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Disturbance and Sustainability in Forests of the Western United States

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Editors

Tara M. Barrett is a research forester, Pacific Northwest Research Station, 1133 N Western Avenue, Wenatchee, WA 98801; **Guy C. Robertson** is national program leader for sustainability assessment, Research and Development, 201 14th Street SW, 2nd floor, Washington, DC 20250.

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Disturbance and Sustainability in Forests of the Western United States

Tara M. Barrett and Guy C. Robertson, Editors

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Abstract

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This report assesses recent forest disturbance in the Western United States and discusses implications for sustainability. Individual chapters focus on fire, drought, insects, disease, invasive plants, and socioeconomic impacts. Disturbance data came from a variety of sources, including the Forest Inventory and Analysis program, Forest Health Protection, and the National Interagency Fire Center. Disturbance trends with the potential to affect forest sustainability include alterations in fire regimes, periods of drought in some parts of the region, and increases in invasive plants, insects, and disease. Climate affects most disturbance processes, particularly drought, fire, and biotic disturbances, and climate change is expected to continue to affect disturbance processes in various ways and degrees.

Keywords: Disturbances, climate change, vegetation, pathogens, wildfire, resilience.

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Chapter 1: Introduction

Guy C. Robertson[1]

The Issue

Citing a threefold increase in insect-induced tree mortality and increases in fire extent and severity, the U.S. Forest Service's 2010 "National Report on Sustainable Forests" identified forest disturbance processes as a major cause for concern, maintaining that "in many areas, particularly the West, confronting the complex causes and effects of disturbance will remain a dominant, if not the dominant, consideration in forest management and policy for the foreseeable future" (USDA FS 2011). Since then, with extensive tree mortality resulting from the mountain pine beetle (*Dendroctonus ponderosae*), and with wildfires of increasing intensity and lethality appearing in the news every year, forest disturbance processes have emerged as a central issue confronting forest managers, rural and urban communities, and the public at large in the Western United States. The most visible and destructive disturbances often entail large expenditures of public funds for fire suppression, compromise the quality of forest-based public goods ranging from clean water to biodiversity conservation, threaten the health and lives of residents, and lead to extremely large post-disturbance outlays. Moreover, because forest disturbance is often a pathway to fundamental ecosystem change (Hessburg et al. 2019, Johnstone et al. 2016), we are confronted with the possibility that large swaths of insect-killed trees or burnt-over forests are a harbinger of radical change in forest structure, composition, and extent at the regional scale, especially if anticipated changes in climate exacerbate the frequency, severity, and extent of these disturbances.

Large swaths of insect-killed trees or burnt-over forests could be a harbinger of radical change in forest structure, composition, and extent at the regional scale.

Describing current conditions and recent trends in the Western States for forest disturbances—drought, fire, insects, diseases, and invasive species—is the central aim of this report. Numerous studies have addressed specific disturbance processes at various spatial scales, and a smaller number of reports have considered the combined effects of multiple disturbance processes at regional or national scales (e.g., Masek et al. 2013), but few have provided the breadth and depth of information presented here for the Western States. In doing so, we attempt to answer two basic questions: (1) Are forest disturbances increasing in frequency, severity, and extent? and (2) How will disturbances affect forest sustainability now and in the future?

The first question can readily be affirmed in recent years, at least as regards fire, a disturbance category for which we have the most extensive statistics. Note, however, that fire frequency and extent were considerably higher in the more

[1] **Guy C. Robertson** is the national sustainability assessment program leader, Research and Development, 201 14th Street Southwest, Washington DC 20250.

distant past, before the advent of fire suppression, than we currently experience. Also evident is increased biotic disturbance from insects and disease, the most extensive disturbance type in the West, although the historical record needed to identify long-term trends for insects and disease is not as well established as that for fire. Drought and invasive species are likewise increasing in extent and intensity, but they are often considered as contributing factors to forest disturbance rather than as proximate causes, and as such are not directly comparable to the tree damage and mortality statistics reported for fire, insects, and diseases. Although we can determine recent trends in disturbance activity, the identification of longer term trends is hampered by a lack of adequate historical data that are comparable across time and space for many disturbance processes. Moreover, determining the ultimate causes and resulting implications of these recent increases is more difficult.

It is tempting to identify recent increases in disturbance activity as undesirable deviations from a largely stable set of forest conditions historically prevalent in the region, but this interpretation faces several challenges. The first arises from the dynamic nature of forest ecosystems and the role disturbance plays in them. Most of the forest disturbance processes considered in this report are endemic to the region, having been present in relative degrees for thousands of years, over which time they have been essential to the development of the forests we see today. Moreover, human activity, mainly in the form of intentionally setting fires and then more recently suppressing them (Ryan et al. 2013), has shaped the region's forests for centuries, so that a natural (or non-anthropogenic) baseline is neither evident nor (perhaps) appropriate to apply. This fact confounds the determination of normal or desired amounts of disturbance activity as well as the identification of underlying causes, be they climate change, a legacy of past management, or other factors. Another confounding factor is that levels of disturbance types demonstrate a high degree of variability from year to year and over longer timespans, and this variability similarly extends over space, precluding the identification of clear signals of deviation, at least from the limited set of consistent statistics we currently possess. Another challenge arises from the complexity of forest disturbance. The disturbance types considered here, and their respective impacts, are extremely heterogeneous, as are the landscapes in which they occur. This is particularly the case in regard to insects, diseases, and invasive species but also pertains to fire in terms of severity and burn pattern, complicating the aggregation of statistics across disturbance type and space. Finally, many of the disturbance types active in the region are closely interrelated, adding to their complexity and complicating analysis and the attribution of cause.

Nonetheless, we can say much about specific disturbance agents and disturbance in general, mainly through the synthesis of many disparate pieces of information rather than through formal statistical analysis of aggregate quantitative data. Also, considerable progress has been made in determining historical ranges of variation (HRV) that can serve as a replacement for the overly simplistic concept of a static baseline (Keane et al. 2009, Wiens et al. 2012). This technique, often relying on modelled simulations of past conditions and fire regimes of specific forest types, has shed a great deal of light on the role of disturbance in shaping the forests in the West (Keane et al. 2003, Perry et al. 2011). These approaches often take the form of one-off studies with varying underlying assumptions and modelling techniques, but in regard to fire, at least, a general consensus has emerged that fire was at one time much more prevalent in the forested landscapes of the West and that fire suppression has significantly altered forest structure throughout the region, resulting in increasing fuel loads and decreasing resilience to fire and other disturbances (Hessburg et al. 2019).

Fire suppression has significantly altered forest structure throughout the West.

The second question listed above considers "sustainability" and thus invokes the complexities and diversity of human values implied by that term, especially when applied to social-ecological systems as complex and dynamic as forests in the West.[2] Accordingly, the definition for sustainability used here is quite flexible. In particular, we are not looking at sustainability as a set of static thresholds that need to be maintained so much as an admonition to think broadly in terms of the implications of disturbance across the full range of values, services, and characteristics associated with forests both now and in the future (see Singh et al. 2009 for a review of sustainability assessment techniques). Whereas the "National Report on Sustainable Forests" and associated reports (McGinley et al. 2017; USDA FS 2011, 2016) have relied on the full set of the Montréal Process Criteria and Indicators for Sustainable Forest Management (Montréal Process 2015), the bulk of analysis in this report focuses on a single criterion—Criterion 3, with its two indicators on biotic and abiotic disturbance, respectively, though direct reference to these indicators is limited. On the other hand, our treatment of the social and economic impacts of disturbance bridges Criterion 3 and Criterion 6, which includes 20 indicators devoted to social and economic aspects of forests. Our treatment of these aspects, however, is only partial, accomplished mainly by including a single chapter on

[2] "Resilience" is another concept that could have been considered in lieu of "sustainability," but resilience entails its own set of complications and is less focused on human values than on system attributes. See Seidl et al. (2016) for application of resilience concepts to forest disturbance management, and Bone et al. (2016) for discussion of forest resilience in the context of U.S. Forest Service policy and management.

social and economic impacts. Given the potential range and intensity of disturbance impacts in the region, there is still much to be written, discussed, and discovered about this topic.

The Study Area

The study region covers broad ranges of latitude and longitude and includes numerous ecoregions arrayed across extreme temperature and moisture gradients, encompassing a large and diverse set of forest types.

This report considers forest-related disturbance processes in the Western United States. The region covers broad ranges of latitude and longitude and includes numerous ecoregions arrayed across extreme temperature and moisture gradients, and it encompasses a large and diverse set of forest types. The inclusion of Alaska further complicates matters in that it is separated from the contiguous states, and its relative size tends to skew aggregate statistics for the region, especially in high fire years. However, most Western States share a number of characteristics. In particular, they have large areas of public lands managed by federal and state public entities, and they have large areas of sparsely populated and relatively undeveloped lands subject to disturbance. Where human populations are more concentrated, surrounding forest lands are often highly valued by residents for the numerous ecosystem outputs and services they supply. Owing to their beauty or other attributes, certain locations attract visitors from around the nation and the world. In many areas, human settlement is broadly interspersed with forests and trees, and in others, commercial forest management for wood products production prevails. Forest disturbances will directly affect these values, and in the case of fire, they can threaten residents' lives and property. All these factors combine to make the management of forest disturbance an issue of focused concern and heated public policy debate throughout the region.

The National Context

The relationship between broad-scale national statistics on forest disturbance on the one hand, and finer scale or more anecdotal information on disturbances in the West on the other, was a motivating factor in the creation of this report. The statistics on disturbance cited in the 2010 "National Report on Sustainable Forests" as cause for concern were aggregated at the national level, but events in the West were the major drivers behind those aggregate statistics.

For example, the mountain pine beetle (MPB) (*Dendroctonus ponderosae*), an insect native to the West, was responsible for a significant proportion of the increase in insect mortality noted at the national level in the 2010 national report. Figure 1.1 shows annual totals of forested acres in the United States with tree mortality caused by insects and disease. It depicts the same metrics used in the 2010 national report, except in this case they are extended to 2017. Year-on-year variation is quite

high, with levels above 12 million ac (4.8 million ha) occurring three times in the 2000s (a figure well in excess of total acres burned by fire annually). However, in subsequent years, the sharply upward trend identified in the 2010 national report has effectively reversed itself, exhibiting an approximate 50 percent drop in mortality since peak years in the 2000s. At the same time, the 2015 level of 6 million ac (2.4 million ha) is nearly three times the levels reported in the late 1990s (composite measures for mortality are not available for prior years). The MPB accounted for 22 percent of total mortality in 2015 at the national scale and fully 75 percent in 2009 (USDA FS 2010, 2017); declines to 2014 in overall mortality shown in figure 1.1 generally reflect the MPB outbreak as it ran its course, reducing susceptible populations of its host trees in the process. Increases in 2015–2017, on the other hand, are

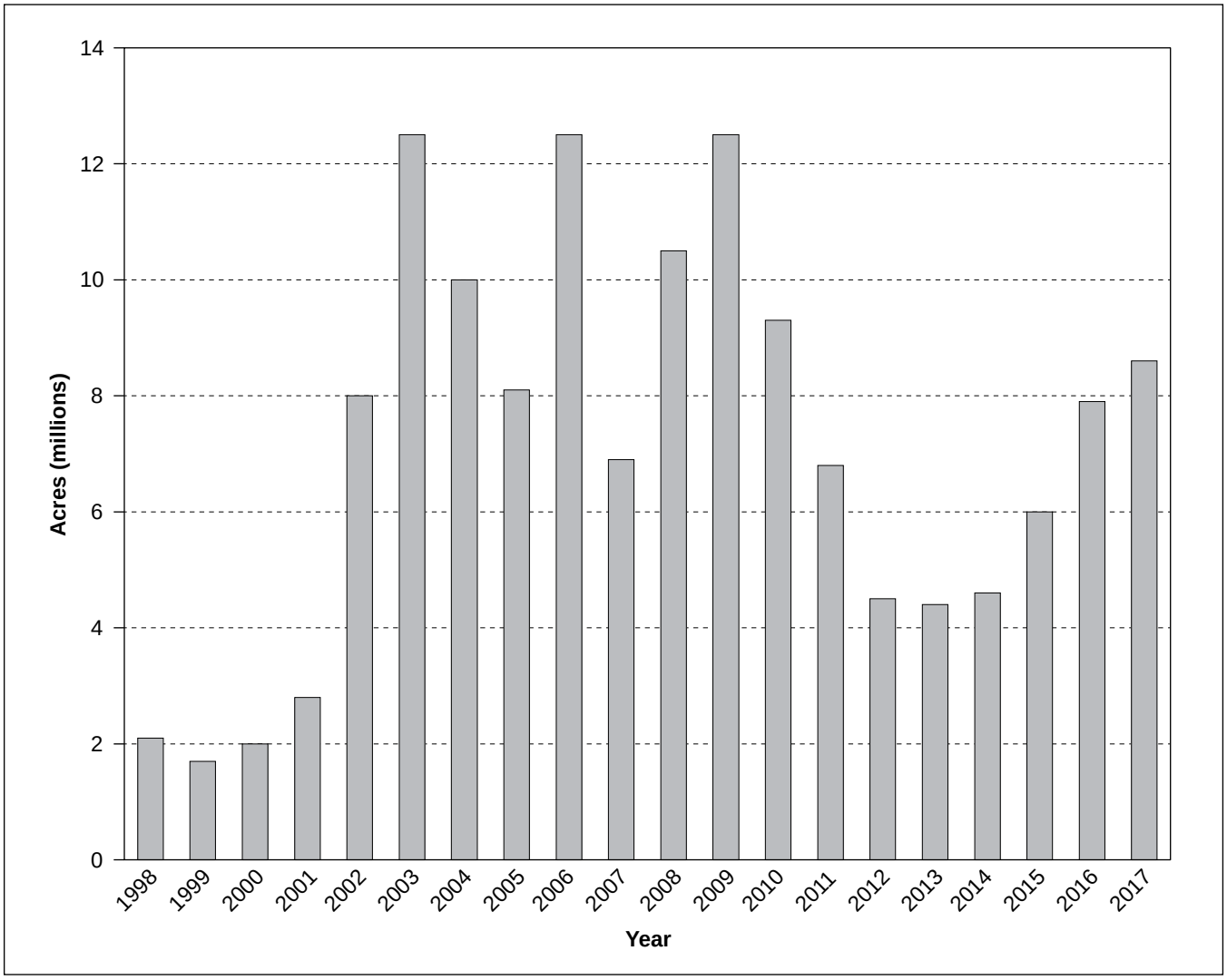

Figure 1.1—Total acres with insect and disease-induced tree mortality in the United States, 1998–2017. (U.S. Department of Agriculture, Forest Service, Forest Health Protection updated from USDA FS 2017).

due in part to expanded activity on the part of another native beetle, the fir engraver (*Scolytus ventralis*), primarily in California, again demonstrating the role of western disturbance in driving national statistics.

Total acres of land burned by fire for the nation as a whole are shown in figure 1.2, using the same statistics as in the 2010 national report but extended to 2018. The figure shows the history of fire extent, both in forests **and** in grasslands, in the United States since the inception of major fire suppression efforts. The reported totals are less reliable the earlier they fall in the historical record, but in terms of simple order of magnitude, they underline two important points: (1) at one time, fire played a

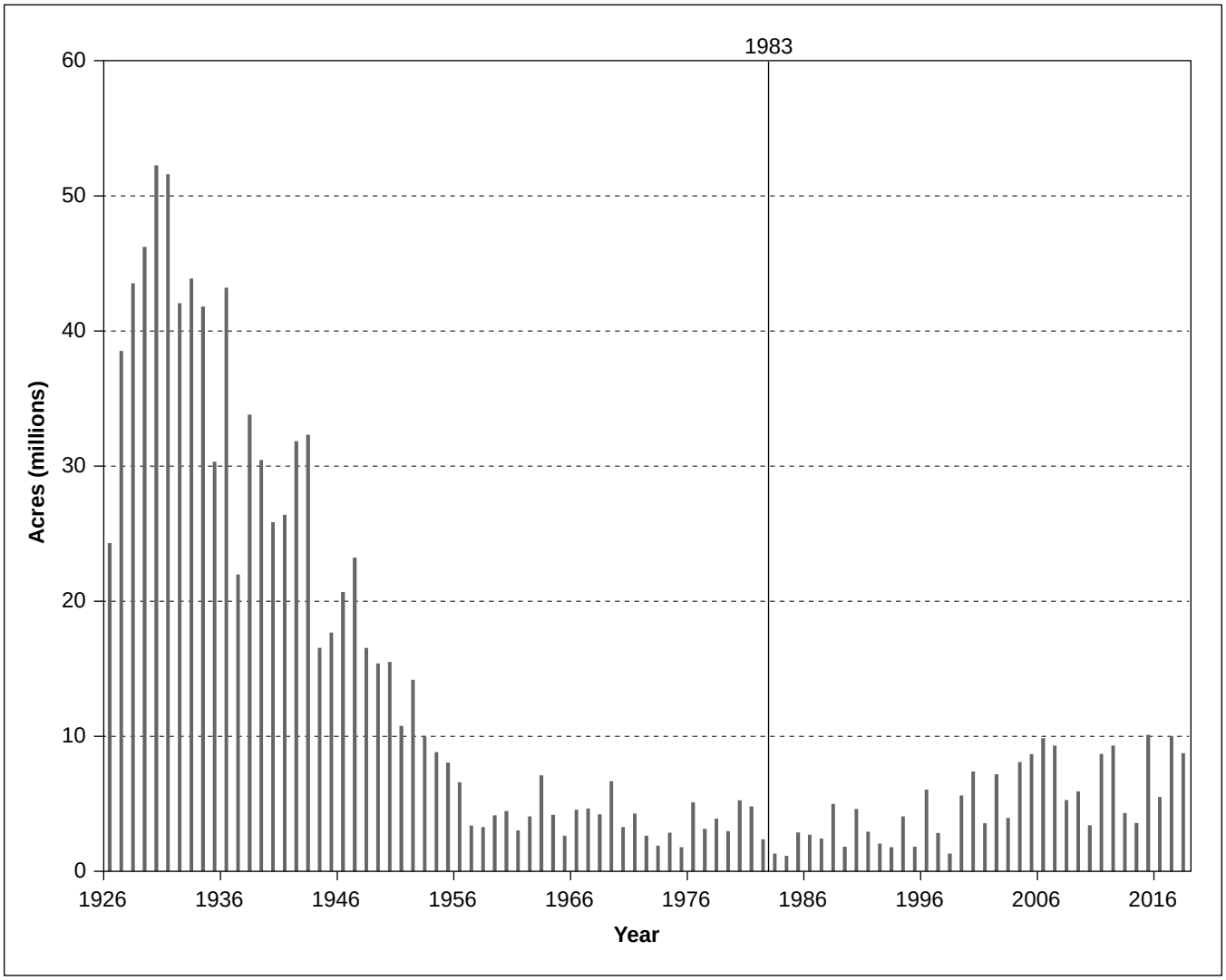

Figure 1.2—Wildland fire in the United States: total acres burned from 1926 through 2018 (includes nonforest acres). Note that the National Interagency Fire Center maintains that "prior to 1983, sources of these figures are not known or cannot be confirmed and were not derived from the current situation reporting process. As a result, the figures prior to 1983 should not be compared to later data." Years prior to 1983 are shown here as general reference. NIFC 2018.

much more significant role in ecosystems at the national scale than it does today, and (2) fire suppression to date has been remarkably successful. In line with this success, we can expect to find significant changes in forest composition and structure relative to pre-suppression conditions. Indeed, a number of studies have documented both the past prevalence of fire in western forests and the resulting changes to these forests from fire-suppression activities, particularly in the drier forests of the interior West (e.g., Allen et al. 2019, Hessburg et al. 2005). This legacy of past management will continue to act as a central driver of disturbance activity in coming years.

Allowing for year-on-year variation, fire extent in the past 20 years is approximately twice that experienced in the 1960 to 2000 period. In the case of forest fires, the vast majority of fire activity originates in the Western and Southern States, with increases in both regions contributing to the recent uptick. However, when viewed in terms of fire intensity, high- and moderate-intensity fires occur almost exclusively in the West.

Figures 1.1 and 1.2 also demonstrate some of the challenges in determining baselines for analysis and detecting departures from historical norms. Both figures display high year-on-year variation. Interpretation of insect and disease activity in figure 1.1 is hampered by a lack of historical data; that the observed changes in mortality are largely driven by a single insect species should give us pause when trying to analyze or project aggregate trends for insect activity as a whole. The fire statistics show a longer history, but the decrease from the extremely high levels reported for the early decades of the 20th century point to a major shift in fire patterns that was undoubtedly the result of anthropogenic forces (i.e., the institution of major fire-suppression efforts and, perhaps, a shift away from broad-scale fire setting in the 1800s), complicating the identification of a "natural" baseline for comparison from these aggregate statistics.

Roadmap for the Report

To serve as a summary, chapter 2 of this report presents a synthesis of findings. In addition to identifying key findings, the chapter discusses interactions between different disturbance types, an essential aspect of disturbance as a whole that is often obscured by separate monitoring and analysis of specific forest disturbance processes.

Chapter 3 provides an analysis of general disturbance data from the U.S. Forest Service's Forest Inventory and Analysis (FIA) program. The FIA data provide measures of forest area damaged by general disturbance categories and thus give us a unique view of disturbance that is comparable across disturbance types. However, the measurement of damaged area and assignment of disturbance cause are undertaken in the course of broader forest inventory activities occurring at

10-year intervals in the Western States, posing several challenges for measurement. Moreover, the FIA data currently lack historical observations sufficient to develop meaningful time series to examine departures from past disturbance behavior.

The next five chapters of the report are devoted to specific disturbance agents, starting with fire (chapter 4), which draws mainly on data from the National Interagency Fire Center to describe fire extent and severity across the region, and drought (chapter 5), which uses temperature and precipitation along with several drought indices to examine recent climate in the study area. Chapters 6, 7, and 8 discuss insects, disease, and invasive species, respectively. Each of these chapters lists and describes the various organisms affecting forests in the region. Perhaps the most striking aspects throughout are the diverse pathways, life cycles, interactions, and effects that these various disturbance processes display.

Finally, chapter 9 addresses socioeconomic effects of disturbance in the Western States, describing some of the many ways that disturbances affect the outputs, ecosystem services, and characteristics of forests that people value (chapter 9). The full list of these benefits is long and difficult to specify completely, and chapter 9 is indicative rather than exhaustive, relying on several case studies of specific disturbance effects along with descriptions of broad categories of impact.

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Chapter 2: Summary of Disturbances and Sustainability

Tara M. Barrett, Christopher J. Fettig, Susan J. Frankel, Andrew N. Gray, I. Blakey Lockman, David W. Peterson, Guy C. Robertson, and José J. Sánchez[1]

Introduction

The 2010 "National Report on Sustainable Forests" (USDA FS 2011) identified forest disturbance as the most immediate threat to forest sustainability in the United States, based on increased levels of insect activity and fire activity relative to levels reported in the 2003 edition of that report. This concern, combined with the prominent role that western forests play in national statistics, was the motivating factor behind this report. Key questions for this project were, are forest disturbances increasing, and will they affect forest sustainability in the Western United States? A simple answer to these questions is "yes." A more complete answer would be "yes, but in a complicated way."

The answer is complicated because disturbances are part of forest ecosystems, and the complex web of interactions between species and their environment has been shaped over millennia by disturbance processes, and more recently by anthropogenic disturbances, including fire use in the pre-European settlement era and fire suppression over the past century (e.g., Hessburg and Agee 2003). It also is complicated because the values and uses that people ascribe to forests are diverse and sometimes contradictory. Social and economic effects of forest disturbances are not evenly distributed, and even disturbances that cause substantial tree mortality can bring some benefits along with costs. Finally, it is complicated because of the heterogeneous and highly variable nature of forest disturbances over time, being the sum of numerous different events and biophysical processes. That said, in this report we highlight recent disturbances that are having the most impact on western forests and the communities that depend on them. We also provide some baseline estimates of disturbance amounts and trends where data were available.

Using a sample of LandSat data from 1985 to 2012, Cohen et al. (2016) estimated annual disturbances rate for the Western United States that ranged from near zero to

[1] **Tara M. Barrett** and **David W. Peterson** are research foresters, Pacific Northwest Research Station, 1133 N Western Avenue, Wenatchee, WA 98801; **Christopher J. Fettig** is a research entomologist, Pacific Southwest Research Station, 1731 Research Park Drive, Davis, CA 95618; **Susan J. Frankel** is a biologist, Pacific Southwest Research Station, 800 Buchanan Street, Albany, CA 94710; **Andrew N. Gray** is a research ecologist, Pacific Northwest Research Station, 3200 SW Jefferson Way, Corvallis, OR 97331; **I. Blakey Lockman** is a regional forest pathologist, Pacific Northwest Region, State and Private Forestry, 1220 SW Third Avenue, Portland, OR 97204; **Guy C. Robertson** is the national sustainability assessment program leader, Research and Development, 201 14th Street Southwest, Washington, DC 20250; and **José J. Sánchez** is a research economist, Pacific Southwest Research Station, 4955 Canyon Crest Drive, Riverside, CA 92507.

About 22 percent of forests in the Western U.S. were affected by disturbance within a five-year time window.

9.61 percent per year and found a trend of increasing disturbance from 1985 to the early 2000s. According to the U.S. Forest Inventory and Analysis (FIA) disturbance protocol for data collected from 2004 to 2016, disturbances in the West affected an average of 22.3 percent of forest lands per 5-year time window. This protocol features a relatively low threshold, recording disturbances that cause damage or mortality to 25 percent of trees in a stand or 50 percent of any individual tree species. Diseases and insects were by far the most common disturbance types using this threshold. Disturbance was highly variable both spatially and temporally, with recent (2004–2016) FIA data indicating that California, Idaho, Montana, Oregon, and Washington each contained more than 2 million ha of disturbance within a 5-year window.

One of the primary motivating questions for this report was "Are disturbances in western forests increasing?" Although FIA disturbance data have not been collected long enough to provide trend information, we do know from a variety of sources that invasive plants, insects, and pathogens are increasing over time. We also know that as the world's climate continues to warm, drought is likely to increasingly affect forests of the Western States (Vose et al. 2016, 2018). Native pathogen and insect trends vary because of complicated relationships with hosts and the environment, and some species are difficult to monitor. Fire is the disturbance with the best and longest historical data records, which also show a complicated temporal pattern.

Fire

Fire disturbances have increased in western forests in recent decades, as increased atmospheric aridity and regional droughts have interacted with high forest stand densities and fuels to increase fuel and forest flammability (Abatzoglou and Williams 2016, Holden et al. 2018). The average area burned in wildfires in the United States has more than doubled in recent decades, with average annual area burned in larger wildfires increasing from 1.2 million ha per year for the decade from 1985 to 1994 to 2.7 million ha for the decade from 2008 to 2017 (NIFC 2018). The area burned at moderate and high severities has similarly increased over this period, as the average proportion of area burned at low, moderate, and high severity has remained relatively stable over time (see chapter 4). Despite these recent trends in annual area burned, forest area burned remains well below historical levels prior to the presettlement and presuppression eras, likely owing to active fire suppression efforts (Marlon et al. 2012, Parks et al. 2015). Western forest area affected by fire remains less than that affected by insects, diseases, timber harvests, or grazing, as wildfire affected only 3.5 percent of western forests per 5-year window using the FIA threshold, but fire typically has the highest per-hectare management cost.

Average annual area burned by wildfires has more than doubled in recent decades.

Wildfire area burned is strongly influenced by climate in both forested and nonforested ecosystems of the Western United States. Wildfire area burned is typically driven by climate and seasonal fire weather. Influential factors include fire season length, fuel moisture levels during the fire season, and the number of days with hot, dry, and windy weather (Abatzoglou and Kolden 2013, Littell et al. 2009). Much of the trend in wildfire area burned over the past 30 years can be attributed to observed trends in elements of climate associated with fire season length and fuel aridity (Holden et al. 2018, Jain et al. 2017, Jolly et al. 2015).

Economic costs associated with wildfire have increased in recent decades along with area burned. In recent decades, steady growth of the wildland-urban interface (WUI) in the West has increased the number of people living close to fire-prone wildlands, thereby increasing the number of human-caused wildfires (Balch et al. 2017, Nagy et al. 2018); putting more lives and property at risk; and increasing social and political pressure to maintain active and aggressive fire-suppression policies (Parisien et al. 2016). Suppression costs have been rising rapidly in response to increased wildfire area burned (Calkin et al. 2005), often reducing resources available for forest restoration and other fuel reduction activities.

Drought, Insects, and Diseases

Drought can increase the probability and severity of wildfires, as well as be a direct cause of tree mortality when drought is severe or prolonged. In recent years (2006–2015), average annual temperature has been higher than the preceding 30-year average (1975–2005) across the Western States, but below-normal precipitation was concentrated in the Southwest (Arizona, California, and Nevada), resulting in spatially variable drought severity across the West.

Higher than normal temperatures in recent years interacted with below-normal precipitation in some areas to increase drought severity, particularly in the southwestern United States.

Drought can also make forests more susceptible to the effects of insects and pathogens. Insects are essential components of western forests, representing most of the biological diversity and affecting virtually all trophic levels and processes. Although most insect species can be viewed as beneficial, others periodically become so abundant that they threaten current ecological, economic, social, or aesthetic values at local to regional scales.

Over the past 30 years, tree mortality caused by bark beetles in the West has exceeded tree mortality caused by wildfires, raising concerns about the sustainability of some western forests to provide certain goods and services over time. Most notably, mountain pine beetle (*Dendroctonus ponderosae*) affected ~10.3 million ha from 2000 to 2016, which represents almost half the total area affected by all bark beetles combined in the West during this period (Krist et al. 2014). Several wood-boring, defoliating, and sap-sucking insects are also important.

Mountain pine beetle impacted about 10.3 million hectares from 2000-2016.

Unlike in the Eastern States, few invasive insects are important disturbance agents in western forests currently. However, with observed and projected shifts in climate, human populations, and trade, invasive insects are expected to cause elevated impacts in the future. The recent establishment of goldspotted oak borer (*Agrilus coxalis*) in California (interstate introduction) and emerald ash borer (*A. planipennis*) in Colorado (intercontinental introduction) is of concern.

White pine blister rust, drought, and bark beetles are driving extensive mortality in high-elevation whitebark pine populations.

In contrast to invasive insects, exotic invasive pathogens, inadvertently introduced as a byproduct of trade, have already substantially affected western forests, and the number of detections continues to rise. Once introduced, pathogens are difficult to control; for example, white pine blister rust, (caused by *Cronartium ribicola*), entered the United States on nursery stock in the early 1900s and continues to spread and kill white pines today, despite extensive eradication efforts in the 1930s. Notably, white pine blister rust, along with drought and bark beetles, is driving extensive mortality in high-elevation whitebark pine (*Pinus albicaulis*) populations. A more recent introduction, *Phytophthora ramorum*, the cause of sudden oak death, was first recognized in the United States in the mid 1990s and has become established along parts of the central California coast into southwest Oregon, where each year it kills an average of more than a million tanoaks (*Notholithocarpus densiflorus*), coastal live oaks (*Quercus agrifolia*), and other oak species.

Native pathogens are also causing widespread impacts in western forests, especially root diseases (Armillaria root disease, Heterobasidion root disease, and laminated root rot [the latter is caused by *Phellinus sulphurascens* syn. *Coniferiporia sulphurascens* and *P. weirii* syn. *C. weirii*]).

Warmer and wetter conditions in some parts of the Western United States are causing outbreaks of foliage diseases.

Climatic influences on forest diseases are complex, as they are an expression of the interaction of the particular pathogen, host plants or trees, and other environmental conditions. Warmer and wetter conditions are causing outbreaks of foliage diseases such as Dothistroma needle blight (caused by *D. septosporum*) and Swiss needle cast (*Phaeocryptopus gaeumannii*). Warming is also responsible for yellow-cedar decline, an abiotic disease associated with a reduction in snow cover, which has caused root damage and mortality of yellow-cedar (*Callitropsis nootkatensis*) in southeast Alaska.

Changes in other forest disturbance patterns, such as drought and wildfire intervals, will change the impacts of pathogens and insects, creating novel forms of disruption to ecosystem services, as well as adding to safety concerns and management problems. Assisted migration and increased restoration plantings may introduce new pathogens to new areas. Tree mortality rates and growth responses will change over time, and pathogens and plant species will adapt to this change and to one another. If drought, fire, and other abiotic disturbances are extreme, the impact

of pathogens may decrease because there will be fewer host plants for microbes to infect, and the weakest organisms in the plant or tree population will have been removed by other mortality agents.

Invasive Plants

Invasive plants are also affecting forests of the Western United States, with impacts on native species diversity, nutrient cycling, the probability of other disturbances like wildfire, and the productivity of desired species. The costs—in value lost in degraded ecosystems and the price for management attempts to control invasive plants—are substantial.

Based on recent FIA data, the proportion of forest covered by invasive plants was highest in Hawaii (46 percent), higher on the Pacific Coast (3.3 percent) than in the Rocky Mountains (0.75 percent), and lowest in coastal Alaska (0.01 percent). Mean cover of invasive plants was related to stand age, with less invasive cover in older stands. Invasive plant cover was also highest in recently disturbed forests and forest types common to drier areas and sparse or open tree canopies. Invasive plants were significantly more abundant in forests that were in proximity to nonforest land uses or ecosystems (e.g., roads or rangelands). Overall, nonnative cover appears to have increased substantially from the 2000s to the 2010s, and further increases are expected.

Invasive plants as a percentage of forest was an order of magnitude higher in Hawaii.

Cheatgrass (*Bromus tectorum*) was by far the most abundant nonnative plant species in western forests, covering 1.2 million ac (500 000 ha), or 0.5 percent of all forested land in the conterminous Western States. Because cheatgrass and other annual grasses dry out earlier and more completely than native perennial vegetation, they promote fire spread and tend to increase fire frequency and extent.

Cheatgrass, which can increase fire spread, was by far the most abundant nonnative plant species in western forests.

Social and Economic Impacts of Disturbances

The increasing frequency and severity of forest disturbances pose significant challenges to the sustainable management of forests on public lands in the Western States and the variety of goods and services they provide (e.g., recreational opportunities, biodiversity protection, climate change mitigation). Estimates of the full economic impacts of forest disturbances could help inform forest management and policy under projected environmental and anthropogenic pressures. To illustrate this point, the economic impact of forest disturbances on outdoor recreation and climate mitigation was assessed in this report.

In the United States in 2016, about 900 million visits were made to federal lands, and visitors spent around $49 billion. The value of recreational activities to each national forest in the West is estimated to range from $11 million to more than

Disturbances affect outdoor recreation benefits through reduced participation, fewer available recreation days, and changes in the quality of outdoor amenities.

$1 billion. Forest disturbances affect outdoor recreation benefits through reduced participation, fewer available recreation days, and changes in the quality of outdoor amenities. For instance, on the Inyo National Forest, the 2007–2009 California drought affected downhill skiing in the mixed-conifer zone, with a 19 percent drop in annual visits between 2006 and 2011, and a corresponding reduction in recreational benefits estimated at $19 million.

Through their contribution to climate change, carbon dioxide (CO_2) emissions have a long-term impact on agricultural and forestry productivity, human health, energy and water security, and managed and unmanaged ecosystems. Recent vegetation and forest cover dynamics in the Western States have resulted in net increases in carbon stocks, mitigating emissions from other economic sectors. The analysis presented in chapter 9 shows that projected increases in net forest carbon are estimated to provide more than $11 trillion in social benefits by 2091. Nevertheless, CO_2 emissions owing to tree mortality from elevated disturbance regimes is still significant. Emissions from trees killed by fire and insect damage between 2003 and 2012 in the West are estimated to have social costs of about $26 billion. More effective management responses to forest disturbances could contribute to higher levels of carbon sequestration and a subsequent increase in social benefits.

Management and Monitoring

Although disturbances are embedded in the ecology of western forests, some management actions can be taken to reduce their impact. Reduction of insect, pathogen, and plant invasions would lessen degradation of western forests. Many options exist to address particular disturbances, such as prescribed burning and other fuel reduction treatments to increase resistance and resilience to fire; thinning to reduce tree mortality from drought; favoring non-host species to address particular insect or disease issues; or altering the timing and duration of recreational site visits. But it is important to recognize that the best choices will differ both by site-specific factors and by management goals. To reduce damage, management approaches should integrate local historical, current, and predicted risks and embed “best practices” into treatment strategies. Homeowner and community education programs, cooperative management and stewardship across multiple agencies and landowners, and local and regional planning can reduce the incidence and impact of disturbances. Because increasing temperatures usually exacerbate some of the most damaging disturbances (wildfire, drought, beetles), actions to ameliorate climate change, such as reducing greenhouse gas emissions or increasing carbon sequestration and storage, are also important.

Year-to-year variability is high for many disturbances, so consistency and duration of monitoring programs will be critical to understanding trends in disturbances. Better methods of integrating disparate data sources would help in gaining leverage from existing monitoring programs such as FACTS (Forest Service Activity Tracking System), Monitoring Trends in Burn Severity (https://www.mtbs.gov), FIA (https://www.fia.fs.fed.us), Insect and Disease Aerial Detection Surveys (https://www.fs.fed.us/foresthealth/aviation/aerialsurvey.shtml), and the National Visitor Use Monitoring program (https://www.fs.fed.us/recreation/programs/nvum/). Some causes of disturbances, notably diseases and pathogens, are only partially covered within existing monitoring programs, and alternative sources and types of monitoring would allow for a better understanding of impacts on forest growth and mortality so that changes can be recognized and addressed. For other disturbances such as drought, the development of better indices and prediction methods specifically tailored to impacts on tree growth and mortality would be helpful. Although we focused on the most recent data for disturbances and did not include predictive modeling, many of the disturbances discussed in this report, along with their associated socioeconomic impacts, are likely to increase in the next few decades given current trends in invasive species, climate change, and changing land use.

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Chapter 3: Overview of Recent Forest Disturbances in the Western United States

Tara M. Barrett and A. Lexine Long[1]

Background

This chapter provides a general overview of forest disturbance levels in the Western United States that relies primarily on data from the U.S. Forest Service's Forest Inventory and Analysis (FIA) program. Subsequent chapters examine specific disturbance types (fire, drought, insects, disease, and invasive plants) and incorporate additional data sources and scientific findings specific to each of these types.

Although disturbances have historically been an integral part of Western U.S. forests, there is concern about recent shifts in disturbance regimes, including alterations to the frequency, duration, and impact of disturbances. In addition, as forest disturbances typically interact, changes in one type of disturbance also alter other disturbances, creating a cascade of changes to forest ecosystems. Stand-replacing disturbances, such as high-severity wildfire, can change the type of forest (predominant tree species) found at a particular location, or even shift vegetation from forest to shrubland or grassland.

Disturbances have historically been an integral part of Western U.S. forests, but changes are occurring in the frequency, duration, and impact of disturbance regimes.

Extensive portions of the drier forest land in the West historically had high frequency but low- or mixed-severity fire regimes (Arno 1980, Swetnam and Baisan 1996). With the advent of effective fire suppression and alterations in forest management, many of these forests went through a multidecade period of reduced fire frequency that resulted in tree densification and a buildup in fuels (Agee 1993, Hagmann et al. 2014). Now the area burned by wildfire is again increasing; when this is combined with an expanding wildland-urban interface and a longer fire season, fire suppression costs increase substantially (see chapter 4).

In addition, high forest density results in greater individual tree stress and increased susceptibility to bark beetles (see chapter 6). Timber harvesting alters species composition and forest structure, changing susceptibility to other disturbances, sometimes negatively and sometimes positively. Invasive plants continue to increase in western forests (see chapter 8), with invasive grasses such as cheatgrass (*Bromus tectorum*) and North Africa grass (*Ventenata dubia*) increasing fire potential (D'Antonio and Vitousek 1992). New invasive diseases such as sudden oak death and white pine blister rust are substantially altering affected forest

[1] **Tara M. Barrett** is a research forester and **A. Lexine Long** was an ORISE research fellow, Pacific Northwest Research Station, 1133 N Western Avenue, Wenatchee, WA 98801; Long currently is an associate planner, City of Wenatchee, Wenatchee, WA 98801.

ecosystems (chapter 7), and recent drought in California provides an example of how disturbances can interact to cause extensive mortality.

Adding to these alterations in disturbance regimes is ongoing change in climate. Global temperature rise is expected to increase tree mortality substantially in parts of the Western States over the next 80 years (McDowell et al. 2015). In drier forests, a longer fire season and decreased snowpack (Mote et al. 2005), reduced summer precipitation (Holden et al. 2018), and higher temperatures (Abatzoglou and Williams 2016) have contributed to increased incidence of fire events. High-elevation and mesic forests in the West that had comparatively low-frequency historical fire regimes have been less affected by the forest densification and fuels buildup. However, changes in snowpack, precipitation, and temperature can still alter disturbance regimes in high-elevation and mesic forests (McKenzie et al. 2004). Moreover, disturbances function as a catalyst for forest ecosystem transitions in response to a changing climate (Thom et al. 2016).

The 2010 "National Report on Sustainable Forests" (USDA FS 2011) raised concerns about whether an increase in disturbances, particularly tree mortality from bark beetles, was affecting the sustainability of U.S. forests. Tracking disturbances consistently is difficult, as remote-sensing methods that are useful for tracking some types of disturbance (e.g., fire) do not work well for other types of disturbance (e.g., some types of disease). However, when the FIA program switched to a nationally consistent system in response to the 1998 Farm Bill (Gillespie 1999), the program began to collect field-level records of disturbance. In this chapter, we use those data to provide information on recent disturbance in the Western States. The strength of the FIA dataset is that it can be used to provide a baseline with a consistent collection methodology for multiple types of disturbance.

Methods

This chapter focuses on the analysis of FIA disturbance data, providing regional estimates of area of disturbance. These data come from on-the-ground measurements of forested field plots sampled as part of FIA's ongoing forest inventory activities (USDA FS 2017a). The sampling design consists of a tessellation of hexagons across all lands, with one plot randomly located in each hexagon. In the Western United States, a systematic sample of one-tenth of the plots are measured each year. The dataset used for this analysis contained information from 172,472 plots (49,715 forested) from 16 Western States (fig. 3.1). Data from 11 states (Alaska, Arizona, California, Colorado, Idaho, Montana, New Mexico, Nevada, Oregon, Utah, and Washington) consisted of a full 10 years of data, or about one plot per 2700 ha, while data from the remaining states consisted of 5 (Wyoming) or 6 (Kansas, Nebraska,

North Dakota, and South Dakota) years of data. Alaska data included only southeast Alaska, and compatible Texas data were not available. Inventory dates and numbers of plots by state are shown in table 3.1.

