

Ecosystem management and ecological restoration in the Anthropocene: integrating global change, soils, and disturbance in boreal and Mediterranean forests

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ABSTRACT

Deforestation, rising temperatures, drought, fire and other ecological disturbances are reducing forest cover on much of the earth, and compromising the ability of forests to supply important ecosystem services. Ecosystem management and ecological restoration are focused on preventing and repairing ecosystem degradation, but the rapidity and pervasiveness of these global transformations threaten to exceed our capacity to plan for or respond to them at sufficient spatiotemporal scales. In this chapter we focus on forests in two contrasting biomes — boreal and Mediterranean — that are anticipated to experience major ecological changes in the 21st century. The relatively species-poor boreal forest covers 11% of the earth's surface and exerts a major influence on the global climate. Mediterranean climate region (MCR) forests are highly fragmented and cover less than 0.5% of the earth's surface, but the MCR regions support nearly 1/5 of the world's flora. Forests in both biomes are dominated by stress-tolerant taxa, but the principal source of ecological stress is very different: low air and soil temperatures in the boreal region, and lack of soil moisture during the warm growing season in the MCRs. Projected climatic changes will ameliorate the central ecological stress in boreal forests, with rising temperatures and increasing precipitation leading to generally better growing conditions for trees. In the MCRs, projected climatic changes will exacerbate hydrological stress, as warming and projected decreases in growing-season soil moisture worsen the annual drought. Overall, the direct effects of climate change on tree survival and growth are likely to be positive in the boreal region and negative in the MCRs, but interactions between these climate drivers and other stressors — changes in precipitation type, fire and pest outbreaks, invasive species, and so on — will have important and potentially contrasting indirect effects on soils and the forests that grow in them. In both biomes the implications for forest composition, structure, function, dynamics and sustainability are profound. We examine how interactions between global change, soils and disturbance are likely to affect boreal and MCR forests, and what the implications of these effects may be for ecosystem management and ecological restoration. We describe general patterns of climate, vegetation, soils, human history, and disturbance ecology in the two biomes and we summarize climate trends and projected future conditions, with focus on effects to soils. We finish with biome-specific summaries of current restoration strategies and practices, and a consideration of how soil responses to global change-related stressors and disturbances might require changes in the way we plan for and implement forest management and restoration. Both boreal and MCR forests and their soils will experience major changes over the next 50–100 years. Adherence to the basic tenets of ecosystem management and ecological restoration will provide the best chance of conserving these forests and sustaining the ecosystem services they provide.

Introduction

Ecosystem management relies on “our best understanding of the ecological interactions and processes necessary to sustain ecosystem composition, structure, and function” (Christensen et al., 1996). The concept of ecosystem management arose in the late 20th century as humans began to question whether biodiversity and important ecosystem services — e.g., water and commodity provision, soil productivity, carbon sequestration, recreational and cultural resources — could be sustained under contemporary societal trends for population rise, and expanding resource use. Key to the concept of ecosystem management are: (1) sustainability of resources and species population viability; (2) the importance of spatiotemporal connectivity, for example among the levels of the ecological hierarchy (species to ecosystems), across large landscapes and administrative boundaries, and through time; (3) ecological integrity, or the degree to which all ecosystem components and their interactions are present and functioning; (4) the maintenance of ecological dynamics, including disturbance regimes; (5) adaptive and cooperative management; and (6) the recognition of humans and their values as integral ecosystem components (Grumbine, 1994; Christensen et al., 1996; Millenium Ecosystem Assessment, 2005).

Ecosystem management is principally focused on large, relatively natural and autonomous landscapes (Aplet, 1999). However, human use of ecosystems usually results in some form of soil and ecosystem degradation, and under mounting human pressure severely fragmented and altered landscapes comprise progressively more of the Earth’s surface. Ecological restoration is “the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed” (SER, 2004). Attributes of successful restoration include: high ecological integrity and resilience, ecological connectivity between the restoration site and the larger landscape, sustenance of native biodiversity and control of non-native species, and a trend toward ecosystem self-sustainability (SER, 2004). The desire to repair and restore degraded landscapes is an innate human trait, and restoration of the composition, structure, and/or function of valuable forested ecosystems has been carried out by human societies for centuries, if not millennia (Vallejo, 2009). The vast majority of these efforts have been private, small-scale, and undocumented, but large-scale forest restoration projects also have a long history. Early brute-force examples include: the medieval reservation of royal forests by the English kings (Young, 1979); the late 19th-century imperially-ordered restoration of the forests around Rio de Janeiro, Brazil (Dean, 1995); and the Spanish general afforestation program, which replanted nearly 3 million ha of forests, mostly between 1940 and 75 during the Franco dictatorship (Valbuena-Carrión et al., 2010).

Today, restoration projects tend to be more focused, more voluntary, and more science-based, but they remain risky and often expensive undertakings. Adding to the complexity, global change is now moving the goal posts for many, if not most restoration projects. The level of human-mediated disturbance on planet earth has reached unprecedented levels, and with 7.6 billion humans adding 230,000 more people per day (World Population Clock, <http://www.worldometers.info/world-population/>), geologists have pronounced a new geological epoch, the Anthropocene (Crutzen, 2002). In the Anthropocene, no ecosystems on earth remain pristine — if they ever really were — as polluted and warming atmosphere and water impact even the most remote bastions of nature.

Designing forest management and restoration prescriptions for rapidly changing conditions are tricky propositions. In traditional ecosystem management and restoration ecology, reference targets are mostly defined as “the way things were before humans mucked them up”, and a premium is placed on divining the proper state of things (ecological integrity, proper functioning condition, etc.), often from historical ecology (Landres et al., 1999; Wiens et al., 2012). Although it can be difficult to reconstruct more than fragments of the putative reference state, a tangible target condition can be ginned up, project plans developed, and a path to success defined (Egan and Howell, 2001). Today, such concrete reference conditions are of progressively more doubtful utility, as the supposedly static environmental baselines that linked the past to the present have turned into directional trends driven by human influences (e.g., atmospheric C content, air temperature, ocean acidity, changed disturbance regimes, etc.; Safford et al., 2012a). Under such conditions, a more nuanced approach becomes necessary, where degree of ecosystem alteration, socio-economic concerns, and the availability of feasible intervention options all interact to drive management and restoration response (Hobbs et al., 2014).

Anthropogenic changes in land use and the climate are major threats to most of the earth’s forested ecosystems. Deforestation, rapidly rising temperatures, drought, fire and other ecological disturbances are reducing forest cover and the ability of forests to sequester carbon, provide habitat, and supply other ecosystem services, and modeling suggests these trends will accelerate (Dale et al., 2001; Settle et al., 2014). Over 5% of global forest cover was lost between 1990 and 2005, and current deforestation rates are nearly 9 million ha per year (Sandker et al., 2017). Coordinated human response will be required to confront the combined threats of deforestation and climate change. Key areas for management response include: synchronizing planning and management across jurisdictional boundaries; addressing numerous threats and climate drivers simultaneously; assisting climate resistance in some cases, building climate resilience or engineering realignment in others; connecting local projects to regional efforts; linking historical and current conditions to future projections; and developing innovative management solutions for novel conditions (Millar et al., 2007; Heller and Zavaleta, 2009; Safford et al., 2012a; Hobbs et al., 2014).

Here we focus on forests in two contrasting biomes — boreal and Mediterranean, both of which are anticipated to experience major ecological changes as a result of climatic warming and other anthropogenic stressors. Forests in the world’s Mediterranean climate regions (“MCRs”) have been heavily used, highly fragmented, and generally degraded from centuries to millennia of intensive land use. However logging and other extractive use of MCR forests have decreased in the last half-century, and forest cover and density have generally been on the increase in the two most forested MCRs, the Mediterranean Basin and California. Boreal forests on the other hand are lightly populated and cover huge parts of the northern hemisphere, but intensifying industrial logging over the last 50–70 years is a major threat to ecosystem sustainability, as is the magnitude of climatic change projected to occur at high latitudes.

Forests in both boreal and MCR biomes are subjected to high levels of ecological stress. In Fig. 12.1, we plot characteristic woody species for both biomes on Grime’s triangle (Grime, 2001), which ordinales plants as to their adaptations to competition (“C”), stress (“S”), and disturbance (“R” for ruderal). Strong competitors are found in the upper third of the triangle, stress tolerators in the lower right corner, and ruderal species (species able to quickly take advantage of post-

disturbance conditions) in the lower left corner. Fig. 12.1 compares the CSR positions of woody boreal and Mediterranean forest taxa with tropical broadleaf forest. Most variation is found along the C–S leg of the triangle, since few trees are ruderal species. The boreal woody flora is strongly dominated by

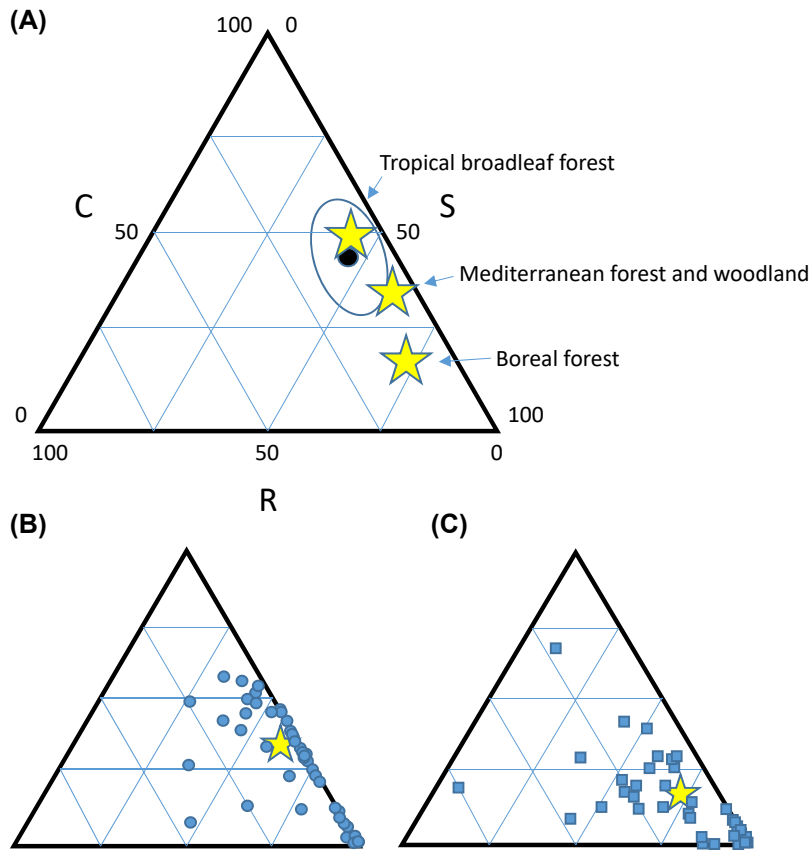


FIG. 12.1

(A) Mean Grime CSR scores for woody species of boreal forests (from [B]), Mediterranean forests and woodlands (from [C]), and tropical broadleaf forest, based on leaf trait data in [Pierce et al. \(2013, 2017\)](#), compared to mean CSR score for all 606 tree species in the [Pierce et al. \(2017\)](#) database (small black circle; blue oval surrounding black circle is boundary of third quartile for those species scores). (B) Raw CSR scores for 50 woody species from Mediterranean Basin forests and woodlands. (C) Raw CSR scores for 40 woody species from boreal forests. All scores based on leaf trait data from [Pierce et al. \(2013, 2017\)](#); yellow stars represent the mean CSR scores for the species represented. Tropical mean score in (A) derived from raw CSR score of 447 species from Neotropical and Afrotropical broadleaf forests ([Pierce et al., 2017](#)). Mediterranean Basin species (B) are from typical Mediterranean Basin lineages that are also represented in California primarily as native taxa or, in some cases, as naturalized, invasive, or ornamental taxa. Boreal species (C) are from genera that are typical boreal lineages and that are (mostly) not shared with the Mediterranean zone.

traits associated with stress toleration (72% vs. 15% associated with competition; Fig. 12.1C). The tropical tree flora is (marginally) dominated by traits associated with strong resource competitors (49% vs. 42% associated with stress toleration). The Mediterranean woody flora is intermediate, but still strongly dominated by stress tolerance-related traits (64% vs. 29% associated with competition) (Fig. 12.1B). The tropical tree flora is similar to the mean global tree flora CSR score (Fig. 12.1A), which is not surprising since most tree species are found in tropical forests. However, both the Mediterranean and boreal forest woody floras fall outside the third quartile of the global CSR scores, highlighting the unique nature of these forest types.

Although both boreal and MCR forests grow in stressful environments, the source of ecological stress in the two biomes is very different. In boreal forests, the major source of ecological stress is suboptimal air and soil temperatures during most of the year. In MCR forests and woodlands, the major source of stress is the long summer drought and the resultant lack of soil moisture during the growing season. Importantly for forest management and restoration in the two biomes, current and projected directions of change in the principle ecological stress are diametrically opposed. Boreal forests will experience an amelioration of their major source of stress – cold – as high latitude temperatures rise 3°–8° or more by the end of this century. In MCRs on the other hand, current and projected trends point to an exacerbation of the major source of ecological stress – drought – as warming and projected increases in rainfall variability reduce water availability to plants. Interactions between these climate-driven trends and other stressors – changes in precipitation type, fire and pest outbreaks, invasive species, and so on – will have important effects on soils. The implications for forest composition, structure, function, dynamics and sustainability in the two biomes are profound.

In this chapter we examine how interactions between global change, soils and disturbance are likely to affect boreal and Mediterranean climate zone forests, and what the implications of these effects may be for ecosystem management and ecological restoration. We dedicate approximately half of the chapter to each biome. We begin each section by providing short descriptions of general patterns of climate, vegetation, soils, and human history in the biome in question; we then outline its disturbance ecology; and subsequently summarize climate trends and projected future conditions, with focus on effects to soils. We finish each biome section with a summary of current restoration strategies and practices, and a consideration of how soil responses to global change-related stressors and disturbances might require changes in the way we plan for and implement forest management and restoration. Finally we summarize our main points and briefly discuss some of the salient issues confronting forest managers and restorationists in the 21st century.

