landscapes by creating canopy openings and groups of trees while retaining individual trees; the proportion of openings, clumps, and individual trees is determined by topography (Figure 16.5). Old, large-diameter trees are protected. Monitoring of implementation, effectiveness, and ecological impacts is paramount to the adaptive management approach of the program.

Implementation of restoration treatments has revealed some obstacles to whole-landscape restoration efforts (Underhill et al. 2014). Early on, most restoration prescriptions were traditional thin-from-below fuels-reduction treatments in which contractors selected the trees to remove in a process called "designation by prescription" (Dickinson and Cadry 2016). The switch to variable tree spacing using the individuals-clumps-openings approach (Churchill et al. 2013) has been challenging, and for now, leave-tree marking in demonstration areas is being used to guide operators and tree-marking crews. Similarly, the introduction of prescribed fire has been stymied by concerns about smoke and escapes. Prescription burning windows have been narrowed by the proximity to major urban areas, travel corridors, and private lands (Ryan et al. 2013). Heavy surface and canopy fuel loads are a major concern because of the potential for high tree mortality or escapes during prescribed burning (Dether and Black 2006).

### Mixed-Conifer and Ponderosa Pine Forests of the Sierra Nevada Range

The mixed-conifer forest of California grows in the rugged mountain ranges of the Sierra Nevada, Cascades, and parts of the coastal ranges. The mixed-conifer forest occurs at mid-elevations, roughly 1000–2000 m. The climate is Mediterranean in seasonality, with most precipitation falling as snow in winter or as rain in spring and autumn; summer droughts can persist from April through September. Average precipitation is about 1000 mm, and average monthly temperatures range from -5°C in January to 30°C in July. The asynchrony between the arrival of moisture and the availability of energy (from increases in air temperature) in the growing season profoundly affects the character of the vegetation (Stephenson 1998).

The vegetation is characterized by tall conifers with a ground cover of shrubs and herbs (Figure 16.6); shrub cover is often larger than that of forbs and grasses. The shrub cover is diverse and includes California lilacs (*Ceanothus* spp.) and manzanita (*Arctostaphylos* spp.). Both conifers and shrubs are sclerophyllous, meaning that they have thick tough leaves that are characteristic of drought-stressed environments. The dominant trees of the mixed-conifer forest (Fites-Kaufman et al. 2007) include California black oak (*Q. kelloggii*), ponderosa pine

**FIGURE 16.6** Mixed-conifer forests of the northern Sierra Nevada in California and Nevada: (a) Large Jeffrey and sugar pines, left middle ground, with dense forest floor of small-diameter white fir and Douglas-fir, background. *(Continued)*

(b)

(c)

**FIGURE 16.6** ***(Continued)*** Mixed-conifer forests of the northern Sierra Nevada in California and Nevada: (b) Ground cover of mountain whitethorn shrubs with overstory of incense cedar; (c) stand of white fir and Douglas-fir after mastication of small-diameter stems. *(Continued)*

(d)

**FIGURE 16.6** ***(Continued)*** Mixed-conifer forests of the northern Sierra Nevada in California and Nevada: (d) Smoke from slash pile burn after fuels-reduction thinning, Lake Tahoe Basin. (Photographs courtesy of Seth Bigelow.)

(ssp. *critchfieldiana*), Jeffrey pine (*P. jeffreyi*), sugar pine (*P. lambertiana*), white fir (*Abies concolor*), incense cedar (*Calocedrus decurrens*), and Douglas-fir (*Pseudotsuga menziesii* var. *menziesii*). The conifers grow readily to diameters of >1 m; the maximum height of the tallest of these trees, the sugar pine, is >60 m.

## Fire Regime

Estimates of median point fire-return interval before settlement from eastern states (before 1849) for mixed-conifer and ponderosa pine forests in the Sierra Nevada and southern Cascades range from 10 to 20 years (Taylor 2000; Stephens and Collins 2004; North et al. 2005; Scholl and Taylor 2010). The historical fire regime was mixed severity, consisting of mostly low-intensity fires that spread slowly through the undergrowth; flare-ups during which flames passed from the surface to the canopy and then spread from tree to tree—resulting in patches of dead trees that eventually became openings or shrub fields—were likely a normal feature of the mixed-severity fire regime (Collins and Stephens 2010; Collins et al. 2015). Estimates of patch size range from 0.2 ha (Scholl and Taylor 2010) to <4 ha, with occasional patches reaching 60 ha (Collins and Stephens 2010).