Disturbance was recorded on forested plots if mortality or damage affected 25 percent of individual trees in the stand or 50 percent of an individual tree species, or, for disturbances such as flooding or grazing, affected at least 25 percent of the vegetation or soil. Crews could record up to three disturbances per plot, with choices including insects, disease, fire, animal damage (including grazing), weather damage, vegetation damage (such as vines), and geologic damage. Disturbances were recorded if they had occurred since the date of the previous inventory, or, for initial installation of a plot, in the past 5 years, based on the judgment of the sampling crew. To create a uniform temporal window for observation of disturbance, we limited disturbances to those occurring within 5 years of the plot visit. Standard national FIA methods (Bechtold and Patterson 2005) were used to make estimates.

Figure 3.1—States included in the study area.

Separate observations are made of silvicultural treatments on FIA plots rather than including them in the disturbance protocol. Although some disturbance research includes only natural disturbance, remote-sensing-based assessment of disturbance (e.g., Masek et al. 2008 or Cohen et al. 2016) typically includes both natural and human-made disturbances. To allow comparisons of disturbance area estimates with results from recent remote-sensing-based research, we included silvicultural treatments with other human-caused vegetation disturbance and reported estimates for it in the category labelled human/silvicultural.

The accuracy of disturbance records for particular disturbance types is unknown. At times, it can be difficult to identify what type of disturbance has occurred, e.g., whether observed damage is from freezing or disease. Where multiple disturbances occur, such as insects and fire, choosing which to list as the primary or secondary disturbance may be difficult. Forests often recover understory vegetation quickly, so it is likely that disturbances such as grazing or low-severity fire are sometimes missed when the disturbance occurs several years before the field inventory. An analysis of repeatability in the first years (2000–2003) of FIA

Table 3.1—Recent forest disturbance for the Western United States

State	Plots	Inventory dates	Forest area	Disturbance area		Disturbance as percentage of forest	Top three disturbance groups		
				Area	SE				
	Number	*Years*	- - - - - - - - -	*Hectares*	- - - - - - - - -	*Percent*			
Alaska	9,057	2004–2015	6 226 000	530 000	33 000	9	Disease	Other[a]	Human[b]
Arizona	12,274	2006–2015	7 483 000	1 080 000	48 000	14	Fire	Insect	Human
California	17,674	2004-2015	12 926 000	3 960 000	85 000	31	Animal[c]	Disease	Human
Colorado	11,222	2006–2015	9 229 000	1 928 000	60 000	21	Insect	Disease	Human
Idaho	8,996	2006–2015	8 737 000	2 021 000	60 000	23	Insect	Fire	Disease
Kansas	8,868	2010–2015	1 023 000	446 000	27 000	44	Animal	Human	Fire
Montana	15,854	2006–2015	10 475 000	2 274 000	66 000	22	Insect	Fire	Disease
Nebraska	8,333	2010–2015	620 000	446 000	24 000	72	Animal	Human	Weather
Nevada	11,428	2006–2015	4 275 000	353 000	28 000	8	Fire	Disease	Insect
New Mexico	13,104	2008–2016	9 956 000	1 767 000	62 000	18	Insect	Animal	Human
North Dakota	7,622	2010–2015	326 000	145 000	17 000	44	Animal	Weather	Insect
Oregon	15,316	2005–2015	12 001 000	3 391 000	66 000	28	Human	Disease	Insect
South Dakota	8,302	2010–2015	789 000	485 000	25 000	61	Animal	Human	Insect
Utah	9,171	2006–2015	7 382 000	869 000	43 000	12	Insect	Disease	Fire
Washington	9,980	2004–2015	8 974 000	2 707 000	61 000	30	Disease	Human	Insect
Wyoming	5,271	2011–2015	4 246 000	977 000	58 000	23	Insect	Fire	Disease
Total	172,472	2004–2015	104 666 000	23 379 000	206 000	22	Insect	Disease	Human

Note: Disturbances were recorded only if a significant threshold was reached: mortality or damage to 25 percent of all trees or 50 percent of an individual species, or, for disturbances such as flooding or grazing, affecting at least 25 percent of the vegetation or soil. Estimates are based on disturbances that occurred within a 5-year window of the year the inventory took place.

[a] "Other" includes disturbance that does not fit into other categories, such as yellow cedar decline, as well as disturbance where it is unclear if the agent is insects, disease, or abiotic.

[b] "Human" includes timber harvest and other silvicultural or vegetation management.

[c] "Animal" includes domestic grazing as well as damage from wildlife.

data collection found that independent crews agreed on whether disturbance had occurred 79 to 86 percent of the time and agreed on whether silvicultural treatment had occurred 87 to 98 percent of the time (Pollard et al. 2006).

Estimated area of disturbance in this chapter is for a 5-year window, to correspond to the measurement protocol. Annual rates cannot be derived by dividing by five. This is because some types of disturbance, such as grazing and disease, would typically occur only for multiple years within that 5-year window. Other types of disturbance, such as fire or weather, are most likely to only occur on the plot once during a 5-year window. Crews recorded a code to indicate whether a disturbance occurred over multiple years but not the specific years in which the disturbance occurred. To illustrate annual variation, we modelled annual disturbance by assuming that the multiple-year code indicated 2 years of disturbance. Model results are

used to depict annual variation for different types of disturbance, but the model would be less accurate in estimating annual disturbance rates or amounts, particularly for chronic disturbances, than would annual measurements.

Results

In the Western United States within a 5-year window, disturbances affected 22.3 percent of forest land, or 23.4 million ha.

Natural and human-caused disturbances affected 23.4 million ha (non-overlapping) in the Western United States within a 5-year window. This constitutes 22.3 percent of forest land in the study area, though it is important to note that the disturbances recorded had highly varied levels of impact. Overall disturbance rates per 5 years was highest in Nebraska (72 percent), South Dakota (61 percent), Kansas (44 percent), and North Dakota (44 percent); all four of these states had animal-damage/grazing as the most common disturbance in forest land (table 3.1). Alaska, Arizona, Nevada, and Utah had the lowest 5-year rates of disturbance, at 9, 14, 8, and 12 percent of forest, respectively. By area, the five states with the largest amount of disturbed forest were California, Idaho, Montana, Oregon, and Washington, each with more than 2 million ha of forest disturbances within a 5-year window. Although only a small portion of Alaska has been inventoried, it is likely that state totals are higher (see inset).

The most common disturbances in forests in the West were insects and disease, each estimated as occurring on 6.1 million ha, followed by human disturbance (including silvicultural management) at 4.4 million ha. Animal-damage/grazing occurred on 3.9 million ha within a 5-year period, with a greater area of disturbance from domestic animals than wildlife (table 3.2). Although fire is the most economically costly forest disturbance, it is not the most common one (fig. 3.2) and was estimated to have occurred on 3.7 million ha of forest within a 5-year window. Not all burned plots had separate records for ground and crown fire; where it was recorded, ground fire was more frequent (table 3.2). Weather damage occurred on 1.7 million ha; within that category, drought and wind were more common sources of damage than ice or floods (table 3.2). Geologic, vegetation, and other damages combined affected only an estimated 0.6 million ha during a 5-year period.

Although the distinction between forest and nonforest (fig. 3.3) controls the overall spatial pattern of forest disturbance in the Western United States, plotting the primary disturbance spatially shows the very strong regional differences in types of disturbance (fig. 3.4). Disease dominates in the high elevations of interior mountain ranges and in the precipitation-heavy regions of western Oregon and Washington. Grazing is the predominant disturbance in low elevations of California's Central Valley. Fire and insect disturbances are found

Table 3-2—Forest area affected by disturbances within a 5-year window, Western United States[a]

Damage type	Specific damage area	Major damage group area
	- - - - - - - - *Hectares*	- - - - - - - -
Animal:		3 869 000
Bear	78 000	
Beaver (includes flooding caused by beaver)	56 000	
Deer/ungulate	108 000	
Domestic animal/livestock (includes grazing)	3 408 000	
Porcupine	137 000	
Rabbit	2 000	
Other/unknown animal damage	81 000	
Disease:		6 076 000
Disease damage to trees, including seedlings and saplings	4 024 000	
Disease damage to understory vegetation	24 000	
Other disease damage	2 028 000	
Fire:		3 692 000
Crown fire damage	848 000	
Ground fire damage	1 365 000	
Fire damage (from crown and ground fire, either prescribed or natural)	1 480 000	
Geologic:		310 000
Avalanche track	140 000	
Earth movement/avalanches	81 000	
Geologic disturbances	6000	
Landslide	75 000	
Other geologic event	7000	
Human:		4 384 000
Silvicultural treatment	4 032 000	
Other human-induced disturbance	352 000	
Insect:		6 054 000
Insect damage to trees, including seedlings and saplings	3 223 000	
Insect damage to understory vegetation	92 000	
Insect damage (unspecified)	2 739 000	
Other:		221 000
Unknown/other disturbance	221 000	

Table 3-2—Forest area affected by disturbances within a 5-year window, Western United States[a] (continued)

Damage type	Specific damage area	Major damage group area
	- - - - - - - - *Hectares*	- - - - - - - -
Vegetation:		116 000
Suppression, competition, vines	116 000	
Weather:		1 748 000
Drought	610 000	
Flooding (weather induced)	177 000	
Ice	66 000	
Weather damage	383 000	
Wind (includes hurricane, tornado)	512 000	
Total—All damages combined		23 379 000

[a] Source are 2006–2015 Forest Inventory and Analysis data for Arizona, California, Colorado, Idaho, Kansas, Montana, Nebraska, Nevada, New Mexico, North Dakota, Oregon, South Dakota, Utah, Washington, Wyoming, and southeast Alaska; includes disturbances from 2001–2015. Multiple disturbances can affect an individual acre, so values should not be totaled. Does not include disturbances that cause permanent conversion to nonforest. Disturbances recorded only if a significant threshold is reached: mortality and/or damage to 25 percent of all trees or 50 percent of an individual species, or, for disturbances such as flooding or grazing, affecting at least 25 percent of the vegetation or soil. Specific damages will not add to totals because of the co-occurrence of multiple damages on individual plots.

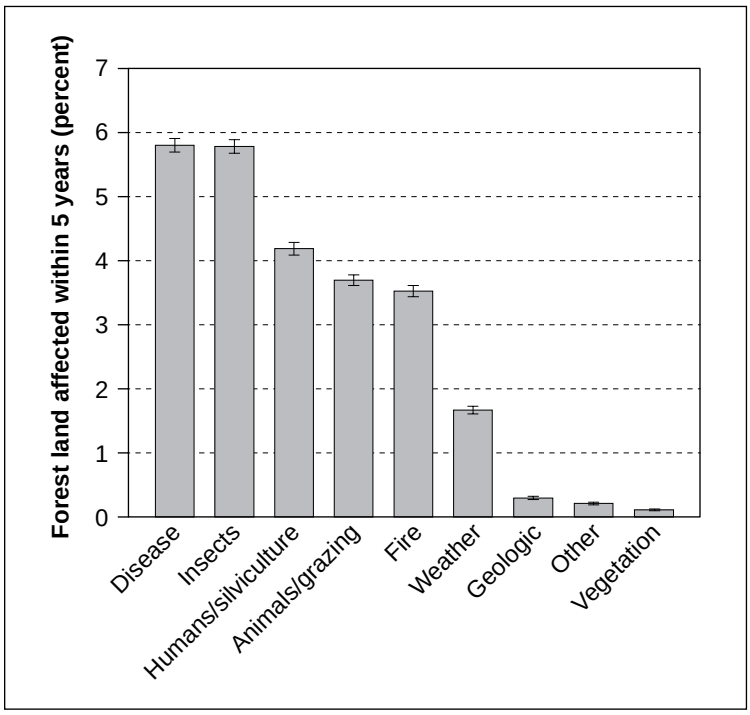

Figure 3.2—Disturbances affecting forests of the Western United States, shown as the percentage of forest affected by a major disturbance group within a 5-year window. (From 2004–2015 Forest Inventory and Analysis data for Arizona, California, Colorado, Idaho, Kansas, Montana, Nebraska, Nevada, New Mexico, North Dakota, Oregon, South Dakota, Utah, Washington, Wyoming, and southeast Alaska; includes disturbances from 2001–2015).

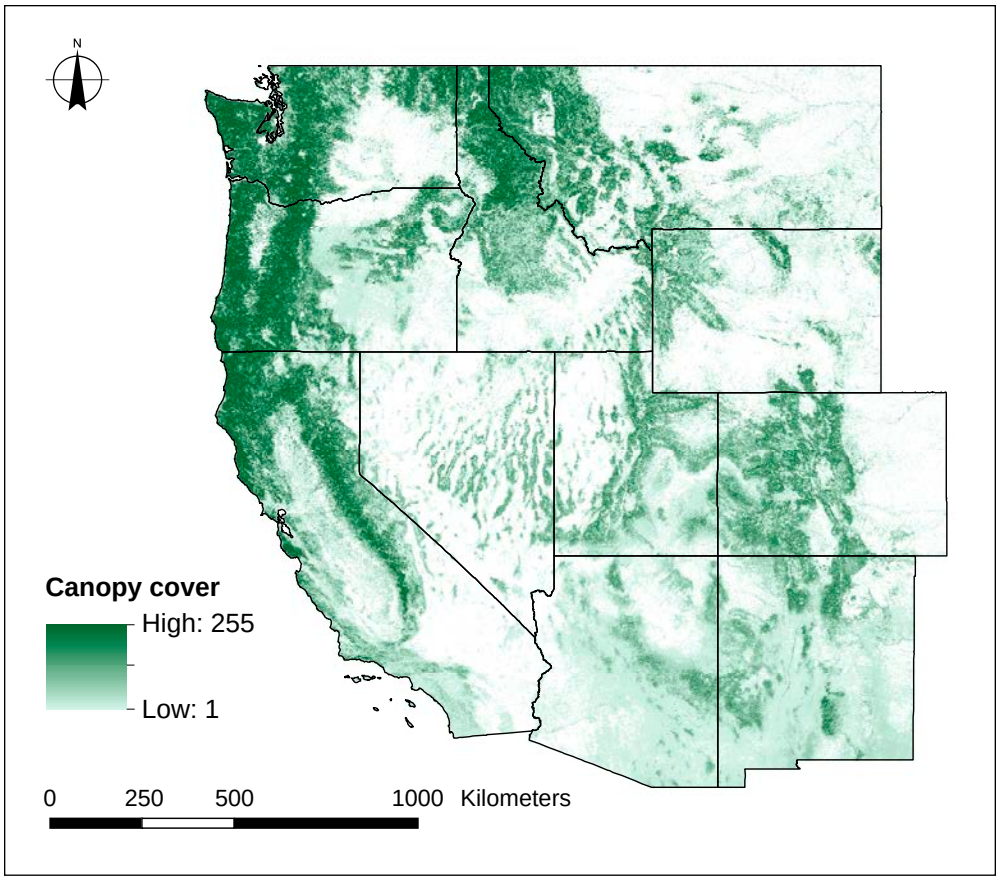

Figure 3.3—Percentage of tree canopy (National Land Cover Database 2011) (https://www.mrlc.gov).

throughout drier portions of the West; although the FIA disturbance protocol does not separate out the type of insect, aerial survey data indicate that most tree damage and mortality are from bark beetles (see chapter 6). Note that while the map can be useful for seeing broad regional differences in disturbance patterns, it can be difficult to discern areas of acute local concern such as sudden oak death, which has led to quarantines in 15 counties of California and 1 county of Oregon (see chapter 7).

Looking at disturbances by the type of forest where found, grazing is most common in hardwood and oak forests (fig. 3.5). Woodlands (forest dominated by juniper, pinyon, mesquite, and mountain brush) had less disturbance overall than other groups of forest types, with fire found on just 1 percent of forest within a 5-year time window. Douglas-fir (*Pseudotsuga menziesii*) forests had the highest rate of human disturbance (7.5 percent per 5 years), followed by pine forests (5.8 percent). Insects were most common in pine forests (9.4 percent per 5 years), while disease was

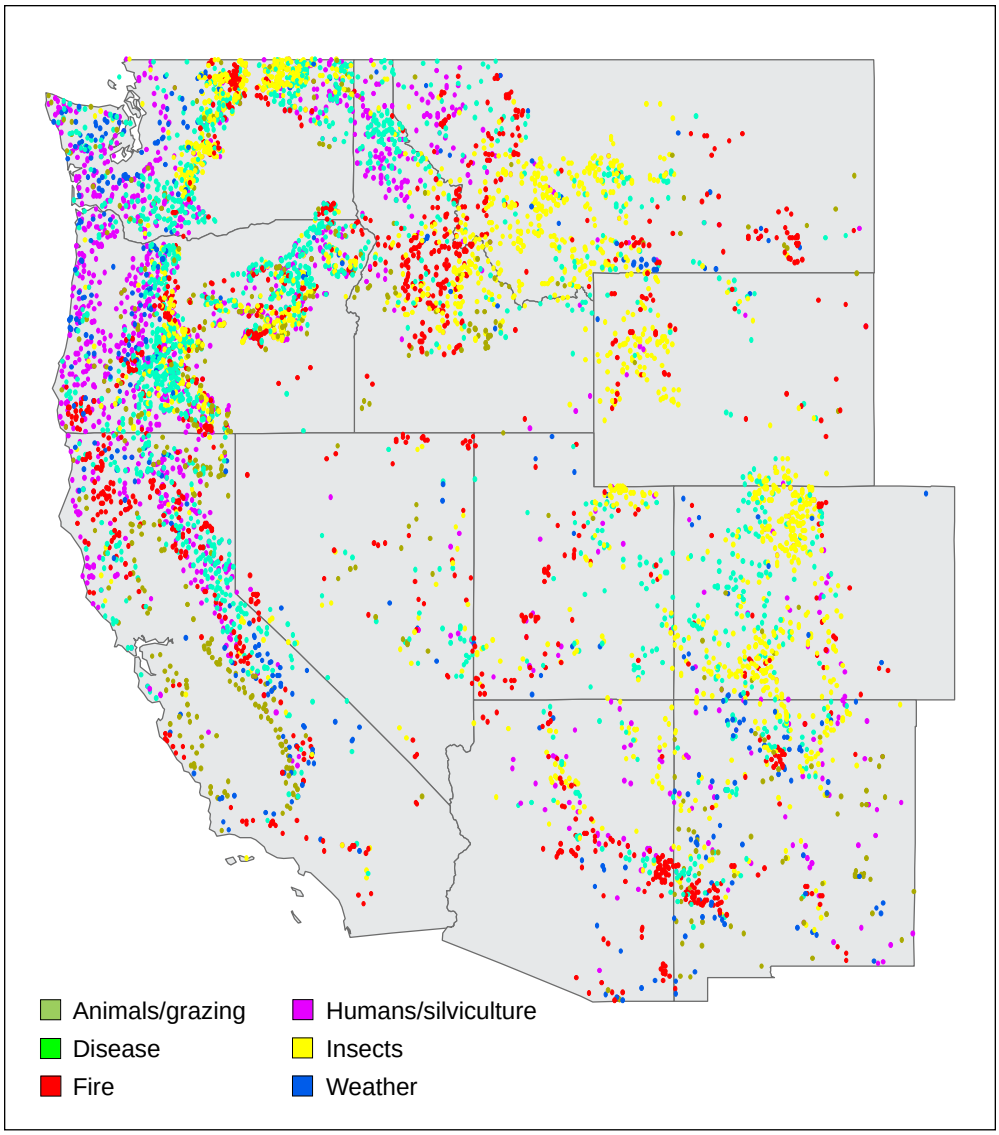

Figure 3.4—Distribution of different disturbance types in Western U.S. forests, using Forest Inventory and Analysis (FIA) plots. For visibility, plot size is enlarged, undisturbed plots are not shown, and only the first (primary) disturbance on a plot is shown. (From 2011–2015 FIA plot data).

common (6.3 to 8.5 percent of forest per 5-year interval) in all forest groups except oak and juniper woodlands (2.5 and 2.2 percent, respectively). For the 5.1 million ha of nonstocked forest, which was forest with less than 10 percent stocking or canopy cover, fire was the leading disturbance (22.3 percent of nonstocked forest) in the preceding 5 years, followed by human disturbance (7.9 percent).

If looked at with a finer resolution of forest types, notable insect disturbance occurred in whitebark pine (*Pinus albicaulis*), Engelmann spruce (*Picea*

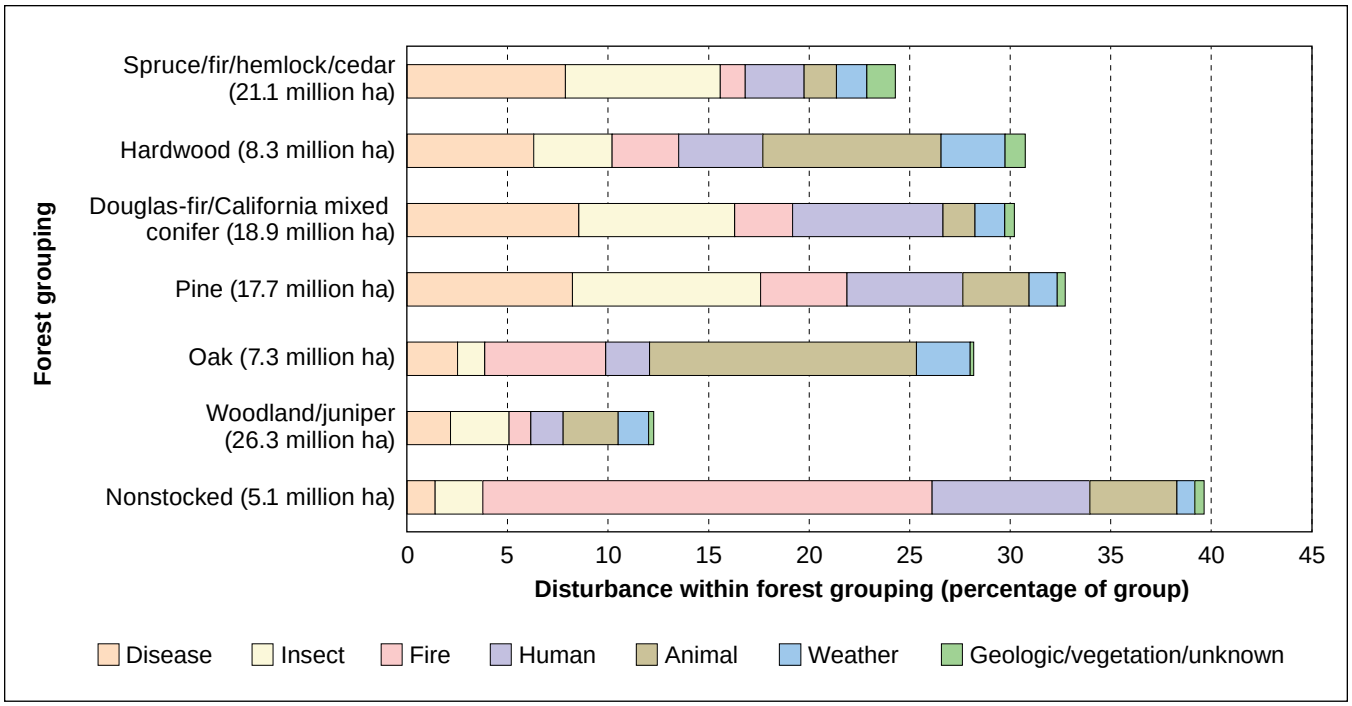

Figure 3.5—Percentage of forest groups affected by disturbance within a 5-year time window. Developed from 2004–2015 Forest Inventory and Analysis data for Arizona, California, Colorado, Idaho, Kansas, Montana, Nebraska, Nevada, New Mexico, North Dakota, Oregon, South Dakota, Utah, Washington, Wyoming, and southeast Alaska. Numbers to the right of forest group names indicate total forest land area within the group.

engelmannii), subalpine fir (*Abies lasiocarpa*), lodgepole pine (*Pinus contorta*), limber pine (*Pinus flexilis*), and western white pine (*Pinus monticola*) forest types, ranging from 21.3 to 10.6 percent of forest per 5 years. Although the FIA disturbance data do not contain information about specific insect species, the period of the inventory corresponded with a large mountain pine beetle outbreak that peaked in 2009. Mountain pine beetle primarily affects lodgepole pine, although it can also attack 14 other North American pine species (see chapter 6). Spruce beetle is a major source of mortality for Engelmann spruce, and subalpine fir is affected by western balsam bark beetle (*Dryocoetes confuses*) and balsam woolly adelgid *(Adelges piceae)* (see chapter 6).

In declining order, the highest rates of fire per 5 years (ranging from 9.6 percent to 4.6 percent) occurred in ponderosa pine (*Pinus ponderosa*), California mixed conifer, and oak woodlands (including canyon live oak [*Quercus chrysolepsis*], tanoak [*Notholithocarpus densiflorus*], interior live oak [*Quercus wislizeni*], Oregon white oak [*Quercus garryana*], and coastal live oak [*Quercus agrifolia*]).

In declining order, the highest rates of disease disturbance per 5 years (ranging from 34.9 to 11.7 percent) occurred in fir (red, white, grand, and noble [*Abies procera*]), California mixed conifer, western white pine, limber pine, and aspen (*Populus tremuloides*).

To look at how shifting from a 5-year disturbance rate to an annual rate might affect results, a model was created for 11 states that had 10 years of measurement. Annualizing disturbances assumed that disturbance that had been coded with a multiple-year code occurred in the 2 years prior to the inventory date, with values also adjusted according to how much forest land was sampled in a particular year. Figure 3.6 shows how this affects results. Compared to the 5-year disturbance rates, disease and grazing, which tend to occur in multiple years, increase more than do weather, fire, and human disturbances, which are less likely to be multiple-year disturbances. Figure 3.6 also illustrates that fire had notably greater year-to-year variability than other disturbances, followed by insects.

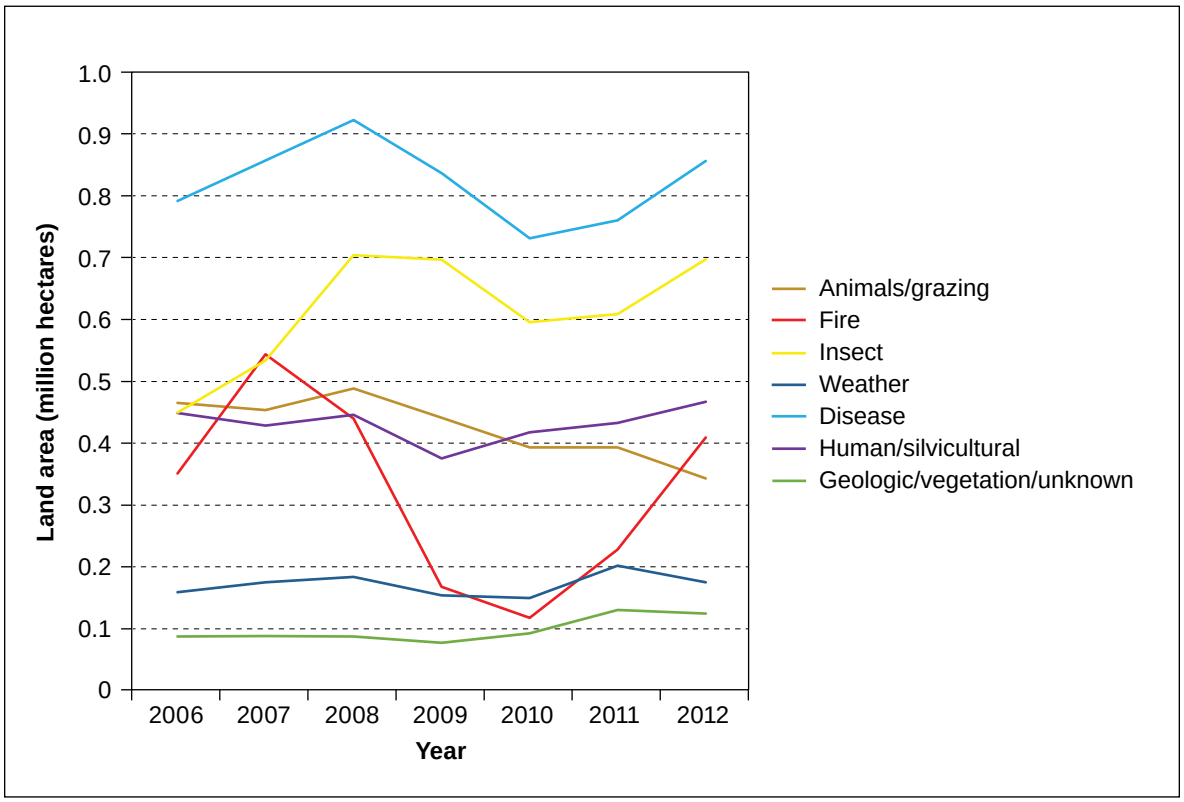

Figure 3.6—Modelled annual variability of different disturbance groups, developed from Forest Inventory and Analysis data for Alaska, Arizona, California, Colorado, Idaho, Montana, Nevada, New Mexico, Oregon, Utah, and Washington with the assumption that disturbances coded as multiple year occurred in the 2 years prior to the date of inventory.

Disturbance in the Boreal Forests of Alaska

Kathryn C. Baer, Sean M.P. Cahoon, and Tara M. Barrett[1]

Inventory of the Boreal Forest

The boreal forests of interior Alaska are vast—comprising 15 percent of all U.S. forest land—and are prone to disturbances such as fire and insect outbreaks. About 45.9 million ha of Alaska's boreal forest were not included in the estimates of the disturbance chapter of this report (chapter 2), because the Forest Inventory and Analysis program's inventories of those forests began in 2014 and are not expected to be completed until 2029. Preliminary 2014–2018 data from 743 forested plots in the Tanana subunit, a 13.9-million-ha section of Alaska, provide some insight into common disturbances in Alaska's boreal forests.

As some plots had more than one forest condition or more than one disturbance in the same stand, the total number of disturbance records for this dataset was 844. Seventy-seven percent of forested conditions encountered on the plots were undisturbed, giving a 5-year disturbance rate (23 percent of forest) equal to the average of the other Western States. Disturbances included damage by insects and other animals, fire, pathogens, weather events, avalanches, or earth movements, such as from melting permafrost, and unknown damage agents, most of which resulted in tree dieback (fig. 3.7).

Herbivory by insects was the most commonly recorded disturbance, with leaf rollers (primarily on birch species) and leaf miners (primarily on aspen) the most frequent damage agents. Both can affect growth (Holsten et al. 2008, Wagner and Doak 2013). Damage from spruce bark beetles was found on less than 1 percent of the Tanana forest plots, but spruce beetles have had a major impact on boreal forest to the south of the Tanana subunit (Burr 2017). Although disturbance by pathogens was relatively uncommon on the Tanana plots, the three most commonly recorded categories were needle rust or blight, spruce bud blight, and leaf rust.

Changing climatic conditions may contribute to altered rates of disease outbreaks in forests such as those of interior Alaska, although the direction of these changes will be specific to the host and pathogen (Chapin et al. 2010, Mulder et al. 2008, Sturrock et al. 2011). Increased insect damage may be related in part to

[1] **Kathryn C. Baer** is a research ecologist (previously an ORISE postdotoral research fellow) and **Sean M.P Cahoon** is a research ecologist, Pacific Northwest Research Station, 161 East 1st Avenue, Door 8, Anchorage, AK 99501; **Tara M. Barrett** is a research forester, Pacific Northwest Research Station, 1133 N Western Avenue, Wenatchee, WA 98801.

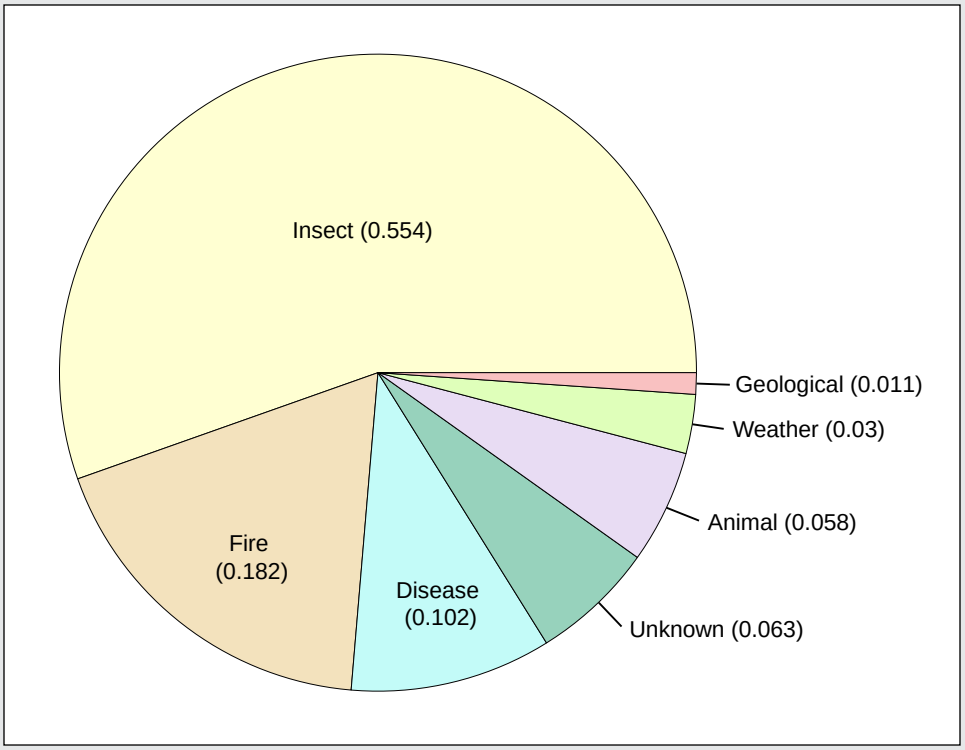

Figure 3.7—Disturbance type as a proportion of disturbance records from Tanana Forest Inventory and Analysis plots.

changing climatic conditions in this region (Berg et al. 2006, Chapin et al. 2010, Dale et al. 2001, Trugman et al. 2017) or to altered fire regimes (McCullough et al. 1998, but see Berg et al. 2006).

Fire was the second most frequent disturbance recorded on the Tanana plots. Fire is the major disturbance agent in the boreal forest of interior Alaska (fig. 3.8), influencing soil thermal and hydrological properties, nutrient dynamics, plant community composition, and greenhouse gas emissions. The fire regime in Alaska is broadly influenced by the interacting factors of global and local climate, landscape characteristics, land management practices, and fire policies (Kasischke et al. 2010, Viereck 1983, Yarie 1981).

Observational records and climate reconstruction of fire history have revealed a substantial increase in the area burned, frequency, and severity of fires throughout Alaska's boreal forest since the 1990s (Kasischke and Turetsky 2006, Kasischke et al. 2010). Although fire suppression policies have certainly influenced the area burned, a warming and drying trend in interior Alaska appears to be an underlying driver of increasing fire activity (Duffy et al. 2005, Young et al. 2016). Indeed, climate-driven shifts in the timing, severity, and

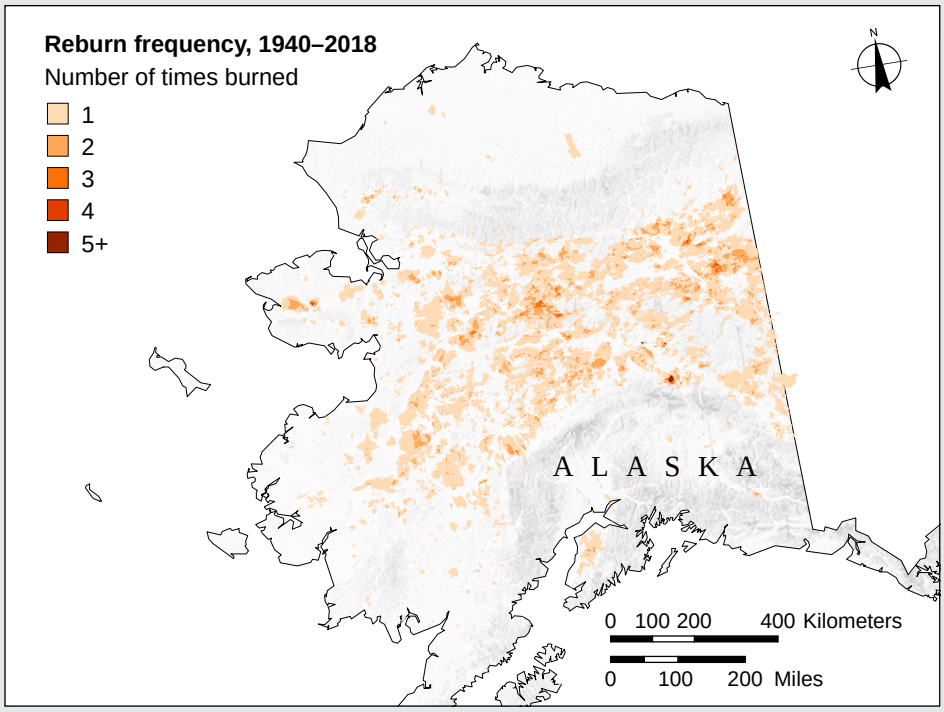

Figure 3.8—Fires in Alaska, 1940–2018.

extent of fire regimes are resulting in changes to carbon storage, climatic feedbacks, and ecosystem properties, with implications for local subsistence communities and land management policies.

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Discussion

We found that about 22 percent of western forests experienced disturbance within a 5-year time window, using recent FIA data. Although that rate may seem high to people, it is important to recognize that this dataset uses a relatively low threshold for disturbance, and that most disturbances recorded on the FIA plots did not result in stand-replacing mortality.

In the Western United States, insect and disease disturbances affected more forest land than fire, but fire has large societal costs.

Many types of diseases and grazing or animal damage have relatively little impact on mortality. Although the FIA disturbance data indicate that disease would be the most common disturbance in western forests on an annual basis, it is not necessarily the most important ecologically. And in terms of economic consequences, fire, though only the fifth most common disturbance per 5-year interval, not only has the potential for high tree mortality but has additional societal costs associated with suppression, structural loss, health costs associated with smoke, displacement, and injury or loss of life.

High tree mortality is most frequently associated with fire and insects, among the natural disturbances (Reilly and Spies 2015). Disturbances with high levels of tree mortality often have negative consequences for people in the short to medium timeframe, as they can reduce land value (Kovacs et al. 2011, Price et al. 2010), reduce timber value (Mills and Flowers 1986), harm endangered species (Lesmeister et al. 2018), increase flooding and erosion (Moody and Martin 2001, Seibert et al. 2010), and facilitate increases in invasive species (Keeley 2006, Kerns and Day 2017). The combined impact of disturbances in the Western States is to reduce gross primary productivity and increase land temperature (Cooper 2017).

The FIA disturbance data provide a way to consistently track disturbances over time and compare one type of disturbance to another. However, the values for amount of disturbance derived from FIA plots are different than estimates from other methods of monitoring. LandSat-based estimates made from 1985 to 2005 put forest disturbance rates in the West at between 0.8 to 1.6 percent per year (Masek et al. 2013), substantially lower than our estimate of 22.3 percent of western forest having disturbance within a 5-year window. The difference is likely due primarily to the different threshold for disturbance, with LandSat most easily detecting stand-clearing disturbance and the FIA definition threshold based on damage or mortality to 25 percent of trees.

Cohen et al. (2016) also interpreted LandSat data to estimate disturbances, but they used a lower threshold for disturbance to capture partial canopy loss. That study tracked disturbance in the United States from 1985 to 2012 by four groups: harvest, fire, decline, and other, where “decline” was likely caused by disease, drought, or insects, and “other” was likely caused by wind, water, land use change,

or geologic forces. Their conclusion that decline was substantially larger than either fire or harvest from 1995 to 2012 in the Western United States is consistent with our findings for an overlapping time period. Estimated annual disturbance rates for the West in Cohen et al. (2016) ranged from near zero to 9.61 percent per year and showed a substantial increase in disturbance from 1985 to the early 2000s.

Although we included harvest in our estimates to allow comparison with other disturbance monitoring, it is worth noting that silvicultural actions are often designed to lessen the impact of other disturbances. That is also true of prescribed fire, which is combined with wildfire in this assessment. Estimated area affected by fire in the data analyzed in this chapter is about one-half of the area tracked by the National Interagency Fire Center (NIFC) (see chapter 4), and the difference would be even larger if we were able to exclude prescribed fire from the FIA dataset. The higher fire levels from the NIFC statistics would primarily be due to inclusion of nonforest shrublands and grasslands in the NIFC dataset, as well as a difference in spatial resolution. An FIA plot is high resolution, covering about an acre in size. In contrast, fire boundaries are often the outer perimeter of areas affected by fire, and the area within the mapped boundaries may include a mix of burn severity along with patches of unburned vegetation.

It is difficult to compare estimates of insect and disease between FIA plots and the Forest Health Protection's (FHP) aerial survey because of differences in time windows, disturbance thresholds, and methods (field vs. air survey). From 2001 to 2015, mortality from insects and disease reported by FHP averaged about 3 million ha per year for the whole United States (USDA FS 2017b). This compares with this report's estimate of 6 million ha each for insects and disease for a 5-year period for the Western United States, and, when modelled as an annual process, about 1.4 million ha per year of insect/disease disturbance for a subset of 10 states. Although beyond the scope of this report, a more detailed comparison could be made by combining 5 years of FHP maps for just the Western States or overlaying plots and FHP maps. Estimates could be expected to differ because of the distinct methodological approaches, but the amounts from FHP reporting and FIA analysis appear to be generally consistent.

The advantages of FHP reporting include easier detection of trends from year to year and attribution of most insect and disease damage to specific agents. Advantages of FIA data include the ability to detect disturbances not easily seen from the air and, over time, the ability to measure processes such as growth, mortality, and regeneration of species before and after disturbance. Animal damages and grazing in forests are typically very difficult to detect with remote sensing, thus the FIA plots provide one of the few sources of information for them for the Western

States. Similarly, some types of disease, such as dwarf mistletoes or blister rust, are difficult to detect through remote sensing. FIA also provides a unique dataset for tracking invasive plants consistently across state boundaries (see chapter 8). In the future, continued collection of FIA data will permit trend analysis.

Most disturbances in western forests cause only partial mortality, and live trees remain.

Disturbances have highly varied impacts. Mortality in stands affected by disease and insects is often relatively low (Reilly and Spies 2016), and it is common for forest disturbances to create multiaged stands with complex structure through selective mortality (O'Hara and Ramage 2013). Although some types of disturbances—fire, harvest, bark beetles, and landslides—may cause stand-replacing mortality, in most cases there is only partial mortality, and live trees remain. When stand-replacing mortality does occur, there may be a shift in forest type, in which the original tree species are replaced by different species.