Boreal forests

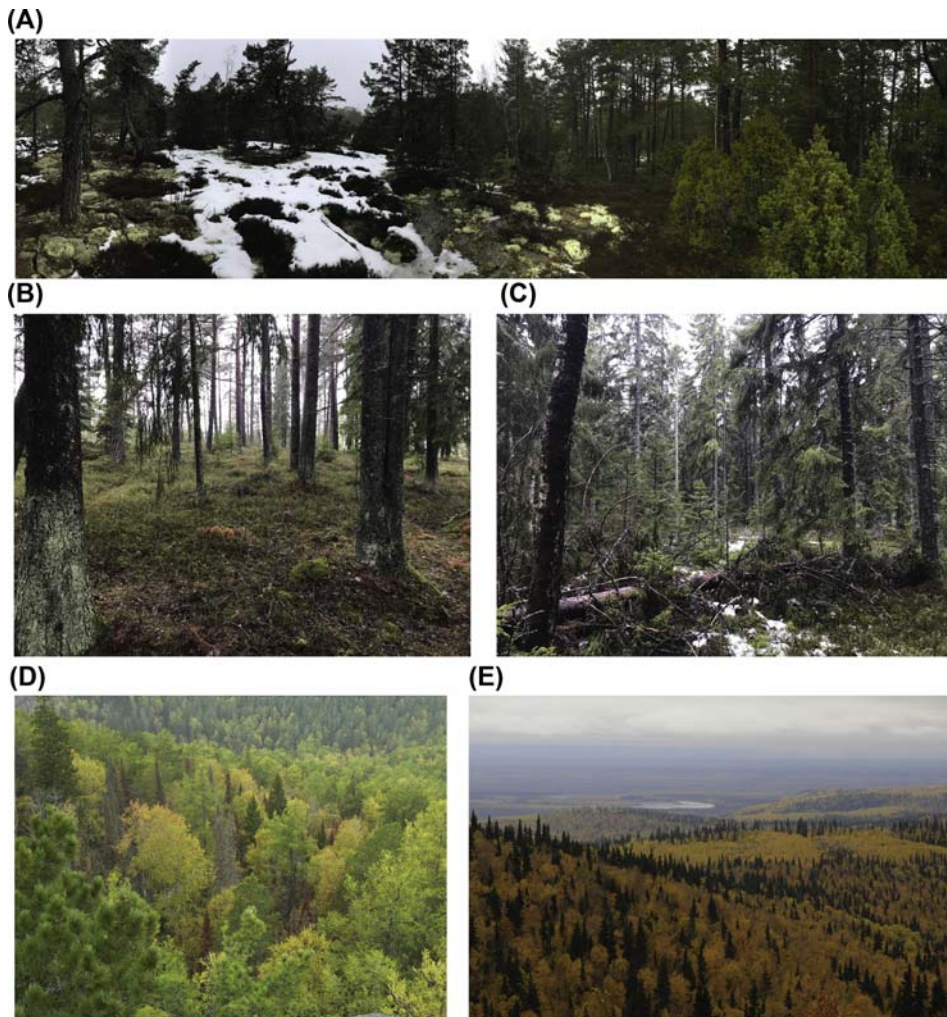
Climate, vegetation, soils, and human history

Boreal forests (also referred to as “taiga”) occupy approximately 11% of the earth’s surface and are found principally in Russia, Canada, Alaska (US) and Fennoscandia, mostly between 45° and 70° latitude. The boreal forest southern boundary approximates the 18 °C mean July isotherm, while the northern boundary approximates the 13° mean July isotherm (Bonan and Shugart,; 1989, Soja et al.

2007). Boreal forests support the bulk of the world's soil organic carbon stocks (Davidson and Janssens, 2006; Gauthier, 2015), and through their effect on high latitude albedo, they exert the greatest biogeophysical effect of all biomes on the global mean temperature (Bonan, 2008). Mean annual precipitation is often surprisingly low (<900 mm/yr in most cases and often less than half of that, especially in continental sites), but low temperatures and high cloud cover lead to low evaporative stress (Binkley and Fisher, 2012). Snow cover persists at least five months in the southern boreal forest, and seven to eight months further north (Shugart et al., 1992). Most boreal forests are found in Köppen's Dfc climate regime, where the mean temperature of the warmest month is $\geq 10^\circ$ but $< 22^\circ$, and mean of the coldest month is $\leq -3^\circ$, while precipitation is relatively evenly distributed through the year (usually with a summer maximum). Some southern boreal sites can be warmer than this, and some locations can have more seasonal precipitation.

Due to difficult growing conditions — cold temperatures, short growing season, acidic often saturated soils, permafrost — woody vegetation is simple and dominated by a few cold-hardy taxa. Typical to any region are a few species of conifer trees in the genera *Picea*, *Pinus*, *Abies*, and *Larix*, broadleaf deciduous trees in the genera *Betula*, *Populus*, *Alnus*, and *Salix*, and shrubs in the genera *Vaccinium*, *Empetrum*, and other cold-hardy genera. Species distributions are often extensive due to high habitat connectivity across large areas of subdued topography (Shugart et al., 1992). For example, the Eurasian species *Pinus sylvestris* is the most widely distributed pine in the world, and *Populus tremuloides*, quaking aspen, is the most widespread tree in North America. Forest productivity in boreal forests is usually correlated with soil temperature and depth. Soil temperature is driven by slope and aspect. Warmer soils increase biological activity and decomposition, releasing more nutrients and permitting faster and more sustained plant growth. As a result, cool (north-facing) slopes and basins that pool cold air tend to support lower biomass than warm (south-facing) slopes. River terraces and floodplains are also sites of high forest productivity, due to the general lack of permafrost and repeated disturbance and sediment deposition (Shugart et al., 1992). Soil depth can vary widely on the landscape, from thin, rocky or sandy soils supporting open woodlands of pines (and often broadleaf species in the south [Fig. 12.2A — left]), to deep, moist to saturated soils supporting high organic content and dense forests of spruce (Fig. 12.2C).

From 30%–40% of boreal forests are underlain by permafrost, and many boreal soils are water saturated for at least part of the growing season (Zimov et al., 2006; Price et al., 2013). Well-drained soils occur on higher landforms, or where local processes (windthrow, treefall, growth of *Sphagnum* mounds) raise the growing surface above the water table. Soils typically include thick O horizons, with well-developed humus layers overlain by moss and lichens. Generally speaking, boreal forest soils tend to be spodosols, histosols, gelisols, or inceptisols (Soil Survey Staff, 1999). Spodosols form under heath or forest vegetation in sandy or coarse-loamy soils, they are acidic and of low fertility. These soils form in well-drained locations or locations where the groundwater levels fluctuate seasonally. In spodosols, organic acids produced in litter decomposition lead to mineral leaching from an eluviated horizon and redeposition of clay and Al and Fe sesquioxides below in the so-called spodic horizon. Histosols are acidic, organic soils that form when fallen plant material decomposes more slowly than it accumulates. This is a common condition in $\pm$ permanently saturated soils found in bogs, fens, moors, and other peatlands. Gelisols are formed where permafrost is found near the soil surface. These soils may be permanently frozen or they may seasonally thaw. Cryoturbation and freeze-thaw cycles are important processes in gelisols. Gelisols can support cold-hardy forests (e.g., of *Picea* or *Larix*) if the soil active layer is deep enough (Soil Survey Staff, 1999; Binkley

**FIG. 12.2**

Boreal forest heterogeneity. (A) Forest variation across a soil toposequence, from a granite hillock dominated by open Scots pine (*Pinus sylvestris*), to moderately deep, more organic soils in a swale, supporting a mixture of Scots pine and Norway spruce (*Picea abies*); such sites are focus areas for thinning and prescribed fire treatments to prevent or stall stand homogenization due to spruce ingrowth; to the right of the photo the forest transitions to a dense spruce forest; Åland, Finland. (B) Traditionally managed stand of Norway spruce with scattered Scots pine near Uppsala, Sweden; both planting and thinning have taken place in this stand, note how the forest understory has been “cleaned”. (C) Structurally heterogeneous, multi-aged Norway spruce forest where fallen trees have been left for ecological purposes; Åland, Finland. Photos H. Safford. (D) Mixed boreal forest on granitic inceptisols (*Populus tremula*, *Betula* sp., *Abies sibirica*, *Picea obovata*, *Pinus sibirica*, *Larix sibirica*), fir mortality caused by *Polygraphus* beetle attack; Stolby Nature Reserve, Krasnoyarsk, Siberia.

(The last part of the 12.2 caption (the part referring to (E)) is found at the bottom of the next page.)

and Fisher, 2012). The inceptisol soil order includes young soils in which pedogenic processes are incipient or have been slowed. In boreal regions, this is often caused by periodic or long-term flooding.

Humans have only been major players in the boreal zone since the end of the last ice age. Human settlement of Fennoscandia and northwestern most Russia began as glacial ice retreated during the Early Holocene, and occurred as boreal plant taxa migrated west and north to reoccupy land lost to glacial advance tens of thousands of years earlier; use and clearing of the forest became more intensive as metallurgy and farming were developed (Blankholm et al., 2017). Most of north-central and north-eastern Russia, on the other hand, escaped glaciation and human interaction with the boreal forest has a much longer history there. In North America, humans arrived from northeastern Asia along the shores of the Bering Strait and Arctic Ocean about 15,000 years ago, and migrated inland as soon as glacial recession permitted (Goebel et al., 2008); the earliest records of humans on the Canadian east coast are from about 10,000 years ago. Today, the world's boreal regions are among the least-densely populated on earth, with densities ranging from 0.5 people per km² (Alaska) to 20/km² (Sweden) and huge swaths of forest remain. The major modern human disturbance to boreal forest is in the form of large-scale industrial logging.

Disturbance

Fire is the principal natural disturbance in boreal forests (Angelstam, 1998; Gromtsev, 2002; Nilsson and Wardle, 2005; Price et al., 2013). Fire frequency and behavior in the boreal zone are driven by interactions between climate, forest type and local soil moisture. In landscapes dominated by pines, multiple studies have documented natural fire rotations (Natural fire rotation (NFR) is the number of years necessary to burn an area equal to the area of study. Also called “burn cycle” in some literature. NFR is a spatial extension of fire return interval (FRI) but is not equivalent to FRI) between 70 and 120 years (Zackrisson, 1977; Heinselman, 1981; Lehtonen and Kolström, 2000; Harvey et al., 2002; Bergeron et al., 2004). Fire resistant species, such as Scots pine (*P. sylvestris*) in the Eurasian boreal and red pine (*P. resinosa*) in eastern North America, are found in drier sites with minimal soil-surface organics, and fires are mostly low to moderate severity; in Russia and Fennoscandia, most boreal fires occur in this forest type (Korovin, 1996; Angelstam, 1998). Some pines (e.g., *Pinus contorta*, *P. banksiana* in North America) are adapted to high severity fire and carry their seeds in serotinous cones that are opened by heat. In the generally more extensive and more mesic landscapes dominated by firs (*Abies* spp.) and spruces (*Picea* spp.), natural fire rotations are much longer, from 200–300 years or more, and fires are more severe, often killing trees over large areas of forest (Bergeron, 1991; Wallenius, 2002); these types of forest dominate most of the boreal landscape in North America. That said, even severe fires with large areas of canopy mortality leave many areas unburned or burned lightly within fire perimeters, due to heterogeneity in vegetation, site moisture, and weather (Angelstam, 1998; Gromtsev, 2002).

◀ (E) Mixed boreal forest (*Betula neoalaskana*, *Populus tremuloides*, *Picea glauca*, *Populus balsamifera*) west of Fairbanks, Alaska; the high density of deciduous broadleaf trees on the landscape is due to a 35-year old fire, the forest will succeed to a spruce forest in the absence of further disturbance.

Photo (D) V.R. Vallejo. (G) Hayward.

Humans play an important role in the modern boreal fire regime, but the scale of the human role is markedly different in North America and Russia. Between 2001 and 2007, the boreal forest in central Siberia experienced >3 times more burned area and >16 times more fires than Canada on an equal area basis (de Groot et al., 2013a). In Russia, 86% of fires were human caused, while 80% of fires in Canada were ignited by lightning. The fire rotation in Canada between 1970 and 2009 was 167 years — not very different from the natural rotation — but in Russia it was only 53 years during the eight years for which data were available. The natural (historical) fire season in the two regions is probably very similar (June and July), but human ignitions in the early spring have moved the peak fire season in Russia to April and May, when there are few lightning strikes but dead understory fuels are abundant and green-up and leaf-out have not yet occurred (de Groot et al., 2013a). In general it can be said that outside Russia, fires in northern and moister boreal sites dominated by spruces, larches, and firs support modern fire regimes that are broadly similar to their pre-settlement regimes, with some reductions in fire frequency but similar patterns of severity, while drier and southern sites supporting pines have generally experienced reduced fire frequencies due to fire suppression policies (Zackrisson, 1977; Bergeron et al., 2004). In recent decades, warming temperatures have led to some enormous fires in Russia, North America and northern China, and these fire regime generalizations and underlying assumptions will need to be updated if the large fire trend continues.

Fire effects on soil depend primarily on fire intensity and duration, proximity of fuels to the soil surface, as well as soil texture and moisture content. In boreal forests, soils are generally covered with a more or less deep organic layer, which insulates the mineral soil and greatly ameliorates effects of soil heating, which even under severe burning rarely penetrate more than a few cm into mineral soil. Long-term smoldering of surface and soil organics leads to the highest levels of soil heating (Wohlgemuth et al., 2018). Heterogeneity in soil moisture and soil organic matter and their interactions with fire are major drivers of fire effects, postfire ecosystem response, soil erosion, nutrient cycling, and ecosystem patterns on the boreal landscape. Postfire soil erosion in boreal forest is often minimal due to incomplete burning of organic materials. Seed germination and survival are higher for most species when the organic layer has been mostly burned off (Johnstone and Chapin, 2006). The ability to resprout, the depth of plant root systems, and the depth of seed placement in the soil all play roles in determining the postfire vegetation (Schimmel and Granström, 1996). Resistant species are those that have roots or rhizomes that extend >5 cm into mineral soil, susceptible species are those whose meristems or propagules are located completely within the organic layer (McLean, 1969). Deep rooted species that can resprout (e.g., shrubs like *Vaccinium*, trees like aspen and birch) are well-positioned to dominate postfire landscapes. In the zone of continuous/permanent permafrost, surface fires often result in the death of forest stands because of the restriction of roots to the upper soil (Masayagina et al., 2015). Boreal fire effects on soil chemistry also depend on heterogeneity in soil organics, soil moisture, and fire intensity (among other things). Soil C and N are lost to burning (although there is often transient increases in the concentration of N at the mineral soil surface), while P and cations like Ca, Mg, and K may increase immediately postfire but decrease relatively rapidly due to ash dispersal by wind or rain. Soil pH rises with the addition of soil cations, which can increase nutrient availability in acidic soils (Harden et al., 2003; Neff et al., 2005; Wohlgemuth et al., 2018). Fire can also influence soil nutrient status by differentially affecting mycorrhizal fungi and influencing patterns of microbial succession (Treseder et al., 2004).

Insect outbreaks, wind, and flooding represent other widespread and ecologically important disturbances in boreal forest. All of these disturbances, especially insect outbreaks and wind, interact with

fire dynamics. For example, augmentation of fuels following insect outbreaks and windstorms may increase the extent and intensity of subsequent fire, whereas fire may weaken live trees and predispose them to subsequent attack by insects (McCullough et al., 1998). Insects with major outbreak dynamics that can affect large areas of forest include species of budworms, tent caterpillars, sawflies, and pine beetles (Neuvonen et al., 1999; Volney and Fleming, 2000). Volney and Fleming (2000) note that under fire suppression, tree mortality due to insects is currently greater than that due to fire in Canada, and climate warming will likely increase frequency and severity of outbreaks, especially at the edge of host ranges. Windthrow can also affect large areas of forest, in some areas on similar time scales to fire (Gromtsev, 2002; Rich et al., 2007). Flooding is a more localized disturbance, and results from heavy precipitation years as well as permafrost melting and thermokarst development. Flooded forest will often die and may convert over time to peatland (paludification) (Price et al., 2013).