## Changes Resulting from Fire Exclusion, Grazing, and Logging

The structure of the mixed-conifer forest has changed dramatically from the onset of settlement to the present day. Historical accounts (Muir 1894) and reconstructions of stand structure from early timber surveys and other sources suggest groves of immense trees with open ground cover, characterized by low tree densities (2–315 per hectare). Canopy tree cover ranged from 20% to 30% or 45% (Collins et al. 2011; Lydersen et al. 2013; Barth et al. 2015; Stephens et al. 2015), compared to the present-day range of about 50%–80%. Increases in canopy tree cover have been accompanied by decreases in ground cover; for example, a site in the central Sierra Nevada has undergone a decrease in shrub cover since 1929, down from 30% to 2.5% (E. Knapp et al. 2013).

The higher-density canopy cover of the present-day forest is accompanied by a change in canopy structure—with many more branches and much more leaf area in the midstory. The shade-tolerant white fir, incense cedar, and Douglas-fir have become more common. Because frequent surface fires are no longer a part of the disturbance regimes in most of the forest, regeneration is no longer regulated by fire; consequently, a density-dependent, competition-driven mortality regime prevails. Diameter-frequency distributions more closely resemble the negative-exponential J-shaped curve of closed forests than the flatter curve of open forests, because of numerous small-diameter, shade-tolerant trees (North et al. 2005; Youngblood 2010).

Most dramatic among recent forest changes is the prevalence of large high-severity fires known as "megafires" (Miller et al. 2008). Increasingly, evidence points to a warming climate as the major driver for increased fire extent, intensity, and severity. The proximate mechanisms include a longer fire season owing to the earlier disappearance of the snowpack in spring and later appearance of rains in autumn (Westerling et al. 2006). If the new, high-severity fire regime persists and burns over areas that have not recovered since the last severe burn, the forest may be driven toward a lower-stature, shrubbier, open woodland condition (Collins et al. 2009).

### Restoration, Management, and Conservation

Forest restoration management, driven by the need to reduce fire hazard, has been underway at large scales in publicly owned forests of the Sierra Nevada for more than a decade. Fuels reduction generally involves thinning trees from the midstory to remove ladder fuels and removal of some codominant trees from the canopy (Agee and Skinner 2005). The resulting forest has lower canopy cover, lower density of foliage in the upper canopy (lower canopy bulk density), increased spacing among tree crowns, and—because shade-tolerant species are targeted—fewer firs (Figure 16.6). Post fuels-reduction forests still have higher stem density, canopy cover, and abundance of shade-tolerant trees than historical reconstructions suggest they ought to have, which could limit recruitment of shade-intolerant trees (North et al. 2007; Bigelow et al. 2011).

Prescribed fire is required for reestablishing a diverse ground cover of herbs and shrubs (Keeley and Fotheringham 2000; Wayman and North 2007); indeed many of these species have seeds that require smoke for germination, meaning that they will not establish under surrogate treatments. Prescribed fire is integral to reduction of surface fuels (live or dead flammable material within 2 m of the forest floor). Although prescribed fire is a common treatment in the national forests of the Sierra Nevada, benefits could be achieved by applying it—along with managed wildfire—over much larger areas. Constraints on wider use of prescribed fire and managed wildfire on public lands in California have been identified as air quality (Figure 16.6), rural house density, and a risk-averse agency culture (North et al. 2012).

## IMPLICATIONS FOR FREQUENT-FIRE FOREST RESTORATION

Restoration and management practices in the frequent-fire longleaf pine forests of the Southeast and other frequent-fire forests have the potential to inform each other. Fire exclusion, heavy logging, and grazing have left their mark everywhere, and degraded frequent-fire forests share functional similarities regardless of location. Potential impacts from climate change are omnipresent, and although novel ecosystems may be created as a result, a better understanding of the historical range of variation is needed in all forests. Although some restoration objectives may be similar among frequent-fire forests degraded by fire suppression, basic differences in the ecological processes of individual systems may require strategic variations from established approaches for changing successional trajectories. Following are common themes and landscape-based variations on needs for restoration research.