Estimates for disturbances in this report included instances of stand replacement, in which all trees on the plot have been killed, but these include only land that is expected to remain forested. Although we did not make estimates of deforestation (permanent removal of tree cover) rates, these have been relatively low in the Western United States in recent decades (fig. 3.9). Fire was recorded as

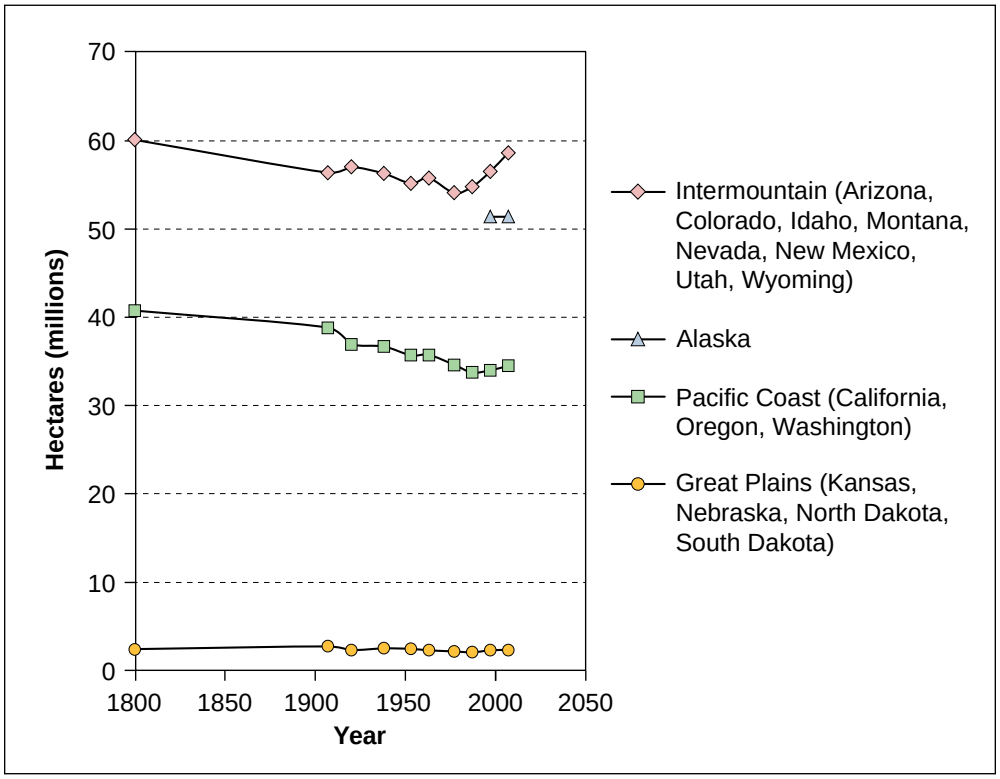

Figure 3.9—Forest area by region for the continental Western United States. Adapted from Smith et al. (2009). Alaska trend data are not available prior to 1995.

a disturbance for nonstocked forest land more frequently than harvest; this may be due to fire occurring on more xeric lands, or perhaps differences in replanting practices. As climate changes, there is increasing concern that after disturbance in dryer forest lands, tree regeneration will not occur (Davis et al. 2019, Savage and Mast 2005, Tepley et al. 2017).

We do not have long-term monitoring data for all types of disturbances. However, we do know that invasive species (plants, insects, and disease) are having an increasing impact on forests. We also know from fire statistics that in recent years, fire has been increasing in extent and resulting in more high-severity impacts. We know that climate has been warming in the West, and as this continues, it is expected to increase the effects on forests from drought, fire, and bark beetles. In the following chapters of this report, five types of disturbances (fire, drought, insects, disease, and invasive plants) are explored in more depth, along with assessment of their impacts on forest sustainability.

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Chapter 4: Fire Disturbances

David W. Peterson and Tara M. Barrett[1]

Introduction

Fire is a common disturbance in western forests that can either threaten or promote sustainability, depending on the forest type, climate, recent fire history, and fire behavior and burn severity. High-severity wildfires can alter forest structure, soil physical properties, and soil water and nutrient cycles, and serve as catalysts for long-term changes in forest structure, species composition, and biodiversity, particularly in a changing climate. In contrast, low-severity wildfires consume surface woody fuels and reduce ladder fuel development, thereby reducing risks of high-severity wildfires. Fire exclusion (absence of fire) has little effect on mesic forests in which wildfires are naturally uncommon (Parks et al. 2015b), but it can destabilize fire-prone forests by increasing fuels and the potential for high-severity wildfire (Hessburg et al. 2005).

The Montréal Process prescribes monitoring of wildfire impacts on forests under criterion 3, "Maintenance of forest ecosystem health and vitality." This criterion recognizes and affirms the importance of natural disturbances in forests, but also recognizes that uncharacteristic disturbances can disrupt normal ecosystem functions, threaten important ecosystem services, and alter future landscape structure and function. Monitoring wildfire impacts is important because fire, where present, influences many aspects of forest conditions and outputs, including biological diversity (criterion 1), productive capacity of forests (criterion 2), conservation and maintenance of soil and water resources (criterion 4), global carbon cycles (criterion 5) and human society (criterion 6). Increasingly costly expenditures for wildfire suppression are promoting changes in policies and approaches for funding wildfire risk mitigation activities, and are encouraging the use of partnerships and an "all lands" approach to wildfire management and forest restoration (criterion 7).

Finding and monitoring appropriate indicators of fire impacts on forest health, sustainability, and ecosystem services presents challenges. Criterion 3b prescribes using the area and percentage of forest lands affected by fire beyond reference conditions as an indicator for maintenance of forest health and vitality. Annual area burned—a single year representation of fire frequency—is effective as an indicator of fire activity over large areas, is relatively easy to monitor and report, and varies sufficiently in space and time to provide clues about the social and ecological drivers of fire activity. Annual area burned is a poor descriptor of fire effects on forest health,

[1] **David W. Peterson** and **Tara M. Barrett** are research foresters, Pacific Northwest Research Station, 1133 N Western Avenue, Wenatchee, WA 98801.

however. It does not account for the considerable variability in vegetation types and natural fire regimes within the Western United States and does not capture changes in fire severity and mean fire sizes relative to reference conditions (Littell et al. 2009).

Wildfire activity varies through time in response to variability in climate, ignitions, and fire suppression effectiveness. Field plot data from the U.S. Forest Service Forest Inventory and Analysis (FIA) monitoring program (chapter 3; fig. 3.6 in this report) show that fire has the greatest year-to-year variability of the common forest disturbances within Western U.S. forests. Because of this high variability, continual monitoring over many years is needed to properly identify longer term trends in disturbance activity and assess departures from baseline conditions.

Wildfire occurrences and effects also vary spatially in response to climate interactions with topography, vegetation, fuels, and land ownership patterns (Dillon et al. 2011, Heyerdahl et al. 2001, Littell et al. 2009, Parks et al. 2012). Regional monitoring programs at broad spatial scales—like the FIA monitoring program and satellite-based fire monitoring programs—can capture much of this variability but do not directly explain it. High spatial variability in fire regimes and fire effects also means that regional assessments of fire trends and impacts on sustainability will not apply equally to all landscapes and forest types (Littell et al. 2009). Similarly, spatial and temporal variability in land ownership and associated management objectives and strategies add complexity to wildfire activity and effects. Recent research has addressed these problems by evaluating the apparent effects of temporal and spatial variability in climate, topography, vegetation, fuels, and human activity on wildfire area burned and burn severity to infer the causal mechanisms driving variability in wildfire area and severity in the Western United States (Abatzoglou et al. 2017, 2018; Balch et al. 2013; Bradley et al. 2018; Harris and Taylor 2017). Results from these studies can be used to project longer term wildfire responses to climatic changes (Abatzoglou and Williams 2016, Dennison et al. 2014), forest restoration and fuel reduction efforts, and changes in wildfire management policies (Barros et al. 2018).

In this chapter, we present recent temporal trends and year-to-year variability for indicators of annual fire effects in forests of the Western United States, including the number of wildfire ignitions, wildfire area burned, wildfire burn severity, and wildfire suppression costs. We use analyses of these data and a review of peer-reviewed scientific literature to address the following questions:

1. What are the recent trends in wildfire area burned, burn severity, and suppression costs?
2. How does current fire activity compare to fire activity in recent decades (1983–2018) and the more distant past (early 20th century)?
3. What are the ecological and social drivers of trends and variability in fire activity?

Recent Trends and Variability in Fire Activity

Annual Area Burned and Number of Wildfires

The average annual area burned by wildfires has more than doubled in the United States in recent decades. Wildfire incident data from the National Interagency Fire Center (http://www.nifc.gov) shows that the annual area burned in large wildfires increased from 1.2 million ha per year for the decade from 1985 to 1994 to 2.8 million ha for the decade from 2009 to 2018 (fig. 4.1A). Much of this increase in mean annual area burned was realized in the decade between 1996 and 2005, and subsequent increases have been more modest (fig. 4.1A) (Hawbaker et al. 2017). In the Western United States (excluding Alaska), wildfire area burned increased from 0.7 million ha per year for the decade from 1985 to 1994 to 1.6 million ha for the decade from 2009 to 2018 (fig. 4.2A).

The trend toward increasing area burned is mostly due to increases in the frequency and size of large wildfires (Dennison et al. 2014, Westerling et al. 2006). The number of recorded wildfire "incidents" in United States was highly variable from 1985 to 2018, ranging between 48,000 and 96,000 fires per year (fig. 4.1B). Long-term trends in the annual number of wildfires are less pronounced than those for areas burned, but there is some evidence for a reduction in the mean annual number of wildfires over roughly the past decade (2009–2018), compared to earlier years (fig. 4.1B). Average wildfire size (in hectares) increased from 17 ha per fire during the first decade (1985–1994) to 42 ha per fire during the most recent decade (2009–2018) (fig. 4.1C).

Although large wildfire activity has increased across most of the Western United States, the increase has not been evenly distributed among ecoregions and vegetation types or through time. Year-to-year variability in area burned is high

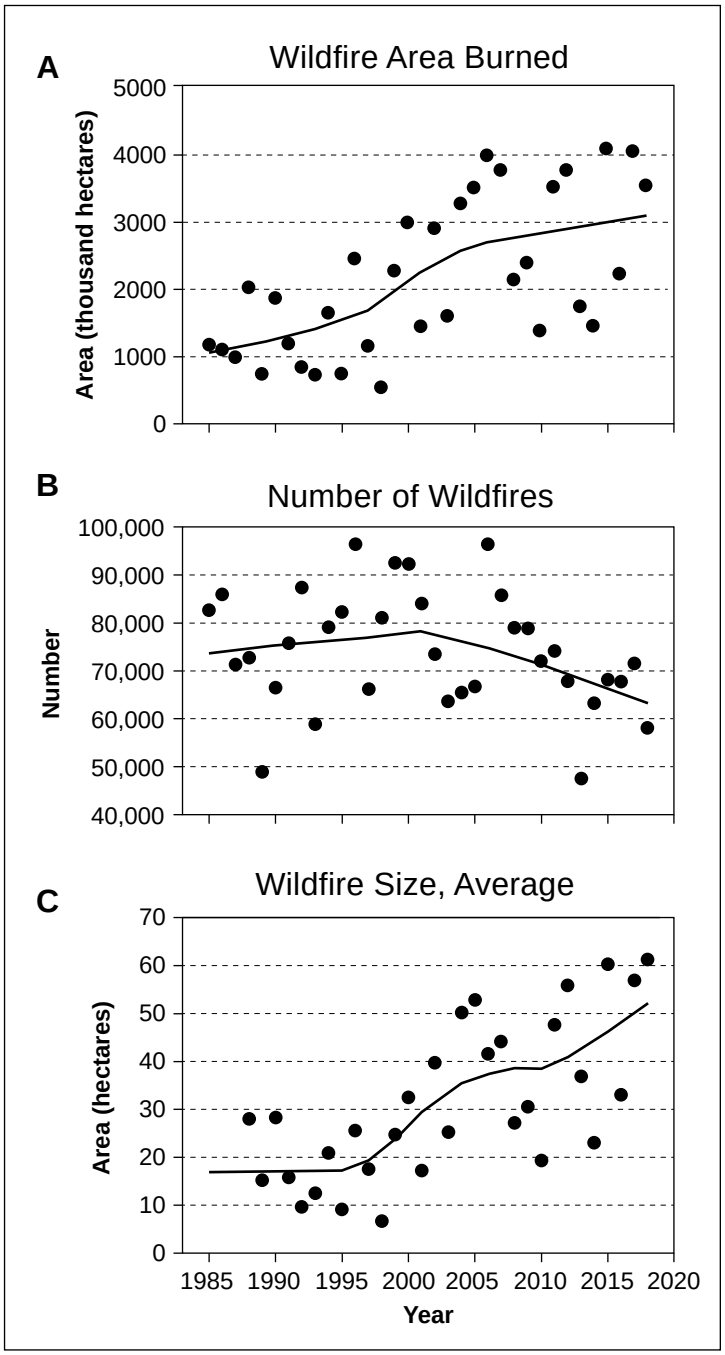

Figure 4.1—Annual wildland fire activity across all owners in the United States, 1983–2018, including (A) the area burned (kilohectares), (B) the number of wildfires recorded, and (C) the average wildfire size (hectares). Source of statistics: NIFC 2019. (https://www.nifc.gov/fireInfo/fireInfo_statistics.html). Trend lines were fitted to data using locally weighted scatterplot smoothing as implemented in SAS PROC LOWESS (SAS Version 9.4) (SAS Software, Cary, N.C.).

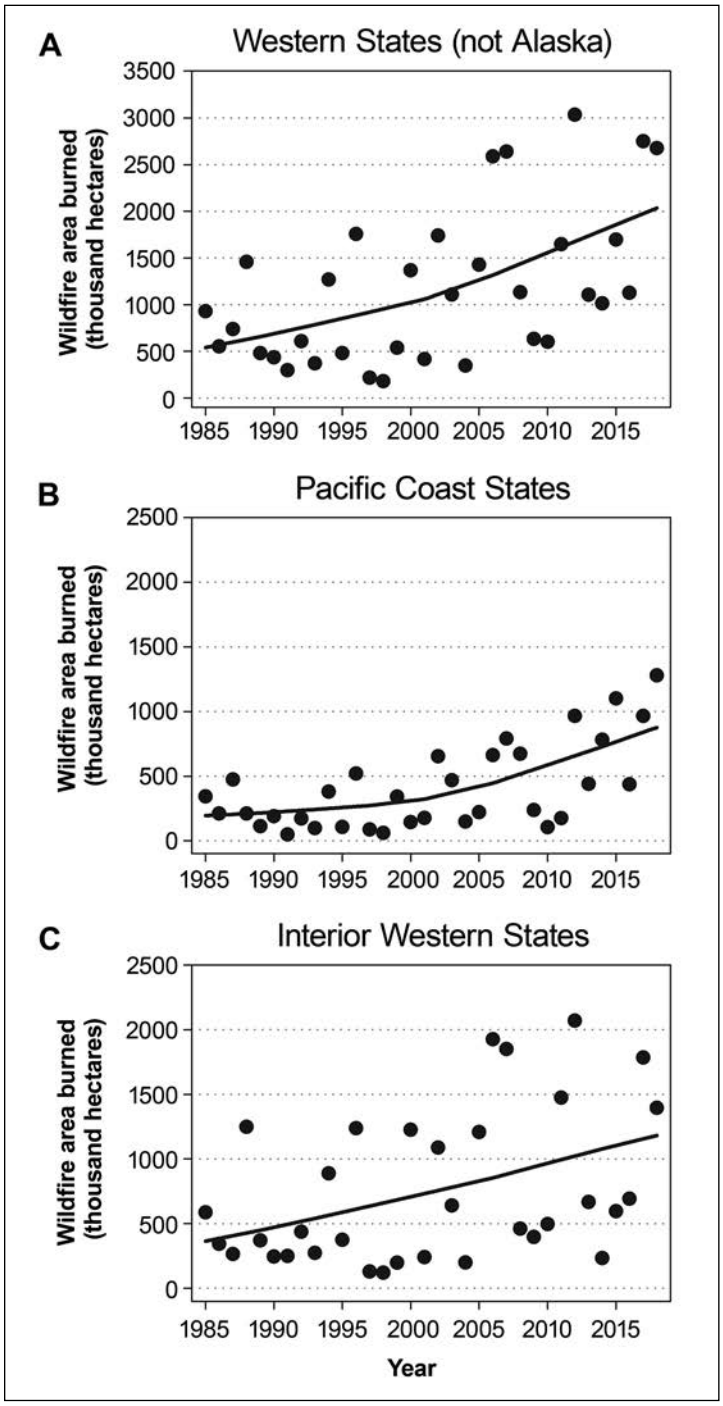

Figure 4.2—Annual area burned in (A) 11 Western States, not including Alaska; (B) Pacific Coast states (California, Oregon, and Washington); and (C) interior Western States (Arizona, Colorado, Idaho, Montana, Nevada, New Mexico, Utah, and Wyoming). Trend lines were fitted to data using locally weighted scatterplot smoothing as implemented in SAS PROC LOWESS (SAS version 9.4) (SAS Software, Cary, N.C.).

for most ecoregions of the Western States, and ecoregions have experienced multiyear periods of high fire activity at different times (Dennison et al. 2014). For example, Westerling et al. (2006) found that the Northern Rocky Mountain region accounted for about 60 percent of the increase in large wildfire frequency in the West between 1987 and 2003, and Picotte et al. (2016) found significant increases in wildfire area burned for only 8 of 27 fire-prone vegetation groups between 1984 and 2010. Similarly, a series of large wildfires have increased mean wildfire area burned in three Western coastal states (California, Oregon, and Washington) over the past decade (fig. 4.2C).

Wildfire Burn Severity

Temporal trends in wildfire burn severity are generally consistent with those for wildfire area burned in the Western United States. Average annual area burned at high severity has roughly tripled over the past three decades in the Western States, excluding Alaska (fig. 4.3A). Area burned at moderate severity has also increased over this period but may have stabilized in the past decade (fig. 4.3A). Wildfire area burned at high severity is highly correlated with total wildfire area burned in the West, however, so area burned at high severity tends to increase with total area burned (Abatzoglou et al. 2017), and the percentage of area burned at high severity has not changed much over the past three decades (fig. 4.3C). Wildfire burn severity trends for Alaska are similar to those of the other Western States for area burned (fig. 4.3B), though the trends are more strongly influenced by a few extreme years (e.g., 2004, 2005, and 2015). As with the other Western States, percentages of area burned at different severities is highly variable from year to year but has not changed markedly over time (fig. 4.3D).

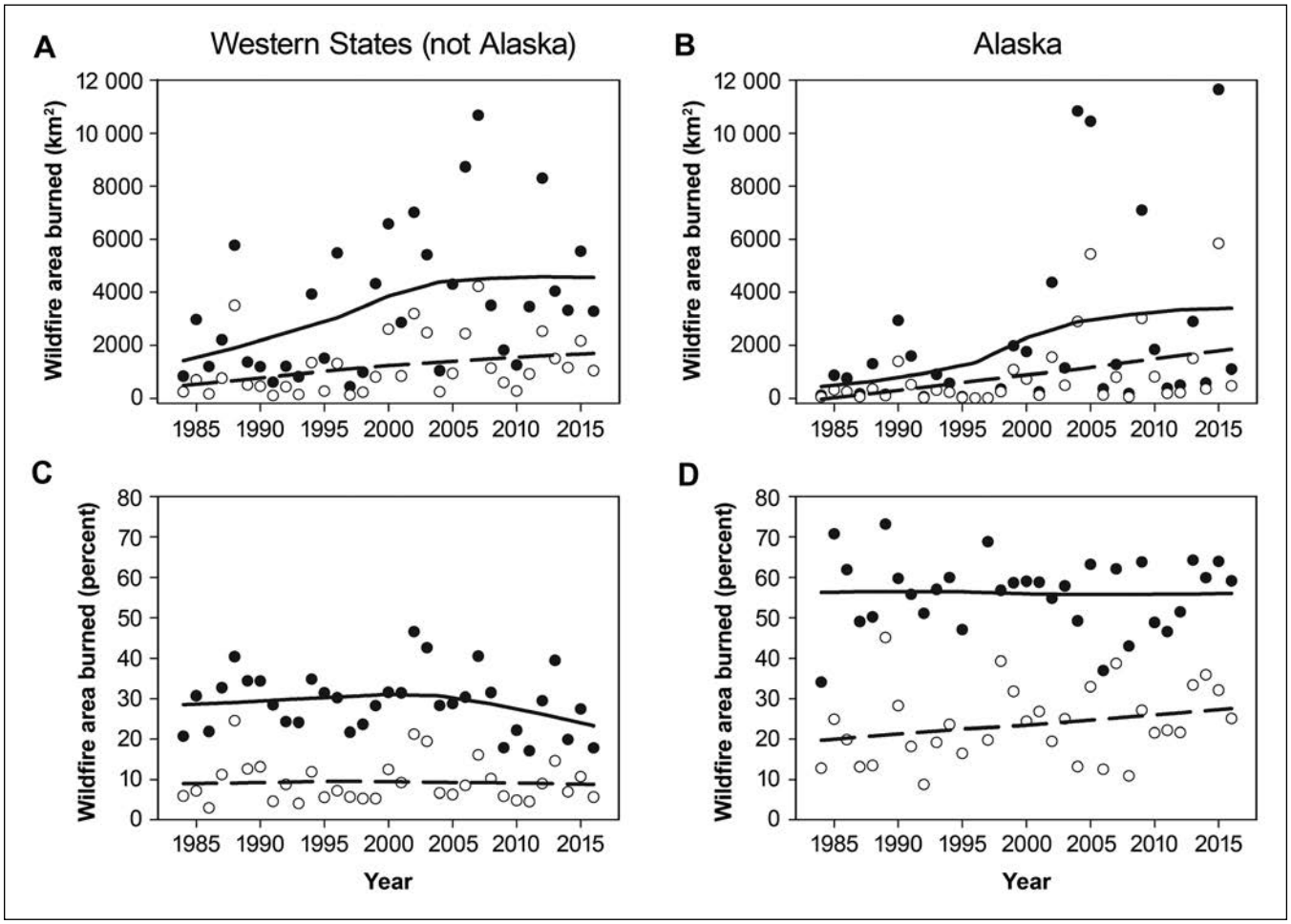

Figure 4.3—Wildfire burn severity in the Western United States. Graphs show wildfire area burned (A and B) and percentage of total wildfire area burned (C and D) for Alaska (B and D) and the other 11 Western States (A and C). Open circles and broken lines show annual variability and trends for high wildfire burn severity, whereas closed circles and solid lines show annual variability and trends for moderate and high wildfire burn severity combined. Trend lines were fitted to data using locally weighted scatterplot smoothing as implemented in SAS PROC LOWESS (SAS version 9.4) (SAS Software, Cary, N.C.).

Wildfire Suppression Costs

Wildfire suppression costs have also increased in recent decades, similar to wildfire area burned. Average annual wildfire suppression expenditures by the U.S. Forest Service and agencies within the U.S. Department of the Interior have more than tripled in the past 30 years, from an average of about $700 million per year in the mid-1980s to more than $2.1 billion per year from 2011 to 2018 (fig. 4.4A).[2] Within this increasing trend, suppression costs are highly variable from year to year, and are positively correlated with annual area burned (r = 0.74). Inflation-adjusted suppression costs per hectare burned were highly variable but did not change significantly between 1985 and 2018 (fig. 4.4B).

[2] Real costs after adjusting for inflation using the Consumer Price Index base 2017.

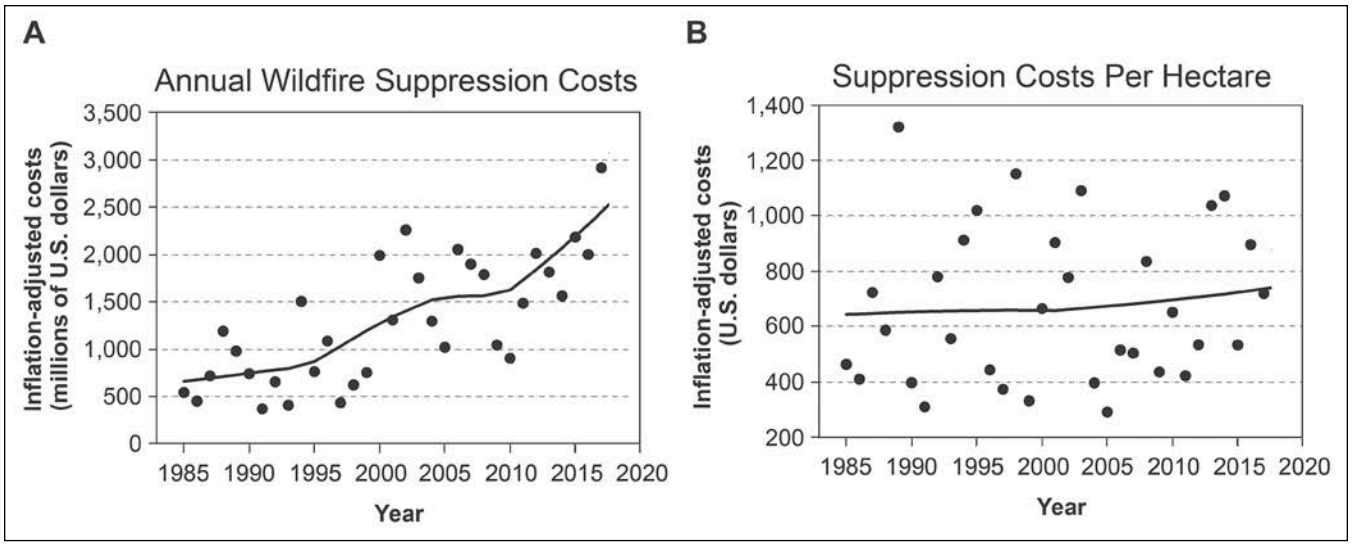

Figure 4.4—Federal wildland firefighting suppression costs 1985–2018 (NIFC 2019), adjusted for inflation using the 2017 Consumer Price Index. Graphs show (A) total annual suppression costs and (B) costs per hectare burned.

Current Wildfire Activity Compared to Historical Conditions

Average annual area burned has more than doubled in recent decades, although it still remains at or below historical norms derived from historical records or reconstructed fire frequencies.

Although area burned in western forests has been increasing in recent decades, area burned still remains at or below historical norms derived from long-term records or reconstructed fire frequencies because of human influences on wildfire ignitions and spread, including active fire suppression, livestock grazing, logging, land type conversion (e.g., to agriculture), and road building (Marlon et al. 2012, North et al. 2012, Parks et al. 2015b). Long-term records of wildfire area burned in 11 Western States suggests that total area burned in recent decades is similar to levels recorded in the early 20th century (Littell et al. 2009). However, fire-scar and charcoal records suggest that human-influenced reductions in fire frequency and area burned began in the late 19th century in response to livestock grazing and Euro-American settlement (Heyerdahl et al. 2001, Marlon et al. 2012), so records from the early 20th century do not necessarily provide reference conditions for natural fire regimes.

Models based on climate-wildfire relationships in minimally managed areas suggest that current departures in area burned from references conditions (i.e., fire deficits and surpluses) differ among vegetation types, with western forest lands still experiencing a fire deficit in recent decades, while nonforest lands are experiencing a fire surplus (Parks et al. 2015b). Departures in area burned also vary among forest types, with wet coastal forests exhibiting little or no departure from reference conditions, while dry forests that historically supported fire regimes with frequent, low-severity fires are still experiencing a fire deficit (Mallek et al. 2013, Parks et al. 2015b).

Ecological and Social Drivers of Wildfire Activity

Wildfires are initiated when wildfire ignitions, typically from lightning or human sources, coincide with flammable fuels and appropriate environmental conditions (Krawchuk and Moritz 2011). Following ignition, wildfire area burned is largely determined by fuel continuity in space and time. Fuel continuity requires sufficient flammable fuel to carry fire, so fuel continuity can be disrupted in areas with little or no fuel (e.g., rocky ridgetops or recently burned areas) or in areas where fuels are not flammable (e.g., wet fuels). Indeed, wildfire suppression tactics generally seek to disrupt fuel continuity by digging fire lines, setting backfires, or dropping water or fire retardant from aircraft.

Climatic Influences on Wildfire Activity

Wildfire area burned is strongly influenced by climate in forested and nonforested ecosystems of the Western United States, as fuel production, accumulation, and flammability commonly vary with mean annual precipitation and temperature. Climate influences natural fire regimes—including fire frequency, severity, and extent—through its effects on vegetation structure and composition (fuel type), fire season length, fuel flammability and connectivity, ignitions, and fire weather. Climate also influences year-to-year variability in area burned, so trends in climate can produce corresponding trends in wildfire area burned and burn severity.

The warmest and driest areas in the West support primarily grass- and shrub-dominated vegetation types. These ecosystems typically have long fire seasons during which fuels are flammable, but low soil water availability limits vegetation productivity, fuel accumulations, and fuel connectivity. Wildfire area burned in these nonforested vegetation types is positively correlated with prior growing season precipitation (Abatzoglou and Kolden 2013, Krawchuk and Moritz 2011, Littell et al. 2009). Elevated soil water in wet years promotes growth of grasses and forbs that subsequently dry out and become fuel, increasing total fuel amounts, fuel connectivity, and the potential for large wildfires.

Forests occupy a broad range of precipitation and temperature conditions in the Western States. In the warmest and driest coniferous forests—such as ponderosa pine (*Pinus ponderosa*) forests—trees and understory plants produce enough fine fuels to carry fires every few years, fire seasons are long, and tree and understory plant species are able to survive frequent, low-severity wildfires (Wright and Bailey 1982). Ignitions and fuel continuity were historically the primary constraints on fire frequency in these forests, and fires generally burned often enough to prevent fuels from accumulating to levels that would promote higher fire severities. Over the past 100 to 150 years, however, many of these forests have undergone changes in forest structure, species composition, and fuels in response to reduced fire frequencies produced by livestock grazing, land cover changes, and active fire suppression (Cooper 1960, Covington and Moore 1994, Hessburg et al. 2005, Keane et al. 2002). Elevated surface and ladder

fuels, higher stand densities, and increased abundance of fire-sensitive species now promote more mixed- and high-severity fire than was common under historical fire frequencies (Allen et al. 2002, Hessburg et al. 2005).

At the opposite end of the annual precipitation gradient, high soil water availability in maritime forests promotes high forest productivity and fuel accumulations, but high fuel moisture limits fuel flammability and spatial continuity throughout much of the year (Agee 1993, Krawchuk and Moritz 2011, Wright and Bailey 1982). Fire frequencies are typically very low in these forests (e.g., centuries between fires), but wildfire burn severity can be high if wildfires occur when fuels are dry.

Wildfire area burned in forested ecosystems is typically correlated with fire season length, fuel moisture levels during the fire season, and the number of days with hot, dry, and windy fire weather (Abatzoglou and Kolden 2013; Abatzoglou and Williams 2016; Holden et al. 2018; Jain et al. 2017; Jolly et al. 2015; Krawchuk and Moritz 2011; Littell et al. 2009; Westerling 2006, 2016). Climatic influences on fire season length are most pronounced in wetter and higher elevation forests where high fuel moisture limits fire ignition and spread throughout much of the year (Abatzoglou and Kolden 2013, Krawchuk and Moritz 2011). Warmer temperatures, reduced snowpack, and drought associated with longer fire seasons favor greater area burned in these forests by increasing ignition potential, increasing the time during which established wildfires can burn, reducing fuel moisture, and increasing fuel connectivity (Jain et al. 2017, Westerling et al. 2006).

Low fuel moisture (high fuel aridity) and extreme fire weather during the fire season can also influence area burned in both forested and nonforested ecosystems in the Western United States (Abatzoglou and Kolden 2013, Williams et al. 2015). In forested ecosystems, low fuel moisture and extreme weather increase fuel continuity, wildfire intensity, and rates of spread, while reducing fire suppression effectiveness. These same conditions promote higher area burned in nonforested grasslands and shrub-dominated communities, but it is unclear if the higher area burned is the direct result of altered fire behavior or because extreme conditions create high regional demand for fire suppression resources, resulting in reduced fire suppression effort in areas of lower priority (Abatzoglou and Kolden 2013).

Much of the trend in wildfire area burned over the past 30 years is associated with observed trends in elements of climate associated with fire season length and fuel aridity.

Much of the trend in wildfire area burned over the past 30 years is associated with observed trends in elements of climate associated with fire season length and fuel aridity. Globally, fire season lengths increased significantly from 1979 to 2013 on about 25 percent of the vegetated surface of the earth, while declining significantly on only 11 percent of those lands (Jolly et al. 2015). In North America, average fire season lengths increased by almost 14 days between 1979 and 2015 (Jain et al. 2017). Warmer spring temperatures and earlier snowmelt have promoted the earlier onset of fire weather conditions throughout much of the West (Westerling et al. 2016), while warmer fall temperatures and later arrival of autumn precipitation have extended the end of the fire season (Jain et al. 2017). Climatic influences on fire season length are most important in

more mesic forests, where higher moisture availability promotes high fuel production, but also limits fire spread throughout much of the year (Abatzoglou and Kolden 2013).

Similar trends have been observed in climatic variables associated with fuel aridity and extreme fire weather. Increasing summer temperatures and vapor pressure deficits over the past several decades have contributed to increasing area burned by reducing mean fuel moisture and increasing the number of days of extreme fire weather (Abatzoglou and Williams 2016, Jain et al. 2017). Declining summer precipitation in recent decades may be further increasing vapor pressure deficits and fuel aridity (Holden et al. 2018).

Human Influences on Wildfire Activity

Humans influence wildfire activity—wildfire ignitions, extent, pattern, and severity—in complex ways. Humans ignite wildfires, suppress wildfires, and alter landscapes and vegetation in ways that influence wildfire extent, pattern, and severity. Humans also drive changes in fire policy based on concerns about wildfire impacts on human health and property.

Humans are the dominant source of wildfire ignitions in the United States as a whole, and are second only to lightning in areas of the West in which lightning ignitions are the dominant source of ignitions (Balch et al. 2017, Nagy et al. 2018). Over a 21-year period from 1992 to 2012, human-ignited wildfires accounted for 84 percent of wildfire starts and 44 percent of wildfire area burned nationally (Balch et al. 2017). Human-ignited wildfires have tripled the average length of the fire season relative to lightning-ignited wildfires, often producing large wildfires that burn under higher fuel moisture conditions than lightning fires (Balch et al. 2017, Nagy et al. 2018). Humans are a significant source of wildfire ignitions even in sparsely populated mountain regions of the Western States (e.g., Northern Rocky Mountains), accounting for more than one third of wildfire ignitions and one quarter of area burned (Balch et al. 2017).

High levels of human influence on wildfire ignitions are offset to varying degrees by human wildfire suppression efforts and human influences on landscape connectivity and vegetation, however, so wildfire activity generally declines with increasing human population densities and other measures of human influence (Parisien et al. 2016). As human density increases, wildfire threats to human lives and property also rise and motivate increasing fire prevention and active fire suppression efforts. The vast majority of wildfire ignitions are therefore extinguished quickly while they are still small, while only a small minority of wildfires (about 1 percent) become large and account for most of the wildfire area burned and wildfire suppression expenditures (Calkin et al. 2005). Humans also reduce fire activity through land-cover changes, such as building roads, converting wildland vegetation to irrigated agriculture, and other landscape changes that alter vegetation cover and disrupt fuel continuity and wildfire spread potential (Parisien et al. 2016). Increasing human influence in an area also reduces climatic influences on fire activity, as humans supplant lightning as the dominant ignition source and active

fuels management, landuse conversions, and suppression activities reduce the influence of temperature and aridity on fire extent and severity (Syphard et al. 2017).

The expanding wildland-urban interface (WUI) is where human influences on fire activity may be most clearly observed. The WUI has been growing steadily in the Western United States in recent decades (fig. 4.5) with more homes and other human structures being built adjacent to fire-prone wildlands, thereby putting more

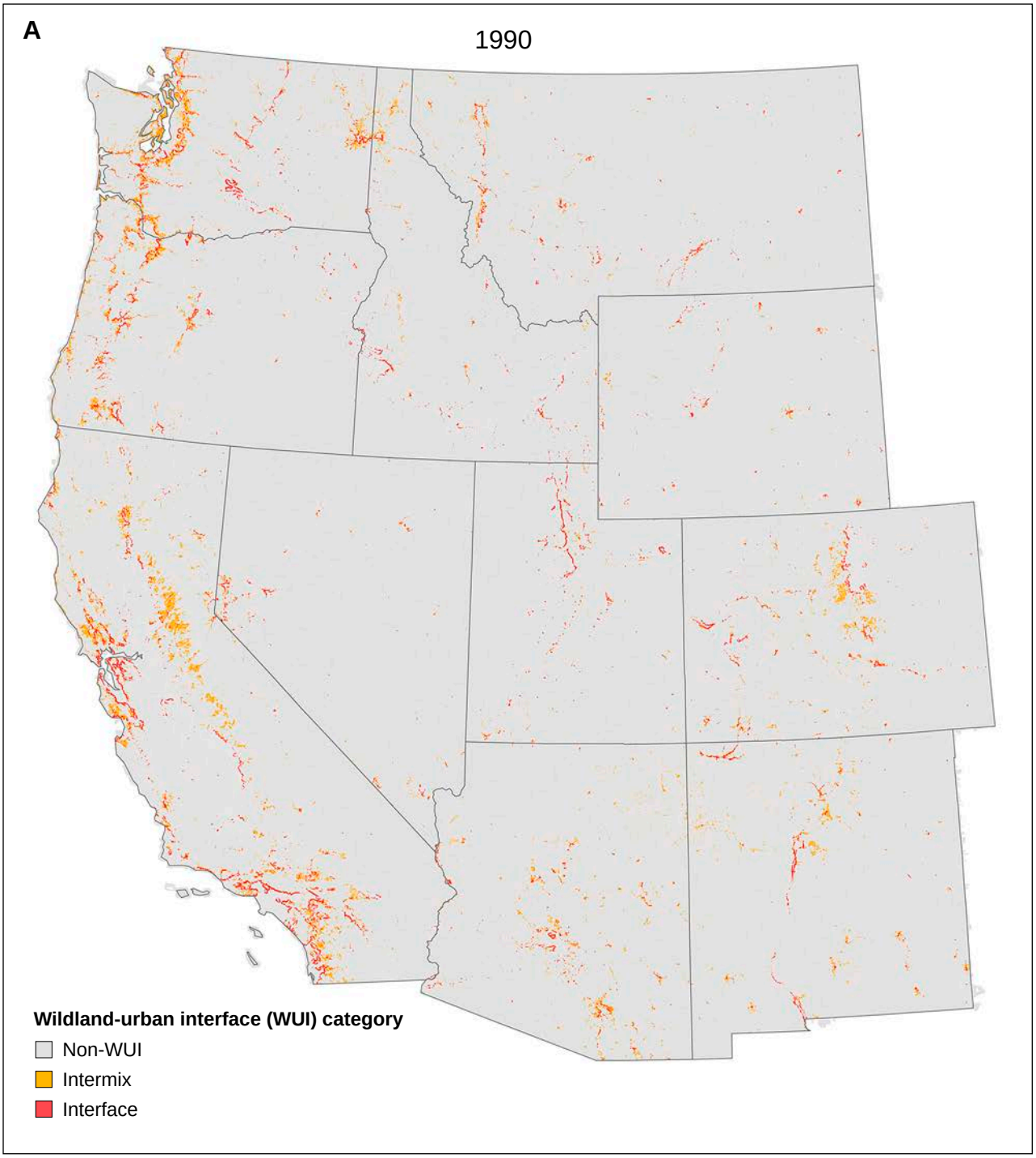

Figure 4.5—The wildland-urban interface and intermix in (A) 1990 and (B) 2010. Data are from Radeloff et al. (2017).

lives and houses at risk from wildfires (Radeloff et al. 2018). The presence of more lives and homes at risk increases social and political pressure for investment in and use of wildland fire suppression resources (Calkin et al. 2005, Donovan et al. 2011, Kramer et al. 2018), though it has also motivated wildfire prevention and hazard mitigation programs, such as Firewise communities (Kramer et al. 2018).

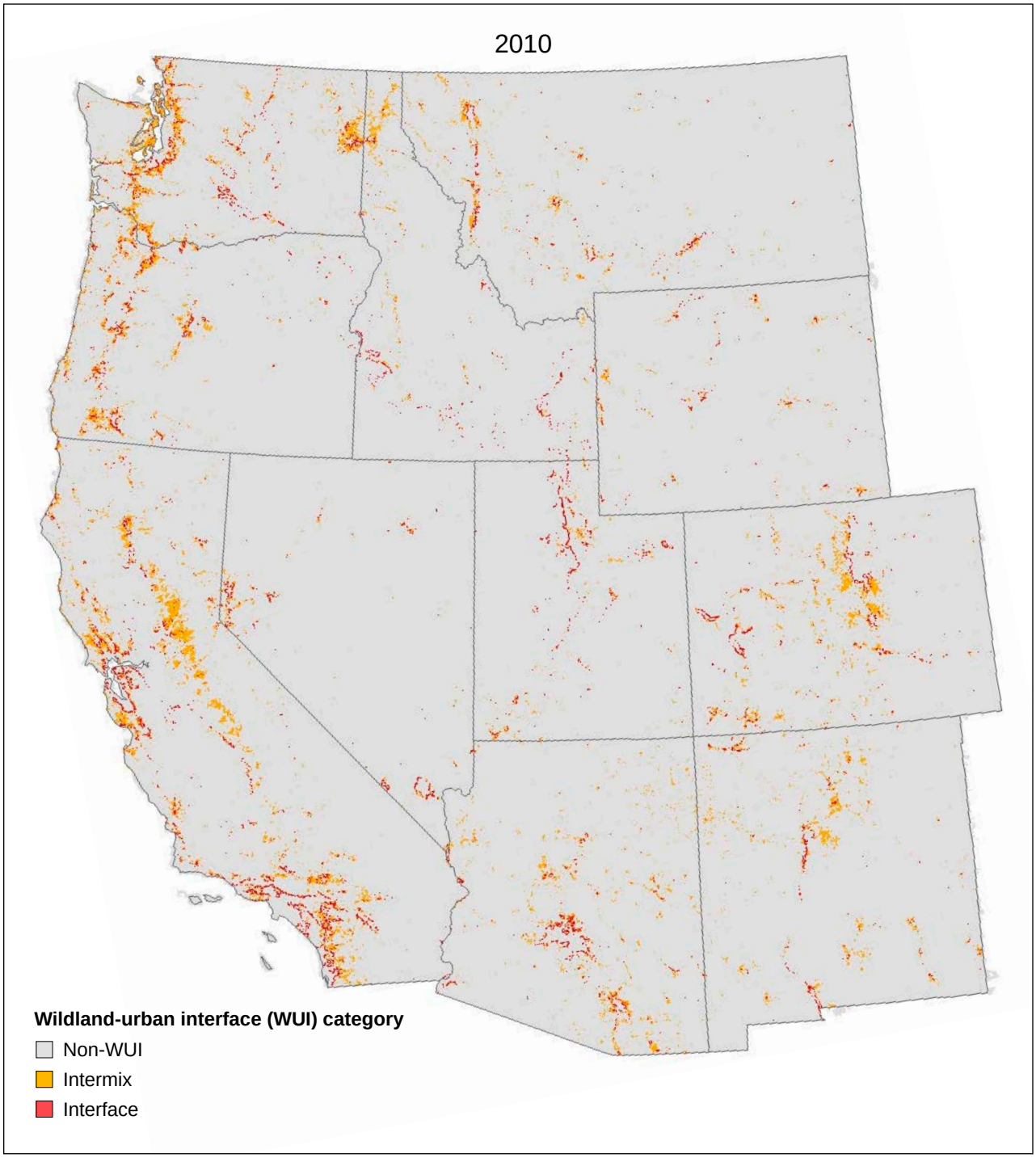

Figure 4.5—continued.