Invasive species have historically not been a major problem in the boreal zone, but this is changing with climate warming and human economic globalization. Soils in the southern boreal zone of North America do not support native earthworms, but multiple introduced Eurasian earthworm taxa have recently invaded the region and precipitated ecosystem changes, including decreased soil microbial biomass, soil respiration, and soil moisture, incorporation of organic matter into deeper soil horizons, and lower herbaceous species diversity (Holdsworth et al., 2007; Eisenhauer et al., 2011). The emerald ash borer (*Agilus planipennis*) is another recent invader, and is devastating ash (*Fraxinus*) populations of multiple species in temperate and southern boreal forests of North America (Hermes and McCullough, 2014). In Sweden, North American lodgepole pine (*P. contorta*) has been introduced as a commercial tree on almost 600,000 ha. Although the species has not yet spread significantly into surrounding wildlands, lodgepole pine is considered to be a highly invasive pine species and there are major concerns about its potential to alter Fennoscandian native forests (Engelmark et al., 2001).

Although human population densities are low at high latitudes, much boreal forest has experienced some level of human management, especially in Eurasia. Most tree cutting in boreal forests before the 20th century was selective, with preference for pine species. Use of fire for forest clearing and improving agricultural ground was widespread until the early 20th century. A major transition in logging methods occurred in the early to mid-1900s, due to a variety of economic and timber-supply factors (Lundmark et al., 2013). Today, timber harvest in boreal forests is industrial in scale and based primarily on the principle of even-aged management, i.e., clear-cutting or group-selection where all or nearly all trees are harvested at one time and the desired species mix is promoted by planting; harvest is typically carried out on a rotational basis. Until recently the concept of sustained yield dominated the industry, where the prescribed rotation was the oldest desired age of a stand and commercial stands reaching that age were preferentially cut. However, the widespread application of sustained yield harvest led to extensive loss of old trees and forest structural heterogeneity and negative impacts to species, habitats, and ecological function (Berg et al., 1994; Ericsson et al., 2000; Cyr et al., 2009). In response, in Fennoscandia and parts of Canada managers and researchers have worked together to modify timber harvest practices to better emulate patterns of natural disturbance and increase ecosystem heterogeneity. According to Gauthier et al. (2015), on the global scale about 40% of the modern boreal forest has been cut at least once. Today, large areas of boreal forest are subject to industrial tree harvest, including up to 90% of the forest in Fennoscandia, and perhaps 40% and 60% of Canadian and Russian forests, respectively (Gauthier et al., 2015).

Climate change impacts

Global increases in air temperature have been — and are projected to continue to be — most marked at high latitudes. Studies in the boreal zone have documented air temperature changes over the last 6–10 decades on the order of $+0.5^{\circ}$ to $+3^{\circ}$ (e.g. Price et al., 2013). Air temperature projections for the period ending in 2035 suggest further increases of 1.5° – 2° in winter and 1° – 1.5° in summer (RCPs 4.5 and 6.0; Kirtman et al., 2013); longer-term projections to 2100 suggest that increases in mean annual temperature are likely to range from 3 to 8° , depending on latitude (Collins et al., 2013). Precipitation has been gradually increasing across much of the boreal zone over the last century, and is projected to increase further by 2035 (increases of 5%–15% in most areas) and 2100 (+20%–40% in most areas) (Collins et al., 2013; Kirtman et al., 2013). The decades-long increase in the rain to snow proportion is also projected to accelerate, and the length of the snow-free period will also continue to increase. Chapin et al. (2005) noted that the date of snowmelt is trending earlier by 3–9 days per decade in interior Alaska, and projections suggest that the duration of consistent snowpack in northern Sweden will shorten by 7–13 weeks by 2100 (Mellander et al., 2007).

Using a dynamic global vegetation model, Gonzalez et al. (2010) found that boreal forest was among the most vulnerable biomes to global warming and future fire activity. Regional climate and vegetation models project a northward expansion of forests into tundra, and a shift in forest composition toward plant functional types from more southerly latitudes. For example, evergreen conifers at the southern edge of their distributions are projected to be at least partially replaced by deciduous broadleaf species (ash, oak [*Quercus* spp.], maple [*Acer* spp.], etc.), and deciduous conifers (*Larix*, larch) are projected to be replaced by more shade- and moisture-tolerant evergreen conifers (e.g., spruce, Siberian pine [*Pinus sibirica*]) moving northward (Kharuk et al., 2009; Ravenscroft et al. 2010; Pearson et al., 2013; Settele et al., 2014). Fire will clearly play an important role in any transformations that occur, as will interactions with herbivores, diseases, and other disturbances (Dale et al., 2001; de Groot, 2013b).

Increases in air temperature and interactions with changing precipitation will have major effects on boreal forest soils, especially in warmer locations. The interactions between air temperature, precipitation, snow cover, and permafrost are key to understanding the impacts of climate change on the boreal soil resource and the vegetation growing in it. Snow insulates boreal soils, and high variations in air temperature are greatly attenuated in snow-covered soil (Sturm et al., 1997). Projected warming in southern and maritime boreal regions will result in average monthly temperatures remaining above 0° for most of the year, which will dramatically reduce snow cover duration and have major effects on hydrology, permafrost, soil frost patterns, soil productivity, and vegetation composition and structure with many more subtle and downstream effects (Price et al., 2013; Jungqvist et al., 2014). In these warmer sites, where snow duration and thickness are greatly reduced and freezing air temperatures are ephemeral, mean soil temperatures will rise. Increased soil temperatures in the (expanding) growing season will increase the volume of thawed soil, and escalate soil respiration, the decomposition of soil organic matter, and C release to the atmosphere (Henry, 2008). Increasing precipitation, especially as rain, will amplify the reduction in depth and duration of snowpack. Together, increasing temperature and moisture will increase soil respiration, N mineralization, and C loss in cases where there is at least some soil drainage. Waterlogged, anoxic soils provide an exception to this rule (Goulden et al., 1998), and climate change-driven flooding of soils could result in local decreases in all of these processes (see below; Davidson and Janssen, 2006).

It has been shown that reduction of winter snow pack can increase seasonal soil freezing, and it has been theorized that climate warming-induced snowpack reductions in the boreal zone could similarly decrease average soil temperatures (e.g., [Groffman et al., 2001](#)). However, such an effect is only possible where air temperature remains at or below 0° (i.e., between late fall and early spring, and/or at very northerly sites) and as [Henry \(2008\)](#) demonstrated with data from Canada, warming is happening at such a rate that the time period during which even these sites can experience this effect is rapidly diminishing. It is well-established however that warming and associated snowpack loss can lead to greater diurnal fluctuations in soil temperatures, and when such fluctuations cross the freezing line, soil freeze-thaw activity can increase. The physical action of repeated ice formation and melting has important effects on soils, including destruction of soil aggregates, increased fine root mortality (which can increase loss of N and P ([Fitzhugh et al., 2001](#))), alterations to hydrological processes, and changes to SOM decomposition ([Jungqvist et al., 2014](#)).

The boundary between continuous and discontinuous permafrost (PF) is found approximately at the mean annual air temperature isotherm of -6° to -8° ([Price et al., 2013](#)). Current climate change velocity in most of the boreal zone is between 50 and 100 km per decade ([Burrows et al., 2011](#)), so the transition zone is moving rapidly northward, and central and southern boreal sites are increasingly found in the sporadic discontinuous PF zone; recent measurements from Canada, Alaska, and Fennoscandia indicate that current PF warming rates are up to 0.7° per decade and rising over time ([Isaksen et al., 2007](#); [Price et al., 2013](#)). PF loss in the boreal zone will have major ecosystem consequences. In areas of low topographic relief underlain by PF, forest tends to occur on upland “plateaus” (these may be as low as a half-meter high) created by centuries of *Sphagnum* growth or other processes that can raise the ground surface above surrounding waterlogged peatlands. Such plateaus collapse as PF melts, creating a so-called “thermokarst” landscape and fragmenting and reducing forest cover ([Baltzer et al., 2014](#)). Overall, poorly drained areas will become even more waterlogged, and many currently well-drained soils will become at least seasonally saturated.

In areas where topography is more complex and high ground is due to more permanent features of the landscape like geological landforms, climate warming, increased precipitation and CO₂ will probably increase soil and ecosystem productivity (although much warmer temperatures are apparently leading to photosynthetic down-regulation in cold-adapted conifers ([Settele et al., 2014](#))). In well-drained locations, soils could become seasonally drier even under moderate precipitation increases, leading to major changes in soil microbial communities and processes, as well as major changes in forest and understory composition ([Högberg et al., 2007](#); [Allison and Treseder, 2008](#); [Price et al., 2013](#)).

Boreal forests in both North America and Eurasia have experienced abnormally large fires in the last few decades, and climate and ignition conditions leading to these extreme events are projected to become more prevalent ([de Groot et al., 2013b](#); [Veraverbeke et al., 2017](#)). Modeling points to large potential increases in annual burned area, fire frequency, and possibly fire severity. [Weber and Flannigan \(1997\)](#) suggested that the rate and magnitude of fire-induced changes to the boreal forest are likely to greatly exceed anything expected due to atmospheric warming alone. Warming climates will allow higher survival of insect larvae during the winter, likely increasing the prevalence and magnitude of insect outbreaks and large scale forest mortality ([Neuvonen et al., 1999](#); [Volney and Fleming, 2000](#)). Flooding disturbance is also expected to increase in prevalence, as melting permafrost leads to development of thermokarst, soil subsidence and saturation.

BOX 12.1 The importance of soil and landscape heterogeneity

Although boreal and Mediterranean forests are different in many ways, management and restoration strategies in both regions have begun to focus on ecological heterogeneity as a way to increase ecosystem resilience and sustain native biodiversity. For terrestrial ecosystems, the most basic form of heterogeneity is in the soil. Soils are naturally variable at all spatial scales, from soil micro-structure to the landscape. This variability plays a major role in driving habitat diversity, from the soil biota, to vegetation structure and composition at local and regional scales, to landscape-scale patterns. Diversity of soil types, structures, depths, and productivities is driven by diversity in the soil-forming factors (climate, biota, topography, geologic substrate, time; [Jenny, 1946](#)), and ecological theory suggests that areas of high spatial environmental heterogeneity should support high biodiversity ([Tilman 1982](#); [Huston 1994](#)) and high biodiversity should feedback to increase habitat diversity ([Rosenzweig 1995](#)).

Landscape heterogeneity is important in other ways as well. Heterogeneous forest landscapes including a mix of species, structural classes, and patch types are much less likely to succumb to large, synchronous disturbances that homogenize ecological conditions and reset successional clocks at the landscape scale. For example, bark beetle outbreaks are abetted by conditions of low tree carbon balance, which depends to a great extent on the distribution of soil nutrients and water as well as competition for both, which is much more intense in homogeneous landscapes dominated by dense forest stands of older trees ([Christiansen et al., 1987](#)). Spatial heterogeneity in forest density and tree size also reduces fire severity and extent by breaking up crown continuity and creating highly variable surface fuel loads. Because of their resilience, heterogeneous forests can continue to provide important ecosystem services even after some areas or some dominant species have been disturbed or extirpated ([Turner et al., 2013](#)).

In the MCRs, modern forests are primarily found in areas of rugged topography, due to expropriation of more level and more fertile forestland for agriculture, pasture, or urban development, but also due to the beneficial soil and moisture conditions provided by mountainous landforms during the dry warm season. Soils in mountain areas are particularly heterogeneous in terms of soil depth, stoniness, and water holding capacity, all of which play major roles in driving vegetation patterns. In California, selective logging and a century of fire suppression have greatly homogenized montane forests, masking the natural pattern of forest heterogeneity as driven by soils and disturbance and greatly increasing the risk of large, high severity fires (in the Mediterranean Basin, rural land abandonment has resulted in the same trend). In response, forest management and restoration on National Forest lands are largely focused on reducing fuels and forest density and increasing heterogeneity in forest structure and composition, using landform and microtopography (essentially surrogates for soil heterogeneity) as templates for types and intensities of treatments ([Fig. 12.3](#); [North et al., 2009](#)).

In boreal regions forests are far more widespread than in the MCRs due to generally more benign water relations and shorter and less intensive history of land use, but the roles of soil and disturbance in driving forest heterogeneity are still important. In the last 50–70 years, industrial sustained-yield forestry has greatly homogenized large areas of boreal forest, leading to a new management focus on restoring forest heterogeneity ([Kuuluvainen, 2002](#)). As in the MCR example above, this work is largely based on spatial patterns in soils. In Fennoscandia, areas of thin, rocky or sandy soils are focus areas for prescribed fire, thinning of shade-tolerant conifers, and planting of broadleaf species; in areas of deeper soils dominated by spruce, dead trees are often left on site and the forest floor is left uncleaned ([Fig. 12.2C](#)); in wet depressions, drainage ditches are filled in order to restore peat-forming processes and the herbaceous flora common to undisturbed bogs and mires ([Larsson and Danell, 2001](#)).

In restoration, the identification of soil “resource islands” is important to ensure success in highly heterogeneous habitats ([Vallejo et al. 2006](#)). Resources in locations subject to high levels of stress or repeated and/or uncharacteristically severe perturbations are often highly clumped in space. Spatial gradients in resources and soil conditions are correlated with gradients in species recruitment and growth ([North et al. 2006](#); [Johnstone and Chapin, 2006](#)). Ideally the gradients are at coarse scales and easy to recognize in the field, but in some systems soil variation occurs at very fine scales and restoration success is associated with apparently minor differences in soil moisture, soil depth, stoniness, or texture ([Maestre et al. 2003](#)). Spatial variation in abiotic conditions can

Continued

BOX 12.1 The importance of soil and landscape heterogeneity—cont'd

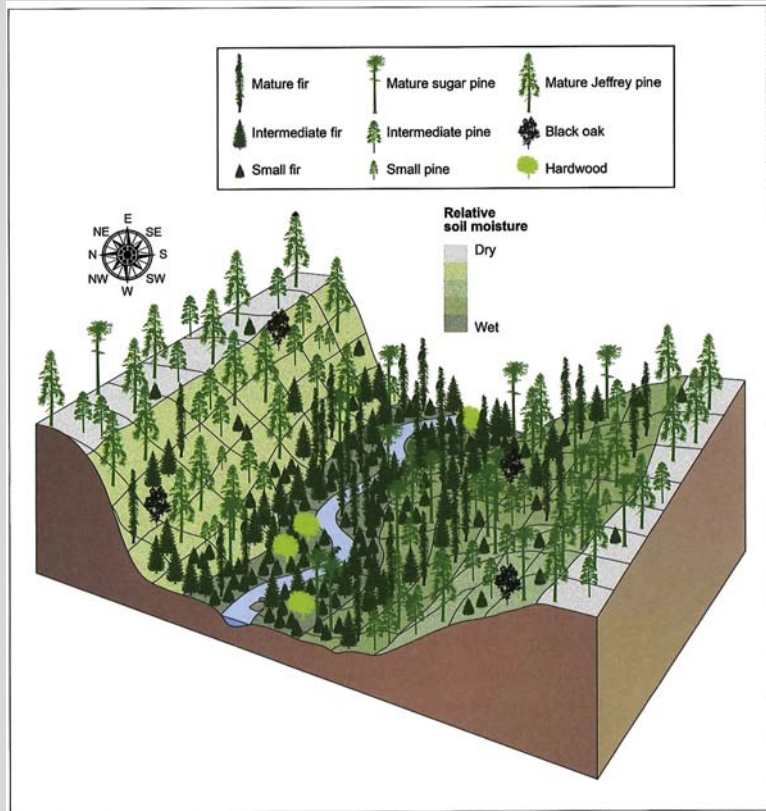


FIG. 12.3

Idealized schematic of heterogeneous forest conditions produced by management or restoration treatments that vary by topographic factors and soil moisture, which both influence fire severity. Driest locations (ridgetops) are managed for low-density pines (fire and drought tolerant), riparian areas and deeper soils can support higher density shade-tolerant/fire-intolerant species like fir (*Abies*) and riparian broadleaf species (*Acer*, *Populus*, *Alnus*, *Salix*). Midslope density and composition vary by aspect, with density and fir component higher on cooler aspects and flatter slopes; oaks (*Quercus*) are important components of the slope forest, as is *Arbutus* on cool slopes. From ecosystem management strategic document for Sierra Nevada, California.