## Thinning

Thinning is frequently used to restore tree species composition, diameter class distribution, density, and spacing in degraded frequent-fire forests. Midstory trees are conspicuously absent from most frequent-fire forests that have active fire regimes, and prescribing the removal by harvesting of such trees is common for fire-excluded forests (Agee and Skinner 2005). But the results of decades of fire exclusion cannot be undone with a single treatment. Structural and species composition targets can be challenging to meet in stands where minor-component species have become significant parts of the stand. Removing all undesired trees—often shade-tolerant species—can result in a canopy cover that is too sparse to support some wildlife species (Stephens et al. 2014) or provides too little flammable litter to perpetuate a continuous fire regime (Jack, Mitchell, et al. 2006).

When less-desired tree species are retained to maintain a continuous source of fuels, regeneration from these trees can be abundant, overwhelming the seed production of large-seeded pine species. For example, after a restoration thinning experiment in a Sierra Nevada mixed-conifer forest, seed rain of residual shade-tolerant white fir and incense cedar was 5–26 times larger than Jeffrey and sugar pine, compromising efforts to shift stands toward increased pine abundance (Zald et al. 2008). In shortleaf and longleaf pine forests, midstory hardwood removal can leave behind rootstocks that sprout readily, requiring the application of herbicides or a rigorous burn schedule to keep pruning back the new growth by top-killing the sprouts, preventing them from getting large enough to become fire resistant (Jack, Mitchell, et al. 2006).

In forests that have very long-lived trees such as longleaf pine and ponderosa pine, stable populations can be maintained with highly infrequent and episodic reproduction and size distributions in which large trees frequently outnumber small trees (O'Hara 2009). This broad irregular distribution of tree sizes represents a striking contrast to the negative exponential curve that is commonly used as the target stand structure in uneven-aged management systems such as the BDq system in which stocking is controlled by a basal area level, maximum diameter, and a q factor (Guldin 1991; Guldin and Baker 1998). These observations suggest that cutting/thinning to create an exponentially decreasing diameter distribution may not be an appropriate approach for restoring many frequent-fire forests (Franklin et al. 2007). Of particular concern is that the choice of model parameters for the BDq system can result in the cutting of large legacy trees that are both irreplaceable (Hessburg et al. 2015) and critically needed for wildlife habitat, carbon sequestration, and other ecosystem services (Brockway et al. 2014).

## Restoring Spatial Pattern

Longleaf pine ecosystems do not have the aridity and topographic complexity that would predispose them to a mixed-severity fire regime, nor do they possess the characteristic patch size of even-aged trees that is sometimes found in conifer forests of semi-arid western landscapes (Palik and Pederson 1996; Pederson et al. 2008; Collins and Stephens 2010). Historical reconstructions are useful for developing silvicultural prescriptions, establishing tree-marking guidelines, and monitoring benchmarks when restoring fine-scale stand structure and tree spatial patterns (Larson and Churchill 2012; Clyatt et al. 2016). As mentioned in the vignette on Ponderosa Pine and Mixed-Conifer Forests of the Interior Northwest, spatial patterns of trees ought to reflect expected fine-scale heterogeneity given natural disturbance regimes and biophysical settings (Hessburg et al. 2015). Intensive exploitation of the easily accessible forests of the southeastern Coastal Plain have left few intact old-growth longleaf pine forests available for study (Varner and Kush 2004; Mitchell, Engstrom, et al. 2009), but spatial analysis of one of the remaining few suggested clumps of seedlings and saplings in a mosaic that is superimposed on a matrix of widely spaced (loosely aggregated) mature trees (Platt, Evans, and Rathbun 1988; Noel et al. 1998).

In most present-day fire-excluded forests, small- and medium-diameter trees occur in larger, denser clumps than they did under historical conditions. Producing more small clumps and small openings (for example, openings with a 25 m radius) is a common treatment goal, which is often achieved by breaking up large clumps (Taylor 2004; Dickinson 2014). In arid western forests, spatial targets include increasing within-stand heterogeneity by creating tree patterns that are composed of local tree clumps, openings, and widely spaced single trees (Knapp et al. 2012; Larson and Churchill 2012; Underhill et al. 2014). Approaches for restoring a historical spatial structure involve increasing stand heterogeneity at several spatial scales (for example, from clumps of trees to stands with gaps of varying sizes). In contrast, the usual fuels-reduction thinning approach strives to maximize space around each residual tree and thereby minimize the risk of crown fire and maximize growth. This approach, although of demonstrated effectiveness in changing fire behavior, deters regeneration of shade-intolerant, fire-tolerant tree species and fails to provide high-quality habitat for wildlife (Bigelow et al. 2011; Stephens et al. 2014). In the western forests, where fire models in current use cannot adequately simulate within-stand heterogeneity, more stand-level research is needed on fire-behavior responses to within-stand heterogeneous spacing treatments. Models that can simulate within-stand heterogeneity and fire behavior are under development (Parsons et al. 2011; Hoffman et al. 2016) but their effectiveness is not yet known (Alexander and Cruz 2013).