Vegetation and fuel changes—
Vegetation changes—including changes in overstory forest structure, understory vegetation, and plant species composition—also contribute to changes in fire activity over time by altering the amount, type, and connectivity of fuels within landscapes. These changes often occur slowly, making their effects on fire activity difficult to assess. However, the effects of exotic grass invasions and forest structural changes have been well documented.

Exotic grass invasions can significantly alter fuels and fire regimes in shrub-dominated and open forest communities by increasing fuel continuity in time and space (Brooks et al. 2004, D'Antonio and Vitousek 1992). Exotic grasses like cheatgrass (*Bromus tectorum*) and North Africa grass (*Ventenata dubia*) promote fire by growing quickly each year and generating highly flammable litter. In arid shrub-dominated lands, they can fill in gaps between shrubs, promoting wildfire spread and increasing fire frequency and area burned, particularly in years with ample precipitation (Balch et al. 2013, Brooks et al. 2004). In the Western States, cheatgrass cover exceeded 15 percent on 210 000 km^2 (31 percent) of the Great Basin and was associated with large wildfire occurrence, a doubling of wildfire area burned, and a quadrupling of area burned multiple times in a 16-year period (Balch et al. 2013, Bradley et al. 2018).

Long-term reductions in fire frequency have altered forest structure and fuel profiles in most regions of the West where local climates historically supported dry coniferous forests with frequent, low-severity fire regimes.

Long-term reductions in fire frequency have altered forest structure and fuel profiles in most regions of the West where local climates historically supported dry coniferous forests with frequent, low-severity fire regimes (Allen et al. 2002, Covington and Moore 1994, Hessburg et al. 2005). Humans have reduced fire frequency through active fire suppression and by disrupting fuel continuity through extensive grazing, land use changes, road building, and other forms of landscape fragmentation (Hessburg and Agee 2003, Richardson et al. 2007, Taylor et al. 2016). Longer fire-free intervals facilitated increases in forest stand density, development of multilayer forest canopies, increased abundance of fire-sensitive tree species, and greater accumulation of surface, ladder, and canopy fuels (Cooper 1960, Covington and Moore 1994, Hessburg and Agee 2003, Keane et al. 2002, Moore et al. 2004, Peterson et al. 2005). These changes in forest structure and fuels have greatly increased potential wildfire burn severity in many western forests (Hessburg et al. 2005, Peterson et al. 2005).

Wildfire and Other Disturbances

Natural and human disturbances influence future wildfire behavior and effects through their effects on fuels and forest structure. Wildfires, insect outbreaks, and severe droughts all injure or kill forest trees and alter fuels in ways that may influence future wildfire behavior and effects. Forest management practices can similarly alter forest structure and surface fuels, though outcomes will vary depending on management objectives and methods used.

Wildfires (and prescribed fires) in forests can inhibit future wildfires for up to a decade or more by consuming fuels, disrupting fuel continuity, and altering forest structure (Parks et al. 2015a). Inhibitory effects of fires on subsequent fire occurrence fade as postfire vegetation recovery and snag decay restore fuel continuity by replacing litter, fine woody fuels, and live biomass.

Wildfires also influence future fire severity through feedbacks on live and surface woody fuels. High-severity wildfires kill large numbers of trees, creating dead woody fuels that are deposited on the forest floor over time as the dead trees decay, thereby increasing potential fire severity (Peterson et al. 2015, Ritchie et al. 2013). In contrast, low-severity fires—and most prescribed fires—consume surface fuels without generating large amounts of new woody fuels, thereby limiting future fire severity (Safford et al. 2012, Schwilk et al. 2009).

Severe droughts and insect outbreaks kill trees without consuming surface fuels, so they have little or no effect on the probability of future wildfires or area burned (Hart et al. 2015, Mietkiewicz and Kulakowski 2016) and may reduce crown fire potential for a decade or more by reducing overstory canopy fuels (Hicke et al. 2012, Meigs et al. 2016). Over longer time frames, however, severe drought or insect outbreaks may increase potential wildfire severity, particularly in regard to forest soils, as dead trees decay and increase surface woody fuels (Monsanto and Agee 2008, Smith et al. 2017).

Forest restoration treatments—particularly those with fuel reduction and fire resilience objectives—often seek to modify overstory stand structure and surface woody fuels with the goal of reducing future wildfire severity (Agee and Skinner 2005, Allen et al. 2002). These treatments typically employ mechanical thinning of overstory trees and prescribed burning, alone or in combination (Allen et al. 2002, Schwilk et al. 2009). As noted earlier, prescribed fires are generally low-severity fires that consume surface fuels and thereby limit future fire severity for 10 to 15 years (Prichard et al. 2010, Safford et al. 2012). Mechanical thinning treatments can reduce potential fire severity by reducing the likelihood of crown fire occurrence but may also increase surface woody fuels if not combined with prescribed fire (Schwilk et al. 2009).

Wildfire Effects on Forest Resilience and Sustainability

Resilient forests can absorb disturbance impacts without incurring fundamental changes to their structure and function. In the case of fire disturbances, resilient forests can either avoid or resist (minimize) fire effects or endure and recover from fire effects over time without major, persistent changes in forest species composition or ecosystem processes. In many western forests, fire played an essential role in shaping historical forest conditions and was fully incorporated in ecosystem processes and species life cycles, so that the type of fire, rather than its simple presence or

absence, is the principal issue in regard to resilience and sustainability. Fire effects on forests—including tree survival, understory vegetation, soil properties and processes, and wildlife habitat functions—are more closely related to wildfire severity than to area burned in most cases. Natural postfire forest recovery depends on successful tree regeneration, which requires local seed sources and is therefore influenced by fire size and severity effects on residual tree densities and seed source distances.

Because most forest types are resilient to their natural fire regimes, uncharacteristic wildfires are what most threatens forest resilience and the sustainability of ecosystem services.

Because most forest types are resilient to their natural fire regimes, uncharacteristic wildfires are what most threaten forest resilience and the sustainability of forest ecosystem services. In fire-prone forests of the Western United States, particularly those dominated by ponderosa pines, the biggest threats to ecosystem resiliency are (1) uncharacteristically long fire return intervals, (2) uncharacteristically high wildfire severity, and (3) uncharacteristically large fire patches and area burned at high severity. Long periods of fire exclusion can allow establishment and canopy ingrowth of more shade-tolerant and fire-sensitive species, thereby increasing crown fire risks and tree mortality during wildfires. High-severity, stand-replacing wildfires can convert forests to grass- or shrub-dominated communities for decades to centuries by killing forest trees; reducing or removing local seed sources; and altering site microclimates, soil properties and processes, and future fuels. These high-severity fires may also serve as catalysts of long-term changes in forest structure and composition if low seed availability extends the forest establishment period and allows new species to become established, and if altered microclimate or the presence of new competitors reduces growth and survival rates of forest tree species that were present before the fire.

Wildfire and Delivery of Ecosystem Services

Wildfire directly affects several ecosystem goods and services, including forest products, carbon storage, recreation and tourism, wildlife habitat, and water and air quality (Seidl et al. 2016). Impacts can often be complicated, differing over time or affecting different segments of a community. For example, wildfire alters the sustainable production of timber products through inventory loss, which can reduce future supply and in turn increase future prices for owners of unburned timber, but it can also result in depressed prices in the short term, if salvaged log quantity is sufficient to affect local markets (Prestemon and Holmes 2008). Socioeconomic effects on local communities are often mixed, with costs associated with reductions in recreation and tourism but benefits from local capture of firefighting expenditures (Davis et al. 2014). Wildfire benefits some wildlife species, such as black-backed woodpeckers (*Picoides arcticus*) that depend on fire-killed snags (Hutto 1995), but wildfire can adversely affect other species, such as northern

spotted owls (*Strix occidentalis caurina*), when late-successional habitat is lost in high-severity fires (Lesmeister et al. 2018).

In other cases, impacts of wildfire on ecosystem services are more consistently negative. For example, forests help to ameliorate climate change through carbon sequestration, but wildfire events release large amounts of carbon dioxide into the atmosphere, with substantial associated economic impacts (see chapter 9). The removal of surface vegetation in wildfire can result in postfire flooding and debris flows. Wildfire smoke contains high amounts of very small particles, which are associated with a variety of human respiratory health ailments (Reid et al. 2016). Although it is difficult to quantify the amounts of exposure and health impacts (Kochi et al. 2010), during recent years, millions of people in the Western States have experienced reduced air quality from wildfire smoke.

Economic costs from wildfire have been increased not just by the increase in wildfire, but also by an expanding WUI, which occupies only about 2 percent of the area within fires that threaten buildings, but accounts for 69 percent of building loss (Kramer et al. 2018). Although suppression costs for fires in the WUI can be substantial, direct economic losses can greatly exceed them, and last for years after the fire. For example, in October 2017, a series of wildfires in California's wine country led to more than $10 billion in insured losses, destroyed 9,000 structures, and caused 44 deaths to become the most destructive fires in the state's history (Mass and Owens 2019). A little more than 1 year later, the Camp Fire broke that record when it overran the town of Paradise in the Sierra Nevada foothills, killing more than 80 people and destroying more than 18,000 structures (CAL FIRE 2018). Strategies that have been suggested for creating more fire-resilient communities include changes in building practices, community planning, emergency response systems, and fuels treatment (Calkin et al. 2014).

Conclusions

Wildfire area burned has increased in forests of the Western United States in recent decades, reflecting an increase in very large wildfires. Many factors are contributing to this increase in mean annual forest area burned, including (1) long-term changes in forest structure and fuels caused by reduced fire frequency over the past century or more; (2) human-caused wildfire ignitions that have expanded fire season lengths in many areas; and (3) increased frequency and severity of drought and severe fire weather. Wildfire area burned is still limited by active fire suppression efforts—for which annual expenditures continue to rise—and landscape fragmentation. As a result, most forest types in the West continue to experience a fire deficit relative to historical and natural fire regimes.

Factors contributing to the increase in area burned in forests of the Western United States include (1) long-term changes in fuels and forest structure; (2) human-caused wildfire ignitions; and (3) increased frequency and severity of drought and severe fire weather.

Resilience and sustainability of Western U.S. forests to wildfire depends on wildfire severity, area burned, and local climate, as these factors influence fire-induced tree mortality and postfire regeneration success. Restoration-based forest management can promote forest resilience and sustainability by using prescribed fires to reduce fuels and limit future fire severity, thinning overly dense forest stands to reduce crown fire potential and tree mortality, and using targeted postfire planting to supplement natural seed sources following high-severity wildfires. Such activities may be particularly useful within and around expanding WUI areas, where increasing human activity is increasing future wildfire potential (ignitions) and unacceptable wildfire effects on human health, property, and ecosystem services. Allowing wildfires to burn under moderate fire weather conditions may also help to reduce fuels in sparsely populated areas with natural fire regimes favoring low-severity fires. In mesic forests with low-frequency, high-severity fire regimes, continued fire suppression (limiting ignitions) and landscape fragmentation (limiting fire size) may be the most effective means for promoting forest sustainability and maintaining ecosystem services.

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Chapter 5: Drought

A. Lexine Long[1]

Introduction

Drought is generally defined as a moisture deficit over a given period of time. Pronounced drought conditions have occurred in the Western United States during recent years, affecting many aspects of ecosystems. Drought as a disturbance can play a major role in forest ecosystems, especially in the Western States (Cohen et al. 2016, Millar and Stephenson 2015, Teskey et al. 2015). Forest ecosystems are resilient to some level of drought; however, drought impacts on forest health may be becoming more severe (Clark et al. 2016a, Millar and Stephenson 2015, Vose et al. 2016). Understanding the role of drought as a disturbance in forest ecosystems will be increasingly important for forest management and sustainability. In this chapter, I review data on and ways of measuring drought and its impacts and provide several summary indices for recent drought in forests of the West.

Drought as a disturbance can play a major role in forest ecosystems in the Western States.

I define drought as a moisture limitation resulting from precipitation that is below the 30-year meteorological average for a region, temperatures above the 30-year meteorological average, or both. Drought is often characterized as one of four types: meteorological, hydrological, agricultural, or socioeconomic (Wilhite and Glantz 1985). Meteorological drought is when dry weather patterns occur in an area over a period of time and is usually defined by the duration of the dry period and the degree of dryness compared to average conditions. Hydrologic drought refers to when a shortage in water supply, such as in streams, ground water, or reservoirs, occurs because of a lack of precipitation. Agricultural drought refers to soil moisture deficits and is tied to crop productivity and mortality. Socioeconomic drought usually follows from both agricultural and hydrologic drought and refers to a drought's economic and societal impacts. In this chapter, I will primarily discuss impacts of meteorological drought on forest ecosystems.

Historical and Current Data

Documented increases in drier conditions and drought in the Western States have been noted in recent years, along with drought-related forest mortality (Allen et al. 2010, Littell et al. 2016, Mishra and Singh 2010, van Mantgem et al. 2013, Young et al. 2017). Quantifying the magnitude of drought and subsequent impacts on forest ecosystems can be complex (Vicente-Serrano et al. 2014). Measures of drought

[1] **A. Lexine Long** was an ORISE research fellow, Pacific Northwest Research Station, 1133 N Western Avenue, Wenatchee, WA 98801. Long currently is an associate planner, City of Wenatchee, Wenatchee, WA 98801.

One of the difficulties in measuring effects from drought on forests and trees is uncertainty in how much the drought impacts are due to evaporative demand as opposed to precipitation deficits.

are usually characterized by duration, temperature, and a precipitation deficit. However, the effects of drought on ecological processes in forests are multifaceted, and drought metrics that are based only on temperature and precipitation can be misleading in forests (Luce et al. 2016). One of the difficulties in measuring effects from drought on forests and trees is the uncertainty in how much the drought impacts are due to evaporative demand as opposed to precipitation deficits. Additionally, when one is looking at drought impacts, multiyear data showing the cumulative impacts are often needed. Here I review commonly used metrics to quantify drought in the West and the benefits and shortcomings of each.

Drought Indices

The standard precipitation index (SPI) and the Palmer drought severity index (PDSI) are two of the more common meteorological drought indices and incorporate both precipitation and duration using a surface air temperature model. SPI uses precipitation data only and calculates precipitation as a departure from modeled expected precipitation (Edwards 1997). Benefits of SPI are that it is a fairly straightforward metric, easier to calculate than PDSI, and more comparable across regions with different climates; however, it does not account for evapotranspiration. SPI is usually calculated over a 30-year period, and anomalies can be calculated for periods of 1 month up to multiple years. PDSI includes previous months to take cumulative drought effects into account; however, it is not able to capture how droughts change as a function of timescale (Palmer 1965, Vicente-Serrano et al. 2014). Another shortcoming of PDSI is that it calculates potential evapotranspiration (PE) solely from temperature data using the Thornwaite equation, which does not account for other factors that may affect PE such as radiation, wind changes, and humidity (Donohue et al. 2010, Sheffield et al. 2012). PDSI may not always be useful in demonstrating a soil moisture deficit for woody plants because estimates of evapotranspiration can vary regionally and are often developed with agricultural systems in mind (Anderegg et al. 2013a). Additionally, PDSI may not be very accurate in topographically complex systems (Anderegg et al. 2013a). A number of other drought indices are related to or can be derived from PDSI. Some studies have used PDSI to create a cumulative drought severity index (CDSI) (Peters et al. 2015). CDSI is a weighted sum of PDSI and can show cumulative drought over longer periods than PDSI (fig. 5.1). Other indices derived from PDSI include the Palmer Z index and the self-calibrated Palmer drought severity index (sc-PDSI), both of which are variations of PDSI. The standardized precipitation evapotranspiration index (SPEI) is similar to SPI, but it takes into account potential evapotranspiration in addition to precipitation (Vicente-Serrano et al. 2010). SPEI can be calculated on different time scales, ranging from 1 to 48 months.

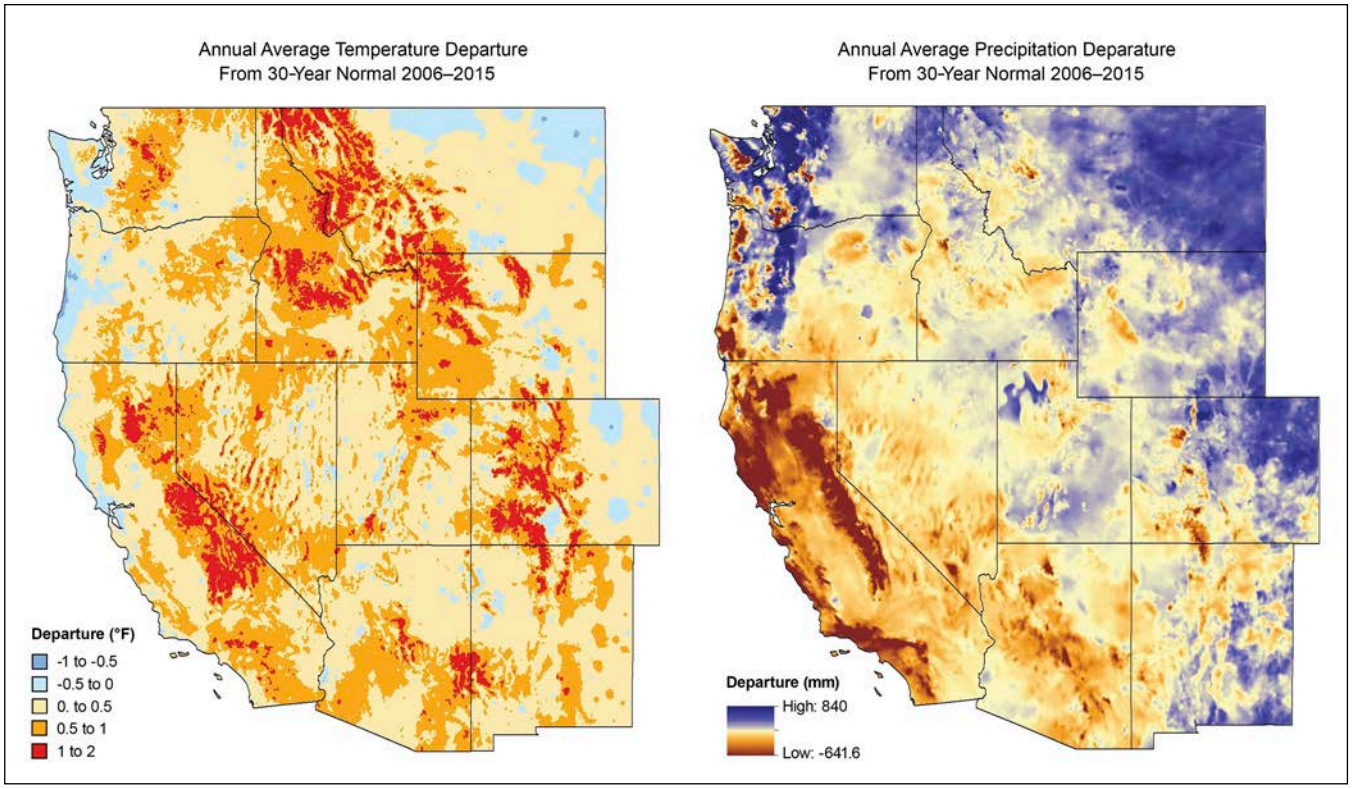

Figure 5.1—Temperature and precipitation departures for the study period (2006–2015) from the 30-year 20th century average (1975–2005) using PRISM climate data (http://www.prism.oregonstate.edu/).

Other Methods of Measuring Drought

In addition to using meteorological drought indices that rely on temperature and precipitation, there are other ways to quantify and assess the magnitude and impact of drought. Many drought indices are not able to show the role of precipitation versus evaporative demand in determining drought. Williams et al. (2012) used tree rings to compile a database of forest stress for trees in the Southwest to create a forest drought severity index (FDSI). By examining tree rings, one can examine the variability in environmental conditions that cause forest stress and limit growth. Specifically, the FDSI shows the contributions to drought stress not only from precipitation but also from vapor-pressure deficits.

Mildrexler et al. (2016) developed a forest vulnerability index (FVI) across a 10-year period in the Pacific Northwest region to assess where and when forests are most vulnerable to drought-related stress during the growing season. To calculate the FVI, MODIS (moderate-resolution imaging spectroradiometer) satellite imagery and PRISM (parameter-elevation relationships on independent slopes model) precipitation data were used, along with a water balance model. Although the FVI can be useful for identifying where forests may experience drought stress, it was

developed only for Oregon and Washington and is not available for the rest of the Western United States.

The Forest Inventory and Analysis (FIA) program collects data on forested lands in the United States for use in forest management and planning. FIA data can be useful in assessing broad-scale forest health and impacts resulting from disturbances such as drought. FIA data for the West does not yet include complete remeasurement data, so information on drought-related tree mortality is limited (Clark et al. 2016a). FIA field crews recorded drought as a disturbance for an estimated 0.53 million ha of forest between 2006 and 2015, with the highest levels recorded in 2014 and 2015. States with the highest level of drought-affected forest were New Mexico (about 0.2 million ha), and Arizona and California (0.1 million ha each). This is likely to be a substantial underestimate of acres affected, particularly for California, as there is a time lag in the estimates owing to the decadal measurement cycle (Clark et al. 2016a).

The U.S. Forest Service Pacific Southwest Region Aerial Detection Monitoring program annually maps areas of recent tree mortality in California using aerial observation (Young et al. 2017). As the data resolution is approximately 3.5 km, this dataset can be used only to analyze patterns of broad-scale tree mortality. Young et al. found that regionally in California, tree mortality was greater in areas that were hotter and drier. Although these data can be useful for looking at broad-scale patterns of tree mortality, they are available only for California; however, the results may be applicable to other parts of the West with similar climates.

Trends in Recent Climate and Drought Indices for the Western United States

In the 2006-2015 decade, average annual temperature was higher across the Western United States, but below-normal precipitation was more regionally diverse.

In the last decade (2006–-2015), the average annual temperature has been higher than the preceding normal (1975–2005) across the Western United States (fig. 5.1). However, average annual precipitation in the last decade has been more variable, with below-normal precipitation concentrated in the Southwest (Arizona, California, and Nevada) (fig. 5.1). These regional differences are also reflected in the PDSI, which indicates repeated instances of severe and extreme drought in the past decade for the southwest region (Arizona, California, Nevada, and Utah) and the interior West (Colorado, Montana, New Mexico, and Wyoming), but not for the Pacific Northwest (Idaho, Oregon, and Washington) (fig. 5.2). The CDSI also indicates that recent drought impacts have been spatially variable, with the highest impacts in northern Arizona, southern California, northern Nevada, and southeast Oregon (fig. 5.3). In contrast, most of Idaho, Oregon, Montana, and Washington have had relatively low CDSI scores (fig. 5.3). In summary, drought has been an important forest disturbance in the past decade, but regionally rather than throughout the West.

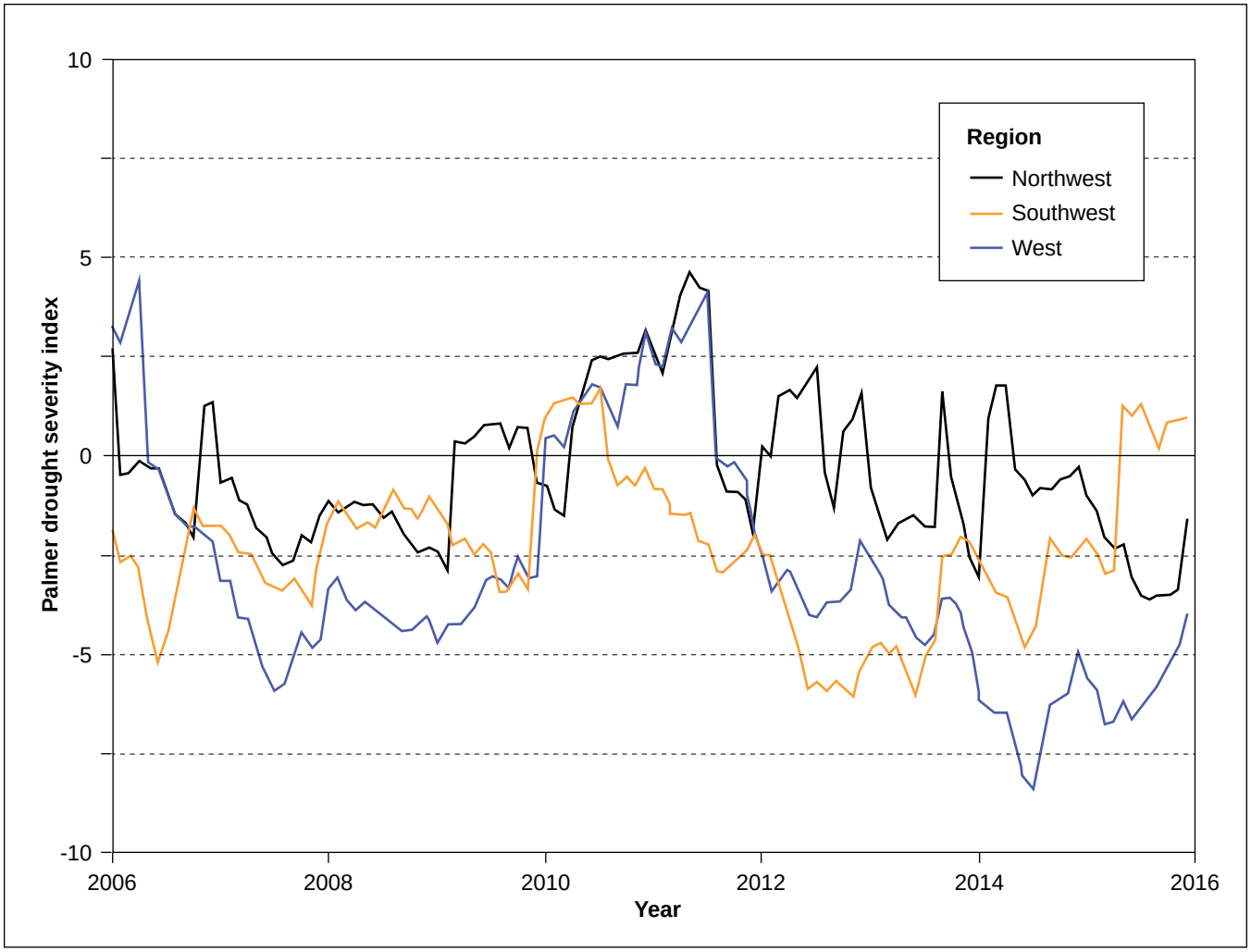

Figure 5.2—Palmer drought severity index by region for the 10-year study period, (2006–2015). Northwest region includes Idaho, Oregon, and Washington; the Southwest region includes Arizona, California, Nevada, and Utah, and the West region includes Colorado, Montana, New Mexico, and Wyoming. Positive values indicate wet conditions, whereas negative values indicate dry conditions. Values between -3.0 to -3.99 and below -4 are considered severe and extreme drought, respectively.

Implications and Impacts

Direct impacts to trees and forest stands from drought can include drought-induced mortality (Adams et al. 2009, 2012; Allen et al. 2015; Michaelian et al. 2011; Wang et al. 2012), reduced growth, leaf area decline, altered forest bio-geochemical cycling, altered hydrologic processes, altered tree recruitment, and altered forest structure and composition (Allen et al. 2010, Anderegg et al. 2013b, Baker and Williams 2015, Bigler et al. 2007, Breshears et al. 2005, Clark et al. 2016a). Hotter temperatures and decreased snowpack or earlier melt associated with drought can increase tree water stress (Millar and Step 2016). If drought occurs early in the growing season, trees may lose their leaves prematurely,

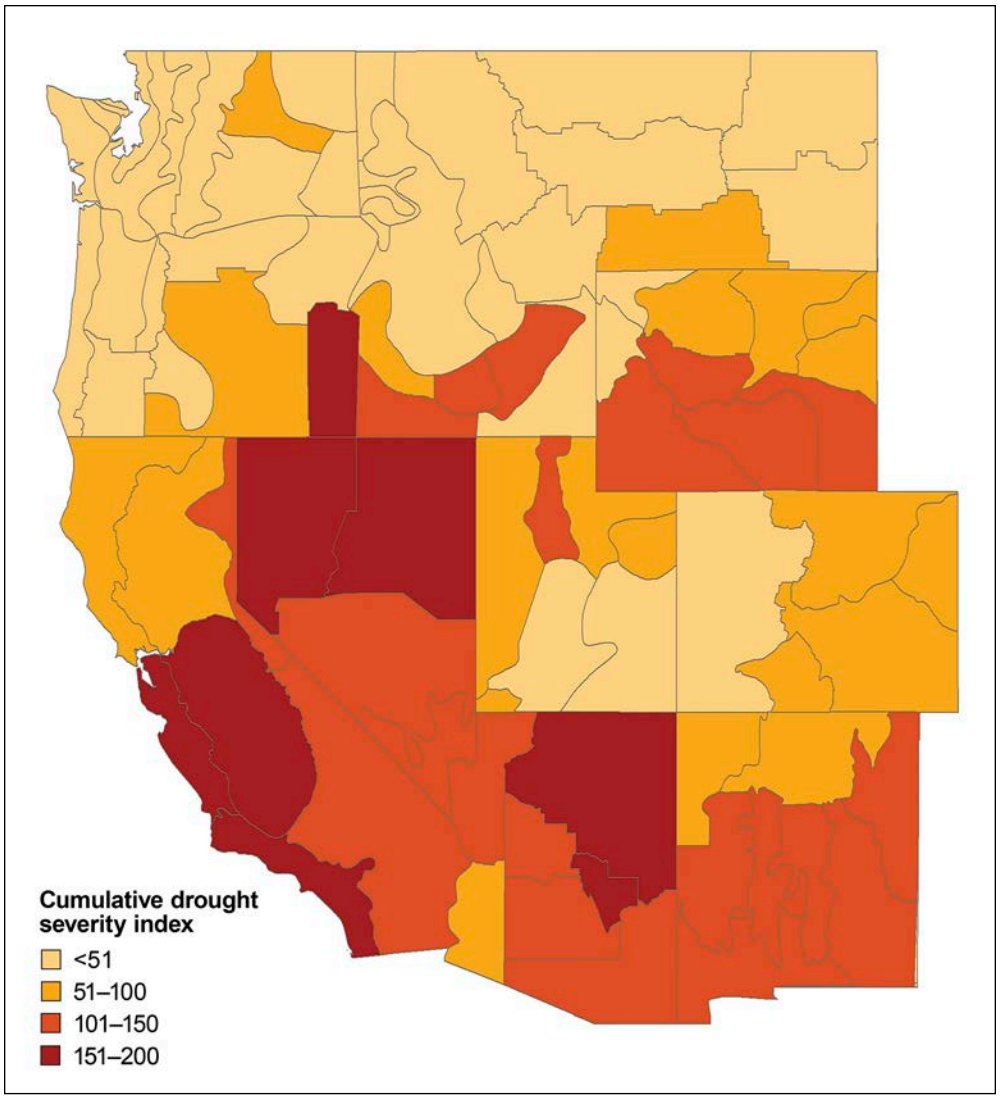

Figure 5.3—Cumulative drought severity index (CDSI) by climate division for the study period (2006–2015). Re-created using methods from Clark et al. 2016 and Peters et al. 2015.

Both the Palmer Drought Severity Index and the Cumulative Drought Severity Index show increased drought during 2006-2015 but with varying severity for different regions within the Western United States.

which can reduce nutrient uptake (Schlesinger et al. 2016). Owing to the Pacific Northwest's Mediterranean climate of wet winters and dry summers, its forests rely on snowpack for summer moisture. Snowpack that melts earlier in the season will result in longer dry periods during the summer for forests in the Western States (Luce et al. 2014). Allen et al. 2010 reviewed global literature on drought-related tree mortality and found more than 150 references, with 88 documented examples of forest mortality resulting from either heat or water stress (Allen et al. 2010). Increases in mortality can reduce the ecosystem services provided by forests, such as carbon storage (Allen et al. 2015, Schlesinger et al. 2016). Because forests store carbon, increases in forest mortality can result in a rise in

carbon released in the atmosphere as trees die off (Allen et al. 2010, Schwalm et al. 2012).

Assessing impacts of drought on forests can be challenging, and much uncertainty remains, especially in some of the mechanisms that lead to drought-induced tree mortality (Allen et al. 2010, McDowell 2011). The physiological mechanisms and relationship to climate variables that drive tree mortality under drought are complex, often involve multiple factors, and result in a nonlinear threshold process (Adams et al. 2009, Allen et al. 2010, Anderegg et al. 2013a, McDowell et al. 2008). Because of this complexity, predicting patterns of regional tree and stand mortality resulting from drought can be difficult (Allen et al. 2010). Additionally, the susceptibility of forests to drought is largely determined by site characteristics (soil texture, etc.), and stand characteristics, which can differ greatly and can often be hard to assess across broad scales (Seidl et al. 2011). Furthermore, field observations of tree mortality that also have associated data showing water stress leading to the mortality is uncommon, thus it can also be difficult to show direct linkages between mortality and drought (Breshears et al. 2009). Trees are slow growing, but mortality can occur quickly, sometimes causing rapid ecosystem changes (Allen et al. 2010). These changes can include changes in tree species composition, understory species composition, hydrologic processes, forest biogeochemical cycling, and carbon budgets (Allen et al. 2010).

Most drought research is on impacts to individual trees, and fewer studies focus on stand-level impacts (Clark et al. 2016a). Drought can have different effects on individual trees as opposed to stands, and predicting stand-level impacts may often be more useful for forest management purposes. Drought-induced tree mortality can have effects on community composition (Anderegg et al. 2012). Understanding and predicting how drought affects hydrological processes is also challenging, as it is necessary to understand the physiological processes happening in individual trees, but also stand dynamics at a watershed scale (Vose et al. 2016). Another area of uncertainty is the effect of drought timing and seasonality of forest mortality (Anderegg et al. 2013a). Although there is research in arid ecosystems showing that seasonality of precipitation can have effects on the net primary production and biomass in grassland and herbaceous-dominated systems, little research exists on these effects on woody plants (Anderegg et al. 2013a, Zeppel et al. 2014). Using dendrochronological studies to look at when mortality occurred can assist with investigating the impacts of drought seasonality on tree mortality (Bigler et al. 2007). However, studies such as these are limited and more research on the effects of drought seasonality on woody plants is needed (Anderegg et al. 2013a).

Interactions With Other Disturbances

Drought stress and increased temperatures can make forests more susceptible to insects and pathogens, and extended drought conditions can increase the chances and severity of wildfire.

Drought interacts with other forest disturbances and stressors, often in ways that can compound each other (Millar and Stephenson 2015). Drought stress and increased temperature can make forests more susceptible to insects and pathogens (Breshears et al. 2009, Gaylord et al. 2013, Hart et al. 2014, Millar and Stephenson 2015), and increases in bark beetle outbreaks in the Western United States in recent years may be partially related to intense drought conditions (Hart et al. 2014, Kolb et al. 2016, Kurz et al. 2008, Sherriff et al. 2011) Warming temperatures often associated with drought can result in increases on insect populations (Millar and Stephenson 2016). Drought can increase plant tissue concentrations of nitrogen compounds, and nitrogen is important in insect metabolism (Kolb et al. 2016). Bark beetles have become the most common biotic agent of tree mortality in the West, and have affected a larger area than wildfires. The relationship between drought and bark beetle outbreaks may be nonlinear, in which moderate droughts can reduce outbreaks, but longer, sustained droughts can increase outbreaks (Kolb et al. 2016).

Extended drought conditions can increase the chances and severity of wildfires (Bigler et al. 2005, Littell et al. 2016). For example, van Mantgem et al. (2013) found that high pre-wildfire water deficits were related to post-wildfire mortality rates. Mortality resulting from drought can increase fuel loads as dry conditions can result in abundant dry litter (Clark et al. 2016a, Williams et al. 2012). Additionally, the seasonality of precipitation can have effects on wildfire cycles. For example, an unusually wet spring in grass-dominated ecosystems can produce a surplus of fine fuels from grasses, and if a wet spring is followed by a dry period, this can increase wildfire risk (Littell et al. 2009).

Severe drought has also been associated with increases in atmospheric pollution, such as surface ozone and $PM_{2.5}$; however, research directly linking drought and increases in atmospheric pollution is relatively limited (Wang et al. 2017). Atmospheric pollutants can affect forest ecosystems in various ways. Pollutants such as nitrogen and carbon dioxide can act as fertilizers and sometimes assist in tree growth. However, in excessive concentrations, atmospheric pollutants can also negatively affect forest health by reducing growth, defoliating trees, and contributing to mortality (Millar and Stephenson 2015).

Impacts to Forest Sustainability

More severe droughts may continue to cause impacts to forest health and sustainability. Understanding the impacts that increased drought can have on forest health and how drought interacts with other disturbance agents will be critical for forest management. Although it is clear that drought is having impacts on forest ecosystems,

large knowledge gaps remain on the impact of drought on forest stands. Forests will likely continue to experience prolonged or more severe drought in the future. For future research, a better understanding is needed of how multiple stressors and disturbance interact with forest drought responses, impacts from drought seasonality, mechanisms leading to drought mortality, and stand-level impacts (Clark et al. 2016b, Vose et al. 2016). Forest management practices, including thinning and other silvicultural treatments, and wildfire and fuel management, may need to be adapted to account for changes from prolonged or more severe drought.

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Chapter 6: Insects

Christopher J. Fettig, Robert A. Progar, Jeanine Paschke, and Frank J. Sapio[1]

Introduction

Insects are essential components of forest ecosystems, representing most of the biological diversity and affecting virtually all ecological processes (Schowalter 1994). Most species are beneficial (Coulson and Witter 1984, Haack and Byler 1993), yet others periodically become so abundant that they threaten ecological, economic, social or aesthetic values at local to regional scales (tables 6.1 through 6.3). Insects influence forest ecosystem structure and function in complex and dynamic ways, for example, by regulating certain aspects of primary production; nutrient cycling; ecological succession; and the size, distribution and abundance of plants and other animals (Mattson 1977, Mattson and Addy 1975). Effects on forest vegetation range from being undetectable, to short-term reductions in crown cover, to modest increases in background levels of tree mortality, to extensive amounts of tree mortality observed at regional scales.

Table 6.1—Bark beetles regarded as primary disturbance agents in western coniferous forests

Common name	Scientific name	Common host(s)
California fivespined ips	*Ips paraconfusus*	*Pinus contorta, P. lambertiana, P. ponderosa*
Douglas-fir beetle	*Dendroctonus pseudotsugae*	*Pseudotsuga menziesii*
Fir engraver	*Scolytus ventralis*	*Abies concolor, A. grandis, A. magnifica*
Jeffrey pine beetle	*Dendroctonus jeffreyi*	*P. jeffreyi*
Mountain pine beetle	*Dendroctonus ponderosae*	*P. albicaulis, P. contorta, P. flexilis, P. lambertiana, P. monticola, P. ponderosa*
Northern spruce engraver	*Ips perturbatus*	*Picea glauca, Pi. × lutzii*
Pine engraver	*Ips pini*	*P. contorta, P. jeffreyi, P. lambertiana, P. ponderosa*
Pinyon ips	*Ips confusus*	*P. edulis, P. monophylla*
Spruce beetle	*Dendroctonus rufipennis*	*Pi. engelmannii, Pi. glauca, Pi. × lutzii*
Western balsam bark beetle	*Dryocoetes confusus*	*A. lasiocarpa*
Western pine beetle	*Dendroctonus brevicomis*	*P. coulteri, P. ponderosa*

Note: all insect species listed above are native to western North America.

[1] **Christopher J. Fettig** is a research entomologist, Pacific Southwest Research Station, 1731 Research Park Drive, Davis, CA 95618; **Robert A. Progar** is a research entomologist, Pacific Northwest Research Station, 1401 Gekeler Lane, La Grande, OR 97850; **Jeanine Paschke** is a contract geographic information system analyst, Forest Health Assessment and Applied Science Team, and **Frank J. Sapio** is assistant director (retired), Forest Health Protection, 2150 Centre Avenue, Building A, Suite 331, Fort Collins, CO 80526. **Rob Progar** is now the national entomology and pathology program leader, Research and Development, 201 14th Street Southwest, Washington, DC 20250.

Interactions With Other Disturbances

Forest insects are directly and indirectly influenced by other biotic (e.g., forest insects and diseases), abiotic (e.g., weather, wildfires, avalanches and windstorms), and anthropogenic (e.g., management activities and land use patterns) disturbances; other disturbances are often influenced by forest insects. As an example, consider bark beetles and wildfire, the principal drivers of change in western coniferous forests. Bark beetles alter forest fuels by killing varying numbers of trees of specific age classes, size classes, and species with consequences to the frequency, intensity, and severity of subsequent wildfires (Jenkins et al. 2014a). Conversely, sublethal heating of critical tree tissues by low- to moderate-intensity wildfires predispose trees to successful colonization by bark beetles because of weakening of their defensive mechanisms (Parker et al. 2006). This, in turn, may cause localized increases in bark beetle populations that lead to additional levels of tree mortality, with feedback loops that further influence fire risk and severity (Jenkins et al. 2014a). Furthermore, some bark beetle epidemics have been so severe that they have converted forests from carbon sinks to carbon sources, further contributing to climate change (Kurz et al. 2008). Climate change and fire suppression activities, among other factors, have resulted in substantial changes in forest conditions over millions of hectares in the Western United States, expressed primarily as increases in the density of shade-tolerant and fire-intolerant trees (e.g., true firs, *Abies* spp.) (Stine et al. 2014), further exacerbating both disturbances (Fettig et al. 2007, 2013; Westerling et al. 2006). These interactions have sparked concerns about the sustainability of some western forests to maintain the numerous ecological goods and services on which human societies have come to rely (Morris et al. 2017, 2018), and have motivated increases in the pace and scale of treatments designed to increase forest resilience to these disturbances (Stephens et al. 2018).