Figure with permission from North et al., 2009.

also shift the relationship between species from competition to facilitation or vice versa (Maestre and Cortina 2004). Heterogeneity in soil resources and conditions varies through time, with especially strong gradients after disturbances like fire, which greatly alter resource availability and promote certain species groups in the soil and above it (Fig. 12.4; Hart and Chen 2006).

BOX 12.1 The importance of soil and landscape heterogeneity–cont'd



FIG. 12.4

Changes over 25 months in soil surface characteristics in the same location following an experimental heathland fire. Gestosa, Portugal.

Photos: V.R. Vallejo.

Boreal forest management and restoration under global change

Concerns about boreal forest ecosystem sustainability developed from studies that demonstrated major biodiversity and ecosystem impacts of large-scale sustained yield timber harvesting (Berg et al., 1994; Larsson and Danell, 2001). Studies conducted in the 1970s and 1980s documented the natural fire regime of boreal forests and highlighted the tremendous structural differences between logged and unlogged landscapes (especially the lack of old growth forest in the latter; Zackrisson, 1977; Heinselman, 1981, Van Wagner, 1983). Since the 1990s, concepts based on or related to the Natural Range of Variation (NRV) and Natural Disturbance Emulation (NDE) (Hunter, 1993; Angelstam, 1998; Landres et al., 1999; Kuuluvainen and Grenfell, 2012) have led to marked changes in timber harvest systems and policy in the boreal zone, especially in Fennoscandia and Canada (Titler et al., 2001; but see Cyr et al., 2009). The basic idea is to balance socioeconomics and ecology to provide a reasonable, but affordable, emulation of the spatial and temporal patterns of natural forest disturbance. For example, cutting rotations have been lengthened, retention of live trees promoted, some uneven-aged management practices adopted, and more focus has been given to the importance of dead wood and forest floor residues (Fig. 12.2C), as well as to the ecological

importance of fire and other disturbances (Liski, 2001; CBFA, 2010; Pukkala et al., 2012; Kaukonen et al., 2018).

In Fennoscandia, extensive commercial forestry has greatly reduced the area of “natural” forest. Managed forests are mostly mosaics of single-aged stands, include few or no trees older than the harvest rotation age, and generally lack forest gaps, standing dead trees and coarse woody debris on the forest floor (Fig. 12.2B); in addition, areas of permanently or seasonally flooded soils have often been drained (Kuuluvainen, 2002). The lack of large, relatively pristine forest blocks (only about 3% of Finland and Sweden are protected in conservation units), the overwhelmingly private ownership of forestlands, and the ubiquity of timber harvest mostly obviates the widespread implementation of ecosystem management. Instead, based on NRV/NDE concepts, restoration projects focus on magnifying heterogeneity in the forest and in the forest soil at the local scale (Box 12.1), dozens to hundreds of hectares at a time. A major focus is the restoration of open habitats that have been lost due to fire suppression and commercial forestry (e.g., the Finnish Light and Fire project [<http://www.metsa.fi/web/en/lightandfirelife>], Sweden’s Taiga project [<http://www.lifetaiga.se/>]). Tactics include felling, girdling, and exploding trees; carrying out prescribed burns; using forest thinning to create gaps and favor broadleaf species; and damming and filling drainage ditches (Larsson and Danell, 2001; Kaukonen et al., 2018).

In Alaska, most boreal forest is under federal management, much of it in wilderness areas or otherwise protected; 60% of the state is in federal ownership, with half of that in strictly protected lands. With so much pristine or near-pristine forest, little restoration work is carried out (other than local wildlife habitat improvement, for example), but general principles of ecosystem management guide broad-scale considerations about how forest habitats can be made more resilient to climate change. Naturally ignited fires are permitted to burn in most Alaska national parks. Major concerns are the impacts of permafrost melting on soils, vegetation, and infrastructure, as well as climate change threats to wildlife habitat and traditional hunting and fishing grounds (B. Schulz, US Forest Service Pacific Northwest Research Station, pers. comm.). In Alaska, much attention is also being paid to ensuring connectivity between relatively pristine federal management units across the working lands that separate them, so as to reduce genetic isolation and to permit migration in response to climate change (Magness et al.; 2018).

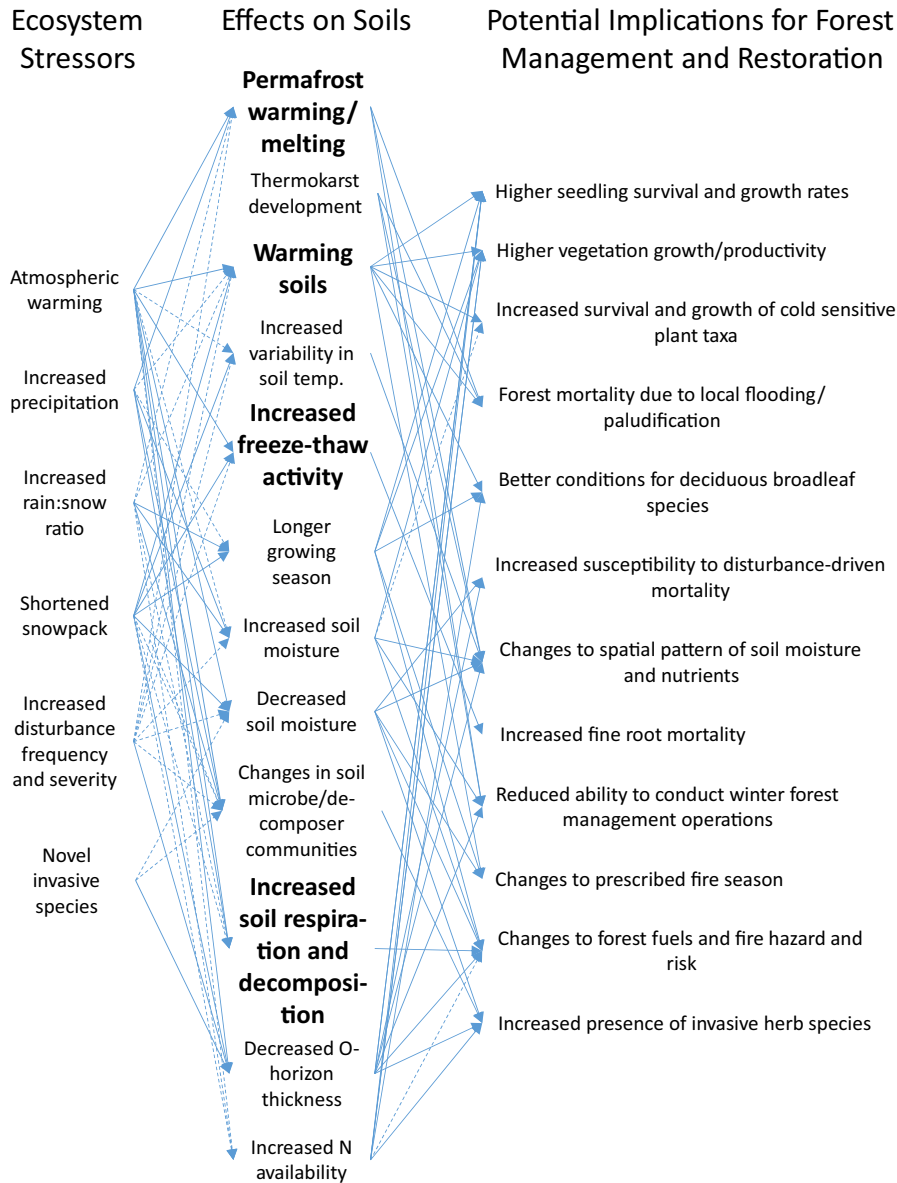
In Canada, more than 90% of boreal forest is state-owned (“crown land”), but little of this is in conservation units (about 6% of Canada is protected in national or provincial parks). Most of the southern boreal forest is leased to logging companies, and forest management policies governing logging practices are developed and implemented at the province level (Tittler et al., 2001). Conflict created by boreal forest degradation and variance in regulations and practices across the country led in 2010 to the Canadian Boreal Forest Agreement (CBFA), a pact between major environmental groups and the Forest Products Association of Canada. The agreement applies to over 70 million hectares of forestland and sets goals related to protected areas, sensitive species conservation, greenhouse gas mitigation, economics, and sustainable forest management practices (CBFA, 2010). The CBFA commits forest management signatories to conduct their business under the guidance of NRV-based targets in forest composition and structure that best emulate natural disturbances in their patterns and ecological effects. Within mapped ecological units, targets are defined for seral stage composition, tree retention, and patch size. Spatial and temporal variation is ensured by targeting a range of conditions that fall between the 25th and 75th percentiles of the NRV-defined range (CBFA, 2015).

Boreal restoration is also happening in Canada, but at a much smaller scale. A good example of the complexity of ecological restoration under global change is the “Back to the Boreal” initiative in Cape Breton Highlands National Park in Nova Scotia (<https://www.pc.gc.ca/en/pn-np/ns/cbreton/decouvrir-discover/conservation/foret-forest>). A spruce budworm (*Choristoneura* spp.) outbreak between the mid-1970s and mid-1980s killed more than 90% of the balsam fir (*Abies balsamifera*) in parts of the park. Such levels of mortality are not unheard of, but normal successional pathways through a birch (*Betula*)-dominated stage back to fir and spruce were halted by the presence of moose, which had been re-introduced to Nova Scotia without its chief predator, wolves, in the 1940s. Moose browsing of the highly palatable and nutritious young birch and fir led to a population explosion, and moose numbers reached densities that were 2–10 times higher than “healthy” mainland levels (Smith et al., 2010). The arrested development of forest permitted tall rhizomatous grass and ferns to expand through the park, which strongly suppressed tree seedling recruitment. Warmer summers have combined with browsing and grass competition to kill many of the stunted trees. The restoration initiative has been multi-pronged: installing large moose exclosures, planting conifer seedlings with public help, removing grass, and culling the moose herd in collaboration with local indigenous peoples.

Boreal ecosystems are generally thought to be relatively resilient to the direct effects of climatic change. This is partly because the biome has been subjected to repeated periods of glaciation and interglacial warming and ice retreat throughout the late Cenozoic. As a result, most dominant species in the boreal forest are wind-pollinated, and broadleaf species have wind-dispersed seeds. Over the Holocene, climatic perturbations less dramatic than continental-scale glacial advance — e.g., the Medieval Warm Period and the Little Ice Age — appear to have resulted in little or no floristic change in boreal forests (Chapin et al., 2010). Because of the strong climatic filters over time, the boreal flora is species poor and the dominant species have huge ranges and relatively high genetic diversity. In addition, a number of the dominant tree species are at least partly serotinous (e.g., black spruce, jack pine, lodgepole pine, larch) and can quickly recolonize severely burned areas (Price et al., 2013); dominant deciduous broadleaf trees have easily dispersed seeds and resprout prolifically after fire.

Major global change-related stressors and their implications in boreal forests

Nonetheless, global change stressors will have major effects on boreal forests, largely through their effects on soils and disturbance regimes. In Fig. 12.4 we conceptualize some of the key connections between global change stressors, soils, and forest management and restoration practices in the boreal forest. The diagram is highly simplified and lacks feedbacks as well as interactions among the factors in each column, and also ignores the direct, non-soil mediated impacts of global change on management and restoration, some of which are very important (e.g., atmospheric warming, changes in precipitation, increased forest disturbance and invasive species will directly affect forest management and restoration activities in many ways; the literature is full of examples). Nonetheless, the spider web of arrows underlines the multivariate, interactive and complex nature of the relationship among factors. Important right-to-left feedbacks in Fig. 12.5 include the impacts of increased deciduous broadleaf litter and increased fine root mortality (in areas of increased freeze-thaw activity) on soil respiration, decomposition, the O-horizon, and N availability; the impact of increasing fire and live and dead biomass removal — as well as forest mortality due to other disturbances like soil flooding or insect outbreaks — on permafrost melting, thermokarst development, soil temperature and moisture, the decomposer communities, respiration, and nutrient cycling. There are also nearly innumerable

**FIG. 12.5**

Major global change-related stressors on the boreal forest, their salient effects on soils, and implications — through soil pathways — for forest management and restoration practices. Most important soils effects are in **bold**. Dotted lines are indirect effects. Stressors, effects, and management implications are not comprehensive. Direct impacts of global change stressors on management and restoration are not depicted, nor are feedbacks from management and restoration practices on soils or stressors, or interactions between factors within categories (but see discussion in text).

feedbacks among the management and restoration implications. Perhaps most important among these are the various ways in which changing vegetation and mortality patterns will alter forest fuels and fire risk.

Overall, boreal forest managers and restorationists will mostly see positive survival and growth effects in their reforestation/revegetation projects (Fig. 12.5). Seed stock, genotypes and species from warmer, more southern latitudes will be especially favored under the high magnitude of warming and increasing precipitation that are projected for most of the boreal. For example, the climate space for deciduous broadleaf species from temperate forests is expanding rapidly into the southern and central boreal. It will be more difficult to conduct winter over-snow harvest where tree cutting and thinning is employed, which will impact how, when, and where large trees can be removed. Drier organic layers will be more combustible, and upland locations will often dry out sufficiently in the warm season to greatly increase the probability of fire. Fire management and fire use (natural as well as prescribed) will need to become a more integral part of resource management and restoration in the future; the period in which prescribed fire is used will shift to earlier in the spring and later in the summer. Permafrost melting in the central and northern boreal will greatly complicate forest management and restoration (Fig. 12.5). Many current areas of forest on biogenic high ground will collapse and fill with water, and current spatial patterns of soil moisture and nutrients will re-organize. The roles of large-scale, synchronous disturbances (e.g., fires, pest outbreaks) and invasive species will greatly increase over the next half-century. Heterogeneous forest landscapes — mixtures of open and closed habitats, mixtures of evergreen and deciduous species, etc. — will be more resilient to large, synchronous disturbances, and more likely to sustain high levels of biological diversity.