Research on the historical structure and disturbance regime of frequent-fire forests suggests that gaps are a pervasive structural element (Palik and Pederson 1996; Noel et al. 1998; Larson and Churchill 2012; Dickinson 2014) that may be necessary for the successful regeneration of light-demanding fire-tolerant species (Stambaugh et al. 2002; Palik et al. 2003; Bigelow et al. 2011); considerations specific to longleaf pine are discussed in Chapters 4 and 7. Gap-based, group-selection silviculture has been suggested as a management system for longleaf pine forests, but results have been mixed. The optimal opening size is unclear despite substantial research on the question (Palik et al. 1997; Pecot et al. 2007; McGuire et al. 2001), and securing consistent regeneration and fire behavior in experimental openings has been difficult (Jack, Mitchell, et al. 2006; Mitchell, Engstrom, et al. 2009). Some have suggested possible gap sizes (Grace and Platt 1995b; Brockway and Outcalt 1998; McGuire et al. 2001; Mitchell et al. 2006) but there is not agreement among the different studies (Pecot et al. 2007). It is probable that there is no one consistent group opening size that can be uniformly applied across the natural range of longleaf pine, and appropriate group opening size should be based on specific site and vegetation conditions. An alternative to gap-based approaches—single-tree selection—has been suggested as an effective silvicultural system that ensures adequate regeneration to sustainably manage multiaged longleaf forests with modest timber yields (Pecot et al. 2007; Neel et al. 2010).

Tree marking is increasingly recognized as an important element of forest management that is worthy of study in its own right (Vitková et al. 2016). Variable-retention or multiscale thinning designs require adjustments to traditional tree-marking systems (Churchill et al. 2013; Brockway et al. 2014). The individuals-clumps-openings (ICO) system (Churchill et al. 2013) is a practical method to manage and monitor heterogeneous tree patterns within patches. Substantial training is required for tree markers to faithfully translate complex prescriptions into marked stands (Underhill et al. 2014). Rapid assessment of tree marking, timely monitoring, and providing feedback to markers are crucial. A team of two markers can be more effective than a larger team that is deployed in the traditional manner of a straight-line formation; markers are encouraged to collaborate on which natural clumps of trees should be retained (Knapp et al. 2012). Designation by prescription may save the time of foresters, but this approach risks becoming a zero-sum game if time saved by foresters becomes time lost by loggers (Dickinson and Cadry 2016). Finally, although research approaches exist for assessing the effectiveness of silvicultural interventions to restore historical spatial structure (North et al. 2007; Churchill et al. 2013), managers lack practical methods for assessing whether the application of spatially heterogeneous prescriptions has succeeded.

## Reintroducing Fire to Fire-Excluded Landscapes

Reintroduction of fire is a universally advocated measure for restoration of frequent-fire forests, but the effort is not without many associated challenges (Brown et al. 2004; Taylor 2004; Dey and Hartman 2005; Mitchell et al. 2006; Ryan et al. 2013). Successful reintroduction often requires management of fuel amount and configuration and careful consideration of weather conditions and firing techniques to avoid undesired fire behavior and consequent damage to vegetation. Silvicultural measures such as eliminating ladder fuels and decreasing canopy bulk density (Agee and Skinner 2005) are more likely to be necessary in arid western forests than in central or eastern ones. The humid climate of the Southeast means that extreme fire behavior and escaped fires are lesser concerns than in western forests (Dether and Black 2006); but in all situations, restoration and management of frequent-fire forests benefit from close coordination of fire and vegetation-management specialists.

Successful reintroduction of fire in the Southeast and elsewhere may require that initial applications be conducted under cool-burning conditions to gradually reduce fuels, particularly duff, without causing excessive tree mortality (Stephens and Finney 2002; Varner et al. 2005). Mortality of legacy trees is a common concern when fire is reintroduced to ecosystems that have a long history of fire-exclusion (Maloney et al. 2008; Varner et al. 2009; Harrington 2012). Raking duff away from tree bases is an effective yet time-consuming method of forestalling mortality of large trees; a better understanding of the controls on duff moisture would likely provide more efficient ways of managing tree mortality that results from smoldering duff (Banwell et al. 2013).