Current and Projected Trends in the Activity and Impacts of Forest Insects

This chapter reviews the ecology and impacts of 22 prominent insect species in the Western United States.

In the following sections, we review the ecology, recent impacts (2000–2016), and projected future impacts (based on the "2013–2027 National Insect and Disease Forest Risk Assessment") of forest insects. We limit discussion to what we consider the 22 most prominent species in the Western United States in three feeding guilds (tables 6.1 through 6.3) based on their impacts to forest structure and composition. Most are native to western North America, with a few notable exceptions (tables 6.2 and 6.3).

Table 6.2—Defoliating insects regarded as primary disturbance agents in western coniferous forests

Common name	Scientific name	Common host(s)
Douglas-fir tussock moth	*Orgyia pseudotsugata*	*Pseudotsuga menziesii, Abies* spp., *Picea* spp.
Forest tent caterpillar	*Malacosoma disstria*	*Populus tremuloides,* other hardwoods
Larch casebearer	*Coleophora laricella*	*Larix occidentalis*
Pandora moth	*Coloradia pandora*	*Pinus contorta, P. ponderosa*
Pine butterfly	*Neophasia menapia*	*P. ponderosa*
Pine sawflies	*Neodipridon* spp., *Zadiprion* spp.	*P. ponderosa, P. contorta*
Western hemlock looper	*Lambdina fiscellaria lugubrosa*	*Tsuga heterophylla, Abies balsamea, Picea* spp.
Western spruce budworm	*Choristoneura freemani*	*Ps. menziesii, Abies* spp., *Picea* spp.
Western tent caterpillar	*Malacosoma californicum*	*Po. tremuloides,* other hardwoods

Note: *Malacosoma* spp. also occur in deciduous forests and shrublands. All insect species listed above are native to western North America except larch casebearer.

Table 6.3—Sap-feeding insects regarded as primary disturbance agents in western coniferous forests

Common name	Scientific name	Common hosts
Balsam woolly adelgid	*Adelges piceae*	*Abies* spp.
Spruce aphid	*Elatobium abietinum*	*Picea engelmanni, Pi. sitchensis*

Note: both species are invasive to western North America.

Bark Beetles

Trees of all species, ages, and size classes may be colonized and killed by bark beetles, with the greatest impacts observed in conifers (table 6.1). Each bark beetle species exhibits unique host preferences, life history traits, and impacts. Some cause extensive levels of tree mortality as demonstrated by mountain pine beetle (*Dendroctonus ponderosae*) in several pines (most notably lodgepole pine (*Pinus contorta* var. *latifolia*) (fig. 6.1); western pine beetle (*Dendroctonus brevicomis*) in ponderosa pine (*Pinus ponderosa*); Douglas-fir beetle (*Dendroctonus pseudotsugae*) in Douglas-fir (*Pseudotsuga menziesii*); and spruce beetle (*Dendroctonus rufipennis* Kirby) in spruce (*Picea* spp.). Others, such as several species in the genus *Ips* (Wood 1982), are secondary agents that usually colonize stressed, dead, or dying trees.

Bark beetles often inflict density-dependent mortality (i.e., population growth rates and associated levels of tree mortality are partially regulated by the density of suitable hosts) (fig. 6.2), and help maintain a diversity of tree species, ages, sizes, and spatial heterogeneity (Fettig 2012). At endemic populations, bark beetles create small gaps in the forest canopy by killing trees stressed by age, drought, defoliation, or other factors. In this context, few negative impacts are observed

C. Fettig, Pacific Southwest Research Station

Figure 6.1—The mountain pine beetle is the most significant forest insect in North America. Most large-scale epidemics occur in lodgepole pine forests in a nearly contiguous pattern across extensive areas. British Columbia, Canada, 2005.

C. Fettig, Pacific Southwest Research Station

Figure 6.2—Thinning has long been advocated as a preventive measure to alleviate or reduce the amount of ponderosa pine mortality attributed to bark beetles, Shasta County, California, 2005. Among other factors, thinning reduces host density; reduces competition among trees for nutrients, water, and other resources, thereby increasing vigor; and affects microclimate, decreasing the effectiveness of chemical cues used in host finding, selection, and colonization.

(Morris et al. 2017). This differs from the impacts associated with epidemics, which may negatively affect timber and fiber production, water quality and quantity, fish and wildlife populations, recreation, grazing capacity, biodiversity, endangered species, carbon sequestration and storage, and cultural resources, among others (Morris et al. 2018). During the last three decades, the amount of tree mortality attributed to bark beetles in the Western States has exceeded that of wildfire (Hicke et al. 2016), and several recent epidemics are recognized as being among the most severe in recorded history (Bentz et al. 2009). Because bark beetles are highly sensitive to thermal conditions conducive to population survival and growth (Bentz et al. 2010), and drought stress negatively affects host tree vigor (Kolb et al. 2016), epidemics have been correlated with recent shifts in temperature and precipitation attributed to climate change. Forest densification has exacerbated the effect (Fettig et al. 2007).

The impact of wood borers, a related group, pales in comparison to that of bark beetles, but there are a few species, such as the California flatheaded borer (*Phaenops californica*), and flatheaded fir borer (*Phaenops drummondi*), that can cause noticeable levels of tree mortality in western coniferous forests (Furniss and Carolin 1977). Wood borers commonly infest trees stressed or killed by other agents and serve a very important ecological function by helping to facilitate wood decomposition and nutrient cycling (fig. 6.3), but given their minor role as primary disturbances will not be considered further.

Mountain pine beetle, *Dendroctonus ponderosae*—
Mountain pine beetle is regarded as the most significant forest insect in North America, and colonizes 15 tree species, primarily lodgepole pine, ponderosa pine, sugar pine (*P. lambertiana*), limber pine (*P. flexilis*), western white pine (*P. monticola*) and whitebark pine (*P. albicaulis*) (Negrón and Fettig 2014). The geographic distribution generally reflects the range of its primary hosts, although lodgepole pine extends farther to the north and ponderosa pine and other pines farther to the south than where mountain pine beetle populations currently exist (Bentz et al. 2010). Historically, the range of mountain pine beetle was restricted by climatic conditions unfavorable to brood development, but is expanding as a result of climate change and other factors. Populations were detected in Nebraska in 2009 (Costello and Schaupp 2011); in Alberta, Canada, in 2003 (Cudmore et al. 2010); and in the Northwest Territories, Canada, in 2012 (Natural Resources Canada 2013). Scientists have expressed a concern that mountain pine beetle could expand farther eastward across the boreal forest of Canada and into the Eastern United States (Safranyik et al. 2010). However, Bentz et al. (2010) described the probability of such a range expansion as low during this century.

The range of mountain pine beetle, regarded as the most significant forest insect in North America, is expanding.

C. Fettig, Pacific Southwest Research Station

Figure 6.3—Bark beetles serve as keystone species that facilitate colonization of trees by other organisms. Of note, subsequent colonization by wood borers helps expedite decomposition and nutrient cycling. Lassen National Forest, California, 2005.

With few notable exceptions (Hopkins 1905), most large-scale epidemics occur in lodgepole pine forests. Mountain pine beetle initially colonizes the largest lodgepole pine in a stand (e.g., >23 cm diameter at breast height [d.b.h.]), which provides a higher reproductive potential and probability of beetle survival (Graf et al. 2012), with progressively smaller trees being colonized over time (Klein et

al. 1978). In ponderosa pine, the small- to mid-diameter classes (e.g., 20 to 40 cm d.b.h.) tend to be colonized most frequently (Olsen et al. 1996), but this may be an artifact of these forests being less dense, less continuous, and exhibiting a higher diversity of stand ages and tree sizes than is observed in lodgepole pine forests (Fettig et al. 2014). There is usually one generation per year (Bentz et al. 2014).

Since 2000, ~10.3 million ha have been affected by mountain pine beetle (fig. 6.4), which represents almost half of the total area affected by all bark beetles combined in the Western States. Activity began to increase substantially in 2004, peaked in 2009 when 3 578 591 ha were affected, and has declined since then (fig. 6.5). Despite this, mountain pine beetle is still ranked the most damaging forest insect in the 2013−2027 risk assessment (Krist et al. 2014), with a projected loss of ~65.8 million m^2 of basal area occurring between 2013 and 2027 (fig. 6.6). Historically, mountain pine beetle epidemics were not usual, but the magnitude and extent of recent events may have exceeded the range of historic variability in some cases (Bentz et al. 2009), and have occurred at high elevations where they formerly were uncommon (see "Impacts to Sustainability").

Western pine beetle, *Dendroctonus brevicomis*—
Western pine beetle is a significant cause of ponderosa pine mortality. The only other primary host in the West is Coulter pine (*P. coulteri*), a species indigenous to the Transverse and Peninsular Ranges of southern California (Miller and Keen 1960). Western pine beetle generally exhibits a preference for larger diameter (>50 cm d.b.h.) trees, but under certain conditions, such as extended drought, may colonize and kill trees of all ages and size classes (Fettig 2016). There are usually two to four generations per year.

Since 2000, there have been only 3 years when <40 469 ha were affected by western pine beetle. In response to a severe, prolonged drought, activity recently increased in 2014 and peaked in 2016 when 892 041 ha were affected (fig. 6.5), mostly in California (846 580 ha) (fig. 6.7). In some areas of the central and southern Sierra Nevada, tree mortality has exceeded 90 percent, and type conversions have been observed (Fettig et al. 2019), prompting California Governor Jerry Brown to declare a state of emergency over concerns about public health and safety. This event may foreshadow future impacts of western pine beetle as the intensity and duration of droughts, important inciting factors (Kolb et al. 2016), and the distribution of ponderosa pine are expected to increase in the future (Rehfeldt et al. 2006). Western pine beetle is ranked the eighth most damaging forest insect in the 2013−2027 risk assessment (Krist et al. 2014), with a projected loss of ~15.4 million m^2 of basal area (fig. 6.6).

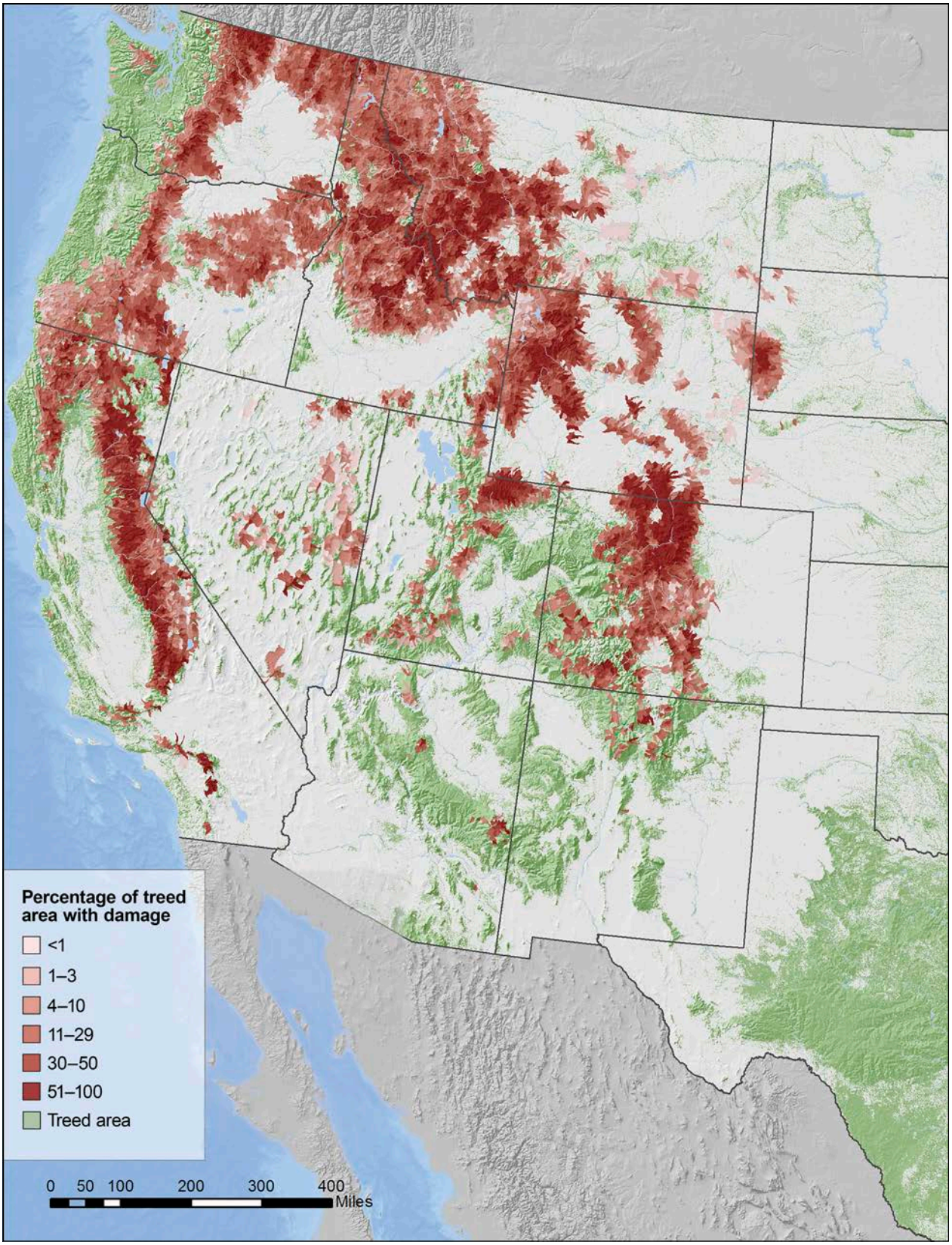

Figure 6.4—Area and severity of tree mortality attributed to mountain pine beetle, 2000–2016 (percentage of treed area by sub-watersheds, 6th-level hydrologic unit codes). Data are from the U.S. Forest Service National Insect and Disease Survey database.

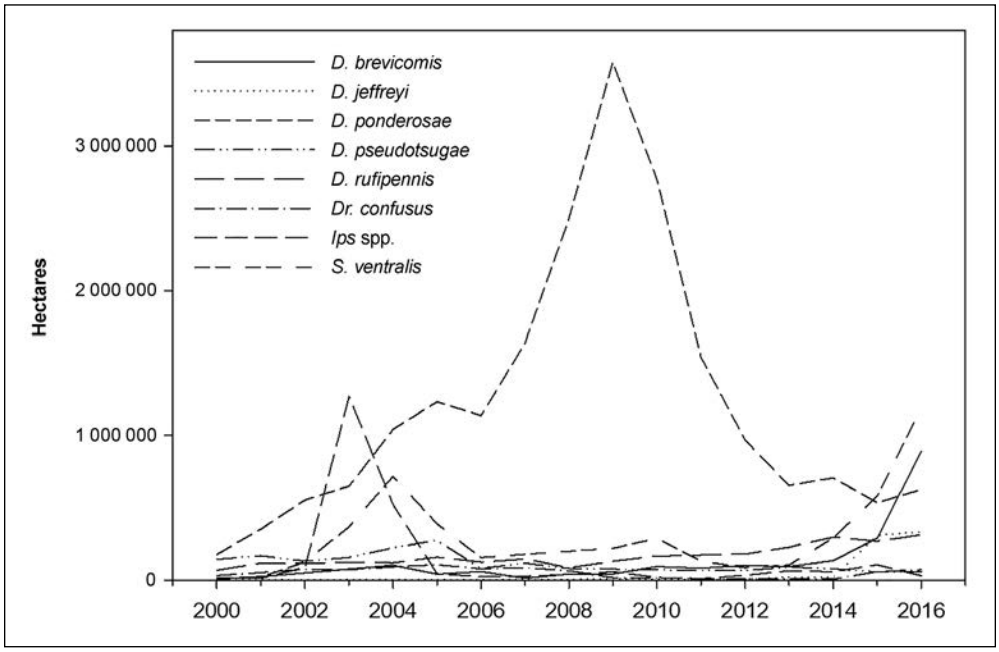

Figure 6.5—Area affected by year, 2000–2016. Values represent the impact observed each individual year and should not be summed across years (i.e., there may be overlap in areas affected from year to year). Data are from the U.S. Forest Service National Insect and Disease Survey database.

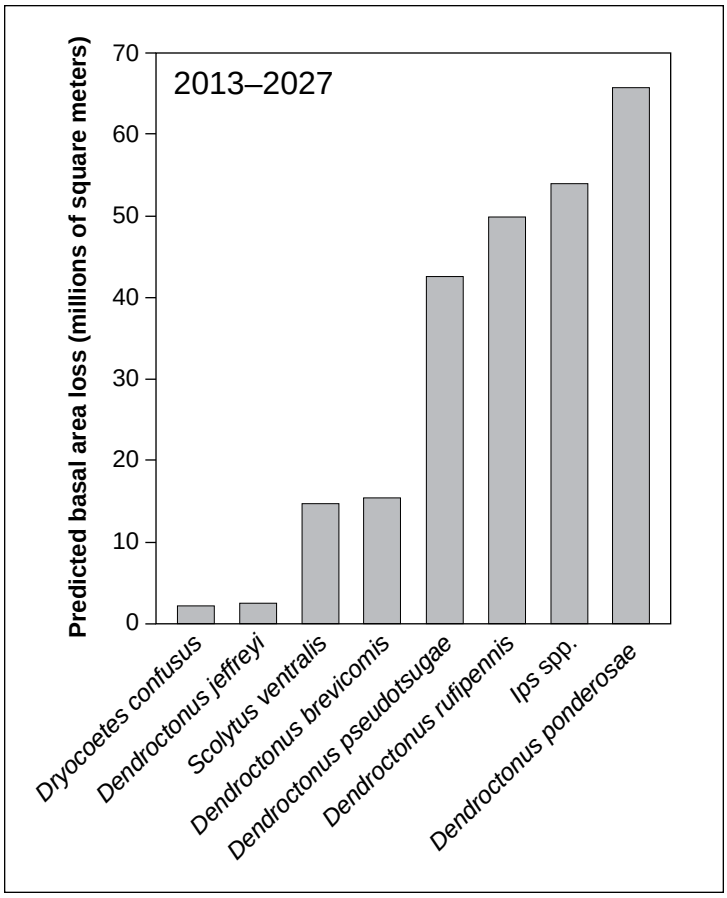

Figure 6.6—Projected losses, 2013–2027. *Ips* spp. including some species from the Eastern United States. Data are from the National Insect and Disease Forest Risk Assessment, 2013–2027 (Krist et al. 2014).

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Figure 6.7—Western pine beetle is a primary disturbance agent in ponderosa pine forests, Sierra National Forest. California, 2016. In 2016, 846 580 ha were affected by western pine beetle in California alone.

Jeffrey pine beetle, *Dendroctonus jeffreyi*—

Jeffrey pine beetle colonizes only Jeffrey pine (*P. jeffreyi*), a species that ranges from the Klamath Mountains in southwestern Oregon to throughout much of the Sierra Nevada and the Transverse and Peninsular Ranges in southern California, to the Sierra San Pedro Mártir in Baja California, Mexico. The species usually colonizes individual trees, and its activity often goes unnoticed, but during extended droughts large groups of trees may be killed (Fettig 2016). Although Jeffrey pine beetle is a significant source of disturbance in forests containing Jeffrey pine, the limited distribution and abundance of the host marginalizes the beetle's overall impact. There are one to two generations per year (Furniss and Carolin 1977). In most years, fewer than a few thousand hectares are affected (fig. 6.5), and consequently Jeffrey pine beetle is ranked the 15th most damaging forest insect in the

2013−2027 risk assessment (Krist et al. 2014), with a projected loss of ~2.6 million m^2 of basal area (fig. 6.6). Based on research focused on the Lake Tahoe Basin, California, Scheller et al. (2018) suggested that epidemics of Jeffrey pine beetle will increase in severity in the future.

Spruce beetle, *Dendroctonus rufipennis*—

Spruce beetle is the most significant mortality agent of mature spruce in the West and ranges throughout western North America, across the boreal forest of Canada, and into the Northeastern United States. Primary hosts include Engelmann spruce (*Picea engelmannii*) in the Intermountain West (fig. 6.8), and white spruce (*Pi. glauca*), Lutz spruce (*Pi.* × *lutzii*), Sitka spruce (*Pi. sitchensis*), and occasionally black spruce (*Pi. mariana*) in Alaska. Disturbances that produce an abundance of downed spruce, including timber harvests, blowdowns, landslides, and snow avalanches, have been implicated in the occurrence of most epidemics (Jenkins et al. 2014b). There are 0.3 to 1 generations per year (Furniss and Carolin 1977).

William M. Ciesla

Figure 6-8—Spruce beetle is the most significant mortality agent of mature spruce in the Western United States. Bridger-Teton National Forest, Wyoming, 2016.

Spruce beetle is the most significant mortality agent of mature spruce in western North America.

During the 1990s, a spruce beetle outbreak occurred on the Kenai Peninsula in south-central Alaska of a magnitude that had not been experienced in recorded history in North America for any bark beetle (Werner 1996), only to be surpassed by mountain pine beetle (fig. 6.5). At the peak, >485 000 ha were affected in a single year. In more recent years, activity increased in 2013 and peaked in 2016, when 313 260 ha were affected (fig. 6.5), mostly in Colorado (141 122 ha), south-central Alaska (76 096 ha), and Utah (50 292 ha). At the time of this writing, spruce beetle populations are undergoing a dramatic increase in Alaska.

Spruce beetle is ranked the fourth most damaging forest insect in the 2013–2027 risk assessment (Krist et al. 2014), with a projected loss of ~49.8 million m^2 of basal area (fig. 6.6). High summer temperatures are correlated with an increase in the proportion of spruce beetle that complete a generation in a single year (Hansen and Bentz 2003), contributing to population growth and observed levels of tree mortality in some populations (Bentz et al. 2010, Berg et al. 2006). Overall, models suggest that future epidemics will be favored by increasing temperatures (Bentz et al. 2010), but projected reductions in the range of Engelmann spruce in the Intermountain West (Rehfeldt et al. 2006) could result in reduced impacts in that region.

Douglas-fir beetle, *Dendroctonus pseudotsugae*—
Douglas-fir beetle is the most important biotic disturbance affecting Douglas-fir forests in the Rocky Mountains (fig. 6.9). Occasionally western larch (*Larix occidentalis*) is colonized. When populations are at endemic levels, Douglas-fir beetle colonizes recently killed trees or trees weakened by defoliation, root disease, ice, fire, or wind. Under normal environmental conditions, small groups of trees are killed, but when large numbers of stressed trees occur, populations may increase in these trees and spread to adjacent healthy trees (Furniss et al. 1979, Furniss and Kegley 2014). The coastal Douglas-fir region, ranging from northern California to British Columbia, has sporadic epidemics of short duration that usually develop following extensive windthrow or large fires. There is one generation per year (Furniss and Carolin 1977). In most years, 80 000 to 100 000 ha are affected (fig. 6.5). In recent years, activity peaked in 2005 when 273 826 ha were affected (fig. 6.5). Douglas-fir beetle is ranked the fifth most damaging forest insect in the 2013–2027 risk assessment (Krist et al. 2014), with a projected loss of ~42.5 million m^2 of basal area (fig. 6.6).

Western balsam bark beetle, *Dryocoetes confusus*—
Western balsam bark beetle is a significant source of disturbance in spruce-fir forests. Its range extends from British Columbia and Alberta south to Arizona and New Mexico (Wood 1982). This species primarily colonizes subalpine fir

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Figure 6.9—Douglas-fir beetle is a primary disturbance agent in Douglas-fir forests in the Western United States. Boise National Forest, Idaho, 2017.

(*Abies lasiocarpa*), although Pacific silver fir (*A. amabilis*), white fir (*A. concolor*), and Engelmann spruce are also hosts (Bright 1963). Western balsam bark beetle preferentially colonizes trees of reduced vigor, and blowdowns appear important in inciting epidemics (McMillin et al. 2003). There is 0.5 generation per year (Negrón and Popp 2009). Activity peaked in 2004 when 105 771 ha were affected (fig. 6.5), primarily in Montana (66 908 ha). Western balsam bark beetle is ranked the 18th most damaging forest insect in the 2013−2027 risk assessment (Krist et al. 2014), with a projected loss of ~2.1 million m^2 of basal area (fig. 6.6).

Fir engraver, *Scolytus ventralis*—

Fir engraver colonizes true firs, particularly white fir. Trees of all sizes may be colonized and killed, but epidemics are typically associated with trees stressed by drought, defoliation (e.g., by Douglas-fir tussock moth (*Orgyia pseudotsugata*), root pathogens, or other factors (Berryman and Ferrell 1988, Ferrell et al. 1994). Large numbers of trees may be killed by fir engraver following prescribed fire (Fettig and McKelvey 2014). There is one generation per year throughout much of the range (Furniss and Carolin 1977). As was observed for western pine beetle, fir engraver activity increased in 2014 in response to a prolonged, severe drought, and peaked in 2016 when 1 186 737 ha were affected (fig. 6.5). Most of this activity occurred in

California (1 070 920 ha) (Fettig et al. 2019). Fir engraver is ranked the ninth most damaging forest insect in the 2013–2027 risk assessment (Krist et al. 2014), with a projected loss of ~14.8 million m^2 of basal area (fig. 6.6).

Engraver beetles, *Ips* spp.—
There are about 25 species of *Ips* in the Western United States (Furniss and Carolin 1977). In most years, fewer than 40 000 ha are affected, but activity peaked in 2003 when 1 271 139 ha were affected in the West (by all *Ips* species) (fig. 6.5) owing largely to pinyon ips (*I. confusus*) in the Southwest. Collectively, *Ips* spp. are ranked the third most damaging forest insect(s) in the 2013–2027 risk assessment (Krist et al. 2014), with a projected loss of ~53.9 million m^2 of basal area (fig. 6.6). However, this figure also includes contributions of several species from the Eastern United States.

In the Western States, most notable is the pine engraver (*I. pini*), which has a transcontinental distribution and is one of the most common bark beetles in North America (Kegley et al. 1997). Like most *Ips*, this species usually infests slash, saplings, and weakened trees (table 6.1). Top killing of ponderosa pine is common, and often facilitates colonization by western pine beetle or mountain pine beetle. Colonization rates are negatively correlated with tree diameter in ponderosa pine (Kolb et al. 2006), and trees 5 to 20 cm d.b.h. are most frequently colonized. Infestations of pine engraver are often short lived but may increase in scale and duration when suitable host material is plentiful and populations grow to sufficient numbers to kill apparently healthy trees. There are one to two generations per year (Furniss and Carolin 1977).

The distribution of the northern spruce engraver (*I. perturbatus*), generally coincides with that of its primary host, white spruce. Other hosts include Engelmann spruce, Lutz spruce, and black spruce. Although populations have been recorded in Idaho, Maine, Michigan, Minnesota, Montana, Washington, and nearly all the Canadian provinces, northern spruce engraver exerts its greatest impacts in Alaska (Burnside et al. 2011). In most years, endemic populations infest forest debris, widely scattered individual trees, or small groups of trees. However, natural (e.g., flooding, wildfire, and windstorms) and anthropogenic (e.g., road building, construction of utility rights-of-way, and logging) disturbances may produce large quantities of stressed, dead, and dying spruce that serve as ideal hosts for northern spruce engraver. Spring drought conditions led to epidemics on the Kenai Peninsula in the 1990s (Holsten 1998). Increased use of mechanical fuel reduction treatments and low-cost wood energy systems, and elevated summer temperatures, have been implicated in epidemics in Interior Alaska (Fettig et al. 2013a). Furthermore, the mean temperature in interior Alaska is projected to increase an additional 3 to 7 °C by the end of this century (Walsh et al. 2008) with only modest increases in precipi-

tation that likely will be insufficient to offset increases in evapotranspiration. As a result, levels of drought stress in white spruce are expected to increase (Wolken et al. 2011) enhancing conditions favorable to northern spruce engraver. There is one generation per year (Burnside et al. 2011).

California fivespined ips (*I. paraconfusus*) occurs from southern Oregon to southern California and east to the crest of the Sierra Nevada and Cascade mountain ranges (Furniss and Carolin 1977) (fig. 6.10). Recently, populations were recorded in Washington (Murray et al. 2013). All pines occurring within the range

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Figure 6.10—Like most *Ips* spp., California fivespined ips readily colonizes logging slash where populations may rapidly increase and then emerge to colonize adjacent trees. Tahoe National Forest, California, 2004.

of California fivespined ips are susceptible to colonization, especially ponderosa pine. Like other *Ips*, endemic populations infest forest debris, widely scattered individual trees or small groups of trees. Occasionally, epidemics result in mortality of large numbers of trees but are usually associated with improper slash management or drought. California fivespined ips is also a vector of the fungus *Fusarium circinatum*, which causes pitch canker disease in Monterey pine (*P. radiata*) in California and other pines worldwide (Wingfield et al. 2008). There are two to five generations per year (Furniss and Carolin 1977). In most years, a few thousand hectares are affected. Activity peaked in 2008, when 24 130 ha were affected.

Pinyon ips (*I. confusus*) is a major cause of mortality in pinyon pine (*P. edulis* and *P. monophylla*) (Furniss and Carolin 1977). Epidemics frequently occur in the Great Basin and Southwestern United States and are usually associated with forest densification and drought (Kleinman et al. 2012, Negrón and Wilson 2003, Shaw et al. 2005). There are two to four generations per year (Furniss and Carolin 1977). In most years, fewer than 20 235 ha are affected, but activity peaked in 2003 when 1 188 785 ha were affected. These numbers likely under report the amount of activity as aerial detection surveys typically do not cover most of the pinyon-juniper woodlands.

Defoliators

Forest defoliators consume, mine, or skeletonize the foliage of trees, but their impacts pale in comparison to bark beetles. In most cases, eggs are laid on the buds, foliage, branches or boles of trees. Larval feeding may result in tree mortality depending on the timing, frequency, and extent of feeding (table 6.2). Most frequently defoliation retards growth, decreases tree vigor, and increases the amount of light reaching the forest floor, influencing understory and mid-story dynamics. Trees of all species, ages, and size classes may be defoliated, but each defoliator exhibits unique host preferences, life history traits, and impacts. Only two species, western spruce budworm (*Choristoneura freemani*) (also *C. occidentalis*) (see Gilligan and Brown 2014) and Douglas-fir tussock moth cause extensive levels of tree mortality. Predators and parasitoids have a strong regulatory effect on their populations, resulting in long time lags between epidemics.

Unlike the case of bark beetles, there appear to be few consistent trends among factors that incite forest defoliators. The cyclic nature of epidemics has prompted the formulation of a number of hypotheses that attempt to explain associated mechanisms, including genetic variation, qualitative variation, climatic release, food (host) quality deterioration or improvement, and diseases (parasitoids and pathogens), among others (Berryman 1996, Myers 1988). Drought has been implicated as an inciting factor (Mattson and Haack 1987), but a recent synthesis

concluded that defoliators exhibit no consistent response to drought (Kolb et al. 2016). The projected future impacts of some notable species (e.g., pine butterfly [*Neophasia menapia*]) were not considered in the 2013−2027 risk assessment (Krist et al. 2014), and thus are absent in the discussion below.

Western spruce budworm, *Choristoneura freemani*—

Western spruce budworm is the most important defoliator in western North America (Brookes et al. 1987) (fig. 6.11). The species feeds on Douglas-fir, true firs, and occasionally spruce and western larch (Furniss and Carolin 1977). Its geographic range coincides with these hosts from British Columbia and Alberta south to Arizona and New Mexico and east to Colorado. Budworm populations erupt episodically over large regions. Defoliation continues for several years to more than a decade, then the insect becomes relatively rare for years to decades (Swetnam and Lynch 1993). This synchrony seems to be associated with favorable weather conditions, warm dry summers in conjunction with synchrony of larval emergence and bud flush, and the influences of natural enemies (Peltonen et al. 2002, Thomson et al. 2012). In mature stands, the most common impact is reduced growth, although repeated defoliation sometimes results in top kill and tree mortality. Severely defoliated trees are often colonized and killed by Douglas-fir beetle and fir engraver.

Outbreaks of western spruce budworm, the most important defoliator in western North America, are episodic.

L. Spiegel, USDA Forest Service Forest Health Protection

Figure 6.11—Western spruce budworm larvae are important defoliators of true firs and Douglas-fir.

Larvae also feed on staminate flowers and developing cones affecting regeneration. There is one generation per year (Furniss and Carolin 1977).

In recent years, western spruce budworm activity peaked in 2009 when 2 066 266 ha were affected (fig. 6.12), mostly in Idaho and Montana. The impact of western spruce budworm may be increasing in some areas (e.g., eastern Oregon) because of extensive logging of ponderosa pine, which favors Douglas-fir (Swetnam et al. 1995). Other research suggests that the duration and intensity of epidemics in western Montana have increased as a result of a decrease in the frequency of wildfires attributed to fire suppression efforts, causing increased host species abundance and multistoried stands (Anderson et al. 1987). Epidemics have also been linked to drought or synchrony of larval development with foliage phenology (Campbell et al. 2006, Williams and Liebhold 1995), although other studies have found that epidemics were associated with wetter conditions at the end of droughts that increase food

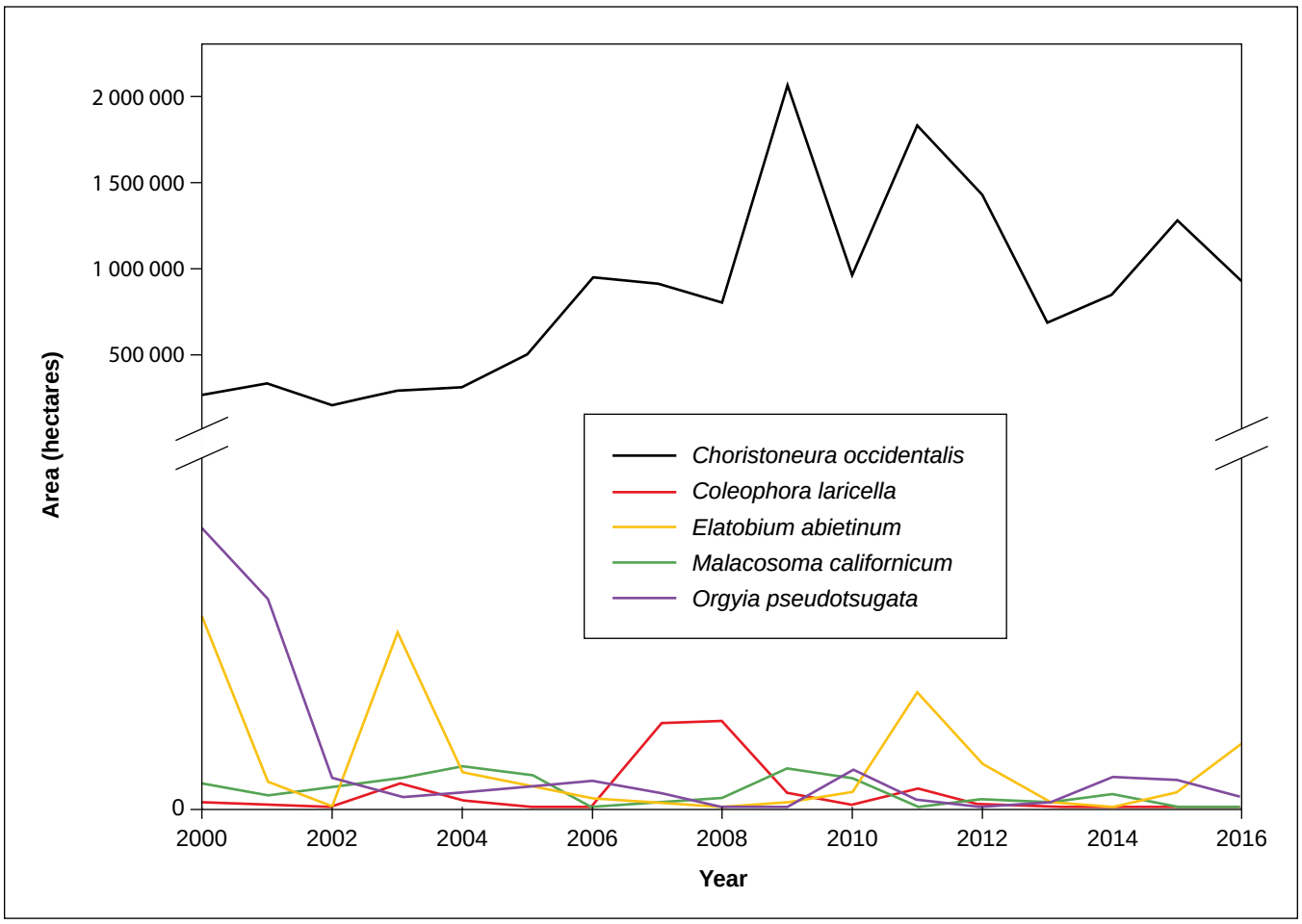

Figure 6.12—Area affected by year, 2000–2016. Values represent the impact observed each individual year and should not be summed across years (i.e., there may be overlap in areas affected from year to year). Data are from the USDA Forest Service National Insect and Disease Survey database. Several other species discussed in the text are not included owing to incomplete data.

resources (Flower et al. 2014, Ryerson et al. 2003, Swetnam and Lynch 1993). Western spruce budworm is ranked the 11th most damaging forest insect, second among native defoliators to eastern spruce budworm (*C. fumiferana*) in the 2013–2027 risk assessment (Krist et al. 2014), with a projected loss of ~10.8 million m^2 of basal area. As observed for some bark beetles, western spruce budworm epidemics may increase fire risk and severity in affected stands (Hummel and Agee 2003, Ryerson et al. 2003, Schowalter 1986).

Douglas-fir tussock moth, *Orgyia pseudotsugata*—
Douglas-fir tussock moth is an important defoliator of true firs and Douglas-fir (fig. 6.13). The species consumes the foliage of several tree species (table 6.2), but only Douglas-fir, white fir, and grand fir (*A. grandis*) are considered primary hosts (Brooks et al. 1978). Epidemics develop quickly and then subside usually in 1 to 2 years, but some have persisted for longer periods (Schaupp et al. 2008). Defoliation by Douglas-fir tussock moth kills or top kills many trees, making them more susceptible to colonization by Douglas-fir beetle and fir engraver. There is one generation per year (Furniss and Carolin 1977).

T. Coleman, USDA Forest Service Forest Health Protection

Figure 6.13—Douglas-fir tussock moth is an important defoliator of true firs and Douglas-fir. Santa Fe National Forest, New Mexico, 2016.

Activity peaked in 2000 when 116 000 ha were affected (fig. 6.12), primarily in Idaho, Oregon, and Washington. Since 2001, the amount of defoliation attributed to Douglas-fir tussock moth has declined substantially, with the exception of 2011 and 2012 when 47 600 and 19 270 ha were affected, respectively. Historically, epidemics of Douglas-fir tussock moth in California were limited to the Sierra Nevada. However, an epidemic occurred in the Transverse Mountain Ranges of southern California in 2009 that was primarily attributed to fire suppression efforts resulting in significant increases in the density and continuity of white fir (Coleman et al. 2014). As a result, the potential for elevated impacts in southern California is of concern. Douglas-fir tussock moth is ranked the 16th most damaging forest insect in the 2013–2027 risk assessment (Krist et al. 2014), with a projected loss of ~2.6 million m^2 of basal area.

Pine butterfly, *Neophasia menapia*—
Pine butterfly is a periodic defoliator of ponderosa pine in the Pacific Northwest (fig. 6.14) and Rocky Mountains, although Douglas-fir, western white pine, and lodgepole pine may also be colonized (Keen 1952). In most years, a few adults can be seen fluttering around the tops of pines in late summer and early fall, but evidence of defoliation from larval feeding is absent or inconspicuous. Repeated defoliation can reduce tree growth, making trees more susceptible to other disturbances, such as western pine beetle. Occasionally, severe (>75 percent) defoliation may result in tree mortality. There is one generation per year (Furniss and Carolin 1977). Historically, epidemics have been recorded in British Columbia, Idaho, Montana, Oregon, and Washington (Bousfield and Meyer 1972, Evenden 1926, Keen 1952). In recent years, activity peaked in 2011 when 101 343 ha were affected, mostly in Oregon.

Larch casebearer, *Coleophora laricella*—
Larch casebearer is native to Europe (Jagsch 1973), but was introduced into North America, likely on nursery stock. The species was first detected in Northampton, Massachusetts, in 1896, in Ottawa, Canada, in 1905 (Otvos and Quednau 1981), and in western North America near St. Maries, Idaho, in 1957 (Tunnock and Ryan 1983), and has since spread to Alberta, British Columbia, Montana, Oregon, and Washington. Larch casebearer feeds on the internal tissue of western larch needles, favoring younger trees growing in the open or along forest edges (Tunnock and Ryan 1983). Repeated defoliation can result in growth loss and tree mortality (Ryan et al. 1987). There is one generation per year (Furniss and Carolin 1977).

In eastern North America, larch casebearer was the subject of a very successful classical biological control program (i.e., the introduction of a natural enemy of

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Figure 6.14—Pine butterfly is a periodic defoliator of ponderosa pine in the Pacific Northwest and Intermountain West. Malheur National Forest, Oregon, 2010.

exotic origin to control an invasive species). As such, several nonnative parasitoids were released in the Western United States for control of larch casebearer starting in the 1960s (Ryan 1990). By the 1980s, two species, *Agathis pumila* and *Chrysocharis laricinellae*, were well established throughout the region, significantly reducing populations of larch casebearer (Ryan 1997). In recent years, activity peaked in 2008 when 36 130 ha were affected in the Western United States (fig. 6.12), but usually only 10 000 to 20 000 ha are affected annually.

Western hemlock looper, *Lambdina fiscellaria lugubrosa*—
Western hemlock looper colonizes western hemlock (*Tsuga heterophylla*). Epidemics usually occur in coastal and interior wet belt regions of the Pacific Northwest in mature hemlock and hemlock-cedar stands. During an epidemic, western hemlock looper will feed on other tree species (table 6.2), and some broad-leaved trees and shrubs (Jardine 1969). Populations periodically increase and persist for 1 to 2 years, resulting in the death of large numbers of trees over limited but well-defined areas. There is one generation per year. Of note, 7571 ha were affected in Idaho in 2011.

Pine sawflies, *Neodiprion* spp.—
Pine sawflies are common defoliating insects of pines, consisting of 16 species in the Western States (Ciesla and Smith 2011, Ross 1955). Sawflies normally occur at low densities (e.g., a few individuals per tree); however, on occasion some species become epidemic, causing defoliation on a vast scale for one or more years (Furniss and Carolin 1977). Most species exhibit a preference for open-grown trees. For example, a study in Arizona showed that defoliation by *Neodiprion autumnalis,* the most widely distributed species that feeds on ponderosa pine, was limited to stands with <4.6 m^2/ha of basal area (McMillin et al. 1996). There is one generation per year. Although good records are not available on the impact of sawflies in the West, most recent activity is reported from Arizona and Colorado.