Mediterranean climate-zone forests and woodlands

Climate, vegetation, soils, and human history

The world's Mediterranean Climate Regions ("MCRs") cover only about 2% of the earth's surface, and forests and woodlands contribute only a fraction of this area (e.g., c. 30% of California, 9% of the Mediterranean Basin [but rising in the northern Mediterranean, see below], less in the other MCRs). MCRs are found in five disjunct parts of the world — the Mediterranean Basin ("MB"), California, coastal central Chile, southwestern Australia, and southwestern South Africa — at temperate latitudes between about 30° and 45°, generally where cold ocean currents wash the west coasts of the continents. Climatically, the world's Mediterranean-climate regions are unique because the wet season is concurrent with the cold season, and the warm, dry season is akin to an annual drought of 3–6 months. This pattern leads to the growing season being mostly out-of-phase with the wet season, and plants either must grow in locations with reliable surface or ground water supply, or develop adaptations to survive long and warm dry seasons. Most MCR ecosystems fall in the Csa or Csb Köppen climate zones, where: (1) the mean temperature of the warmest month is $> 10^{\circ}\text{C}$, and the mean temperature of the coldest month is $< 18^{\circ}$ but $> -3^{\circ}\text{C}$; (2) precipitation in the driest month of the summer is $< 30\text{ mm}$ and less than one-third of the wettest winter month; and (3) in the case of Csa, the mean temperature of the warmest month is $> 22^{\circ}$, or in the case of Csb, the mean temperature of the warmest month is $< 22^{\circ}$. In California, Chile, and the Mediterranean Basin, there are also montane Mediterranean systems, where the mean temperature of the coldest month falls below -3°C .

Although the MCRs are widely separated, their vegetation is very similar, providing a classic example of convergent evolution (Di Castri and Mooney, 2012). Widespread vegetation types include sclerophyllous (hard-leaved, with a high weight-to-area ratio) evergreen shrublands with widespread adaptations to intense fire (*maquis/macchia* in the MB, *chaparral* in California, *fynbos* in South Africa, *mallee* in Australia, and *matorral* in Chile [but here lacking the strong fire association]); lower, more xeric, sometimes drought-deciduous shrublands called, e.g., *garrique* in the MB and *coastal sage scrub* in California; woodlands with drought-adapted deciduous and evergreen species (oaks, pines, cypresses in the MB and California, acacias in Chile and South Africa, eucalyptus and acacias in Australia); and forests in moister, often higher locations, such as the pine and fir dominated forests of California and the MB, *Nothofagus* forests in Chile, and *jarrah* forests of Australia (Cowling et al., 1996; Rundel et al., 2016). The MCRs support extremely high plant diversity, nearly 20% of the world's cataloged flora (Cowling et al., 1996). Fig. 12.6 depicts soil and landscape driven heterogeneity in California forests.

Contrary to the boreal zone, cool slopes and basins in the MCRs tend to support higher biomass than warm slopes, due to the overwhelming influence of water availability and transpiration demand on plant and soil productivity. Where meteorological water availability is low, there can be major changes in vegetation physiognomy from cool to warm slopes. Because drought during the growing season is the principle ecological stress in the MCRs, spatial heterogeneity in soil depth, water infiltration and water holding capacity is a major driver of vegetation pattern (Fig. 12.4; Meyer et al., 2007; Padilla and Pugnaire, 2007).

General characteristics of MCR soils include a xeric moisture regime, moderate weathering, and moderate leaching leading to shallow, weathering-limited soils (except in foothills and lowlands where conditions are better for soil development), and the widespread presence of paleosols, especially in South Africa and Australia (Zinke, 1973; Bradbury, 1977; Yaalon, 1997). In the MCRs, slow soil development makes soil characteristics highly dependent on bedrock (Bradbury, 1981; Vallejo, 1983). Mediterranean forests and woodlands are found mostly on alfisols and inceptisols, and sometimes on entisols (Soil Survey Staff, 1999), but jarrah forests in southwestern Australia occur on deep, lateritic oxisols. Entisols are new or underdeveloped soils found in locations where the formation of pedogenic horizons has not yet occurred, or where it cannot occur. Often entisols are found in uplands where soil formation rate is very low or lower than soil loss (shallow soils), and locations where sediment accumulation is faster than soil development, for example on steep slopes with active erosion or in flood plains or dune fields; in MCRs other than the MB the parent material is often quartz-rich and the soils are often excessively well-drained. Where trees can grow on entisols, they are typically scattered in low productivity woodlands (Fig. 12.5B — S slopes; and 12.5C — left side), but forests can occur if clay content and water holding capacity are high. If conditions permit, inceptisols will develop from entisols over time. In MCRs inceptisols often include a high volume of rock fragments. Soils developed on siliceous bedrock usually show low profile differentiation and leaching and are moderately acidic. Inceptisols often support conifer forests, typically dominated by *Pinus* (Fig. 12.5E and F). Alfisols occur where soil development has led to accumulations of clay in the B horizon. Alfisols are fertile soils that have experienced low levels of leaching, they form more rapidly on calcareous parent materials because such substrates break down to clay minerals more efficiently. In the MCRs, alfisols are the soils most likely to support broadleaf forests, but they will also support productive conifer forest Fig. 12.5A and D; Soil Survey Staff, 1999, Binkley and Fisher, 2012).

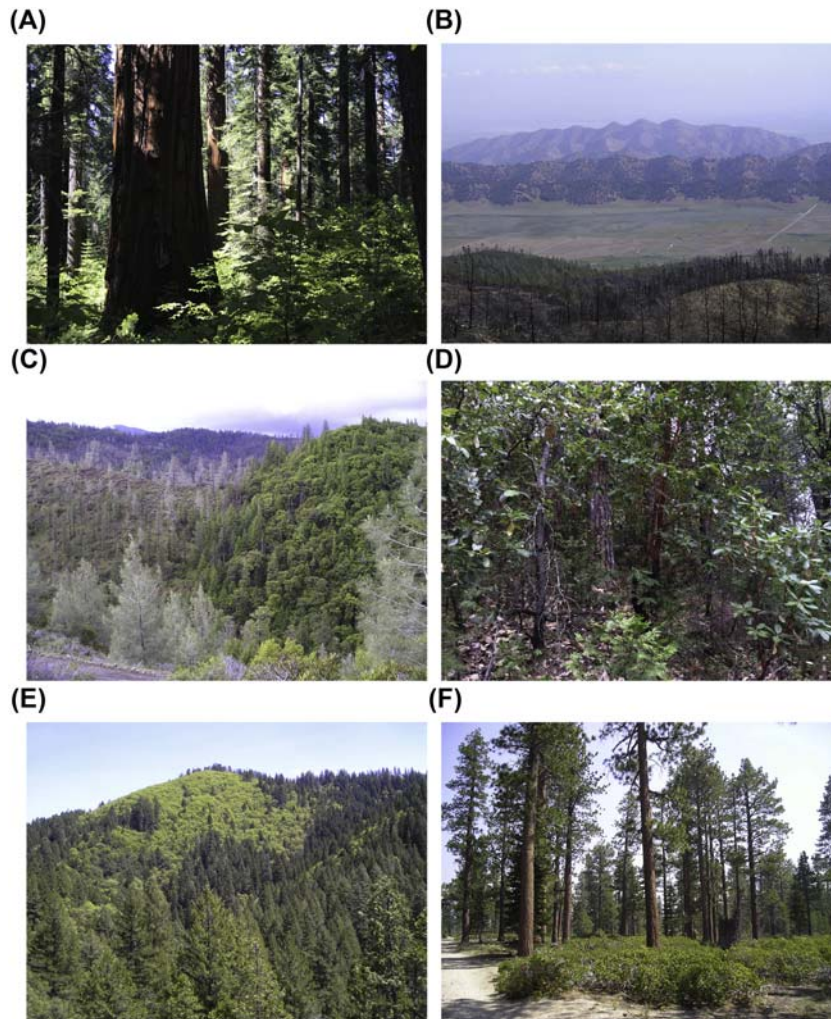


FIG. 12.6

MCR forest heterogeneity in California. (A) Giant sequoia (*Sequoiadendron giganteum*) grove in moist mixed conifer forest in the South Sierra Nevada on deep soils of high water-holding capacity; prescribed fire is used as a restoration treatment in this site to reduce fuels and provide mineral soil for sequoia seed germination. (B) In North Coast Ranges, cool, north slopes with deeper soils support dense broadleaf forest; warm, south slopes with shallow soils support blue oak (*Q. douglasii*) woodland; foreground is serotinous conifer woodland on serpentine soils. (C) Forest heterogeneity driven by soil type: dense mixed conifer and broadleaf forest on relatively fertile metasediments adjacent to open conifer woodland on serpentine, N Sierra Nevada. (D) Dense, highly fire-prone forest, result of 100 + yrs fire suppression and logging on fertile soils in Sierra Nevada foothills; dominant genera are *Pinus*, *Calocedrus*, *Quercus*, and *Arbutus*. Due to high density and water stress, this forest condition is highly susceptible to severe disturbances like fire and insect and disease outbreaks.

(The last part of the 12.6 caption (the part referring to (E and F)) is found at the bottom of the next page.)

One distinct feature of the MB as compared to the other Mediterranean-climate regions is the abundance of limestones. On this parent material, it is common to find red and brown-reddish Mediterranean soils called *terra rossa*. Directly on this bedrock, especially in the uplands, soils are shallow, stony and mostly carbonate-free in the particles finer than coarse sand (mostly entisols and inceptisols, or alfisols in deeper soil profiles; Vallejo, 1983). The origin of these soils is attributed, at least in part, to Saharan dust accumulated during the Quaternary (Muhs et al., 2010). Like in other drylands in the world, saline soils are widespread in poorly drained and coastal areas in the MCRs. Soil salinization is largely an agricultural issue and it is not tackled in this chapter. An exception is the deep lateritic soils underlying jarrah forests in SW Australia, where salinization is a direct result of forest canopy disturbance and the only known guarantor of potable water is maintenance of intact forest (Dell et al., 1989). Low productivity soils that have developed on chemically severe bedrock types, such as ultramafic (“serpentine”) rocks and gypsum, are low in macronutrients necessary for plant growth and support often stunted vegetation with high numbers of endemic plant species (Fig. 12.6C – left; Harrison and Rajakaruna, 2011).

Young orogenic systems and geomorphic dynamism in the MB, California and Chile contrast with stable, older systems in the Australian and South African Mediterranean regions (Thrower and Bradbury, 1973). In general, Mediterranean soils from Australia and South Africa are oligotrophic as compared to the more nutrient-rich soils in the other regions (Rundel et al., 2016). High mountain landscapes produce relatively high erosion rates in the MB, California and Chile; the much older landscapes in South Africa and Australia are less topographically diverse and less erosion-prone. Overall however, soil erosion is a major issue in all of the MCRs, due to the frequently skeletal soils, high disturbance rates, and the often heavy rainfall events that signal the end of the dry season. Compared to Chile and California, which were not deforested centuries ago and which are tectonically more active, watershed sediment yields tend to be relatively low in the MB, probably due to the early loss of the most vulnerable soils (Yaalon, 1997).

Human occupation of the Mediterranean climate zones has a long history, especially in Africa, Europe, and Australia. In the MCRs, winter rains and — near the coasts — pleasant, dry summers result in easy living conditions and high terrestrial productivity, and cold upwelling currents offshore lead to high ocean productivity as well. South Africa has supported hominids for millions of years, and Australia was settled as early as 65,000 years ago (Clarkson et al., 2017). The first human civilizations developed along the eastern edges of the Eurasian Mediterranean climate zone and long human influence on MB ecosystems leads to difficulty in discerning the extent to which modern vegetation is the result of “natural” processes versus human agency (Blondel and Aronson, 1995). Humans did not settle the Americas until the late Pleistocene, probably between 13,000 and 15,000 years ago, however by

(E) Under warming temperatures, increasing N deposition, and recurrent disturbance, broadleaf species are increasing their density in montane forests: resprouting black oak (*Q. kelloggii*) dominates an area of timber harvest; more mixed forests are often goals of modern restoration work, since broadleaf species are more resilient to fire and are not affected by bark beetles. (F) Restored stand of Jeffrey pine-white fir (*P. jeffreyi*-*Abies concolor*) forest near Lake Tahoe; site was similar to (D) before restoration, which involved a mechanical thinning, followed by hand thinning, and then fuel pile burn and prescribed fire over 9 year-period. Sites like this survived the 2007 Angora Fire largely intact while neighboring untreated sites suffered 90%–100% mortality.

Photos: H Safford.

the time of Euroamerican arrival in California in the 18th and 19th centuries, California's native populations were the densest in North America (Ubelaker, 1988). One outstanding difference between the MB and the rest of MCRs is its longer history of widespread and intense land use (Keeley et al., 2011), especially with respect to agriculture. Among the other MCRs, only central Chile showed significant agricultural development before European settlement. By the late 20th century, profound changes in land use and population migration to urban areas in the MB greatly reduced the dependence of the human population on forest resources. Today, as in the other MCRs, management focus in MB forests is more centered on recreation, ecological, cultural, and landscape values.

Disturbance

Most Mediterranean climate areas receive sufficient precipitation in the winter and early spring to produce a crop of fuel just in time for the hot, dry summer. Where ignition sources are at hand, fire is the inevitable outcome, and Mediterranean vegetation is among the most fire-prone and fire-shaped in the world. The origin of the Mediterranean-type climate in the middle Miocene led to high levels of diversification in sclerophyllous and other species, with fire being a major driver of speciation in four of the five MCRs. Fire adaptations in the MCR floras are widespread, and range from adaptations such as seed banking, serotiny, and fire-cued germination in high severity fire regimes, to fire resistant traits (e.g., thick bark, self-pruning of lower branches) in low severity fire regimes (Keeley et al., 2011). Because of their dense crowns, highly-combustible foliage, and very low fuel moistures in the late dry season, sclerophyllous shrublands in all of the MCRs support high severity fire regimes, but in Chile there is almost no natural ignition source so fire adaptations are essentially absent. Serotiny is common in these shrublands in Australia and South Africa, but in the northern hemisphere MCRs serotiny is restricted to conifer taxa (pines and cypresses) that coexist with the sclerophyll species (Keeley et al., 2011). Low severity fire regimes are concentrated in woodlands and open forest systems, dominated by oaks and certain fire-resistant pine species in California and the MB, and acacia and/or eucalyptus woodlands in the other MCRs.

The causes of fires vary among MCRs, and the balance and density of ignitions has changed over time. Climatic conditions in the MCRs during the dry season (dominated by stable, high pressure systems) lead to relatively low lightning strike densities (Manry and Knight, 1986; Safford and Van de Water, 2014). As a result, humans have played and continue to play the major role in fire ignition. In Chile the height of the Andes prevents westward advection of summer storms from the South American interior and nearly all fires are human caused; fires were nearly unknown before human settlement. In California, fires in shrub-dominated and oak woodland landscapes are almost entirely anthropogenic, but montane forests experience relatively frequent lightning ignitions and the modern balance of natural to human fires is closer to 50:50. Although mountainous areas in eastern California have always supported sufficiently high lightning strike densities to account for most of the historical area burned (natural fire rotations of 20–35 years, dominated by high frequency, low severity fire (Safford and Stevens, 2017)), lowland areas in western California would have experienced very little fire before the arrival of people (Keeley and Safford, 2016). In the MB, 95% of fires are human caused (Ganteaume et al., 2013); this pattern has probably not changed greatly over time. In South Africa and Australia, lightning and human ignitions have not been assessed at the MCR scale, but local studies suggest balances between 30:70 and 70:30 for natural versus human caused fires, depending on the proximity to human communities (e.g., Horne, 1981).