Understanding public perceptions—and correcting misperceptions—is vital for successful fire reintroduction. Smoke management, the problem of decreasing burn windows that are enforced to address air quality concerns, and a changing climate (Mitchell et al. 2014) are widespread challenges (see Chapter 13). Research indicates that public approval increases as people become more familiar with the goals of prescribed fire application (Jacobson et al. 2001; Winter et al. 2004). Southeastern states are fortunate in this regard because their long history of forest burning and large area that is burned annually contribute to public acceptance (McCaffrey 2009; Way 2011; Melvin 2015). Effective communication about the goals and procedures of prescribed burning becomes even more important in regions that do not have this history.

Because no pine seedling is fire-tolerant immediately after it germinates, understanding the developmental schedule of fire tolerance is essential for creating prescribed fire regimens that foster pine regeneration and recruitment while selectively reducing competing plants. Prescribed fire that is applied either too often or not often enough for the target species can result in regeneration failure and can sometimes cause pine forests to be replaced by grassland (Myers and Rodríguez-Trejo 2009) or other vegetation types. The development of fire tolerance in longleaf pine is well known (see Chapter 4): newly germinated seedlings become fire tolerant after a year by entering a grass stage in which their long needles insulate the apical meristem; after 5–15 years, the juveniles rapidly grow taller (known as bolting) for several years and are again vulnerable to fire (Grace and Platt 1995a). With continued growth, they eventually become much less vulnerable to fire-induced mortality. The same developmental sequence occurs in tropical pine (de las Heras et al. 2005). Shortleaf pine does not have a grass stage and—depending on site conditions—can take up to 7 years to develop significant fire tolerance (Weddell and Ware 1935). The long fire-free period required for shortleaf pine establishment and recruitment can provide an advantage to competing hardwoods, which can become impossible to control without applications of herbicides. Thus, even with knowledge of the fire-tolerance developmental schedule for a species, achieving management goals using only fire as a tool may still be difficult.

## Ground Cover Restoration

In frequent-fire forests, fire is carried by leaves and other ground fuels, by surface fuels such as grasses and shrubs, and sometimes by canopy fuels. The characteristic vegetation that carries fire changes in different areas according to climate and plant life form distribution. In the humid Southeast, the interaction between ground fuels (such as needles) and surface fuels takes on particular importance because high decomposition rates can quickly render needles inflammable (Mitchell, Hiers, et al. 2009; Platt et al. 2016). For example, wiregrass (*Aristida stricta*), with its spreading crown of rigid leaves, provides perches that allow fallen leaves to dry (Hendricks et al. 2002; Nelson and Hiers 2008). When fire is excluded, species richness of the ground cover declines (Moore et al. 2006; Peterson and Reich 2008; Kirkman et al. 2016) and the growth of wiregrass and other surface fuels is curtailed.

Spatial continuity between ground and surface fuels is required for prescribed fire to carry (Miller and Urban 2000). Anything that disrupts fuels continuity (such as logging, roads, or a change in species composition) changes fire behavior and the effectiveness of prescribed fire (Loudermilk et al. 2012). Longleaf pine restoration guidelines emphasize the need for evenly distributed overstory pines to provide the needle cast that is necessary for the spatial continuity of surface fuels (Mitchell, Hiers, et al. 2009). The complete removal of longleaf pine from extensive parts of its former range has provided an opportunity to study the establishment of the species in agricultural fields from the ground up (without recourse to remnant stands or trees). Such restoration involves challenges of establishing a scaffolding of trees, suppressing old-field vegetation, introducing native ground cover, and applying fire (Addington, Greene, et al. 2015). If longleaf pines and native ground cover are planted at the same time, frequent thinning is required to prevent the development of a high, uniform canopy cover that can shade out the ground cover (Harrington and Edwards 1999). Mulligan et al. (2002) propose an alternative: establish longleaf pines first, wait until a dense canopy develops that suppresses old-field plants, harvest some trees for economic returns, and then plant native ground cover (see Chapter 11). Although an even thinning (usually every third to fifth row) would typically be applied, the creation of larger openings—as would be done in a group selection—might be preferable when forest floor restoration is the primary consideration (Sharma et al. 2012). When larger openings are created, however, less needle cast reaches the center of openings, prescribed fire intensity diminishes, and control of hardwoods becomes tenuous (Jack, Mitchell, et al. 2006).