Pandora moth, *Coloradia pandora*—
Pandora moth larvae feed on the foliage of several pines, primarily ponderosa, lodgepole, and Jeffrey pines throughout much of the West (Furniss and Carolin 1977). Epidemics usually occur in mature stands and cause extensive defoliation, leading to growth loss and tree mortality. When tree mortality occurs, it is often associated with prolonged drought, dwarf mistletoe infection, or colonization by bark beetles (Wagner and Mathiasen 1985). Some epidemics encompass large areas but occur only where soils are loose enough for larvae to bury themselves where they pupate. Based on observations of 20th-century epidemics and interviews with American Indians who use the larvae and pupae as food, epidemics typically recur every 20 to 30 years and last about 6 to 8 years (Furniss and Carolin 1977). However, Clark et al. (2017) reconstructed epidemics from a 1,572-year (435 to 2006) ponderosa pine chronology in central Oregon and reported that epidemics occurred on average every 104 years. Pandora moth has 0.5 generation per year, with feeding and moth flight occurring in alternate years so that most of the defoliation occurs every other year. Although good records are not available on the impact of pandora moth, most recent activity is reported from Arizona.

Forest tent caterpillar, *Malacosoma disstria*, and western tent caterpillar, *Malacosoma californicum*—

Forest tent caterpillar is found throughout the United States and Canada, where preferred hosts include numerous broad-leaved trees (Batzer and Morris 1978). This species often defoliates extensive areas during epidemics, resulting in severe (>75 percent) growth loss, but tree morality is generally rare. Interestingly, analyses of historical data (1950–1984) from northern Ontario, Canada, indicate that host abundance such as that of aspen (*Populus tremuloides*) has no effect on the duration of epidemics (Roland 1993). Instead, the duration of epidemics was best predicted by the amount of forest edge per unit area (Roland 1993). Because forest tent caterpillar populations are regulated largely by parasitoids and pathogens (Witter and Kulman 1979), this suggests that forest fragmentation negatively affects interactions between these natural enemies and forest tent caterpillar (Roland and Kaupp 1995). If so, increased fragmentation of forests in the Western United States may exacerbate future epidemics. There is one generation per year (Batzer and Morris 1978). Forest tent caterpillar is ranked the 14th most damaging forest insect in the 2013−2027 risk assessment (Krist et al. 2014), with a projected loss of ~2.7 million m^2 of basal area nationwide. Among Western States, activity peaked in 2004 when 4938 ha were affected in Utah.

Like forest tent caterpillar, western forest tent caterpillar colonizes a wide range of tree and shrub species, but aspen is preferred (Furniss and Carolin 1977). Successive years of defoliation causes reduced growth and fruit production, branch dieback, top kill, and in rare cases, tree mortality. In Colorado, entire aspen stands across large areas have been defoliated. During epidemics, western forest tent caterpillar may only be one of several disturbance agents (including other insect defoliators and leaf diseases) that cause widespread defoliation of aspen. There is one generation per year. In recent years, activity peaked in 2003 when 16 828 ha were affected (fig. 6.12), mostly in New Mexico.

Sap-Sucking Insects

Aphids and adelgids comprise a large group of small (usually <4 mm in length), soft-bodied insects that are frequently found sucking sap from leaves and stems of plants. Most overwinter as eggs, which hatch in spring into females that reproduce parthenogenetically (asexually) and birth live young. This unique reproductive trait results in rapid changes in population densities over time (Minks and Harrewijn 1987). Sap-feeding insects are particularly susceptible to desiccation in drier climates. More mesic habitats are found to support higher populations (Progar and

Schowalter 2002), which are usually kept in check by numerous natural enemies and abiotic factors. Generally, feeding is manifested as needle stunting, needle chlorosis, and stem deformation. Several aphid species are also regarded as vectors of notable plant viruses (Minks and Harrewijn 1987). Aphids often go unnoticed in forests because of their small size and cryptic behavior. Adelgids are important pests of several conifers and are closely related to aphids (Havill and Foottit 2007).

Balsam woolly adelgid, *Adelges piceae*—
Balsam woolly adelgid is a nonnative, invasive insect that threatens true fir species throughout North America. The species was introduced into eastern North America from Europe before 1900 (Foottit and Mackauer 1980), and subsequently near San Francisco, California, in 1928 (Ragenovich and Mitchell 2006). In the Western States, it is now well established in Oregon and Washington, but continues to move eastward through Idaho and into Montana, threatening subalpine fir (fig. 6.15). A similar trend is observed in British Columbia (Zilahi-Balogh et al. 2016). In 2017, balsam woolly adelgid was detected for the first time in Utah, and now populations have been confirmed in several counties within the state (Alston et al. 2018). Hrinkevich et al. (2016) developed a climatic risk model for balsam woolly adelgid, and reported

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Figure 6.15—Balsam woolly adelgid is a nonnative, invasive insect that threatens fir species throughout North America. Feeding causes swelling or gouting on new growth. Wallowa-Whitman National Forest, Oregon, 2012.

climatic susceptibility decreases from the Olympic Peninsula in Washington and the Cascade Range in Oregon and Washington eastward, with the exception of some high-risk areas in northern Idaho and western Montana. There is also a pattern of decreasing climatic suitability from north to south in the Rocky Mountains.

In response to feeding by balsam woolly adelgid, the host produces a type of compression wood in the sapwood that inhibits waterflow within the tree, eventually leading to tree death. There are usually two generations per year (McMullen and Skovsgaard 1972). Balsam woolly adelgid is ranked the 10th most damaging forest insect, first among invasives in the Western United States in the 2013–2027 risk assessment (Krist et al. 2014), with a projected loss of ~11.3 million m^2 of basal area. In recent years, activity in the West peaked in 2010 when 79 757 ha were affected (fig. 6.12), mostly in Oregon. Balsam woolly adelgid is likely to become a more important disturbance agent in forests of Idaho, Montana, and Utah (Hrinkevich et al. 2016).

Spruce aphid, *Elatobium abietinum*—

Spruce aphid causes chlorosis, defoliation, and mortality of spruce (fig. 6.16) but has also been recorded infesting pine and Douglas-fir (Furniss and Carolin 1977). This insect, an exotic invasive from Europe, was first reported in British Columbia in 1916 (Carter and Halldórsson 1998), and has since spread throughout coastal British Columbia, southeast Alaska, Oregon, and Washington as well as several mountain ranges in Arizona and New Mexico (Lynch 2014). The highly dispersive nature of alates (winged individuals) allows populations to spread rapidly to the limits of its climatic tolerances (e.g., temperatures below -10 °C in maritime areas).

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Figure 6.16—Defoliation of Sitka spruce by spruce aphid in southeast Alaska, 2005.

High densities can lead to severe defoliation, but needle loss is generally restricted to 1-year-old and older needles in spring and summer. In the fall and winter, feeding occurs on current as well as older needles, and high populations at this time of year are capable of causing complete defoliation of the tree. Young trees that have been heavily defoliated during the winter often have terminal buds that fail to break the following spring (Carter 1977). Impacts are most significant in coastal southeast Alaska on Sitka spruce and in Arizona on Engelmann spruce. In 2003, 73 039 ha were affected (fig. 6.12), mostly in Arizona.

Host water stress is thought to positively influence sap-sucking insects through an increase in phloem nitrogen content (Kolb et al. 2016). Spruce aphid populations are higher when water stress is intermittent and lowest when water stress is continuous. Lynch (2003) reported that epidemics tend to occur after dry winter and spring conditions in high-elevation forests in Arizona, but do not seem to be influenced by the amount of moisture received during the summer monsoon. Given projected increases in temperature and the frequency of droughts in the West, spruce aphid will likely become a more significant disturbance agent.

Role of Management in Mitigating Impacts

Management to reduce the vulnerability of forests to insects has historically focused on suppression and prevention.

Several tactics are available to reduce the vulnerability of forests to insects, including those that help reduce atmospheric warming (through reductions in carbon dioxide and other greenhouse gas emissions), which increases drought stress and the probability that trees will succumb to colonization by insects (Kolb et al. 2016). Others facilitate transition of forest stands to tree species better adapted to future climates (Fettig et al. 2013b, Millar et al. 2007, Millar and Stephenson 2015). Historically, management has focused on suppression and prevention. Suppression involves short-term tactics designed to reduce current infestations by manipulating insect populations through the use of fire, pesticides (contact and systemic insecticides, microbials [bacteria, viruses, pathogens and nematodes]), insect growth regulators, soaps, and horticultural oils, semiochemicals (chemicals produced by one organism that elicits a response, usually behavioral, in another organisms, e.g., pheromones), sanitation harvests (harvesting currently infested material and destroying living life stages in that material before they emerge), or combinations of these treatments (Coulson and Witter 1984, Fettig and Hilszczański 2015). For suppression to be effective, accurate detection and survey methods are required to identify and delineate infestations prior to treatment. Prevention is designed to reduce the probability and severity of future infestations by manipulating forest conditions through thinning, prescribed burning, or alterations of age classes and species composition. There is considerable support for thinning of conifer forests to

reduce the severity of bark beetle infestations (Fettig and Hilszczański 2015, Fettig et al. 2007, but see Six et al. 2014). In contrast, there is little consensus regarding the efficacy of thinning for reducing vulnerability to most forest defoliators (Muzika and Liebhold 2000). Given the diversity of ecologies and life histories displayed by forest insects, it should be no surprise that the application of many of these tactics must be tailored to specific insect-host associations (e.g., in regard to timing of treatments, use of specific semiochemicals, etc.).

When implementing prevention or suppression, there are opportunities for collaboration with other resource disciplines, allowing additional objectives to be met perhaps with little or no additional cost. For example, fuel reduction treatments are frequently applied in the Western States to reduce the quantity and continuity of forest fuels (Stephens et al. 2012). Although prescriptions generally differ between thinning implemented for fuels reduction and that for prevention of certain insects (e.g., bark beetles), there are opportunities to alter fuel reduction treatments without reducing their efficacy while increasing the effectiveness of these same treatments for reducing the vulnerability of forests to insects.

Impacts to Sustainability

The impacts of insects on forests extend beyond the levels of tree mortality caused and associated cascading ecological effects. For example, bark beetle epidemics may increase water yield yet diminish water quality because of pulses of concentrated nutrients and suspended sediment loads (Mikkelson et al. 2013), as well as degrade air quality by the release of volatile organic compounds and biogenic aerosols (Berg et al. 2013). Landscape aesthetics are important drivers of nature-based tourism and may be negatively affected (Morris et al. 2018). In the wildland-urban interface, concerns are often more practical and focused on property values, hazard trees, and fire risk (Cohen and other 2016). Interestingly, a notable western pine beetle outbreak in southern California affected how homes were advertised and sold (Fettig 2019).

Several recent assessments have concluded that western forests are increasingly vulnerable to mortality associated with the direct and indirect effects of climate change, and that substantial shifts in the geographic distributions of some tree species and forest ecosystems is likely (Fettig et al. 2013b, Williams et al. 2013). Rehfeldt et al. (2006) suggested that about 48 percent of the Western U.S. landscape is likely to experience climate profiles with no contemporary analog for the current coniferous vegetation by the end of this century. Projections show that distributions of grassland, chaparral, and montane forest are likely to increase at the expense of subalpine forest, tundra, and Great Basin woodland. Shifts are expected to be most

Several recent assessments have concluded that western forests are increasingly vulnerabile to mortality associated with the direct and indirect effects of climate change.

rapid along ecotones, particularly in semiarid landscapes (Allen and Breshears 1998). To that end, Krist et al. (2014) projected that most Western States have at least 10 percent of their forested landscapes at risk (defined as without remediation at least 25 percent of standing live basal area greater than 2.54 cm in diameter will be killed in the next 15 years) to forest insects and diseases epidemics. Most notable, in Idaho 28 percent of the forested landscape is considered at risk (Krist et al. 2014).

In the West, seven trees species are expected to suffer substantial levels of tree mortality in the near future, including whitebark pine (58 percent of total basal area), limber pine (44 percent), lodgepole pine (39 percent), ponderosa pine (28 percent), pinyon pine (27 percent), Jeffrey pine (26 percent), and grand fir (25 percent) (Krist et al. 2014). In particular, whitebark pine is of concern as it serves as a keystone species. Whitebark pine seed is a critical food source for birds, small mammals, and bears. Furthermore, the species quickly establishes after disturbance, and is important in maintaining snowpacks and reducing erosion of steep slopes. Significant levels of whitebark pine mortality have been attributed to mountain pine beetle (and its interactions with climate change and white pine blister rust), and the U.S. Fish and Wildlife Service first announced in 2011 that it determined whitebark pine to warrant protection under the Endangered Species Act, but that adding the species to the Federal List of Endangered and Threatened Wildlife and Plants was precluded by the need to address other listing actions of higher priority (Federal Register 2011). Some have questioned if the species can be saved (Neuenschwander et al. 1999).

Conclusions

Most forest insects are beneficial, but a few species periodically become so abundant that they threaten ecological, economic, social, or aesthetic values.

Most forest insects are beneficial (e.g., they help facilitate decomposition, nutrient cycling, and pollination), but a few species periodically become so abundant that they threaten ecological, economic, social, or aesthetic values. In the Western United States, chief among these are several species of bark beetles (table 6.1), most notably mountain pine beetle. Several defoliating and sap-sucking insects are also important (tables 6.2 and 6.3). These disturbance agents have interacted with others such as wildfire for millennia, shaping the structure and composition of forests over time. Unlike the Eastern United States, relatively few exotic insects are the source of important disturbance in western forests, but we expect an increase in their prevalence in the future owing to recent shifts in human populations and trade (Aukema et al. 2010, 2011), among other factors. Relatedly, the recent establishment of goldspotted oak borer (*Agrilus coxalis*) in California (Coleman and Seybold 2008) and emerald ash borer (*A. planipennis*) in Colorado (Berry et al. 2017) are

cause for concern. The heterogeneity of western landscapes and the diversity of climates that occur in relatively small geographic areas (ranging from deserts to subarctic) do provide some buffer against invasion.

We expect that epidemics of the species described herein will occur with relatively predictable frequency and that basic host relationships will remain largely intact. However, with climate change, we expect the severity of most bark beetle epidemics to increase as long as susceptible hosts exist. Overall, the impact of climate change on defoliating and sap-sucking insects is less unclear. Finally, we agree with many experts who have argued to increase the pace and scale of treatments designed to increase forest resilience to insects and other disturbances exacerbated by climate change.

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Chapter 7: Diseases and Abiotic Damage Agents

Susan J. Frankel and I. Blakey Lockman[1]

Background

In what ways do plant diseases affect forest sustainability? How will disease disturbance regimes change given global warming and increasing population pressures? In this chapter, we review how plant diseases, caused by both biological pathogens and abiotic disturbance agents, influence forest sustainability—but note that our findings derive less from data and more from observations and professional judgment. Many forest pathogens (fungi, bacteria, viruses, nematodes) are microscopic, which predisposes them to nondetection because they cannot be seen with the naked eye and their symptoms are indistinctive. Further complicating our understanding of pathogen roles in disturbance are limitations in accurate attribution of the cause(s) of tree mortality. Typically, sampling and laboratory analysis are required to determine whether pathogens are contributing to mortality, even when tests are conducted, technological and financial constraints often result in an incomplete determination of the causal agent. Despite improvements in molecularly based (polymerase chain reaction, PCR) diagnostic methods, our knowledge as to pathogen distributions and ecological impacts remains limited (Das et al. 2016; Wingfield et al. 2015, 2017).

How Are Plant Pathogens, Including Abiotic Agents, Affecting Western Forests?

Western forests host a multitude of plant pathogens: microbes, parasitic plants, and other organisms that can cause disease in vegetation (tables 7.1 and 7.2) (Goheen and Willhite 2006, Hagle et al. 1987, Hepting 1971, Paine and Lieutier 2016). Tree or plant disease is defined as a malfunctioning of host tree or plant tissues by a pathogenic agent or environmental factor that leads to the development of symptoms (Agrios 2005). Forest diseases may be caused by biological organisms (e.g., mistletoes, fungi, Phytophthoras (oomycetes), and other organisms), as well as by direct damage to plants from nonliving, abiotic factors, primarily adverse environmental conditions such as drought, sunburn, or frost injury (Hansen and Lewis 1997, Manion and LaChance 1992, Sinclair et al. 1987). In this report, drought is treated in a separate chapter (see chapter 4). The effects and behavior of other biotic and abiotic diseases as they contribute to forest disturbance are briefly discussed below.

Forest diseases may be caused by both biological organisms and abiotic factors such as drought, sunburn, or frost injury.

[1] **Susan J. Frankel** is a biologist, Pacific Southwest Research Station, 800 Buchanan Street, Albany, CA 94710; **I. Blakey Lockman** is a regional forest pathologist, Pacific Northwest Region, State and Private Forestry, 1220 SW Third Avenue, Portland, OR 97204.

Table 7.1—List of common biotic diseases in Western U.S. forests

Scientific name—causal agent	Common name
Arceuthobium spp.	Dwarf mistletoe
Armillaria ostoyae	Armillaria root disease
Cronartium comandrae	Comandra blister rust
Cronartium ribicola	White pine blister rust
Cryphonectria parasitica	Chestnut blight
Dothistroma septosporum	Red band needle blight
Endocronartium harknessii	Western gall rust
Fusarium circinatum	Pitch canker
Geosmithia morbida	Thousand cankers
Heterobasidium irregulare, H. occidentale	Heterobasidion root disease (Syn. Annosum root disease)
Leptographium wageneri	Black stain root disease
Ophiostoma ulmi, O. novo-ulmi	Dutch elm disease
Phaeocryptopus gaumannii	Swiss needle cast
Phellinus sulphurascens, P. weirii	Laminated root rot
Phoradendron spp.	True mistletoe
Phytophthora cinnamomi	Phytophthora dieback
Phytophthora lateralis	Port-Orford-cedar root disease
Phytophthora ramorum	Sudden oak death, ramorum blight
Valsa sordida	Cytospora canker

Table 7.2—List of common abiotic diseases in the Western United States

Environmental factor	Disease or injury
Drought	Wilting, water stress, lack of transpiration, carbohydrate starvation
Ice accumulation	Freeze burn, branch breakage, and tree failure
Intense heat and drought	Sudden aspen decline, other declines, tree mortality
Flooding	Root suffocation, increased susceptibility to Phytophthoras, tree mortality
Lack of snow and early warming	Yellow-cedar decline, and other declines, freeze injury to new growth
Ozone and air pollution exposure	Leafspots, reduced growth
Snow accumulation	Uprooting from avalanche or crown damage
Storm damage	Flooding, branch breakage, tree failure
Unseasonal cold temperatures	Frost injury, bud damage, leaf burn, top die-back and tree mortality
Volcanism	Heat injury, smothering, or chemical toxicity
Wind damage	Windthrow or branch failure

Despite their cryptic nature, plant diseases (i.e., root diseases, wilts, cankers) profoundly influence forest composition, structure, and function (Boyce 1938, Ramsfield et al. 2016). Although fire and insect attack often appear dramatically across a landscape, damage from pathogens typically occurs more gradually. Nevertheless, plant diseases can be widespread and persistent; cumulatively, they can significantly shape forest openings by shortening tree lifespans and causing mortality. Impairment from plant diseases influences energy and water flux (Anderegg et al. 2013), carbon cycles (Cobb and Metz 2017, Hicke et al. 2012), and wildlife habitat quality (Boyd et al. 2013; Liebhold et al. 2017b, Loo 2009, Lovett et al. 2006). Both native and invasive pathogens can cause significant growth reduction, deformity, and mortality. Some of the more damaging native and nonnative pathogens are predicted to cause 15 percent or greater total basal area loss on more than 7.9 million ha across the West from 2012 to 2027 (fig. 7.1) (Krist et al. 2014). Pathogens interact synergistically with insects and fire to compound their effects on forests (Cobb and Metz 2017, Logan and Powell 2001, Metz et al. 2011, Parker et al. 2006). Disease development is very sensitive to environmental conditions, in particular temperature and moisture, so changes in climate patterns, including storm intensity and shifts in seasonality, will alter the behavior of biological pathogens and the nature of abiotic impacts on growth and mortality (Ayres and Lombardero 2000, Dale et al. 2001, Ramsfield et al. 2016, Sturrock et al. 2011).

Plant diseases profoundly influence forest composition, structure, and function.

In this review, we consider how pathogens influence disturbance and sustainability mainly in term of impacts to forest stocking, species composition, and related measures. Sustainability has many other important dimensions, i.e., provisioning of ecosystem services, which are considered in more detail in other chapters because pathogen impacts on most ecological processes are not well quantified.

Current Trends—Invasive Diseases (Pathogens) Are of Increasing Concern

Damage from invasive pathogens is on the rise as the number of pathogens detected continues to increase (Aukema et al. 2010, 2011; Ayres and Lombardero 2018; Liebhold et al. 2017; Santini et al. 2013; Wingfield et al. 2017). Exotic pathogen introductions are responsible for the most damaging forest pests ever witnessed in North American forests, including chestnut blight (*Cryphonectria parasitica*) (Anagnostakis 1987, Freinkel 2009); white pine blister rust (*Cronartium ribicola*) (Benedict 1981, Boyce 1938, Geils et al. 2010, Vitousek et al. 1996); and Dutch elm disease (*Ophiostoma ulmi* and *O. novo-ulmi*) (Brasier 1991, Strobel and Lanier 1981). Chestnut blight devastated Eastern U.S. forests, but introduced pathogens have killed and continue to destroy millions of trees and limit survival of their hosts in all parts of North America, including the Western United States (Liebhold et al. 1995, Mack et al. 2000).

Damage from invasive pathogens is on the rise as the number of pathogens detected continues to increase.

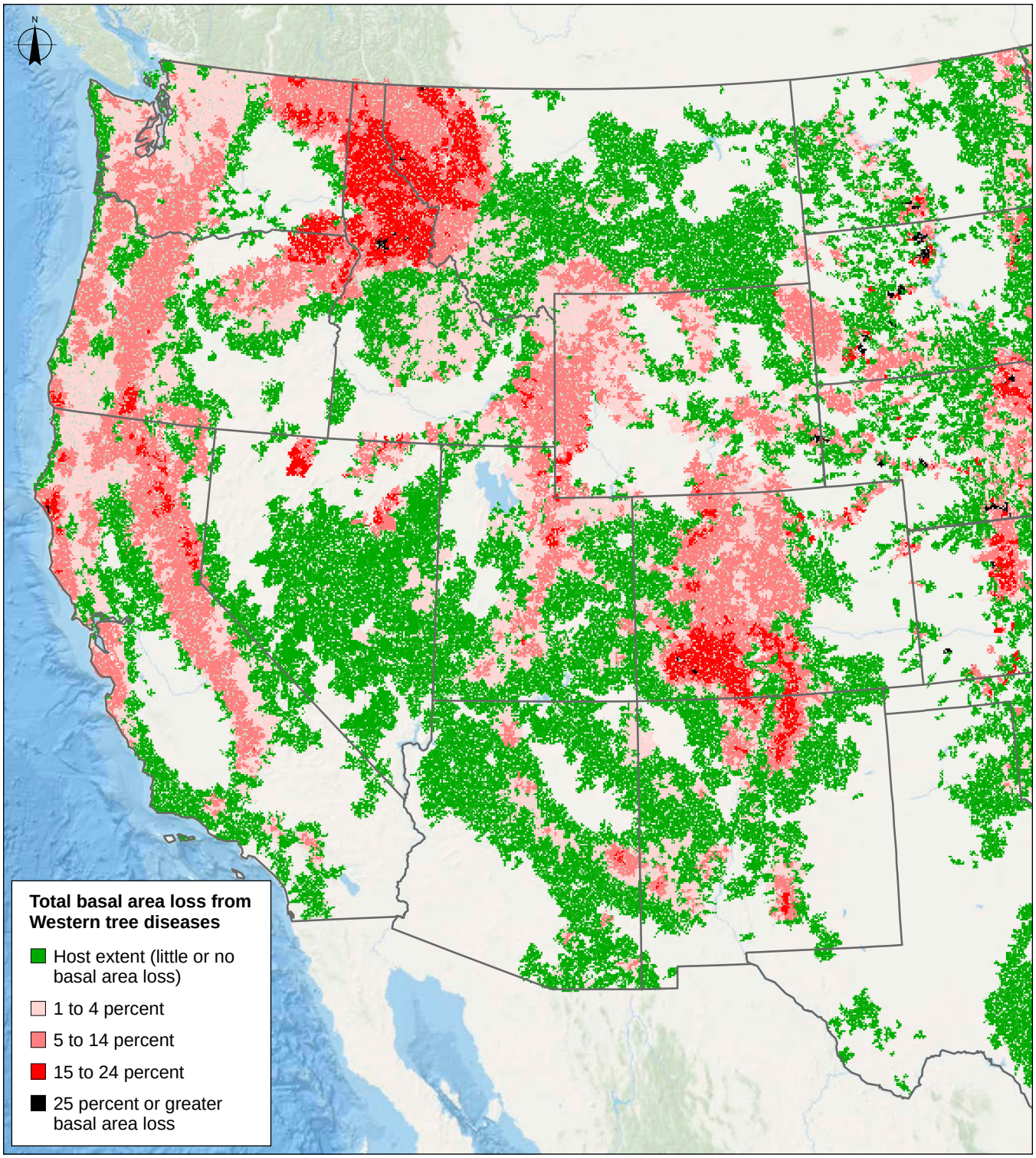

Figure 7.1—Predicted basal area loss from diseases in the Western United States. Diseases modeled are Armillaria root disease, aspen and cottonwood decline, dwarf mistletoes, Heterobasidion root disease, laminated root rot, Port-Orford-cedar root disease, sudden oak death, and white pine blister rust. (Data are from Krist et al. 2014).

Greater volumes of global trade and changes in import patterns that shift to new markets in portions of Europe, Asia, or South America will support continued establishment of new invasive species (Liebhold et al. 2017). Live plant imports are the most significant pathway for pathogen and insect transport into the United States; as nursery plants are increasingly grown overseas, the higher volume of imported plants will result in further pest introductions (Liebhold et al. 2012).

Capsule summaries for two highly damaging invasive pathogens in the West, sudden oak death (*Phytophthora ramorum*) and white pine blister rust (*Cronartium ribicola*), are provided below.

Live plant imports are the most significant pathway for pathogen and insect transport into the United States; as nursery plants are increasingly grown overseas, the higher volume of imported plants will result in further pest introductions.

Native Pathogens

Native pathogens are present throughout Western U.S. forest ecosystems, from coastal forests to high-alpine areas. The most widely dispersed native pathogens in the West are dwarf mistletoes (*Arceuthobium* spp.) (Hawksworth and Wiens 1996, Mathiasen et al. 2008). Obligately (only) parasitic, dwarf mistletoes can reduce growth, wood quality, seed production, and the lifespan of infected host trees. At times, however, they may be considered beneficial because their berries provide food for wildlife, and the witches' brooms they cause serve as favored nesting sites or habitat (Mathiasen et al. 2004).

Other significantly damaging native pathogens include root diseases: Armillaria root disease, Heterobasidion root disease, and laminated root rot (*Phellinus sulphurascens* and *P. weirii*). These root diseases have existed in parts of western forests for thousands of years, dating back to the most recent ice age, and all cause significant ecological impacts in many parts of the West (Shaw and Kile 1991, Washington State Academy of Sciences 2013). Overviews of these native diseases are provided below.

Both native and nonnative pathogens are present in western forests and their impacts depend on the stand management objectives and how they are implemented, or the land uses in place, such as whether the stands are plantations, managed stands, urban forests, or wilderness, in addition to other factors (Tubby and Webber 2010, Wingfield et al. 2015). Tree mortality in wildlands may contribute to biodiversity because reduced forest canopy creates openings that may provide habitat for shade-intolerant plants (Hansen and Goheen 2000). Conversely, plant death may reduce diversity; for example, in rare species habitat, the death of threatened or endangered plant species caused by the inadvertent introduction of pathogens during restoration plantings (e.g., *Phytophthora* spp.) may irreversibly degrade resource values (Rooney-Latham et al. 2015).

Abiotic Diseases

Depending on the environmental conditions, flooding, high winds, or intense heat may result in physical conditions that are beyond the capacity of trees or plants to endure. Abiotic injury is typically dependent upon the interaction of plants and climate, terrain, and geology. The most damaging abiotic injuries are caused by fire, drought, wind damage, snow (accumulation in crowns or as avalanches), ice accumulation in crowns, frost, flooding, ozone injury or air pollution damage, and landslides (Sinclair et al. 1987). Less frequent but significant damage occurs periodically from volcanic eruptions and glaciation (Gardiner and Quine 2000). Land development may also contribute to abiotic damage as a result of construction practices (Hauer et al. 1994), poor planting (Ware 1994), or fire injury from human ignitions (Stephens 2005, Syphard et al. 2007).

Biological pathogens contribute in many ways to abiotically driven die-offs; in particular, they increase stress by reducing physiological function, causing trees to die that could otherwise have recovered, thereby expanding and extending mortality events after the climatically induced stress is relieved (Worrall et al. 2010).

Megadisturbance

The abiotic diseases mentioned above are overshadowed by recent megadisturbances, in which interactions from increasing temperatures, drought, native insects, and pathogens have resulted in unprecedented levels of tree mortality in some forests (Allen et al. 2010, 2015; Millar and Stephenson 2015). It is difficult to predict responses of pathogens and other factors that interact in these extreme climate events (Allen et al. 2015), but widespread tree mortality in the Rocky Mountains (Berner et al. 2017, Wong and Daniels 2017) and California (Tree Mortality Working Group 2017) provides an indication of the severity and types of changes to expect in forest ecosystems.

Unprecedented Drought in the Sierra Nevada Mountains, 2010–2017

Tree mortality in the central Sierra Nevada Mountains in California has increased by an order of magnitude, from tens to hundreds of dead trees per square kilometer (Young et al. 2017); an estimated 129 million trees have died from 2010 to 2017, with more than 3.6 million ha of drought-stricken forests with approximately 62 million trees dying in 2016 alone (USDA FS 2017). Tree mortality—primarily of ponderosa pine (*Pinus ponderosa*), white fir (*Abies concolor*), and sugar pine (*P. lambertiana*)—is attributed to drought, native bark beetles, and other factors (Kolb et al. 2016), including pathogens that contributed an unquantified amount to total

mortality (California Tree Mortality Working Group 2017, USDA FS 2016). The tree mortality is extensive. For example, on the Sierra National Forest at elevations between about 1000 to 1500 m, 80 percent of the ponderosa and sugar pine trees more than 50.8 cm diameter at breast height (d.b.h.) have died, creating extensive landscapes with mostly dead trees (Ballard 2017). California's Tree Mortality Working Group reported the removal of 830,000 hazard trees to maintain safe conditions, with the Forest Service having removed almost 480,000 of these trees along 1355 km of roads, in 343 recreation sites, and around 135 communities, as well as having installed more than 556 ha of fuel breaks on six national forests in 10 counties (Gomes 2017).

This level of mortality is causing some forest type conversion to hardwoods or possibly other conifer species, but it will take decades for stands to reestablish and the impacts to be realized. Root disease fungi, dwarf mistletoes, and other native pathogens in these areas are expected to recolonize these stands, but the high level of uncertainty as to stand trajectory makes it difficult to predict the role that pathogens will play in influencing stand development or future disturbance.

Sudden Aspen Decline Illustrates the Role of Pathogens and Pests Following Drought Injury

In the early 2000s, a severe, hot, multiyear drought in the Southwestern United States incited sudden aspen decline of quaking aspen (*Populus tremuloides*) in Arizona, Colorado, Utah, and other states (Rehfeldt et al. 2009; Worrall et al. 2008, 2010). Sudden aspen decline, first observed in 2004, is characterized by rapid branch dieback and tree mortality on a landscape scale, without the involvement of a primary pest or pathogen. By 2008, it had been observed on more than 220 000 ha, or about 19 percent, of the quaking aspen type in the national forests of southwestern Colorado (Worrall et al. 2010). Marchetti et al. (2011) investigated the role of pathogens and insects in sudden aspen decline and found Cytospora canker (*Valsa sordida*), bronze poplar borer (*Agrilus liragus*), and aspen bark beetles (*Procryphalus mucronatus* and *Trypophloeus populi*) to be the most common pests in damaged areas, correlated with crown loss. They showed that sudden aspen decline fits the pattern of a true forest decline (Manion and LaChance 1992), with a predominance of secondary damage agents and no evidence that the mortality was caused by a primary pest; rather, the outbreak of secondary insects and pathogens increased the rate of mortality in aspen stands that were already affected by drought (the inciting factor). As is typical of responses to drought stress, populations of the pathogens and insects increased, allowing more vigorous trees to be attacked, thereby amplifying the impact of drought by killing trees that may have otherwise recovered.

Historical and Current Data: Management Approaches

To predict pathogens' contributions to forest disturbance, it is helpful to categorize the types of damage as root diseases, dwarf mistletoes, foliage diseases, rusts, Phytophthoras, or abiotic diseases, and consider them as disease groups with somewhat similar characteristics. Short descriptions of each are provided below.

Root Diseases

All forest types and tree species in western forests are affected by one or more root diseases.

Root diseases are the most damaging group of forest diseases in terms of tree volume loss in the United States (Krist et al. 2014). Root diseases are well distributed throughout the West (Lockman and Kearns 2016); all forest types and tree species are affected by one or more root diseases. Root disease pathogens kill trees, slow tree growth, decay wood, predispose trees to other biotic and abiotic agents, and cause trees to break off or uproot. All these impacts affect timber volume (Smith 1984), forest stand composition, and structure, ecosystem function (Hagle et al. 2000), personal safety (Filip and Goheen 1982), and carbon sequestration (Healey et al. 2016, Washington State Academy of Sciences 2013). Nearly all the root diseases affecting western ecosystems are caused by native pathogens, and as such, they have been long-term associates of the tree species they affect and are enduring components of the forest ecosystems they occupy. Climate change, management practices such as fire suppression and selective harvesting, and the introduction of exotic tree pathogens (e.g., white pine blister rust) have changed the historical roles of root diseases and, in many cases, have exacerbated their effects (Byler and Hagle 2000, Kliejunas 2011, Lockman and Kearns 2016).

Root diseases, as a group, are one of the four top contributors to overall risk of tree mortality in the lower 48 states.

Root diseases, as a group, are one of the four top contributors to overall risk of tree mortality in the lower 48 states (along with oak decline/gypsy moth, mountain pine beetles, and southern pine beetles) (Lockman and Kearns 2016). Total basal area mortality rates of 25 percent or more are predicted to occur on nearly 930 777 ha in the Western States over a 15-year period (2012 to 2027) (fig. 7.2) (Krist et al. 2014).

In northern Idaho and western Montana, national forest land affected by root diseases has been measured at more than 2.3 million ha as determined by aboveground symptoms on U.S. Forest Service Forest and Inventory Analysis (FIA) plots (Lockman et al. 2015). This acreage equates to just under 40 percent of the land base on the seven national forests in northern Idaho and western Montana. Based on these area estimates, the volume lost from all root diseases in northern Idaho and western Montana is approximately 4.7 million m^3 per year. Volume growth impacts from root diseases often occur before a tree displays declining crown symptoms of infection aboveground (Cruickshank et al. 2011, Thies 1983), so actual volume lost is likely much more than estimated.

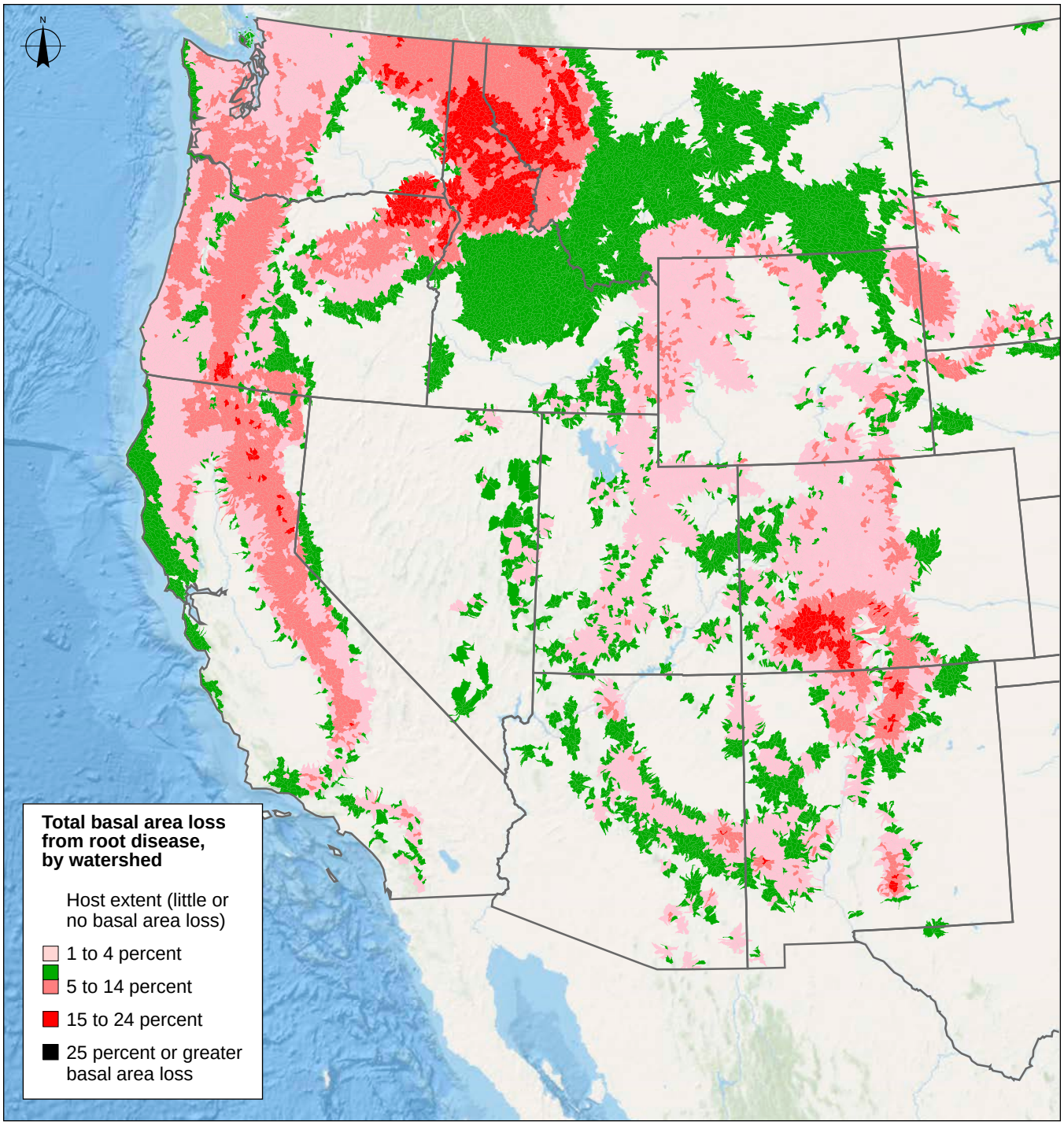

Figure 7.2—Predicted basal area loss from root diseases in the Western United States. Root diseases modeled are Armillaria root diseases, Heterobasidion root diseases, laminated root rot, and Port-Orford-cedar root disease (Data are from Krist et al. 2014).

Root disease pathogens have long-term and persistent impacts on the forest ecosystems they occupy because of their longevity on a site long after infected trees have died (Hagle et al. 2000). Root diseases kill millions of trees across the Western United States each year. This mortality occurs in large groups or as scattered small groups or individual trees, causing long-term openings in the forest canopy. Most of the fungi that cause root diseases survive saprophytically in root systems for years to decades, allowing them to become relatively permanent components on a site. This long-term presence can affect site productivity, along with the edaphic (related to soil) and physiographic features. Productivity is generally reduced by growth loss and mortality of host trees, which subsequently affects carbon sequestration. Root diseases can influence species composition by removing susceptible host trees and leaving tolerant species; and they can also modify successional pathways by changing forest structure and composition. Conversely, gaps and openings created by root diseases also allow for increased diversity in structure and species of plant life (Hansen and Goheen 2000, Kearns and Jacobi 2005). Additionally, root diseases create snags and down woody material, all of which affect wildlife habitat.

Management to minimize the impacts from many of the root diseases, such as Armillaria root disease and laminated root rot, involves converting or maintaining species that are tolerant to root diseases on the site. This can occur by regeneration harvest followed by planting tolerant species or using seed trees of a tolerant species to naturally regenerate a site, and precommercial and commercial thinning to favor root-disease-tolerant species. Other root diseases, such as black stain root disease in Douglas-fir (*Pseudotsuga menziesii*), can also be managed by minimizing damage to residual trees and harvesting during periods when the vectors of the causal fungus are at a minimum (Hansen et al. 1988, Rippy et al. 2005).

Dwarf Mistletoes

Dwarf mistletoes (*Arceuthobium* spp.) are parasitic plants that extract water and nutrients from living conifer trees (Shaw and Mathiasen 2013). Dwarf mistletoes are native components of western coniferous forests, having co-evolved with their hosts for millions of years. They cause reduced growth, decreased cone and seed production, tree mortality, and predisposition to other pathogens and insects (Hawksworth and Wiens 1996). The effects of dwarf mistletoes on individual trees are gradual, leading to measured impacts on ecosystem functions. These effects are generally scattered across a landscape, so significant impacts on ecosystem function may be detectable only across a long temporal scale and at a larger landscape level (Hawksworth and Wiens 1996). Fire suppression and silvicultural practices that favor dwarf mistletoes (e.g., regeneration harvest leaving dwarf mistletoe-infected

residuals over susceptible regeneration) have altered the historic levels of dwarf mistletoes and, in many cases, increased their severity and distribution across landscapes (Zimmerman and Laven 1984).

Dwarf mistletoes tend to be fairly host specific, and their distributions tend to loosely coincide with the distributions of their hosts (fig. 7.3). Mortality rates from dwarf mistletoes tend to be low. One to 15 percent of total basal area mortality resulting from dwarf mistletoes is predicted to occur on more than 1.2 million ha over a 15-year period (2012 to 2027) in the interior West, which includes Arizona, Colorado, Idaho, Montana, Nevada, New Mexico, Utah, and Wyoming (fig. 7.3) (Krist et al. 2014). Although direct mortality rates are low, dwarf mistletoes play a significant role across the West as contributing agents to mortality in trees that are stressed by other factors, such as drought, insects, or root diseases (Byler 1978).