Fire impacts physical, chemical and biological soil properties based largely on fire intensity at the ground level (see [Cerdà and Robichaud 2009](#); and [Caon et al. 2014](#) for reviews). Direct fire impacts are usually concentrated in the top few centimeters of soil. The most important direct changes are net ecosystem losses of nutrients (but ephemeral increases in cations), especially nitrogen through volatilization; development of soil water repellency (hydrophobicity) in coarse-grained soils; and temporary loss of forest floor habitat for soil biota ([Wohlgemuth et al., 2018](#)). The indirect effects of fire are often more damaging ([Vallejo and Alloza, 2015](#)). These are mostly related to the temporary loss of plant biomass and the forest floor cover, and sometimes soil sealing in fine-textured soils. Thick organic layers at the top of the soil pedon are rare in MCR soils, and even low intensity fires can remove much of the soil cover. As a result, soil erosion and runoff risk in the MCRs can increase dramatically after burning, causing further nutrient losses and impacts downstream such as flooding and siltation ([Shakesby, 2011](#)). Peak and storm flows in burned MCR watersheds can radically increase in the years after fire, with higher flows persisting for 5–10 years ([Wohlgemuth et al., 2018](#)). Recurrent high severity fires may cause ecosystem nutrient depletion ([Raison et al., 2009](#)). In MCR forests, stand-replacing fires drastically reduce transpiration at the landscape scale, and ground water and surface water levels typically rise notably until vegetation regrowth is well underway. A novel situation is developing in California, where an extensive drought- and beetle-driven mortality event has killed more than 150 million trees in the southern Sierra Nevada. As trees begin to fall, coarse fuel loads could eventually reach many 100s of tons per ha across hundreds of thousands of hectares. The sudden inputs of massive amounts of large woody debris can spawn conditions for firestorms or “mass fires”, where the contemporaneous and protracted burning of the landscape generates its own atmospheric circulation. Such fuels conditions cannot be accounted for in current wildfire behavior models ([Stephens et al., 2018](#)).

Overall, humans tend to increase the number of ignitions in wildland landscapes in which they settle (at least until settlement has removed most of the natural vegetation), and that pattern is apparent in most of the MCRs that were settled during the Pleistocene (Australia, California, Chile). Divergence from this pattern occurred after Euroamerican settlement in California and Australia, where anthropogenic burning declined after aboriginal populations were decimated and fires banned ([Bradstock et al., 2002](#); [Safford and Stevens, 2017](#)). Since that time, fires in peri-urban MCR settings have tended to increase in frequency. An exception is found in yellow pine and mixed conifer forests in the California mountains, where a century-long policy of fire suppression essentially erased fire as an ecological force until resultant fuel accumulation and forest densification ([Fig. 12.6D](#)) interacted with recent climate warming to promote a growing wave of large and destructive fires ([Safford and Stevens, 2017](#)). The densification and homogenization of California conifer forests has been paralleled by similar trends in the MB, but for different reasons. There, rural land abandonment has drastically reduced management of vegetation, and forest cover is densifying and becoming more contiguous ([Fig. 12.9](#)) across broad areas of southern Europe (except in more arid regions, where ecosystem recovery has been poor). As in California, this increase in fuel amount and continuity has been partially blamed for recent destructive fires in the MB ([Vallejo and Alloza, 1998](#); [Benayas et al., 2007](#)).

Fire is by far the dominant large-scale disturbance process in Mediterranean forests and woodlands, but insect and disease outbreaks are locally important, as are human-driven impacts such as livestock grazing, logging, and land use change. A variety of indigenous beetle taxa have evolved to attack fire- or drought-weakened trees in MCR forests, and the fear of such outbreaks is one of the principle justifications for intensive post-fire tree harvest in much of the MB ([Vallejo et al., 2012b](#)). In California,

interactions between forest densification and drought have led to a number of large-scale tree mortality events, most notably between 2012 and 2016 in the Sierra Nevada (Young et al. 2017). Tree mortality events of this scale have important interactions with fire (Stephens et al., 2018). Direct grazing and browsing influences on MCR forests are most apparent in young forests, where heavy browsing can negatively impact palatable regenerating trees; successful restoration of broadleaf species can depend heavily on browse control. Indirectly, grazing affects forests by reducing understory fuels and potentially reducing fire frequencies, although the general lack of summer precipitation in the MCRs reduces grass biomass in the understory and this phenomenon is not as important as it is in summer precipitation regime savannas (Safford and Stevens, 2017).

Human impacts on MCR forests and woodlands have been extensive and locally intensive. Human populations in the MCRs have always been relatively high, and urban and exurban expansion, agriculture, forestry and other economic activities have greatly transformed many MCR landscapes. The MB is the most extreme case, where deforestation and intensive agricultural use have occurred for millennia. Almost all MB forests are secondary, and seral shrublands and forests tend to be highly flammable. Current MB forest compositions generally include at least some important species moved by man. For example, Phoenician, Greek, and Roman traders greatly expanded the distributions of chestnut (*Castanea sativa*) and walnut (*Juglans regia*) (Blondel et al., 2010). Human introduction of invasive species is a major issue in all of the MCRs. Acacia and eucalyptus species from Australia and pines from the MB and California were introduced to the other MCRs beginning mostly in the 19th century. Acacia is a very invasive genus and it has become a plague in South Africa and is well on its way in Chile; Monterey pine (*Pinus radiata*) and other invasive pines are invading wildlands in all of the southern hemisphere MCRs (Groves et al., 1991). California and Chile have been overrun by weedy plants from the Mediterranean Basin (Rejmanek and Randall, 1994), and lower elevation forests have seen their understories converted largely to exotic species. Invasive pathogens have become major problems in some of the MCRs. For example, *Phytophthora* species introduced from the tropics are causing extensive mortality of oaks in California and the MB, eucalyptus species in Australia, and *Nothofagus* in Chile (e.g., Rizzo and Garbelotto, 2003; Fajardo et al., 2017; Brasier, 1996). *Cronartium ribicola*, introduced from Europe, is having devastating impacts on white pines in western US montane forests.

Climate change impacts

Current trends in temperature are similar across the MCRs. In the MB, mean annual temperatures have risen about 1.5° since 1910, with most of the change in the summer and since 1970 (Mariotti et al., 2015). In southwestern South Africa, temperatures rose an average of 0.13°/decade between 1916 and 2013 (Lakhraj-Govender et al., 2017), southwestern and southern Australia warmed by 0.5°–1.5° between 1910 and 2013 (CSIRO and BOM, 2015), and California temperatures have risen between 0.5° and 1.8° since the beginning of the 20th century (Bedsworth et al., 2018). Current trends in precipitation are more divergent. For example, California shows a slight average increase in annual precipitation over the last century (the drier recent decade notwithstanding), but in southern and western Australia mean annual precipitation is down by 10–20% since 1970 (CSIRO and BOM, 2015). The eastern MB shows a significant decrease in annual precipitation over the last half century, but the trend in the western MB is not as clear (Caloeiro et al., 2018). Future climate projections suggest moderate (compared to the boreal projections) increases in temperatures. Under RCPs 4.5 and 6.0, mean annual temperatures are projected to increase between 0.7° and 1.5° by 2035, and 2°–4° or

more by 2100 (Collins et al., 2013; Kirtman et al., 2013). Precipitation projections suggest drier future conditions for all MCRs except California, where most recent projections are for similar or slightly increased precipitation (Polade et al., 2017).

Klausmeyer and Shaw (2009) projected the future geographical extent of the climate space occupied by the five MCRs based on three climate scenarios. The generic MCR climate space expanded by 6%–11% depending on the scenario, but different MCRs suffered different fates. California experienced $\pm$ no projected change in area, but the MB (+15%–30%) and Chile (+30%–50%) were both projected to see notable expansions in their geographical extent. The South African and Australian MCRs were both projected to shrink considerably, to 60%–80% of current size in South Africa and 50%–75% in Australia, because their poleward boundaries are defined by the ocean. Climate change velocity in the MCRs is mostly between 0 and 50 km per decade, except in Europe, where it is between 0 and 100 km per decade due to the high east-west mountain ranges that constrain northward movement for many taxa (Burrows et al., 2011). Climate change effects on vegetation in the MCRs, in interaction with fire, will be significant. Temperate mixed forests (broadleaf + conifer species), which dominate the forests of California and the MB, were projected by Gonzalez et al. (2010) to be the most vulnerable biome type to the combination of global warming and increased fire activity. Mechanistic models incorporating climate and disturbance project widespread transformation of conifer forests to broadleaf forests (Fig. 12.2E), forests in general to shrubs and sometimes grassland (Fig. 12.7), and shrubs to grassland (Mouillot et al., 2002; Lenihan et al., 2003).

For the MCRs, soil moisture projections are universally down, due primarily to higher potential evapotranspiration but also to projected decreasing precipitation input in some MCRs (e.g. Mariotti et al., 2015; CSIRO and BOM, 2015). At medium to high elevations, the balance of rain and snow will shift strongly toward the former, increasing the potential for soil leaching, but leaching will probably decrease in most MCR forests. The main soil property potentially affected by climate warming is soil organic carbon (SOC). SOC and its decomposition rate affect many other soil properties, such as microbial activity, soil structure (and its impacts on water flow), soil fertility, and the residence time of forest floor residues. In theory, warming should increase the soil organic matter decomposition rate and soil respiration. However, a more profound summer drought – which is universally projected for the MCRs – may reverse this effect, since sufficient soil water is required for microbial respiration to occur (Manzoni et al. 2012). This effect will be especially strong in forest types whose water use has risen due to anthropically increased stem densities, which is the case in much of California and the northern MB (Concilio et al., 2009). In southwestern Australia, jarrah forest SOC is projected to decrease by as much as 30% under future drying (Dean and Wardell-Johnson, 2010).

Increased water stress may also reduce plant productivity, cascading to decreased litter inputs to the soil and reductions in soil microbial activity. Warming will increase soil biological activity in the cold season, but drying is likely to decrease it in the warm season. In California, most soil C decomposition in montane forests occurs between March and June, under the warming snowpack and in the late spring and early summer. Climate warming will move this period to earlier in the year and potentially shorten it as well (M. North, US Forest Service Pacific Southwest Research Station, pers. comm.). Projections of SOC dynamics are affected by high uncertainty regarding microbial community responses to climate change (Hararuk et al., 2015). The decomposing activity of invertebrate detritivores is expected to be reduced by the combination of warming and reduced precipitation (Thakur et al., 2018). Gains and losses of SOC related to climate change are likely to be spatially heterogeneous, depending on gradients in temperature, moisture, soil type, vegetation, and land use (Gottschalk et al., 2012).



FIG. 12.7

Interactions between multiple ecosystem stressors can greatly complicate forest management and restoration. 12 years before the photo, this site on Laguna Mountain near San Diego, California was occupied by a mature Jeffrey pine-black oak (*P. jeffreyi-Quercus kelloggii*) forest. The site was burned by a severe wildfire in 2003 following a two year drought, killing most Jeffrey pine on site. With most adults killed, seedling densities in the year after fire were low, and survival was nearly zero after two hot summers. In response, the US Forest Service made three attempts to artificially plant stands near this site, but all attempts failed. Fortuitously, some patches of forest survived the fire, but a subsequent Jeffrey pine beetle (*Dendroctonus jeffreyi*) outbreak killed many adults, even though prescribed fire had been employed to thin some of the stands and reduce water stress. Given the Jeffrey pine mortality, black oak appeared poised to dominate stands like the one pictured, but recent invasion by a boring insect apparently introduced in firewood from SE Arizona (Gold Spotted Oak Borer, *Agrilus auroguttatus*) has devastated the oak population. Most recently, the area experienced another severe drought from 2012 to 2016. Today sites like this are succeeding to open shrubland with an understory of invasive grasses from the Mediterranean Basin, and accumulating heavy loads of coarse woody fuels. Management and restoration goals for this area are being reevaluated.

Photo: H. Safford.

In the MCRs, soil degradation processes are projected to be reinforced by climate change, largely because of increased soil erosion due to more extreme and more frequent climate and disturbance events (fire activity, periods of intense rainfall, etc.) and soil salinization (Settele et al., 2014). Soil degradation in drylands can lead to major losses in soil fertility and ecosystem productivity. These dry-land processes are often referred to as “desertification” (United Nations Convention to Combat Desertification, <https://www.unccd.int/>), since they involve the poleward migration of desert-like conditions, which dominate the subtropical boundaries of the MCRs. Desertification is driven by interactions between changing climate, disturbance, and land over-exploitation, which is related to the economic dependence of local populations on primary land productivity. The MB provides an example of strongly contrasting situations. In southern MB nations — North Africa and the Levant — contemporary desertification is driven by over-exploitation of the land by rural populations and is magnified by increasing climatic aridity (similar trends are occurring on the eastern borders of the South African MCR). In contrast, over-exploitation in the northern MB (southern Europe) occurred principally prior

to the mid-20th century. Today rural land abandonment and increasing forest cover are the dominant trends in the northern MB (Blondel et al., 2010).

Climate change is projected to increase heat waves and drought in all of the MCRs (e.g., Bedia et al., 2015; Bedsworth et al., 2018). Where precipitation is also projected to decrease (most of the MCRs), this may lead to lower vegetation productivity and less potential for large fires over time, but a large proportion of the current biomass on MCR landscapes is likely to burn in the meantime. In addition, in California and the MB, forest densities are abnormally high due to anthropogenic causes (fire suppression in California, land abandonment in MB), which has greatly increased the potential for large, destructive forest fires. Climate change will influence fire via direct effects on fire weather and ignitions, and via indirect effects mediated through climate change's influence on vegetation and fuels. The influence of climate variables on fire occurrence will be different for different landscapes (for example, in fuels- vs. climate-limited ecosystems), but some general patterns emerge in current trends and models. Restaino and Safford (2018) summarized likely climate change impacts on California fire regimes. Overall, California forests will experience higher fire probabilities (except in fuel-poor ecosystems under drier future scenarios); higher fire frequencies; much greater annual area burned (100%–200% increases in some models); higher fire severities (but dropping over time as woody vegetation increasingly fails to recover); a much lengthened fire season, by as much as 4–8 weeks by 2100 in some models; and possibly increased incidence of lightning. van Mantgem et al. (2013) note that because climate warming and increasing growing season drought interact to decrease tree vigor, fire severity in the future will rise even without a concurrent increase in fire intensity. Projections of climate changes influence on fire are similar for the other MCRs (e.g., Williams et al., 2001; Battlori et al., 2013).