## Landscape and Topographic Approaches to Restoration

The incorporation of landscape topography into planning is a frontier for restoration research (Palik et al. 2000; Reynolds et al. 2013; Hessburg et al. 2015). Geomorphology, topography, and soil characteristics can be used to guide restoration of the longleaf pine forests and wet depressions in the southeastern Coastal Plain: because fire frequency is linked to geomorphological characteristics, maintaining longleaf pine structure and regeneration may require a shorter fire interval in more fertile or mesic landscapes than in the deep sand of xeric sandhills (Gilliam and Platt 1999; Kirkman, Goebel, et al. 2004). In western landscapes, topography can guide restoration of successional and habitat patchworks when patch sizes and stand structures are tailored to a particular ridge, valley, or aspect (Underwood et al. 2010; Hessburg et al. 2015). These topographic settings give rise to contrasting forest communities and fire regimes: sparse pine-dominated forests with lower levels of clumping and more openings on southern aspects and ridges compared to mixed-species stands containing multiple canopy layers and large tree clumps on north aspects and valley bottoms (Heyerdahl et al. 2001; Lydersen and North 2012). Regardless of biophysical setting, independent stand-level restoration treatments need to be developed into integrated, multiscale landscape restoration plans (Hessburg et al. 2015).

Wetlands are topographic landscape features of frequent-fire forests that have a much higher conservation importance than their size would suggest (Cohen et al. 2016). Such wetlands include the wet depressions of longleaf pine forests and the meadows of western mountain forests. Wet depressions and meadows have high plant diversity and provide foraging and breeding grounds for many aquatic and terrestrial organisms (Kaeser and Kirkman 2009; Erwin et al. 2016); they are highly vulnerable to degradation resulting from alterations in fire regimes, hydrological period, and grazing regimes (Miller and Halpern 1998; Dull 1999). When the fire regime is interrupted in the wet depressions of longleaf pine forests in southwest Georgia, mesic oaks become established, displacing wetland grass species and drastically altering wetland hydrology and habitat value (Kirkman et al. 2000; Martin and Kirkman 2009). When shrubs and conifers invade meadows of western mixed-conifer and ponderosa pine forests, the culprit is often an alteration in hydrology but changes in grazing and fire frequency can also be responsible (Weixelman et al. 1997; Berlow et al. 2003; Haugo and Halpern 2007). Identifying and correcting sources of wetland degradation is essential for maintaining biodiversity of frequent-fire ecosystems. Fire often burns into wetlands under the natural fire regime, although perhaps not with the same frequency as the rest of the forest burns; this should be anticipated and planned for under prescribed fire regimens.

### The Fire Next Time: Managing in the Anthropocene

Human-caused global climate change has accelerated the frequency and scale of stand-replacing wildfires in frequent-fire forests of western landscapes (Abatzoglou and Williams, 2016); these effects are projected to increase and spread throughout other U.S. regions over the 21st century (Pechony and Shindell 2010; Liu, Goodrick, et al. 2013). The increase in uncharacteristically high-severity fire has created a need for better landscape-scale restoration strategies in the wake of large wildfires: how do we manage previously fire-excluded, unrestored sites that have experienced uncharacteristically high severity fire? At present, management actions are often limited to short-term emergency response activities (such as culvert replacement) and controversial salvage proposals. Landowners and forest managers critically need models and tools to help them determine where fires have achieved restoration objectives and where additional postfire restoration or climate change adaptation treatments are indicated. The millions of trees killed by bark beetle outbreaks and historic weather events such as the California drought of 2013–2015 (Potter 2016) creates a related need: this raises questions about how to manage the resulting dead biomass and how to foster the development of an ecologically appropriate, climate-resilient forest community. Historic droughts and fires of 2016 in the Southeast suggest that this region is not immune from the cataclysmic ecosystem changes that are occurring in western landscapes.

## CONCLUSIONS

The preservation of a culture that favors prescribed burning and the development of a fire-dependent silvicultural system were key contributions of the Southeast to the restoration and management of frequent-fire forests. The science of frequent-fire forest management has rapidly spread throughout the United States and, despite markedly different landscapes, all regions can benefit from sharing information and analytic approaches. Understanding how regional restoration and fire management practices fit into frameworks for addressing issues—such as land use, climate, and topography—forms the basis for comparing and analyzing different approaches. What must be demonstrated are silvicultural and restoration practices that can be refined to serve specific goals, meet long-term management objectives for specific forest and wildlife habitats, and ensure continuation of ecosystem services such as watershed protection.

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