Direct mortality from dwarf mistletoes tends to be low, but dwarf mistletoes play a significant role as contributing agents to mortality in trees that are stressed by other factors.

Dwarf mistletoe management includes regeneration harvesting that allows for removal of infected overstory, planting of non-host species if maintaining an infected overstory, and removal of infected overstory to avoid infection of a susceptible understory (Hoffman 2010). Pruning to remove dwarf-mistletoe-infected branches in high-value trees can minimize hazards from brooms breaking out, reduce infection of understory trees, and increase the longevity of host trees (Maffei et al. 2016, Scharpf et al. 1988).

Foliage Diseases

Many foliage diseases are caused by numerous leaf- or needle-dwelling fungi, bacteria, and viruses (Sinclair et al. 1987). The symptoms, leafspots—discolored or necrotic areas—look similar, making it hard to identify a causal agent. In early disease stages, it may be difficult to distinguish between a benign spot and a serious pathogen that could lead to defoliation and growth loss. Foliar pathogens infect most all tree species and are particularly problematic in nurseries, plantations, or on ornamental plantings; however, significant damage in commercial forests occurs, as exemplified by Swiss needle cast and Dothistroma (red band) needle blight.

Swiss needle cast, *Phaeocryptopus gaeumannii*—
Swiss needle cast is a native foliar disease specific to Douglas-fir. The causal fungus, *Phaeocryptopus gaeumannii*, infects newly emerged needles in the spring and colonizes the intercellular spaces within needles (Hansen et al. 2000). Disease is caused by the pseudothecia (fungal fruiting body) physically blocking the stomata in the needles, which impedes uptake of carbon dioxide and water vapor needed for photosynthesis (Manter et al. 2005). Significant symptoms in native plantations were first noted in coastal Oregon in the late 1980s and early 1990s and have gradually increased in severity and spatial extent (Lee et al. 2017, Ritokova et al.

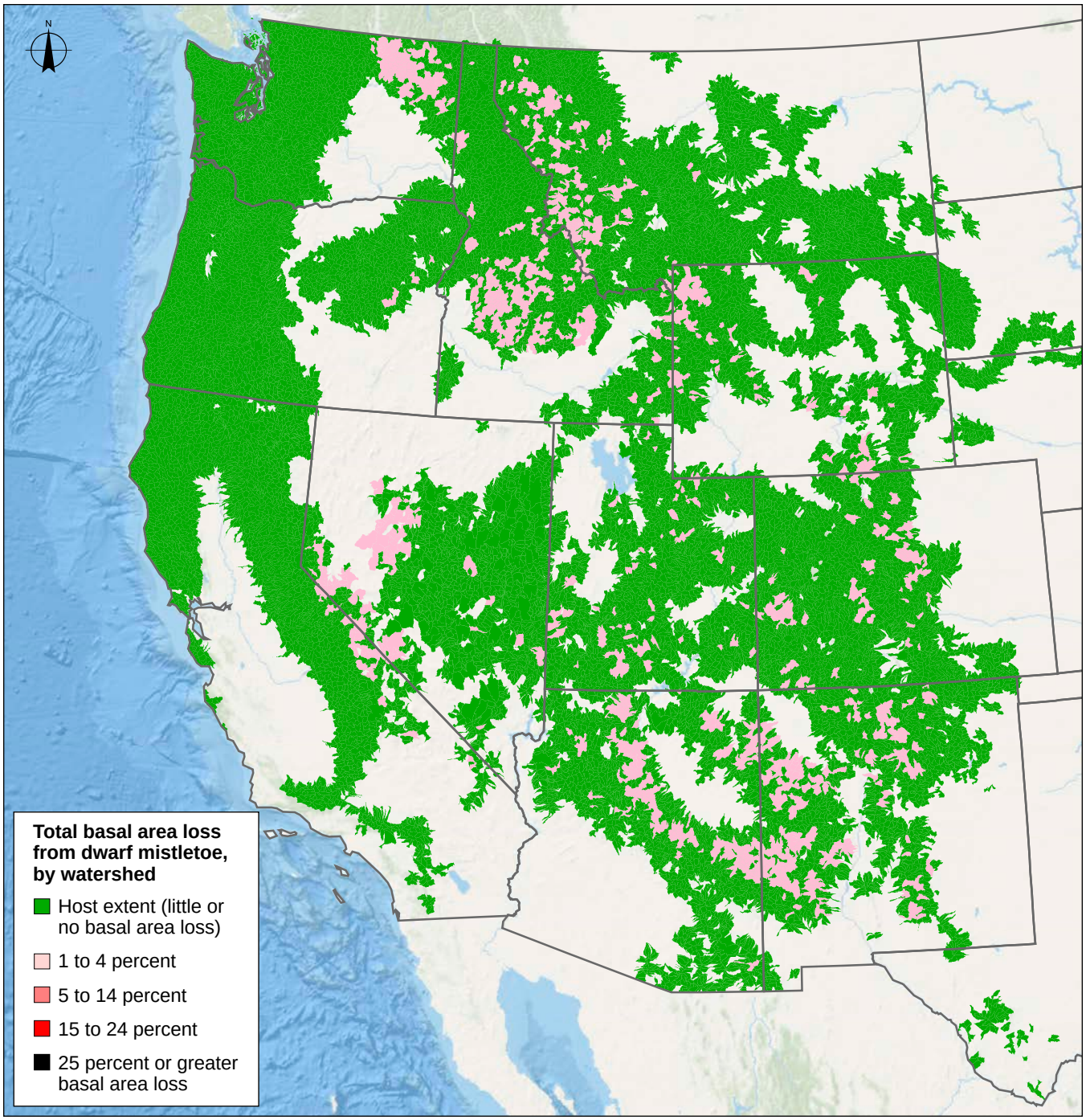

Figure 7.3—Predicted basal area loss from dwarf mistletoes in the Western United States. (Data are from Krist et al. 2014).

2016). Although much of the disease has been associated with young Douglas-fir plantations in the Sitka spruce(*Picea sitchensis*) zone, Swiss needle cast has been shown to affect older trees as well (Black et al. 2010). Aerial surveys have been used to map the extent of the disease, with the first flight occurring in 1996. In Oregon, the area affected has significantly increased in recent years from 53 050 ha in 1996 to 238 705 ha in 2015 (Ritokova et al. 2016). Although mortality is not attributed to this disease, significant growth losses have been documented. A 25-percent reduction in height growth and 35-percent reduction in basal area growth were measured in the more severely infected Douglas-fir plantations in the 1990s (Maguire et al. 2002). Radial growth in older trees can be decreased up to 85 percent in severely affected sites (Black et al. 2010).

A climate- and terrain-based model for Swiss needle cast severity in the western Oregon Coast Range has found strong correlations with mean daily temperature in late winter/early spring, with the best-fit model also including cumulative spring leaf wetness hours (Stone et al. 2008). As winter temperatures and spring/early summer precipitation continue to increase in current climate change scenarios, the area affected by severe Swiss needle cast will likely expand beyond the western Coast Range (Lee et al. 2017).

The increased development of Swiss needle cast along the Oregon Coast has been partially attributed to planting Douglas-fir into sites that typically have supported other species, such as western hemlock (*Tsuga heterophylla*), Sitka spruce, and red alder (*Alnus rubra*), with much less Douglas-fir than is currently present (Zhao et al. 2015). Impacts from Swiss needle cast can be minimized by planting mixed species to lessen potential disease losses, reducing the proportion of Douglas-fir on moderate- and high-severity sites, using seed adapted to the site, and avoiding planting offsite stock (Shaw et al. 2011).

Impacts from Swiss needle cast, a native foliar disease specific to Douglas-fir, can be reduced by planting mixed species and using seed and stock adapted to the site.

Dothistroma (red band) needle blight, *Dothistroma septosporum*— Dothistroma needle blight is one of the most damaging foliar diseases of pines in both the Northern and Southern Hemispheres (Drenkhan et al. 2016, Watt et al. 2009); the pathogen infects 82 pine species as well as other non-pine species, and can cause successive needle defoliation, reduced diameter and height growth, and tree mortality (Barnes et al. 2014). The global distribution of *D. septosporum* is considered an important example of the unintentional, human-assisted movement of a plant pathogen, facilitated by the establishment of *D. septosporum* in plantations planted outside their natural ranges, particularly on Monterey pine (*Pinus radiata* D. Don) (Barnes et al. 2014).

Recent outbreaks of Dothistroma needle blight have led to severe defoliation and subsequent mortality in localized areas of ponderosa pine in central Montana (James 2010); in limber pine, (*P. flexilis*) in eastern Montana, and in an extensive area of lodgepole pine (*P. contorta*) in British Columbia (Woods et al. 2005). Growth impacts are positively correlated to the amount of needles lost and are somewhat linear in younger trees, but a convex-curved relationship exists in older trees (Gibson 1972), in which older needles, which are infected first in disease progression, contribute less to growth than in younger trees. Mortality is a consequence of severe outbreaks and has occurred in limber pines in eastern Montana with greater than 90 percent of crowns affected by Dothistroma needle blight (Jackson and Lockman 2003). Similar infection levels leading to mortality were found in lodgepole pines in British Columbia (Woods 2003).

These outbreaks appear to be tightly tied to periods of warmer and wetter conditions in winter and spring (Jackson and Lockman 2003, Taylor and Schwandt 1998, Welsh et al. 2014, Woods et al. 2005), and if the current climate trend continues to favor warming with an increase in spring rains, there is a moderate likelihood of increased damage from this disease (Kliejunas 2011). Woods et al. (2016) found that, since the 1950s, four of the past five strong El Niño events coincided with reports of increased Dothistroma damage on several continents. The extent and severity of the current outbreak in British Columbia is also linked to an increase in prevalence of the host species there, where young lodgepole pine plantations have increased from 10 to 40 percent in managed stands (Woods et al. 2005).

Impacts from Dothistroma needle blight can be minimized by planting mixed species to lessen potential disease losses (Woods et al. 2005), reducing the proportion of susceptible pines on moderate- and high-severity sites, thinning to favor resistant trees or species, using seed adapted to the site, and avoiding planting offsite stock (James 2010).

Rusts—White Pine Blister Rust, *Cronartium ribicola*

Rusts are obligate parasites and may cause serious canker or foliar diseases. They often have complicated lifecycles that require more than one host plant species for disease to express, and may produce up to five types of spores. One of their spore types, uredial spores, are bright orange, hence during sporulation, the affected plant part appears “rusty” (Geils et al. 2010, Ziller 1974).

In the Western United States, the most damaging rust pathogen is *Cronartium ribicola*, a fungus native to Asia that is the cause of blister rust of white pines (*Pinus* spp.) and which was introduced to North America in the early 1900s (Maloy 1997). Infection by *C. ribicola* causes branch dieback, reproductive failure, and tree mortality. In the West, white pine blister rust is widely distributed (fig. 7.4); over

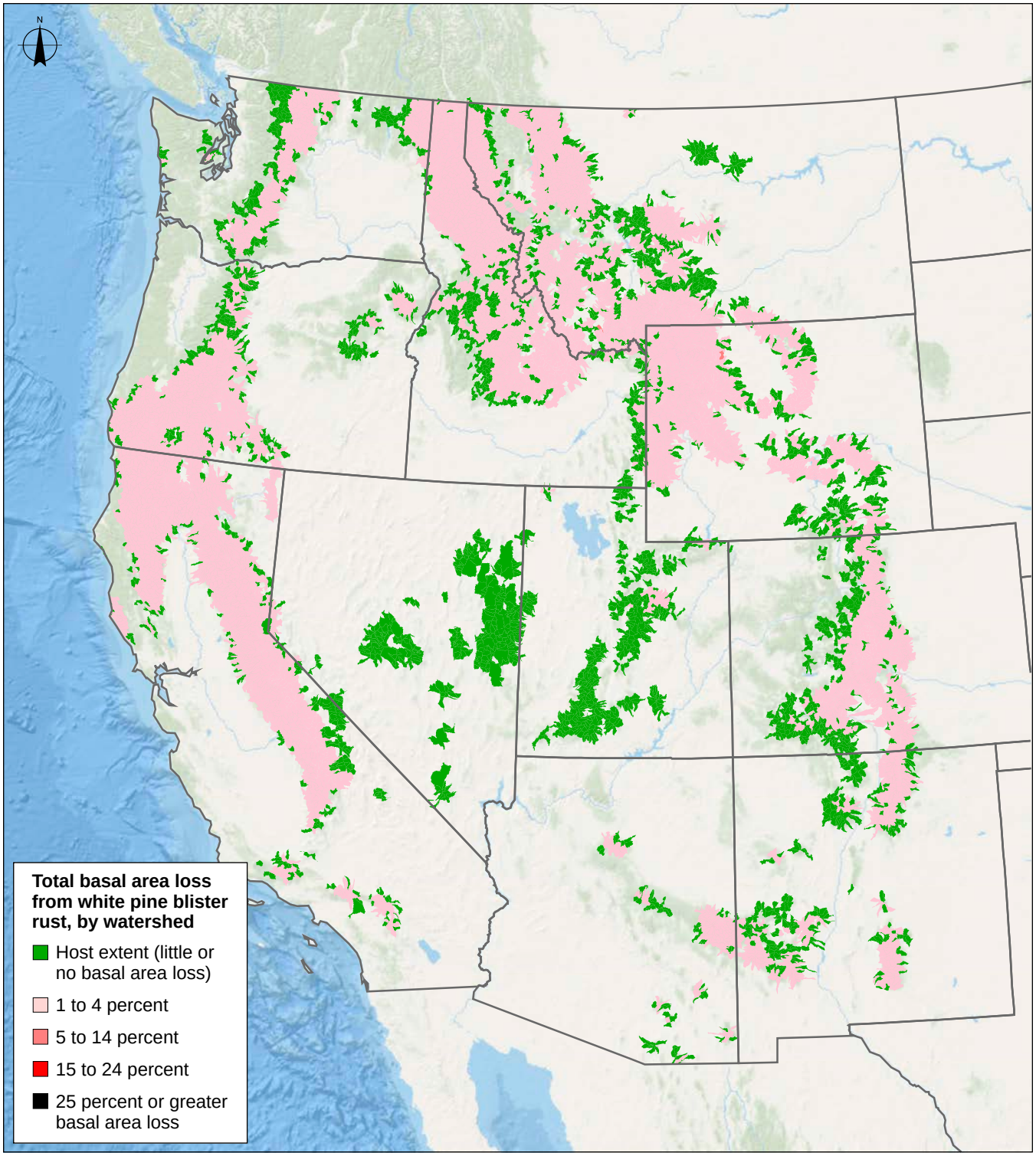

Figure 7.4—Predicted basal area loss from white pine blister rust in the Western United States. (Data are from Krist et al. 2014). Losses are based on current white pine cover type, which has been greatly diminished by past mortality from white pine blister rust, mountain pine beetle (*Dendroctonus ponderosae*), and harvesting (Neuenschwander et al. 1999).

White pine blister rust, caused by a nonnative fungus introduced in the early 1900s, over time has caused more damage than any other conifer disease in western forests.

time, the disease has caused more damage and more has been spent on control than any other conifer disease (Bega 1978). Control started in the 1930s with widespread removal of *Ribes* spp., the alternate host for white pine blister rust (Geils et al. 2010). Over time, individual trees were observed to be surviving white pine blister rust with little to no disease, so control efforts shifted into a program for breeding white pines resistant to white pine blister rust. Second- and third-generation trees bred for resistance are grown and outplanted. Genetically improved trees outperform wild trees when outplanted (Kearns et al. 2012), but they are not immune to the disease. Owing to the likelihood of blister rust damage, the number of western white pine (*P. monticola*) grown for reforestation has been reduced by 95 percent because forest managers consider it too risky to plant (Kinloch 2003), and there are few openings created to allow for outplanting stock. Of particular concern, synergistic effects of *C. ribicola*, drought, increasing temperatures, and outbreaks of mountain pine beetle (*Dendroctonus ponderosae)* are driving extensive mortality in high-elevation whitebark pine, (*P. albicaulis*) populations (Jules et al. 2016, Logan and Powell 2001).

Basal area mortality from white pine blister rust on all white pines is predicted to be low (Krist et al. 2014), but is based on current white pine cover type, which has been greatly diminished by past mortality from white pine blister rust, mountain pine beetle, and harvesting (Neuenschwander et al. 1999). One to 4 percent of total basal area mortality is predicted to occur on more than 4.7 million ha in the Western United States over a 15-year period, and 5 to 15 percent total basal area mortality is predicted to occur on nearly 0.8 million ha during the same period (2012 to 2027) (fig. 7.4) (Krist et al. 2014).

Management strategies to minimize impacts from white pine blister rust include planting of genetically improved stock resistant to white pine blister rust, thinning to improve growing conditions of host species, and pruning to remove the more susceptible lower branches and to remove nonlethal cankers before they are able to girdle the tree (Schwandt et al. 2013).

Phytophthoras, Sudden Oak Death, *Phytophthora ramorum*

Phytophthora is a fungus-like genus of microorganisms (Class Oomycetes) that includes water molds, diatoms, and brown algae. *Phytophthora* species thrive in water or moist conditions and produce swimming, flagellate spores, called zoospores. They are among the most damaging pathogens of forest, horticultural, or agricultural plants (Parke and Eberhart 2017).

A recent invader, *Phytophthora ramorum*, is now the leading biotic cause of tree mortality in California coastal forests and serves as an example of an aggressive

Sudden oak death, caused by a recent invasive pathogen, is now the leading biotic cause of tree mortality in California coastal forests.

pathogen causing disturbance along the Pacific Coast by preferentially attacking the largest, fastest growing host trees (Swiecki and Bernhardt 2006). This exotic, invasive, quarantine pathogen kills tanoak (*Notholithocarpus densiflorus*), coastal live oak, and other red oaks, but is known to infect more than 100 plant species including conifers, herbaceous plants, and ferns (Kliejunas 2010, Rizzo et al. 2005). Estimated to have been introduced sometime around 1980 to California on rhododendron, (*Rhododendron* sp.) nursery stock in landscape plantings (Mascheretti et al. 2008), *P. ramorum* escaped into the forest to kill millions of trees along the Pacific Coast, from the California central coast north to Curry County, Oregon (Grünwald et al. 2012). Tree mortality increased dramatically in the late 1990s in the San Francisco Bay Area, where more than 6 million people reside, so managing hazards (failing trees and falling branches) in residential areas, and along roadways and power lines, is a chronic concern (Frankel 2008). The pathogen is particularly disconcerting to some American Indian tribal members who rely on acorns for ceremonial use or food and may consider the primary host tree species, tanoak, to be sacred (Alexander and Lee 2010).

In heavily infested areas, such as on Mount Tamalpais (Marin County, California), the pathogen has caused type conversion of previously tanoak-dominated stands; tanoak is no longer a primary stand component (Klein et al. 2013). Tanoak stands managed by the Marin Municipal Water District are severely disease-transformed, which threatens the provisioning of drinking water and creates a high fire risk, so the district is conducting restoration trials in redwood-tanoak and Douglas-fir/tanoak forests to re-create an overstory component composed of less susceptible species, e.g., redwood (*Sequoia sempervirens*) and Douglas-fir (Cobb et al. 2017).

In Oregon from 2001 to 2012, an interagency team attempted to eradicate the pathogen, supported by the state's quarantine, which required destruction of infected and nearby host plants (Goheen et al. 2002). Eradication treatments eliminated disease from many infested sites, but the pathogen continued to slowly spread. Since 2001, the wildland quarantine area has expanded from 22 to 1333 km^2 in 2017, covering over 30 percent of Curry County, Oregon. Hundreds of thousands of tanoaks have died from *P. ramorum* in southwest Oregon (Goheen et al. 2017, Kanaskie et al. 2017). Environmental conditions along the Oregon Coast are highly conducive to disease development, and the pathogen, if untreated, is expected to expand north into Coos County and east into Josephine County (Association of Oregon Counties 2017).

In 2015, a second significant introduction of a new lineage of the pathogen was detected in Oregon, the first find of the *P. ramorum* EU1 lineage (European Union Lineage One) in a U.S. forest. This detection, on a tanoak tree about 1 mi (1.6 km)

north of a small private nursery (now closed) near the Pistol River (Curry County), underscores the importance of the nursery pathway for long-distance invasive pathogen movement (Grünwald et al. 2016).

Preventing the transport and outplanting of infested nursery stock is critical to avoid introducing pathogens to new areas.

Prevention of pathogen introduction via transport and outplanting of infested nursery stock is critical to avoid introductions to new areas (Grünwald et al. 2012). Quarantines are in effect in the United States and more than 67 other countries to prevent long-distance spread (Kliejunas 2010). Best management practices and strict sanitation may limit spread within nurseries (Parke and Grünwald 2012). Once the pathogen has been established in a forest, it is very difficult to control. Early detection, (conducted by aerial survey, waterway sampling, and ground surveillance) followed by hotspot eradication has slowed spread in Oregon (Kanaskie et al. 2017). To prevent infection, phosphonate may be injected or applied to the bark of high-risk trees, particularly on high-value coast live oak. High-value trees may also be protected by removing adjacent California bay laurels (*Umbellularia californica*), which serve as reservoirs for inoculum that can move via windblown rain to nearby oaks (Lee et al. 2011, Swiecki and Bernhardt 2013).

Abiotic Disease—Yellow-Cedar Decline

A good example of a species-specific abiotic disease is yellow-cedar decline, which has been observed across southeast Alaska for more than a century (Hennon and Shaw 1994, Sheldon 1912). Until recently, the cause of extensive mortality of Alaska cedar or yellow-cedar (*Callitropsis nootkatensis*), a culturally and economically important species, was unknown (Hennon et al. 2012). Over the past decade, Hennon and colleagues determined that, across more than 250 000 ha, Alaska cedar have died from climatic changes that cause a reduction in snow cover, promote early springtime thaws that trigger dehardening, and make the trees susceptible to root injury from freezing (D'Amore and Hennon 2006, Hennon et al. 2012, 2016).

Yellow-cedar decline is characterized by a slow decline in tree condition over several years. Affected forests are composed of standing, long-dead and recently dead and dying Alaska cedar and other tree species. Insects (*Phloeosinus* beetles) and pathogens (*Armillaria* spp. and other fungi) are contributing factors in yellow-cedar decline. Predisposing factors include landscape, site, and stand conditions (Hennon et al. 2008) that increase the probability that the fine roots of Alaska cedar will freeze during cold weather in late winter and early spring (Schaberg et al. 2008). Alaska cedar trees are healthy where snow persists past the last cold period in spring, or where they are deep-rooted on well-drained soils (Sturrock et al. 2011).

Conservation strategies that define suitable vs. unsuitable areas for Alaska cedar are recommended to determine areas where cedars are likely to survive and

thus are suitable for planting. Higher elevation sites with well-drained soils where snow or deeper rooting protect Alaska cedar roots from cold temperatures are most suitable, but may require thinning to ensure regeneration of Alaska cedar. In declining forests, salvaged wood from dead cedars may be economically valuable (Hennon et al. 2012).

Implications and Impacts of Pathogens on Sustainability

Interactions With Climate Change

The impacts of pathogens to forest sustainability are dependent upon (1) environmental conditions, (2) the distribution and density of host species, and (3) whether pathogens are present at the location. This simple model for disease expression, termed the "disease triangle," is one of the fundamental principles of plant pathology (Agrios 2005). If there are changes to any of these factors, disease patterns and pathogen impacts will change. Climate change and human population pressures alter the environmental influences (such as the amount of forest fragmentation) on plant pathogens and create new disease conditions. Changes in climate alter pathogens' ability to establish, infect, develop, and reproduce. The changes can be extreme, as in the drought that caused unprecedented levels of tree mortality in the Sierra Nevada Mountains from 2010–2017 (Tree Mortality Working Group 2017).

In the Western United States, examples of observed or predicted changes in pathogen behavior that are tied to climate change include increased damage from Swiss needle cast in the Pacific Northwest (Manter et al. 2005, Stone et al. 2008); Armillaria root disease on Douglas-fir (Klopfenstein et al. 2009); *Phytophthora cinnamomi* in Pacific Northwest forests (Burgess et al. 2017); and white pine blister rust on several species of white pines (Sturrock et al. 2011).

Changes in climate alter pathogens' ability to establish, infect, develop, and reproduce.

Similar to accelerated deterioration often observed in offsite plantings, as climate changes, plants growing in environments that are no longer suitable for them may be subject to increased damage from pathogens that attack stressed plants. As landscape-level stress intensifies, increased disturbance area, frequency, and intensity are likely, with pathogens playing a primary or secondary role (following, e.g., drought or heat stress).

Patterns of disturbance are expected to change as pathogens and their hosts adapt to new environmental conditions. The distribution of both hosts and diseases will change and a cascade of adaptations will ensue (Johnstone et al. 2016). The more rapid reproduction cycles of microbes relative to trees may give them an adaptive advantage over their hosts. Fungi and other types of pathogens may spread via wind or infested raindrops and settle in locations where the climate is newly

suitable for their survival and reproduction. Synchronicity between forest plants and pathogens will be disrupted by climate-driven changes in timing of budbreak in tree hosts, or temporal changes of pathogen spore release; these and other expected changes may alter disease incidence and severity given climate change (Sturrock et al. 2011). Drought increases stress, so trees severely infected by dwarf mistletoes may be more susceptible to insect attack, particularly bark beetles and wood borers (Kolb et al. 2016). Foliar fungal diseases and other diseases that require moisture for reproduction or spread are expected to decrease under extended drought periods (Kolb et al. 2016).

Impacts to Sustainability

Recent megadisturbances resulting from unprecedented drought, coupled with experimental findings on tree responses to warming (Adams et al. 2017a, 2017b), demonstrate that as areas warm, many forest disturbances will increase in size and frequency (Allen et al. 2015, Seidl et al. 2017). There remains much uncertainty as to how forest pathogen behavior and forest disease impacts will influence disturbance patterns given a warming and changing climate (Kolb et al. 2016). However, based on recent observations, unprecedented environmental conditions (i.e., extreme extended drought) will cause abiotic effects to increase, which may decrease the impact of biotic pathogens, because the hosts that the microbes attack will be killed by direct effects of heat, water stress, or other abiotic injury. This decrease in pathogen activity is predicted only under extreme situations (i.e., 80 percent tree mortality), such as occurred in parts of the central Sierra Nevada Mountains from 2015 to 2017 (Ballard 2017).

As areas transformed by megadisturbance are reforested, the increase in replanting may raise the potential for pathogen introductions or facilitate the spread of root diseases from living stumps to adjacent seedlings (Lockman and Kearns 2016). The risk of pathogen introduction may be elevated by planting of infested nursery stock, grading with contaminated equipment, or other human-mediated problems (Rooney-Latham et al. 2015).

Interactions Among Disturbance Agents

Under less extreme conditions, pathogens will continue to increase stress and damage trees directly or indirectly through interactions with insects, fire, and other disturbance agents (Sturrock et al. 2011). In areas that become overstocked, experience water-stress, or have been heavily affected by past land uses, trees may be prone to fire, beetle attack, or pathogen injury (Doblas-Miranda et al. 2017). If one of these mortality agents is favored, the mortality from the other agents may decrease because the number of suitable hosts has been reduced. Widespread fire

may reduce damage from pathogens as the heat produced can kill many microbes and dwarf mistletoes (Alexander and Hawksworth 1976, Shaw and Agne 2017) or eliminate the hosts they require for survival (Harrington and Hawksworth 1990). However, conversely, pathogens may influence fuels and stand structure to intensify fires (Metz et al. 2011, 2017).

Many management actions aimed at improving forest conditions may decrease detrimental impacts of forest pathogens, particularly thinning to reduce competition or brush control to enable seedling establishment (Jactel et al. 2009). However, precautions are needed during stand manipulations to avoid wounds that can serve as pathogen infection courts, or the creation of stand conditions that are favorable to mistletoes, root disease, or invasive pathogens (Muzika 2017). Additionally, although silvicultural treatments can reduce stress on residual trees in the short term, they also increase risk given the greater reliance placed on the remaining trees because the reduced density allows for larger trees, which can lead to hydraulic constraints (D'Amato et al. 2013). For each geographic area, prevention of known pathogen problems needs to be integrated into treatment strategies; e.g., in Oregon and Washington, species susceptible to laminated root rot (*Phellinus* spp.) should not be planted into areas with a history of root disease (Agne et al. 2018).

Management approaches to address climate change may inadvertently increase forest disease. In particular, assisted migration of tree species may allow pathogens to be introduced on nursery stock (Rooney-Latham et al. 2015). Introducing new plant species into new areas presents a risk that the environment will create conducive conditions for novel diseases.

Given the uncertainty of managing forests under a changing climate, increased monitoring of forest condition and pathogen distribution would enable wildland managers to respond to new invasive pathogens or newly aggressive native pathogens before they establish so widely that they become too expensive or logistically difficult to manage (Millar and Stephenson 2015). Species diversification and various planting configurations should be tried in new plantations (Thompson et al. 2009). Adaptive management approaches in which vegetation managers plan for the unexpected would increase resiliency against disturbance (Millar et al. 2007).

Increased monitoring of forest condition and pathogen distribution might enable managers to respond before new invasive pathogens become too expensive or difficult to manage.

Most forests will have to adapt to climate change without management intervention (Seidl et al. 2017, Spittlehouse and Stewart 2003), thus, in actively managed forest areas, decisions should be carefully designed to maximize the likelihood of tree survival and ecosystem benefits. Prescriptions for higher planting densities to account for anticipated losses to forest diseases should be avoided (Woods and Coates 2013).

Adapting to changing forest conditions resulting from climate change and other anthropogenic causes will require reconsideration of all aspects of forest management (i.e., yield tables, site quality indices) (D'Aprile et al. 2015), and expectations about the impacts of forest pathogens need to be reexamined as well. Unprecedented tree mortality is raising awareness of the vulnerabilities that forests face (Fettig et al. 2013); to sustain forests, we need a better understanding of the role of forest pathogens under current and anticipated future conditions and disturbance patterns. Needed actions include an increase in forest pathogen and climate change experimental data as well as detailed recording of observations of responses to warming, as well as inventory monitoring that more accurately attributes the cause of tree mortality so that ecological impacts of pathogens can be understood and addressed.

Conclusions

Changes in forest disturbance patterns, such as wildfires, will change the impacts of pathogens, creating novel forms of disruption to ecosystem services, as well as safety and management problems.

Changes in other forest disturbance patterns, such as wildfires, will change the impacts of pathogens, creating novel forms of disruption to ecosystem services, as well as safety and management problems. Tree mortality rates and growth responses will evolve over time, as pathogens and plant species adapt to one another (Zenni et al. 2017). Our ability to characterize pathogen issues under changing disturbance regimes is limited. If drought, fire, and other abiotic disturbances are extreme, the impact of pathogens may decrease because there will be fewer host plants for microbes to infect, and the weakest in the tree population will have been removed by other mortality agents.

Management approaches should consider the need to integrate historical, current, and predicted pathogen risks for their particular geographic area and to embed disease prevention into treatment strategies. Increased monitoring allows us to better understand the current impacts of pathogens on growth and tree mortality so that changes can be recognized and addressed. Prevention of pathogen invasions would lessen degradation of western forests.

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Chapter 8: Invasive Plants

Andrew N. Gray, Kathryn C. Baer, and Christopher Witt[1]

Introduction

Nonnative invasive plants are a potential threat to the sustainability of Western U.S. forests. In the implementation of the Montréal Process Criteria and Indicators, invasive plants are recognized as affecting native species diversity (indicator 1.6), the status of endangered species (indicator 1.7), and the area of forest outside the range of natural variation (indicator 3.15) (Robertson et al. 2011).[2] Invasive plants can affect forest ecosystems by reducing the abundance and functions of native species, and they can affect nutrient cycling and the fitness of native plants and animals (Pyšek et al. 2012). Invasive plants can also change the probability of other disturbances, most notably increasing the probability of wildfire (D'Antonio and Vitousek 1992). Because many invasive species are well suited to disturbed and early-seral vegetation conditions, this can set up a positive feedback loop that further degrades native ecosystems. It is estimated that invasive plants cost the U.S. economy $34 billion per year in lost productivity and control and eradication costs, primarily on agricultural lands (Pimentel et al. 2005); a study of selected plants in Oregon forests suggested a cost of $80 million per year (The Research Group 2014).

This chapter assesses the abundance of nonnative plants in Western U.S. forests, draws from the literature to discuss control and eradication efforts and species' potential impact on forest sustainability, and evaluates options for continued monitoring and understanding of their role in our forests.

U.S. Federal Executive Order 11312 on Invasive Species (Feb. 8, 1999; Executive Order 1999) defines as "invasive" a species alien to a particular ecosystem whose introduction causes or is likely to cause economic or environmental harm, or harm to human health (Moser et al. 2009). Although many nonnative species were introduced accidentally (e.g., as contaminants in imported seed), others were purposely introduced to provide benefits in particular conditions (e.g., agricultural crops or erosion control) but are deemed invasive when they have unanticipated negative impacts or become established in other environments. Identifying which

[1] **Andrew N. Gray** is a research ecologist, Pacific Northwest Research Station, 3200 SW Jefferson Way, Corvallis, OR 97331; **Kathryn C. Baer** is an ORISE postdoctoral research fellow, Pacific Northwest Research Station, 101 East 1st Avenue, Door 8, Anchorage, AK 99501; and **Christopher Witt** is an ecologist, Rocky Mountain Research Station, 507 25th Street, Boise, ID 83702.

[2] Indicator numbers follow the system established in Robertson et al. (2011), in which the first numeral refers to the criterion and the second numeral refers to the indicator. This is a different numbering system than that used by the Montréal Process, although indicator titles are **identical.**

Identifying which plants are nonnative is straightforward, but prioritizing which are invasive depends on the values of interest.

species are "alien" or nonnative is relatively straightforward. Defining and prioritizing plants that are causing or are likely to cause harm depends to a large degree on the ecosystem and values of interest. For example, state noxious weed lists, which require control efforts for listed species, tend to focus on protecting agricultural production, with less emphasis on native forest and nonforest vegetation types. In contrast, organizations like The Nature Conservancy have developed specific criteria for prioritizing invasives based on ecological impact, current distribution, current trend, and management difficulty (Morse et al. 2004).

Usually, an introduction occurs with a few organisms in a limited geographical area, particularly if the introduction is unintentional. Although there are some areas where introductions are more likely than others (e.g., commercial ports, transportation corridors), detecting early introductions is difficult simply because few organisms are present, and the taxonomic expertise to recognize them may be rare (Hulme et al. 2006). Once established, plant invasions tend to follow a logistic population growth curve (Radosevich et al. 2005). However, many species may remain at low levels for years or decades and seem innocuous before they accumulate enough in population or something triggers an eruptive phase (e.g., Rosenthal et al. 2008). Or they may remain at a low frequency indefinitely. Although finding invasive plants becomes easier as they expand, eradicating them becomes more costly. Because of the low density of samples, broad-scale strategic forest monitoring will not be useful for detecting and characterizing invasive populations when they are rare. However, these monitoring efforts may help to identify the location and extent of existing infestations of invasive species and facilitate management responses.

Distribution and Abundance of Invasive Plants in Western U.S. Forests

Datasets

The only comprehensive and consistent source of information on the location and severity of plant invasions in Western U.S. forests are the inventories conducted by the Forest Inventory and Analysis program (FIA) (Gray et al. 2012). FIA samples the nation's lands with a base grid of points at a density of one per 2400 ha. Each year's samples are evenly distributed; under the base cycle in the West, they are remeasured every 10 years (Bechtold and Patterson 2005). Vegetation on grid points falling on forest land (≥10 percent current or recent cover of tree species and ≥0.4 ha) are measured with four 7.3-m-radius (168 m^2) subplots. Because the grid points are systematically located, some plots straddle stand boundaries, roads, and other nonforest conditions (nonforest has either never had 10 percent cover of tree species

or is being currently mowed or otherwise managed for nonforest uses). Inventory crews note the occurrence and type of silvicultural activity or natural disturbance that affected vegetation on the plot since the previous measurement. In the Western United States, FIA crews use the "phase 2" (P2) vegetation protocol to record the identity and cover of the four most abundant species per life form (tree, shrub, forb, graminoid) present on each forested subplot with at least 3 percent areal cover[3] (USDA FS 2018). In addition, the presence in any amount of species on targeted invasive lists is also recorded in most states. In interior Western States, the invasive lists consist of each state government's noxious weed list; in Pacific states, the lists were compiled from literature and consultation with clients and experts to produce a short list of species of greatest concern (the protocol has not been implemented in California, Oregon, or Washington to date). In addition, FIA has a "phase 3" (P3) vegetation indicator protocol in which experienced botanists conduct a full sample of all vascular plants, during the peak phenological period, on a subset of plots (Schulz et al. 2009); the protocol has been implemented only in a few states for a few years in the West. The National Forest System's Northern Region (which includes Montana and northern Idaho), Pacific Southwest Region (which includes California), Pacific Northwest Region (Oregon and Washington), and Alaska Region also fund data collection on nonforest plots using the same FIA protocols and plot grid. Some national forests also fund data collection on a greater density of plots. FIA data and tools to query them are publicly available (https://www.fia.fs.fed.us/tools-data/).

The regional-scale sample and plot-level cover estimates enable calculation of the land area covered by a particular species as well as the mean cover for any domain of interest (e.g., forest type). This estimate of area covered is different than commonly presented statistics of area "infested" by invasive plants, which is based on delineated patches where invasive plant cover may range between a few to 100 percent. Given the number of different criteria that classify nonnative plants as invasive, we decided to analyze the abundance and distribution of any nonnative plant recorded in FIA inventories. We used the nonnative criteria in the PLANTS database (USDA NRCS 2019), which are defined for broad geographic areas (e.g., the conterminous United States), and treated species known to have mixed non-native/native genotypes (e.g., common yarrow) as native. Because invasive lists differ within and across regions, the analyses presented here rely primarily on the

[3] The 3 percent cover and four most abundant criteria result in underestimating species occurrence, particularly of species that typically occur at low densities; an analysis of older data with no maximum number limit indicated that 6.9 percent of species records would be missed, or 2.1 percent of the plant cover, by recording only the four most abundant per life form (unpublished analysis).

P2 dominant species sample and identify only the species found on one or more invasive lists to provide a sense of those identified as problematic. Most of the results present cover measurements taken in the most recent 10-year measurement cycle, 2007–2016. The change estimates are based on when a state began annual inventory and the 10-year intervals the plots have been remeasured since then, referred to generally as "change from the 2000s to 2010s"; for example, 2001–2006 to 2011–2016 for Arizona. For the states with available P3 data (collected irregularly 2001–2010), we compare estimates of nonnative abundance with those from the P2 sample. We report on nonnative plants as a whole and note those species that are found on one or more FIA invasive plant lists. From the cover estimates of nonnative plants on each plot, we calculate the land area covered, the mean cover for the population (referred to as "ground cover"), and the "relative cover," calculated as the sum of nonnative species' cover divided by the sum of all understory (non-tree) plant cover. Standard errors for the estimates reflect inventory sample error. Independent estimates can be compared by multiplying the standard error (SE) by 1.96, with values outside that range considered significantly different. We also provide some estimates of the proportion of plots occupied in the P2-P3 comparison.

Results

Cover of nonnative plants in forests differed dramatically by region and state within region. Nonnative plant cover was highest in Hawaii at 46 ± 2.5 percent (percentage ground cover ± SE) and lowest in coastal Alaska at 0.01 ± 0.01 percent (table 8.1). Nonnative cover was significantly lower in Rocky Mountain (RM) states than in Pacific Coast (PC) states. New Mexico had the lowest cover within RM states at 0.15 ± 0.02 percent, while Nevada had the highest at 1.4 ± 0.14 percent. Within PC states, nonnative cover was highest in California at 5.2 ± 0.19 percent, with cover in both Oregon and Washington at 2.2 ± 0.12 percent. Relative cover of nonnatives was substantially higher than ground cover, particularly in sparsely vegetated more arid states where the sum of all plant cover is substantially less than 100 percent. For example, relative cover of nonnatives was 20 ± 1.7 percent in Nevada and 17 ± 0.55 percent in California (table 8.1).

Cheatgrass covers 480 thousand hectares of forestland in the Western United States.

Cheatgrass was by far the most abundant species in western forests, covering 1.2 million ac [480 thousand ha], or 0.49 ± 0.017 percent cover of all forested land in the conterminous Western United States (fig. 8.1). The area of RM forest land occupied by the other 195 nonnative species found there was less than one-fifth the area of cheatgrass (table 8.2). Several of the most abundant nonnatives in RM states were grasses that were not on any of the individual states' noxious weed lists, including timothy, crested wheatgrass, red brome, and orchardgrass. Of the 15 plants with the

Table 8.1—Forest land area and proportion covered by nonnative plants as total area and percentage (ground), and percentage of total vegetation cover (relative), from Forest Inventory and Analysis sample 2007–2016

		Nonnative cover					
Region/state	**Forest land**	**Total**	**SE**	**Ground**	**SE**	**Relative**	**SE**
	- - - - Thousand hectares - - - -			*- - - - - - - - - - - - Percent - - - - - - - - - - -*			
Rocky Mountain:							
Arizona	7449	46.8	4.9	0.63	0.06	4.82	0.48
Colorado	9229	65.2	5.8	0.71	0.06	3.15	0.27
Idaho	8737	51.9	4.5	0.59	0.05	2.10	0.18
Montana	10 487	124.7	7.0	1.19	0.07	4.04	0.22
Nevada	4257	58.8	5.9	1.38	0.14	20.31	1.69
New Mexico	9956	15.2	2.4	0.15	0.02	1.36	0.22
Utah	7335	59.6	5.5	0.81	0.07	8.21	0.71
Wyoming	4226	39.8	5.8	0.94	0.13	3.31	0.46
Rocky Mountain total	61 675	461.9	15.2	0.75	0.02	3.87	0.12
Pacific Coast							
California	12 894	1647.8	64.1	5.17	0.19	16.95	0.55
Oregon	12 002	639.4	29.9	2.16	0.10	5.52	0.25
Washington	8966	501.0	28.3	2.26	0.13	3.91	0.21
Pacific Coast total	33 862	1128.4	30.8	3.33	0.09	8.17	0.21
Coastal Alaska	6226	0.6	0.3	0.01	0.00	0.01	0.01
Hawaii	595	272.1	18.6	45.70	2.52	50.46	2.41
Total	102 358	1862.6	39.1	1.82	0.04	6.54	0.13

SE = standard error.

most cover, spotted knapweed, Canada thistle, saltcedar, quackgrass, gypsyflower, and redstem stork's bill were on one or more states' noxious weed list. Grasses were also important in the PC states, although the species in the 15 most abundant list were different from those in the RM states (table 8.3). In the PC states, 259 nonnative plant species were recorded. Of the top 15 species, Himalayan blackberry and medusahead (fig. 8.1), as well as bulbous bluegrass and Scotch broom, were on the proposed FIA invasives list (Gray et al. 2011). In coastal Alaska, eight nonnative plant species were recorded (fig. 8.1). Common dandelion was the most abundant nonnative plant, followed by reed canarygrass and clover (table 8.4). In Hawaii, 136 nonnative species were recorded. The most abundant species in Hawaii was strawberry guava, covering 9 percent of the forested area in the state (table 8.5, fig. 8.1). There were more shrubs in the list of most abundant nonnative species in Hawaii than in the other states.