Mediterranean forest and woodland management and restoration under global change

As in the boreal forest, ecosystem management and ecological restoration are relatively new developments in the human relationship with Mediterranean forests. Given the very high level of ecosystem degradation in the MCRs, as well as rapidly growing human populations and economies, the conservation situation is more urgent. Other than obvious ecosystem-level differences, one of the major differences between the boreal and Mediterranean climate regions is the more complex geographic, political, and economic situation in the latter. The boreal zone stretches across three continents and includes land in nine nations whose median GDP per capita ranks 22nd in international rankings (<https://www.imf.org/external/pubs/ft/weo/2018/01/weodata/index.aspx>). The MCRs are found on six continents and in 25 nations whose median GDP per capita ranks 46th. In the MCRs, the outlier in this respect is the MB, where 20 of these nations are found, some of which have been recently or are currently in the throes of major unrest or economic downturns (e.g., Syria, Lebanon, Algeria, Cyprus, Turkey, Greece). The northern MB is in Europe, the southern and eastern parts of the MB are in North Africa and the Middle East, and economic, political, and ecological trends are very different between the subregions. We make these points to underline that almost all money and effort being spent in ecosystem management and restoration in the MB is being spent in Europe — with a few notable exceptions — and the examples we have to draw from come overwhelmingly from there and other wealthier MCRs.

Pausas et al. (2004) summarized the overarching focus areas for forest restoration in the Mediterranean Basin as:

1. Soil and water conservation
2. Increasing the resistance and resilience of ecosystems to climate change and disturbances
3. Increasing the prevalence and stability of mature woody formations

This list encompasses most of the forest management and restoration focus areas in the other MCRs as well. We would add:

4. Biodiversity conservation
5. Fire hazard and risk reduction
6. Other ecosystem services, including aesthetic, recreational, and provisioning services

In southwestern Australia, an area of almost 2 million hectares is covered by *jarrah* forest, so named for its dominant species (*Eucalyptus marginata*), and provides a salient example of the great complexity facing managers and restorationists in the MCRs. Jarrah forest is at the center of one of the world's great biodiversity hotspots (Hopper and Gioia, 2004). Unique to the MCRs, jarrah forest grows on very old and deep oxisols which can harbor large quantities of salt in the subsurface. Jarrah is a fire-resistant resprouting tree whose deep roots permit high transpiration even during the dry summer. Fires are common in jarrah forest and fuel accumulation is rapid, current fuel management strategies involve extensive prescribed burning. Western Australia's population has been rising at a rate of 2% per year since the early 1970s and demand for municipal and agricultural water supplies is increasing. The major management priority in most jarrah watersheds is the maintenance of water quality, but an extraordinary number of interacting and conflicting threats complicate the picture (Shea 1982; Dell et al., 1989). Much of the original forest was cleared in the 20th century for agriculture, and most of the remaining forest outside conservation units has been logged at least once. Forest loss has led to release of the soil salt into waterways in the eastern jarrah region and some river systems are now too brackish for human or even irrigation use. To compound the water problem, precipitation in southwest Australia has dropped by 15%–20% since the 1970s, and the trend is projected to continue. Mining has caused further forest loss: 30% of the jarrah forest may grow on commercial-grade deposits of bauxite (aluminum ore), and bauxite mining is a major forest disturbance factor. To further complicate the picture, an exotic oomycete — *Phytophthora cinnamomi* — was introduced to western Australia in the early 20th century and has expanded throughout much of southwestern Australia. *P. cinnamomi* infects 40% of the plant species in Western Australia, and large areas of jarrah forest and *Banksia* woodlands have experienced high mortality (Dell et al., 2005). Vehicular traffic, soil disturbance and movement, and altered hydrology are principal drivers of *P. cinnamomi* dispersal, so mining, recreation, timber harvest and other forest uses conflict with containment of the outbreak.

Considering all of the threats to the jarrah forest, settling on a coherent and comprehensive regional policy for effective ecosystem management and restoration has proven difficult. Conservation and forest management to increase the area of uneven-aged forest with old-growth attributes has been recommended to increase water yields, reduce salinization, sequester more carbon, and protect more habitat for threatened biota (Shea, 1982; Dean and Wardell-Johnson, 2010; Macfarlane et al., 2010). The use of hotter prescribed fire in the jarrah understory has been recommended to reduce the density of alternate hosts of *P. cinnamomi* (Burrows, 1985), but recent work suggests that fire can actually increase *P. cinnamomi* infection rates (Moore et al., 2015). Sena et al. (2018) recommended focusing

conservation on forest communities in drier soils, which are more resistant to *P. cinnamomi* infection. Wardell-Johnson et al. (2015), noting the climate and fire threats to old forest, proposed focusing forest restoration on understory species, which are highly diverse in jarrah forest and are more likely to be resilient to future conditions.

In California, forest management and restoration are focused on similar priorities. California is the most populous state in the US, its economy is larger than the economies of France or India, and its agricultural sector is the most productive in the world. Water supply for California's 40 million people and massive agricultural infrastructure is mostly provided by the forested watersheds that encircle the Central Valley, especially in the Sierra Nevada. With highly variable precipitation, decreasing snow-pack, and increasing summer drought and propensity for large and severe disturbances (Dettinger et al., 2018), there is much political and management focus on ensuring resilience of montane forests and their ecosystem services. On US Forest Service (USFS) lands, ecosystem management principles underlie national and regional strategies to deal with these and other issues (USDA, 2012). In California, ecosystem management and ecological restoration are both driven by a focus on NRV/NDE concepts — modified to account for projected trends in climate and other ecosystem drivers — and ecosystem resilience (USDA, 2011; Safford and Stevens, 2017). In montane forests, management and restoration practices emphasize the generation of heterogeneous stand structures based on the joint workings of the physical habitat template (soils and topography) and the fire regime (Box 12.1; North et al., 2009, 2018; North, 2012). Multiple large-scale collaborative regional and watershed assessments have been carried out, and the State of California recently passed a law allocating \$1 billion of carbon cap-and-trade funds to an expansion of forest thinning and prescribed fire to abet forest resilience. The USFS, which manages most of California's forests, is mandated to manage for multiple uses, and frequently finds itself caught between competing interests. For example — in a situation similar to SW Australia — a major conservation concern is the sustainability of old, complex forest and biota adapted to those conditions, but such habitat is of doubtful stability under projected future conditions (or even current conditions, as evidenced by the huge areas of severe forest disturbances in the last decade; Stephens et al., 2016). Also like Australia, forest pests and diseases, exotic and native, are a wildcard and will complicate forest management (Fig. 12.7). Some of the other issues to be resolved include: how to deal with timber and small diameter biomass removed in expanded forest thinning with very few saw-mills or biomass energy plants operating in California; how to operationally increase the area of naturally ignited wildfires allowed to burn in frequent-fire forests; how to measure and mitigate soil, water, air and biotic impacts from greatly increased forest management activities and fire; how to carry out monitoring on a huge scale and use results in an adaptive management framework.

Socioeconomically speaking, the Mediterranean Basin (MB) is the most complex of the MCRs. Found on 3 continents and across 20 nations, the MB is a region defined by many cultures, races, languages religions, and political persuasions. The range of private to public ownership is extreme, ranging from 95% private forestlands in Portugal to 0% in Turkey. At the same time the MB is the least biodiverse of the MCRs on a per area basis (yet it is still a world biodiversity hotspot; Cowling et al., 1996), probably due to the effects of tens-of-thousands of years of human occupation and degradation. Although the European Union affords a level of homogeneity to management and restoration efforts in the northern MB, the variability across the MB in policies, science support, political engagement, and even cognizance related to the effects of global change on forests and their soils is tremendous.

BOX 12.2 Degradation problems and restoration opportunities of Mediterranean terraced oldfields

Oldfields on terraced slopes are widespread in the northern MB countries, and in some cases date from well before the Roman period. Terraces were constructed to reduce soil erosion on steep slopes, to deepen available soil for crop roots, and to enhance water infiltration into the soil. Terraced fields provided a large portion of the cereals, grapes, olives, fruits and nuts consumed by people living in mountainous parts of the MB. However, beginning in the mid-20th century, economic and social changes in the MB led to migrations from rural areas to cities and abandonment of most terraced agriculture (Lasanta et al., 2013). Today, the stone walls supporting the terraces are suffering a generalized degradation (Fig. 12.8), as human maintenance no longer counteracts the natural process of slope regularization. In areas with sufficient moisture and moderate slopes, dense forest cover can rapidly develop on previously cultivated slopes, and many forested landscapes in the MB hide extensive terracing under their canopies. Degraded and unmaintained terrace networks are especially sensitive to soil erosion when plant cover is low however, for example under semi-arid conditions (e.g., in southern Spain, Italy, Greece, Turkey), or after wildfires (Fig. 12.8). The northward march of aridification in the MB is a major threat to the stability of unmanaged terraced landscapes (Bautista et al., 2009).



FIG. 12.8

Terraced oldfields after a fire. After abandonment, the site was colonized by obligate seeder shrubs and scattered pines. Stone walls are suffering a degradation process, which is intensified by fire. Castellón, Eastern Spain.

Photo: V.R. Vallejo.

In the MB, secondary succession in oldfields is dominated by highly flammable, obligate seeder shrubs and serotinous pines (especially Aleppo pine, *Pinus halepensis*), which constitute extremely fire-prone ecosystems (Santana et al., 2018). Often, in high risk areas with frequent human ignitions, these plant formations enter high frequency fire cycles that arrest succession (Baeza et al., 2007) and increase post-fire erosion and ecosystem degradation risk. Serotinous pines are locally eradicated when the fire interval is shorter than their maturity age (some 15–20 years for Aleppo pine), and fire-prone shrublands develop instead (Pausas et al., 2004). In these conditions, the recovery of native sclerophyllous vegetation, both tall shrubs and trees, is very slow owing to the low ability of most of these species to disperse to and recruit in new spaces (Vallejo et al., 1999).

When they are abandoned, terraced oldfields can become foci for soil and ecosystem degradation. Soil plowing is a major driver of soil degradation in all of the MCRs and its effects persist for decades (see, e.g., Stromberg and Griffin (1996) from California). Conventional agricultural plowing provokes a dramatic reduction of soil organic matter content and deleteriously impacts soil microbial composition and activity. These effects

Continued

BOX 12.2 Degradation problems and restoration opportunities of Mediterranean terraced oldfields—cont'd

contribute to soil compaction in fine textured soils and soil crusting when plant cover is low, e.g. under semi-arid climate or immediately after a fire. Soil crusting and compaction reduce soil water infiltration and water holding capacity, thereby reducing available soil water for vegetation.

Plowing also causes the eradication of natural vegetation. After abandonment, primary vegetation recovery is through colonizing, obligate-seeding species (e.g., *Cistus*, *Rosmarinus*, *Ulex*) that in the MB are fuel accumulators and facilitate a high frequency fire cycle. Secondary succession can be arrested for decades in this fire-prone community. Significant natural recolonization of sclerophyllous plants (resprouting species in genera like *Arbutus*, *Pistacia*, *Quercus*, *Rhamnus*) only occurs when fire frequencies can be reduced and when adult individuals are present in the vicinity, owing to the low dispersal ability of these species.

Abandoned agricultural terraces slowly disintegrate once the supporting stone walls begin to fall down. Wall failure depends on, among other factors, the quality of the original construction, the soil type, the slope, and the potential for heavy rainfall events. Ironically, over time a structure built originally as a soil and water conservation system can become a source of concentrated soil erosion. This degradation is often reinforced by forest fires (Fig. 12.8).

If properly maintained, oldfield terraces can provide excellent opportunities for forest restoration. Soils of agricultural terraces are relatively deep and the microtopography is favorable for runoff harvesting. These factors facilitate the successful plantation and establishment of woody, later successional species. Low soil organic matter content, poor biological activity and degraded soil structure – results of long-term tilling and cultivation – can be improved through organic soil amendments and soil preparation techniques. Soil microbial activity has been shown to co-vary with plant productivity (Broughton and Gross, 2000), and revegetation of degraded soils improves taxonomic and functional diversity of soil microbial communities (Guo et al., 2018), but the recuperation of native, pre-cultivation soil biota is a challenge that deserves further research. Overall, the probability of restoration success and its cost-effectiveness are relatively high compared to unterraced slopes and shallower soils.

**FIG. 12.9**

Colonization of oldfields. Succession from shrubland (bottom) to Aleppo pine (*P. halepensis*) forest (center). Portion of site was burned earlier in the year. Valencia, Eastern Spain.

Photo: V.R. Vallejo.

Restoration of terraced oldfields also offers the opportunity to convert highly flammable, low resilience plant formations dominated by obligate seeders into less flammable, more fire-resilient plant formations dominated by woody resprouters and sclerophyllous species (Santana et al., 2018). Where successful, oldfield restoration thus serves the additional purpose of improving landscape fire-resistance and resilience.

In the MB, land use practice over millenia has generally selected the most productive soils for agriculture, leaving poor soils, especially on the hillslopes, for forests and shrublands. In periods of food shortage even these poor soils were often cropped and later abandoned. Therefore, contemporary forests are generally developed on steep slopes, on shallow, stony soils, or in places that show some other limitation for cropping, e.g. difficult access, sand dunes, gypsiferous soils, river banks (Fig. 12.9). As a consequence, forest restoration in practice is mostly restricted to low productivity soils. In the MB, terraced oldfields often offer the best soils available for the restoration of quasi-natural habitats (Box 12.2).

In most restoration projects in the MB, irrigation is not available and drought duration exerts strong influence on seedling establishment, especially during the first year after planting or seeding (natural or artificial) (Vallejo et al., 2012a). This is especially the case in the southern MB, where the summer drought can last 4–6 months. In degraded soils, ecosystem recovery requires the enhancement of water availability for plants and soil biological activity. Key factors are soil water infiltration capacity and soil water holding capacity, with shallow soils often strongly limiting the latter. These issues are particularly critical for sclerophyllous species that are deep-rooting (e.g., *Quercus*, *Arbutus*, *Pistacia*), where seedling survival strongly depends on rapid root elongation to avoid summer desiccation of the topsoil. In Eastern Spain, 40 cm soil depth is the minimum required to successfully introduce tall woody species on degraded soils (Vallejo et al., 2012a). In restoration treatments, organic amendments can help to improve soil biological activity, but soil fertility limitations rarely cause direct failure of planting in the MB. Indeed, over fertilization may lead to eutrophication (atmospheric N inputs are already high in much of the MB) and weed invasion, which can hinder natural vegetation recovery (Fuentes et al., 2010). Once woody plants are established on a site, feedbacks to the soil through litter accumulation, root activity, soil pH changes and nutrient inputs, etc. help to further improve soil properties. In summary, forest restoration success in MB drylands is highly dependent on improving soil physical properties and rain-use efficiency.

Reforestation and afforestation of degraded lands is an old practice in the MB. Large areas in the northern MB were planted beginning in the late 19th century, especially with pines and exotic species, with objectives centered on the protection of watersheds, timber production, or dune fixation. With land abandonment and economic changes over the last quarter-century, management and restoration objectives have gradually become more biodiversity- and ecosystem service-based. Today, the focus tends to be on “natural capital” and things like climate change mitigation or adaptation, forestalling desertification, reducing the potential for catastrophic fires, restoring natural disturbance regimes, and cultural and recreational values (Aronson et al., 2007; Bautista et al., 2009). In the MB — as in all of the MCRs — new restoration objectives arise according to new social perceptions of nature, and new regional and global threats. For example, forest restoration is rapidly transitioning from tree-oriented, monospecific plantations — often of exotic pines or eucalypts — with little consideration of provenance, to multi-purpose plantations employing native species and provenances, and including broadleaf species and shrubs. Active management of plantations is often necessary to ensure they provide ecosystem services beyond timber volume (Gómez-Aparicio et al., 2009). Focus has also shifted from green forests to black (burned) forests (Moreira et al., 2011). On severely burned landscapes, short-term soil stabilization measures are implemented to mitigate soil erosion, while over the longer-term native resprouting broadleaf species may be (re)introduced to reduce fire hazard and increase fire resilience (Vallejo and Alloza, 2015; Gavinet et al., 2016).

Major global change-related stressors and their implications in mediterranean climate region forests

Most MCR species are relatively drought-tolerant. However the length and severity of the annual drought and the prevalence of year-long and multiyear droughts is increasing (Polade et al., 2017), putting even highly drought-tolerant species at greater risk. Most dominant MCR species also possess adaptations to fire, but the fire regimes to which they are adapted are changing rapidly (Keeley et al., 2011). Atmospheric warming, population and economic growth, land use change, invasive species, air pollution, and other stressors and disturbances are interacting with drought and fire to increasingly threaten the sustainability of MCR forests (Dettinger et al., 2018). As in Fig. 12.5 for boreal forests, in Fig. 12.10 we conceptualize some of the key connections between global change stressors (excepting direct human impacts), soils, and forest management and restoration practices in the MCR forests. As in Fig. 12.5, the diagram is highly simplified and lacks feedbacks as well as interactions. The most important suite of right-to-left feedbacks in Fig. 12.10 comprises positive impacts on soil erosion (i.e., more erosion) and negative impacts on soil fauna, soil respiration/decomposition, O-horizon thickness, and SOC by all of the forest management implications involving lower vegetation productivity/growth/cover, loss of forest, and increased fire activity. Where salinization occurs, it will also have negative effects on soil fauna, soil respiration/decomposition, and SOC. Again as in Fig. 12.5, there are too many feedbacks among the management and restoration implications to list, but among the most important of these are the ways in which changing vegetation and mortality patterns will alter forest fuels and fire risk, and the ways in which changing fuels and fire will alter vegetation.

In contrast to developments in the boreal forest, MCR forest managers and restorationists will mostly experience negative survival and growth effects in their reforestation/revegetation projects (Fig. 12.10). The magnitude of this effect will be inversely related to latitude, altitude, and water balance. Because the MCRs are found at the poleward border of the subtropical deserts, generally lower, more variable precipitation (and loss of snowpack at higher elevations), coupled with a longer and more profound summer drought, will cause a retraction in the climate space for forests. The desert-proximal location of the MCRs also means that the availability of more drought tolerant tree genotypes and species is relatively limited. Plantation success will be increasingly compromised by climate, soil, and disturbance, and novel approaches to planting — genetics, species, pattern, density — will be required (North et al., 2018). Fire potential in the MCRs is already high, but decreasing soil moisture and decreasing forest vigor will further increase forest mortality due to fire and other disturbances, and forest regeneration will be progressively more difficult. Fire impacts to forest cover will be especially profound in the MB and California, where human socioeconomic influences on forest management (rural land abandonment, fire suppression, exurban development) have greatly increased forest density and continuity, and the risk of large-scale, synchronous disturbances. Forest retraction will lead to major, landscape-level expansion of shrublands and grasslands and ultimately a reduction in fire severity and fire risk (but not fire hazard, the probability of fire) as woody fuels dissipate. As global change stressors multiply and increase in magnitude, climate change adaptation in the MCRs will be forced to focus increasingly on sustaining ecosystem services (vs. preserving single species), realignment-type management (vs. resistance or even resilience; Millar et al., 2007), and engineering novel approaches to novel situations (Hobbs et al., 2014).

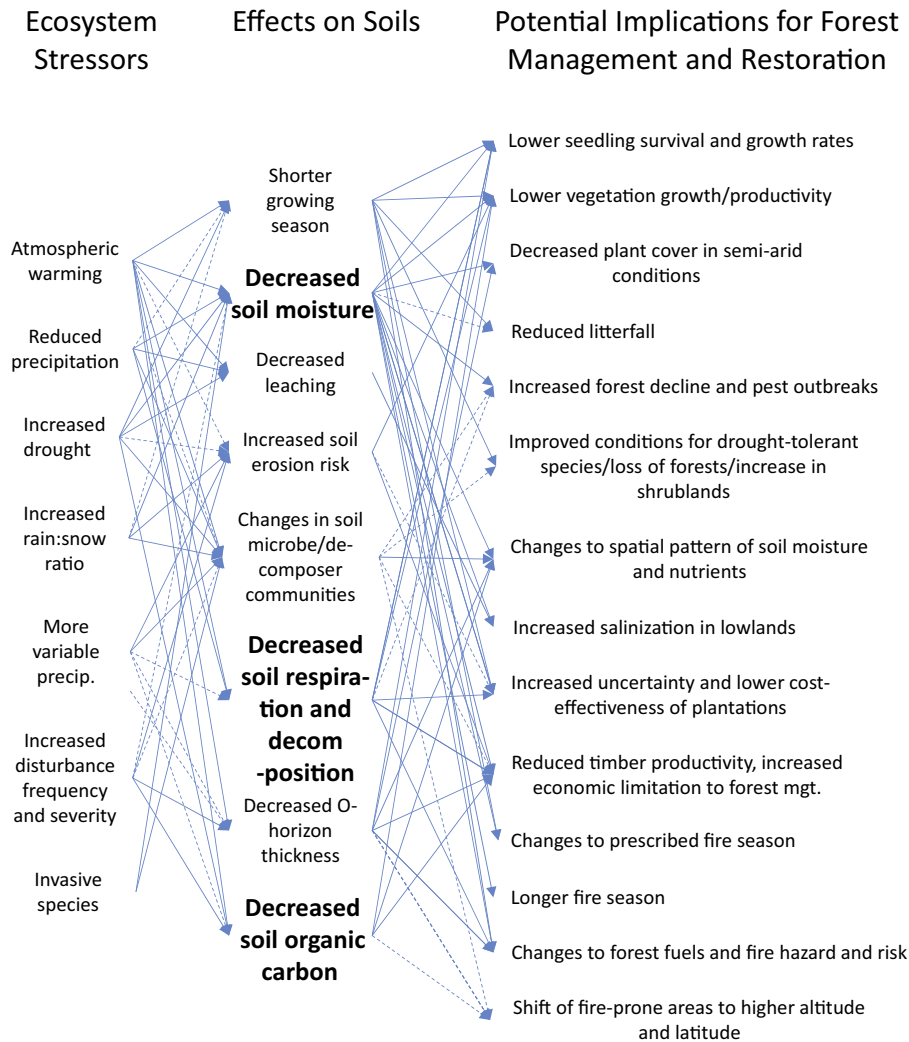


FIG. 12.10

Major global change-related stressors on MCR forests, their salient effects on soils, and implications — through soil pathways — for forest management and restoration practices. Most important soils effects in **bold**. Dotted lines are indirect effects. Direct impacts of global change stressors on management and restoration are not depicted, nor are feedbacks from management and restoration practices on soils or stressors, or interactions between factors within categories (but see discussion in text).

Conclusion

Some of the salient features of the situation currently confronting boreal and MCR forests and their managers are summarized in [Table 12.1](#). Although the types and degrees of global change-related threats facing the two biomes differ in notable ways, the general principles of ecosystem management and ecological restoration provide a common framework to approach the challenges faced by forests in both biomes. In this short conclusion, we outline some of the more prominent issues that forest managers and restorationists will deal with as the 21st century progresses.

Global change stressors will have major effects on boreal and MCR forest ecosystems and their soils, but they will also impact the way we do management and restoration. The strong dependence of restoration efforts on NRV/NDE concepts comes immediately to mind, since global change trends are leading us away from the past, not back to it ([Millar et al., 2007](#)). That said, history provides the only insight we have into ecosystem processes at long time scales, and mechanistic understanding of

Table 12.1 Generalized summary of global change threats and issues in boreal and MCR forests.

Major distinctive features	Boreal	Mediterranean
Forest management and restoration framework	Mostly linked to reducing impacts of timber production. Focus on large landscape connectivity in North America, focus on microhabitat restoration (open-dry habitats, old-growth forest) in Fenno-scandia	Focused on water and soil conservation, maintenance and expansion of old forest habitat, resilience to climate change, reduction of fire risk, provisioning of other ecosystem services
Typical landscape	Large areas of continuous forest	Mostly complex matrix of forest patches (some large areas of forest) and intermixed land uses
Main climate change effects	Extreme warming, increasing moisture	Drying, moderate warming
Main threats (1)	Direct effects of climate change on habitat conditions and C cycling, strong positive feedbacks to climate warming	Decreased and less reliable soil moisture, aridification/desertification
Main threats (2)	Unsustainable, large-scale industrial timber harvest	Increased occurrence of severe, large-scale ecological disturbances (fire, insects, diseases)
Main threats (3)	Increased frequency and area of stand-replacing fires	Increased human population and demands for ecosystem services
Critical soils issues for restoration under climatic change	Warming soil temperature regime, loss of permafrost, increased freeze-thaw activity, increased soil biological activity	Decreased, spatiotemporally more variable soil moisture; loss of soil organic carbon
Technical feasibility of plantations in ecological restoration projects	High, owing to generally improved habitat and climate conditions	Low, owing to increased water stress and high disturbance probability (fire, pests)

relationships between biota, environment, disturbance and climate in the past will only benefit us in the future (Wiens et al., 2012). In most places, static reproductions of historical conditions are no longer a sustainable management option. However, given that human planning horizons generally extend only 10–20 years into the future, NRV/NDE based targets can provide useful and potentially attainable “waypoints” for management or restoration (Safford et al., 2012a), with desired endpoints subject to modification/update as socioecological conditions change over time. Other modulations in management thinking and practice will become necessary as global change progresses. For example, decreased soil moisture in MCRs is leading to higher baseline plant stress and increasing tree mortality from prescribed fire and managed wildfire, and may also increase postfire beetle attack success. Managers will need to adjust fire prescriptions and timing so as to remain within desired mortality ranges. Also in MCR forests, with higher interannual variability in precipitation forecasted, failed tree planting efforts will become more common. New improvements in short-term climate predictability may engender more planting success (Bradford et al., 2018), but planting in some years may need to be skipped altogether. In boreal forests, melting permafrost and more rain will lead to major complications with the road infrastructure and decrease accessibility for machinery to sites of active management and restoration. In both MCRs and boreal forests, the ongoing invasion of novel biota and pathogens will change the playing field multiple times between now and the end of the century.

Given that climatic conditions projected for the late 21st century are likely to be outside the realm of human experience, our reliance on “tried-and-true” methods in forest management and restoration may become progressively less effective and less justifiable. Given the magnitude of projected change, we are in dire need of some novel, “out-of-the-box” thinking. Even where traditional management and restoration methods are reasonably applied, some level of experimentation should be engineered into project plans. Environmental impact documents assume we can reliably predict the long-term outcomes of proposed management actions, but under rapid global change that certainty is a fantasy. Rather than selecting a single management response from many, we suggest that several management responses be employed, with the best option applied to most of the landscape, and less supported — but still potentially viable — options and an unmanaged control area assigned to smaller segments of the landscape. This sort of bet hedging will require changes in the way that society, science, and forest management interact, but it is best suited for learning from unexpected outcomes, especially when scientific experiments can be embedded in the management plan (Gellie et al., 2018). Just as importantly, effectiveness monitoring and adaptive management must become part of the DNA of government, industry, and private efforts to manage resources. Although they have been talked about for decades, neither effectiveness monitoring nor adaptive management have experienced broad or consistent implementation in ecosystem management or restoration (Suding, 2011; Allen and Garmestani, 2015). Without these two foundation stones, the entire process of “learning by doing” is compromised and future changes in course are made more difficult or even impossible.

Ecological heterogeneity is one of the key concepts underlying current thinking in ecosystem management and ecological restoration in boreal and MCR forests (Kuuluvainen, 2002; North et al., 2009). There are various well-documented reasons for promoting heterogeneity in ecological systems (see text and Box 12.1), and local and regional efforts to increase ecological heterogeneity are underway in both biomes. One important question is the extent to which managers should use active versus passive tactics to promote heterogeneity and other desired ecosystem states. Active tactics (cutting, burning, digging, moving, spraying ...) can effect more rapid and predictable change and can be focused on specific habitats or species, but are more expensive in the short term, are rarely of sufficient scale to

solve major environmental problems, and often encounter public resistance. Passive tactics (hands-off management, let-burn policies ...) are relatively cheap and can “treat” large landscapes when permitted, but outcomes are less predictable, species in need of special habitats or sensitive management may suffer, and undesirable ecosystem trajectories may simply be reinforced. In general, the different spatial scales of the boreal and MCR forests and the different degrees of threat they experience make passive management more likely in the former and active management more likely in the latter, but in both biomes there is plenty of opportunity for both approaches.

Human management of ecosystems that is focused on reducing the impacts of climate change (rather than its causes) is referred to as “climate change adaptation”. Management recommendations in the climate change adaptation literature are largely based on the general principles of ecosystem management and ecological restoration. Below, we provide a collated list of such recommendations for forested ecosystems, drawn from [Blate et al. \(2009\)](#), [Heller and Zavaleta \(2009\)](#), [Littell et al. \(2012\)](#), [Safford et al. \(2012a,b\)](#), and [Schwartz et al. \(2012\)](#).

- Increase landscape and habitat heterogeneity
- Maintain biological diversity, attempt to ensure some level of ecological redundancy
- Develop corridors/habitat connectivity for species migration and habitat protection
- Mitigate non-climatic threats (e.g., deforestation, land use change, invasive species, uncharacteristic disturbances, etc.)
- Implement active or passive forest treatments that restore resilience at large spatial scales (e.g., reduce forest density and homogeneity, reduce drought stress and fire risk)
- Update genetic guidelines and seed zones for reforestation
- Only carry out assisted migration/managed relocation of species where the major ecological, legal, political, and ethical questions have been asked and answered
- Where feasible, better align habitats with future conditions by proactively applying forest management tactics from warmer and drier locations in the vicinity; best implemented in experimental fashion
- Where possible, permit natural disturbance processes to operate at natural rates and severities
- Treat large-scale disturbance as a management opportunity and integrate it in planning
- Incorporate ecosystem services into management plans and objectives, more explicitly consider ecosystem processes and the physical habitat template (e.g., geology, soils, topography, the stream network, etc.) in conservation planning
- Use historical data to develop a mechanistic understanding of ecosystem processes and link to future projections
- Use historical reference conditions as “waypoints” rather than endpoints
- Match infrastructure engineering to expected future conditions
- Promote public and employee education and awareness about global change
- Develop collaborative adaptation strategies and ecoregional management plans with stakeholders

Both boreal and MCR forests and their soils will experience major changes over the next 50–100 years. Some of these changes are unavoidable, but some can be prevented, and in both cases ecosystems can be made more resilient. Management tactics will evolve over time, but the general goals of ecosystem management and ecological restoration — integration of spatial and temporal scales; biodiversity conservation; promotion of ecological integrity; provision of ecosystem services;

development of ecosystem resilience; learning by doing; and explicit consideration of the needs and desires of man — will continue to be relevant even as the world changes underfoot.

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