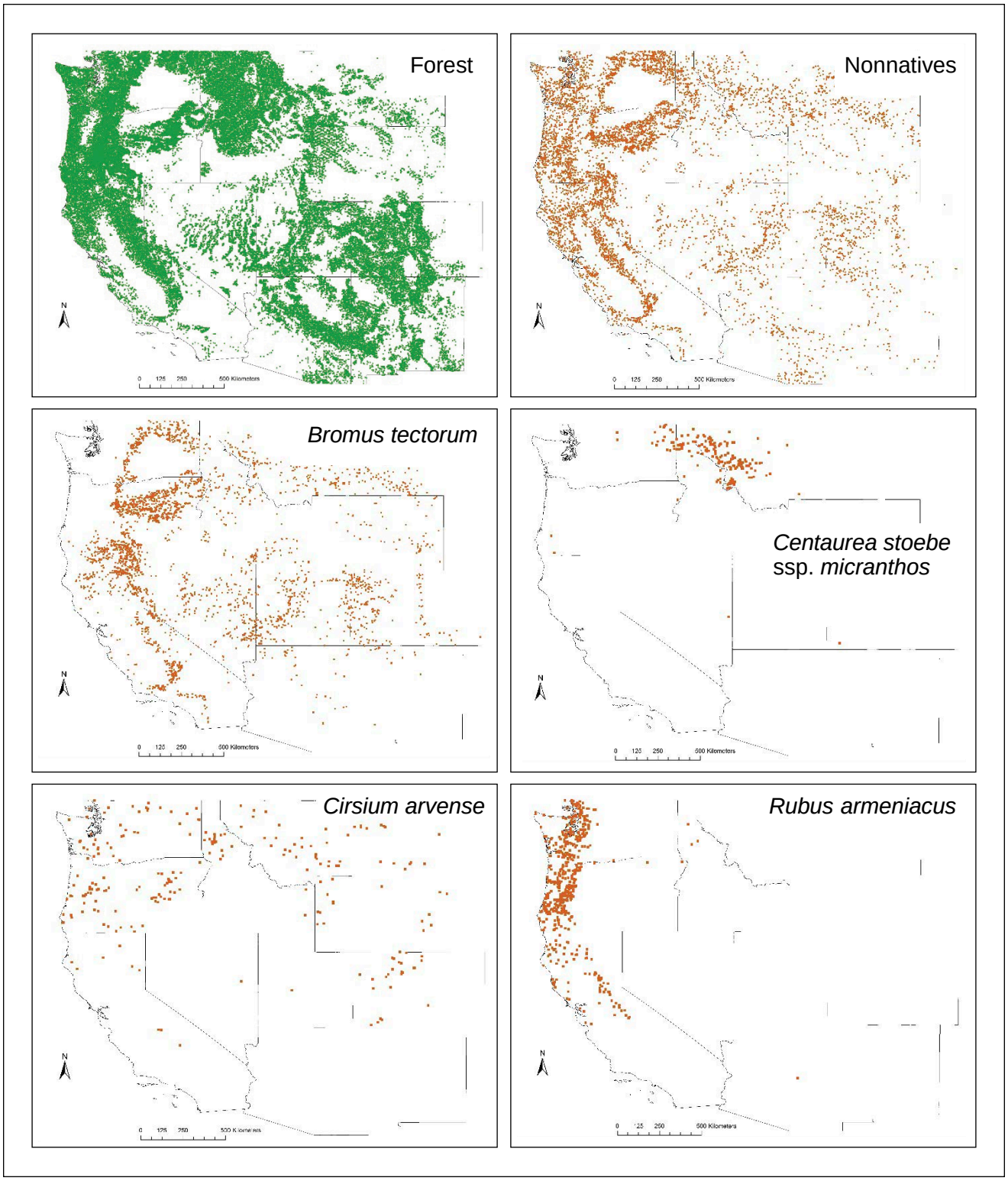

Figure 8.1—Maps of distribution of forested Forest Inventory and Analysis plots and those with cover of any nonnative plants, and of selected species, 2007–2016. Note that plot density is about three times higher on national forests in Oregon and Washington.

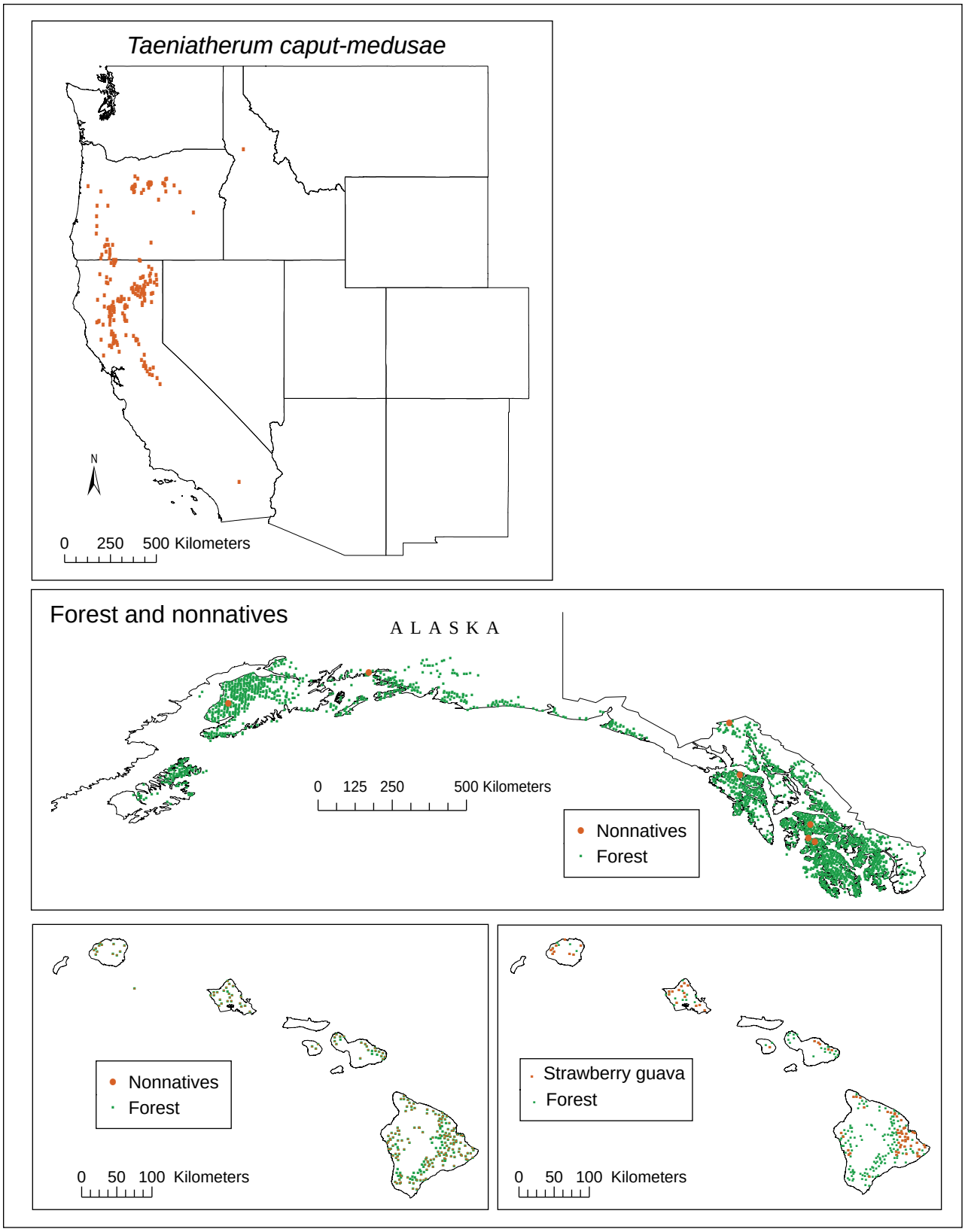

Figure 8.1—Continued.

Table 8.2—The 15 nonnative plants that cover the most forest land in the Rocky Mountain states out of 195 species recorded, 2007–2016

Scientific name	Common name	Area	SE	Cover	SE
		- - - Hectares - - -		*- - - Percent - - -*	
Bromus tectorum[a]	Cheatgrass	212 082	11 022	0.85	0.04
Phleum pratense	Timothy	36 365	3502	0.15	0.01
Centaurea stoebe ssp. *micranthos*[a]	Spotted knapweed	18 969	2443	0.08	0.01
Agropyron cristatum	Crested wheatgrass	16 538	2384	0.07	0.01
Cirsium arvense[a]	Canada thistle	14 098	1082	0.06	0.00
Eragrostis lehmanniana	Lehmann lovegrass	14 047	3202	0.06	0.01
Bromus rubens	Red brome	12 027	1806	0.05	0.01
Dactylis glomerata	Orchardgrass	11 852	2124	0.05	0.01
Bromus arvensis	Field brome	11 456	1751	0.05	0.01
Melilotus officinalis	Yellow sweetclover	5266	1266	0.02	0.01
Tamarix ramosissima[a]	Saltcedar	5250	1859	0.02	0.01
Elymus repens[a]	Quackgrass	5134	1302	0.02	0.01
Cynoglossum officinale[a]	Gypsyflower	4368	391	0.02	0.00
Erodium cicutarium[a]	Redstem stork's bill	4256	1167	0.02	0.00
Thinopyrum intermedium	Intermediate wheatgrass	4184	1159	0.02	0.00

SE = standard error.
[a] Species on one or more Forest Inventory and Analysis state invasive species lists.

Table 8.3—The 15 nonnative plants that cover the most forest land in the Pacific Coast states out of 259 species recorded, 2007–2016

Scientific name	Common name	Area	SE	Cover	SE
		- - - Hectares - - -		*- - - Percent - - -*	
Bromus tectorum[a]	Cheatgrass	256 164	11 810	1.87	0.09
Rubus armeniacus[a]	Himalayan blackberry	137 002	9 603	1.00	0.07
Bromus diandrus	Ripgut brome	97 941	8 869	0.71	0.06
Cynosurus echinatus	Bristly dogstail grass	61 914	5 212	0.45	0.04
Bromus hordeaceus	Soft brome	52 072	5 558	0.38	0.04
Torilis arvensis	Spreading hedgeparsley	46 876	4 358	0.34	0.03
Avena fatua	Wild oat	45 582	6 032	0.33	0.04
Taeniatherum caput-medusae[a]	Medusahead	42 790	4 936	0.31	0.04
Erodium botrys	Longbeak stork's bill	28 636	4 446	0.21	0.03
Poa bulbosa[a]	Bulbous bluegrass	26 999	3 356	0.20	0.02
Brachypodium distachyon	Purple false brome	25 508	4 692	0.19	0.03
Erodium cicutarium	Redstem stork's bill	22 633	4 387	0.17	0.03
Cytisus scoparius[a]	Scotch broom	17 493	3 137	0.13	0.02
Holcus lanatus	Common velvetgrass	15 760	2 608	0.11	0.02
Hypochaeris radicata	Hairy cat's ear	14 695	3 166	0.11	0.02

SE = standard error.
[a] Species on the proposed Forest Inventory and Analysis invasive species list.

Table 8.4—The nine nonnative plants recorded on forest land in coastal Alaska, 2006–2015

Scientific name	Common name	Area	SE	Cover	SE
		- - *Hectares* - -		- - *Percent* - -	
Taraxacum officinale[a]	Common dandelion	292	210	0.00	0.00
Phalaris arundinacea[a]	Reed canarygrass	265	128	0.00	0.00
Trifolium[a]	Clover	43	24	0.00	0.00
Rumex crispus	Curly dock	18	21	0.00	0.00
Vicia cracca ssp. *cracca*[a]	Bird vetch	7	7	0.00	0.00
Matricaria discoidea[a]	Disc mayweed	5	6	0.00	0.00
Phleum pratense[a]	Timothy	5	6	0.00	0.00
Plantago major[a]	Common plantain	5	6	0.00	0.00

SE = standard error.
[a] Species on the Forest Inventory and Analysis invasive species list.

Table 8.5—The 10 nonnative plants that cover the most forest land in Hawaii, out of 136 species recorded, 2010–2015

Scientific name	Common name	Area	SE	Cover	SE
		- - - *Hectares* - - -		- - - *Percent* - - -	
Psidium cattleianum[a]	Strawberry guava	55,858	7,586	9.39	1.27
Urochloa maxima	Guinea grass	18,040	4,614	3.03	0.78
Clidemia hirta[a]	Soapbush	16,873	2,983	2.84	0.50
Pennisetum clandestinum	Kikuyugrass	16,577	4,867	2.79	0.82
Melastoma candidum[a]	Asian melastome	10,213	2,738	1.72	0.46
Schinus terebinthifolius[a]	Brazilian peppertree	10,191	2,973	1.71	0.50
Microlaena stipoides	Weeping grass	9,919	3,239	1.67	0.54
Psidium guajava[a]	Guava	9,088	3,024	1.53	0.51
Ardisia elliptica[a]	Shoebutton	6,973	2,934	1.17	0.49
Dodonaea viscosa	Florida hopbush	6,945	1,582	1.17	0.27

SE = standard error.
[a] Species on the Forest Inventory and Analysis invasive species list.

Nonnative plants were most abundant in dry forest types and stages of stand development with sparse tree cover.

The abundance of nonnative plants varied among forest types. In the RM states, the nonstocked forests (forest land where current tree stocking was <10 percent) had the highest mean cover of nonnative plants at 3.8 ± 0.25 percent (fig. 8.2). Among remaining forest types, mean nonnative cover was between 0.5 and 1.0 percent in hardwood and low-elevation and dry conifer types (e.g., pinyon/juniper, ponderosa pine), and <0.5 percent for high-elevation and moist conifer types (e.g., true fir/Engelmann spruce and western hemlock/redcedar). In the PC states, mean nonnative plant cover was highest in oak-dominated forest types at 14.0 ± 0.57 percent (fig. 8.2). As found in the RM states, nonnative plant cover was

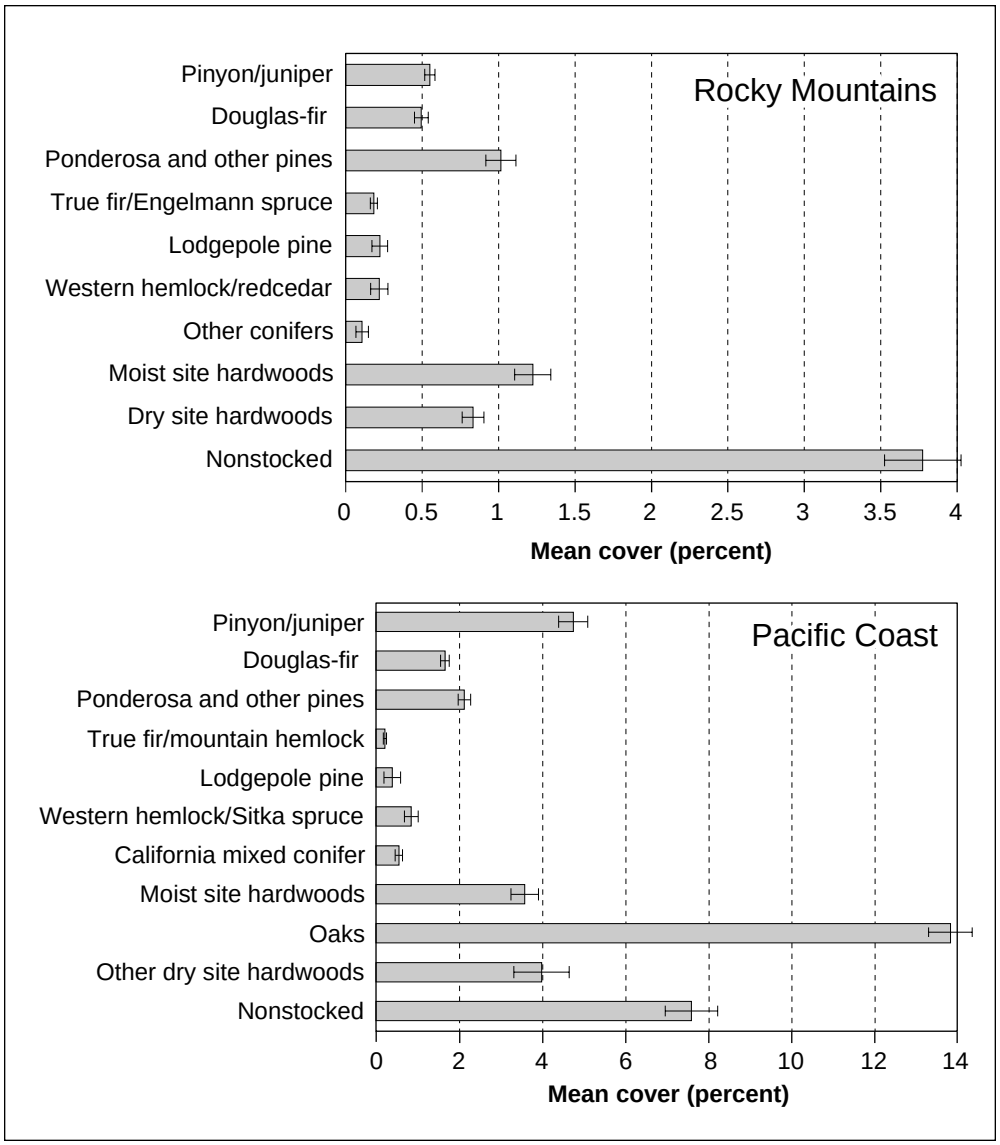

Figure 8.2—Forested area covered by nonnative plants (with standard errors) by forest type group in Rocky Mountain and Pacific Coast states, 2007–2016. Scale of *x*-axis differs between regions.

higher in hardwood and dry conifer forest types than in high-elevation and moist conifer types. Similarly, in coastal Alaska, most of the sites where nonnative plants were found were either nonstocked or hardwood types, and close to roads (data not shown). In Hawaii, mean cover was similar (>40 percent) in all forest types except for cloud forest, where mean cover was 7.8 ± 2.5 percent.

Mean cover of nonnative plants varied by stand age. For both the RM and PC regions, mean cover of nonnative plants in the oldest stands (>200 years) was less than one-fifth the cover of nonnative plants in the youngest stands (0 to 50 years)

(fig. 8.3). Although stand age is clearly related to time since severe disturbance, the incidence of disturbance in the previous 10 years was not a definitive explanation of nonnative plant cover. Although nonnative cover was higher on sites experiencing fire than other disturbances (or no disturbance) in RM states, differences were less among disturbance types in PC states (fig. 8.4).

Proximity to nonforest land conditions, which tend to be preferred habitat for many nonnative species or frequently disturbed (e.g., roads), was significantly associated with cover of nonnative species. Mean nonnative plant cover was significantly lower on plots that were entirely forest than on forested portions of plots that contained both forest and nonforest land, except for coastal Alaska, where low abundance of nonnatives led to a relatively high standard error (table 8.6).

National forests in the Northern Rocky Mountains (U.S. Forest Service Northern Region), coastal Alaska (Alaska Region), and the Pacific Coast (Pacific Southwest and Northwest Regions) measure vegetation on nonforest land on FIA plots. Although cover of nonnative plants was similarly low on forest and nonforest lands on national forests in coastal Alaska, mean cover of nonnative plants on RM and PC national forests was more than five times greater on nonforest conditions than on forested conditions (2.1 ± 0.4 vs. 0.32 ± 0.03, and 5.5 ± 0.4 vs. 0.96 ± 0.06 percent, respectively) (table 8.7). Cheatgrass covered the most area on both forest and nonforest lands on RM and PC national forests (tables 8.8 and 8.9). Two of the top five species on nonforest land in both the RM and PC regions were relatively uncommon on forest land, covering less than 2,000 ac [809 ha]; those species were crested wheatgrass and field brome in the RM region, and North Africa grass and field brome in the PC region.

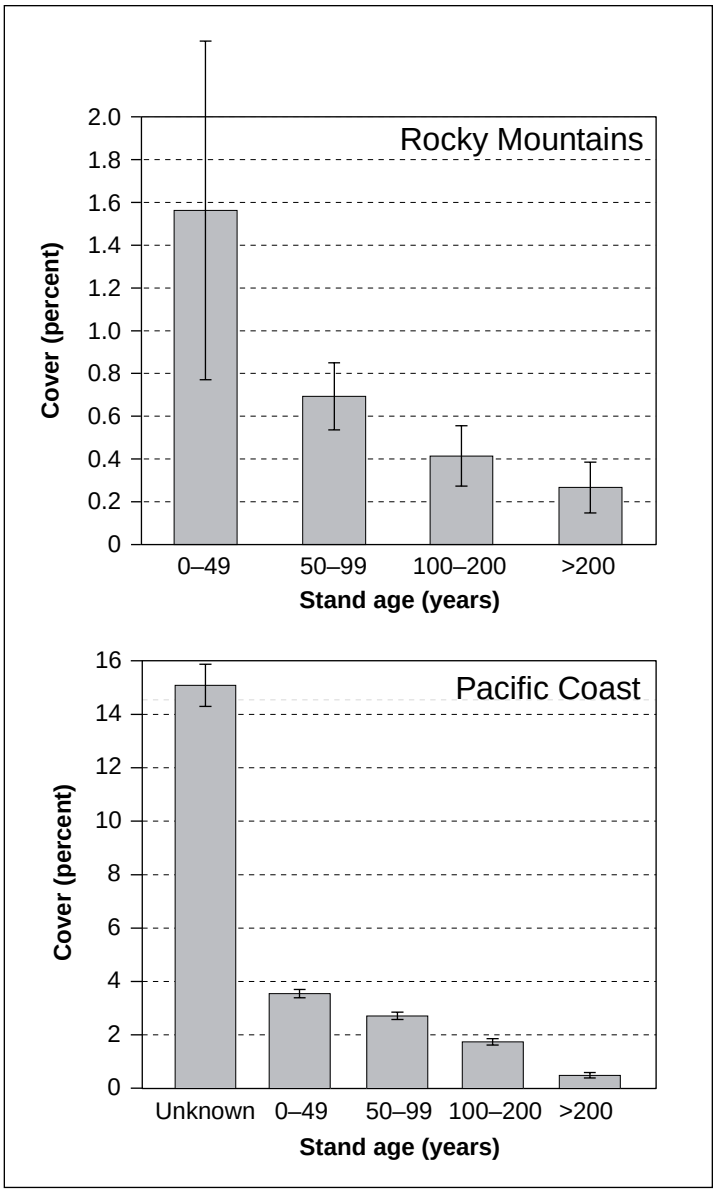

Figure 8.3—Mean cover of nonnative plants by stand age class and region, 2007–2016. The unknown stand ages in the Pacific Coast states are primarily oak-dominated stands where trees were not increment-cored for age.

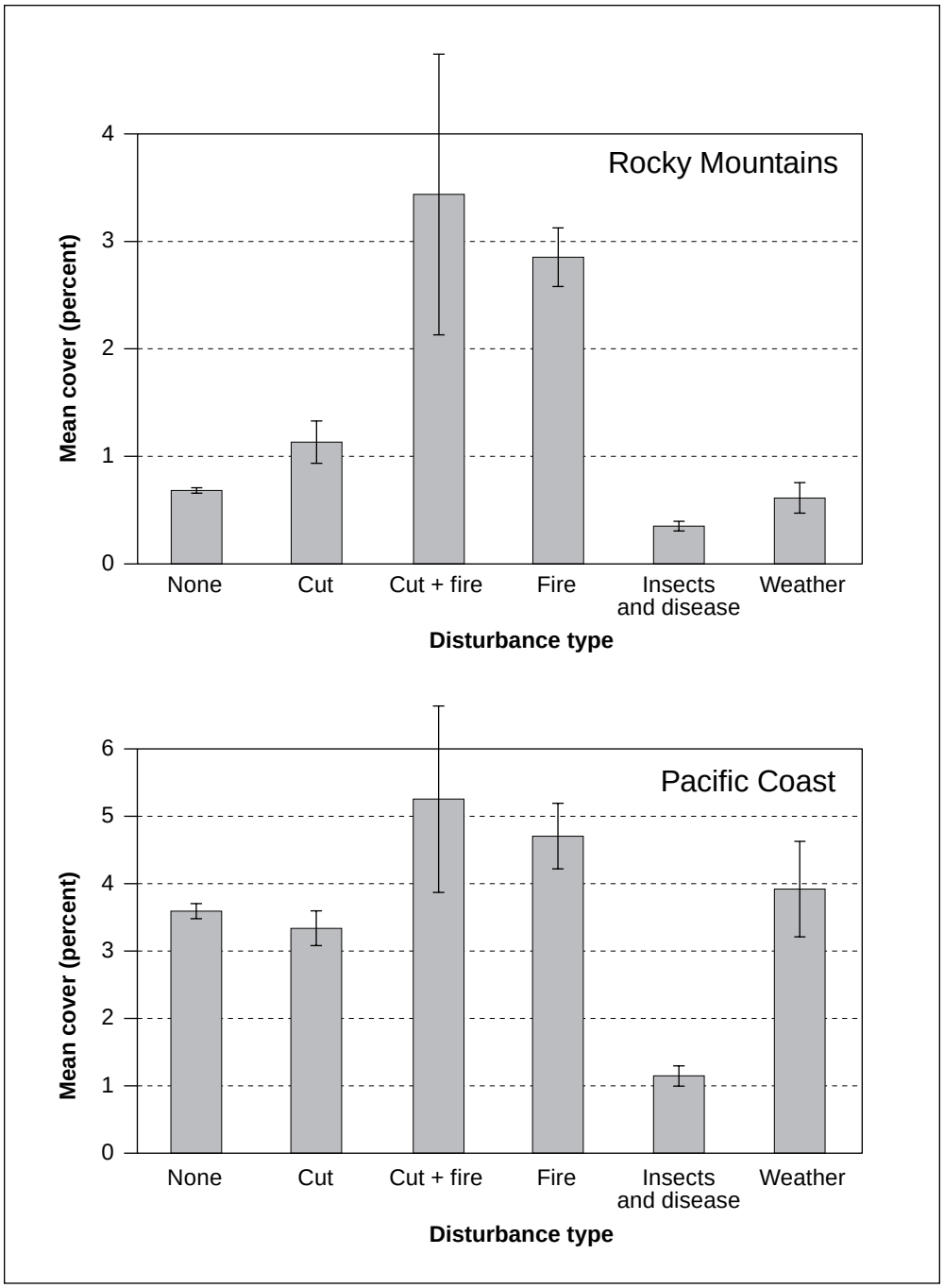

Figure 8.4—Mean cover of nonnative plants by disturbance type occurring in the 10 years prior to measurement, by region, 2007–2016. Insect and disease impacts are grouped under "insectdis"; *y*-axis scale differs between regions.

Table 8.6—Mean cover of nonnative plants comparing fully forested plots to forested portions of plots with forest and nonforest conditions, by region, 2007–2016

	Fully forested		Partially forested	
Region	**Cover**	**SE**	**Cover**	**SE**
	Percent			
Rocky Mountain	0.69	0.03	1.36	0.09
Pacific Coast	2.86	0.09	6.26	0.30
Alaska	0.01	0.00	0.15	0.10
Hawaii	44.67	2.66	59.86	7.62

SE = standard error.

Table 8-7—Comparison of nonnative cover on forest and nonforest lands where measured on national forests by region, showing land area, nonnative area, and mean percentage of cover, 2007–2016

	Forest		Nonforest	
Region	**Estimate**	**SE**	**Estimate**	**SE**
Land area:				
	Hectares			
Rocky Mountain	8 907 242	88 500	882 962	42 838
Pacific Coast	15 076 562	59 856	1 551 951	44 173
Coastal Alaska	2 694 493	100 271	862 130	58 497
Nonnative area:				
	Hectares			
Rocky Mountain	28 635	2 929	18 305	3 435
Pacific Coast	145 093	9 061	85 039	6 632
Coastal Alaska	161	102	16	18
Nonnative cover:				
	Percent			
Rocky Mountain	0.32	0.03	2.07	0.38
Pacific Coast	0.96	0.06	5.47	0.41
Coastal Alaska	0.01	0.00	0.00	0.00

SE = standard error.

Table 8.8—The five nonnative plants that cover the most forest or nonforest land on U.S. Forest Service Northern Region national forests (Montana and northern Idaho), 2007–2016

	Common name	Area	SE	Mean cover	SE
		- - - Hectares - - -		*- - - Percent- - -*	
Forest:					
Bromus tectorum	Cheatgrass	7 864	1 573	0.09	0.02
Phleum pratense	Timothy	6 032	1 451	0.07	0.02
Centaurea stoebe ssp. *micranthos*	Spotted knapweed	3 616	793	0.04	0.01
Hypericum perforatum	Common St. John's wort	1 251	577	0.01	0.01
Cirsium arvense	Canada thistle	1 091	442	0.01	0.00
Nonforest:					
Bromus tectorum	Cheatgrass	3 977	1 042	0.45	0.12
Phleum pratense	Timothy	3 679	1 110	0.42	0.12
Agropyron cristatum	Crested wheatgrass	2 118	2 101	0.24	0.24
Bromus arvensis	Field brome	1 900	775	0.22	0.09
Centaurea stoebe ssp. *micranthos*	Spotted knapweed	1 262	523	0.14	0.06

SE = standard error.

Table 8.9—The five nonnative plants that cover the most forest or nonforest land on Pacific Coast national forests, 2007–2016

Scientific name	Common name	Area	SE	Mean cover	SE
		- - - Hectares - - -		*- - - Percent - - -*	
Forest:					
Bromus tectorum	Cheatgrass	74 201	5 795	0.49	0.04
Bromus diandrus	Ripgut brome	8 068	1 967	0.05	0.01
Torilis arvensis	Spreading hedgeparsley	5 603	1 609	0.04	0.01
Cynosurus echinatus	Bristly dogstail grass	5 200	1 267	0.03	0.01
Avena fatua	Wild oat	4 723	1 918	0.03	0.01
Nonforest:					
Bromus tectorum	Cheatgrass	50 509	4 252	3.25	0.26
Ventenata dubia	North Africa grass	4 771	1 184	0.31	0.08
Bromus diandrus	Ripgut brome	3 152	1 540	0.20	0.10
Bromus arvensis	Field brome	2 785	637	0.18	0.04
Taeniatherum caput-medusae	Medusahead	2 101	712	0.14	0.05

SE = standard error.

Changes in nonnative plant cover suggest significant increases over a 10-year period (fig. 8.5). However, some of the apparent increases may not be real, because rates of plant identification to the species level improved between the first and second measurements. Crews could assign genus or general growth form codes if they were not sure of the correct species, but identification to the species level is needed to determine whether a species is nonnative. For PC states, 83 percent of species cover was coded to the species level at time 1 and 90 percent at time 2. However, in the case of graminoids in California, the change was 20 to 48 percent. Species identification rates were better for the RM states, but still increased from 87 to 96 percent.

Complete vascular plant sampling found that 60 percent or more of the forested plots had at least one nonnative plant on them.

The full vascular plant sample from the P3 protocol indicated that at least 60 percent of forested plots in Utah, Oregon, and Washington had at least one nonnative present, compared to 10 to 20 percent from the P2 protocol (fig. 8.6). The P3 estimates of forested area covered by nonnatives were more than twice the estimates from P2, primarily because of the cover of forbs and graminoids. Fewer differences were seen in relative cover; both the nonnative and total cover estimates differ among protocols. The more complete sample of species in P3 could be part of the reason for greater levels of cover, but species identification rates could be a factor as well; for P3, 95.6 percent of the cover was identified to species (the proportion of those identified to genus was 0.70), while the rate for P2 in the same states was 93.3 percent. Identification rates were lowest for graminoids (87.7 vs. 83.5 percent of cover identified to species for P3 and P2, respectively). Even for P3, identification rates were quite low for forbs in Utah as

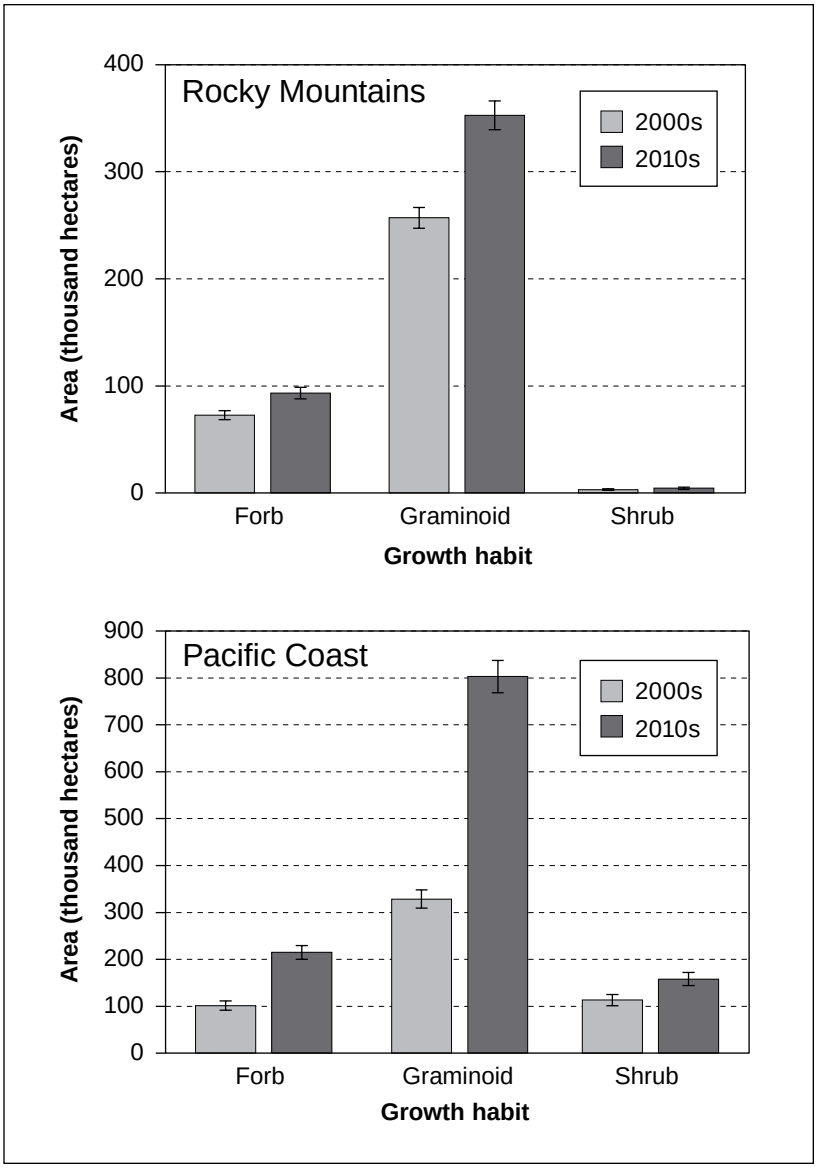

Figure 8.5—Mean change (with standard error) in estimated area covered by nonnative plants between the 2000s and 2010s, by growth habit and region. Some of the increase may be due to better plant identification at the second measurement, particularly for graminoids in the Pacific Coast states.

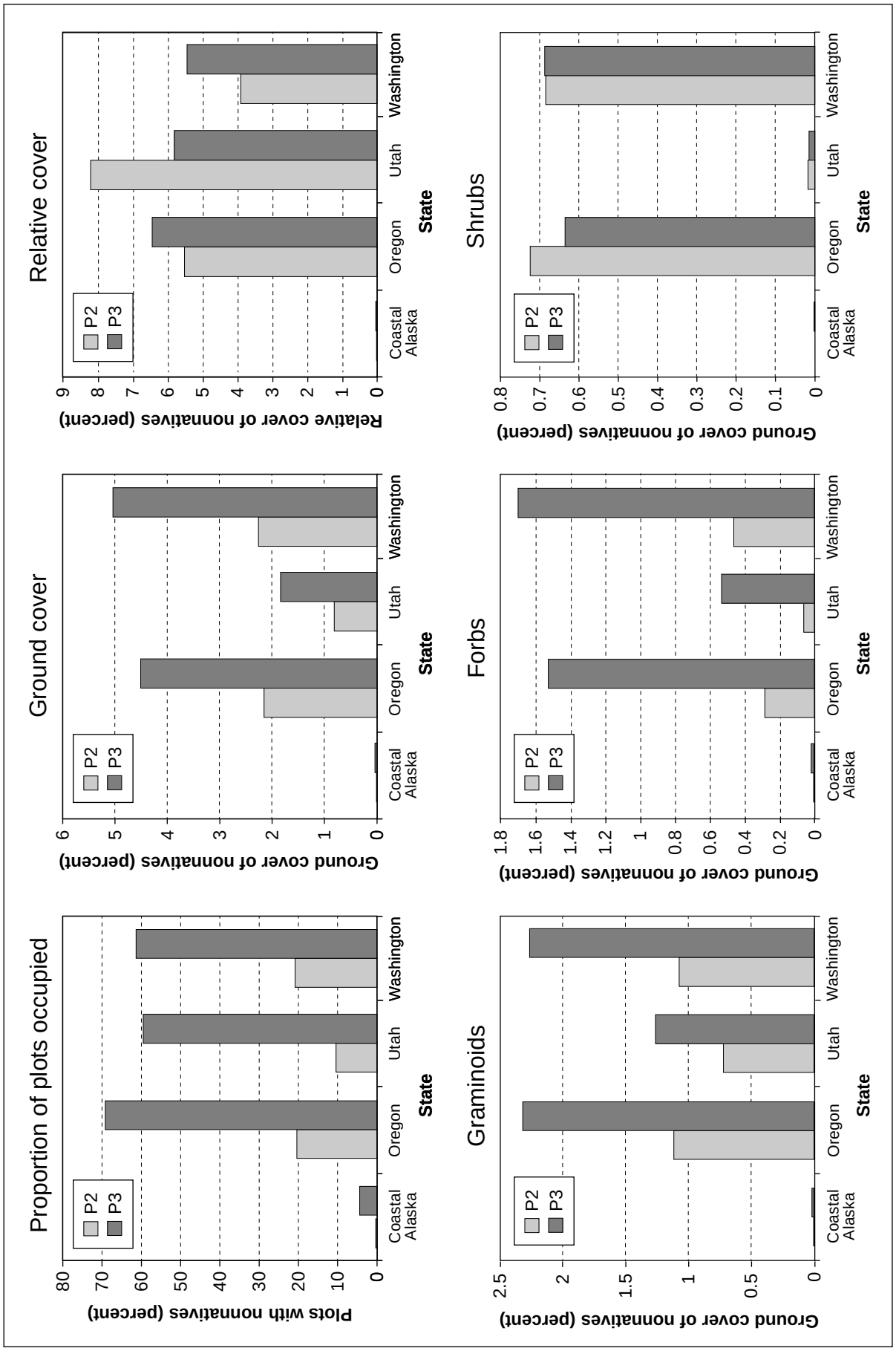

Figure 8.6—Comparison of measures of nonnative plant abundance by Forest Inventory and Analysis protocol type (P2 or P3), illustrating the proportion of plots occupied by at least one nonnative, and mean cover and relative cover (top row), and cover by growth habit (bottom row), by state.

well (76.4 percent); in dry open forest types, many plants are only readily observable and identifiable during short periods.[4] The nonnative species with the most cover from the P3 protocol (table 8.10) were similar to those from the P2 protocol, except that nonnative grasses tended to be ranked with greater abundance in the P3 measurement, and a few species that tended to occur as scattered individuals at low cover were ranked abundant in P3 but were rarely recorded in P2 (e.g., yellow salsify in Utah and wall-lettuce in Washington).

Key Findings

The area of forest covered by invasive plants ranged from 46 percent in Hawaii to 3.3 percent on the Pacific Coast, to 0.75 percent in the Rocky Mountains, to 0.01 percent in Coastal Alaska (table 8.1). Except for Hawaii, the impact of invasive plants is substantially lower than has been found in eastern forests (Oswalt et al. 2015, Schulz and Gray 2013). The best data available from a subset of the region suggest that the area covered by invasives on the Pacific Coast and Rocky Mountains could be twice those values. Nonnative cover appears to have increased substantially from the 2000s to the 2010s.

Most of the invasive species in western forests are early-seral and associated with stands that are nonstocked, young, or fragmented. Roads, in particular, fragment forest land and create suitable sites for invasive plant establishment, thereby providing vectors for invasion into forest land (e.g., Kerns and Day 2017, Lovtang and Riegel 2012). Streams and rivers may also serve as conduits for the dispersal of seeds and plant fragments of invasive plants into resource-rich riparian habitats (Parendes and Jones 2000).

Annual grasses are particularly abundant in dry western forests and woodlands. Because these grasses dry out earlier and more completely than native perennial vegetation, they often increase the spread, severity, and frequency of fire when it occurs, thereby perpetuating conditions ideal for annual grasses (e.g., Balch et al. 2013, D'Antonio and Vitousek 1992).

Invasive plants tolerant of full or partial shade are found in moist western forests and can invade intact stands or those that are partially cut (e.g., Gray 2005). Forests of the Northeastern United States, which have been exposed to nonnative plants longer than Western States, tend to be invaded by many shade-tolerant species. A similar fate may await western forests.

[4] **O'Brien, R. 2004.** Personal communication. Ecologist (retired), U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, 507 25th Street No. 301, Ogden, UT 84401.

Table 8.10—The 10 nonnative plant species with the greatest mean cover in the P3 sample by state, 2001–2010

Scientific name	Common name	Cover
		Percent
Alaska:		
Taraxacum officinale	Common dandelion	0.02
Schedonorus phoenix[a]	Tall fescue	0.01
Phalaris arundinacea	Reed canarygrass	0.01
Vicia cracca ssp. *cracca*	Bird vetch	0
Erysimum cheiranthoides	Wormseed wallflower	0
Oregon:		
Bromus tectorum	Cheatgrass	0.99
Geranium lucidum[a]	Shining geranium	0.57
Rubus armeniacus	Himalayan blackberry	0.46
Holcus lanatus	Common velvetgrass	0.25
Poa bulbosa	Bulbous bluegrass	0.22
Dactylis glomerata	Orchardgrass	0.14
Cytisus scoparius	Scotch broom	0.13
Phleum pratense	Timothy	0.10
Arrhenatherum elatius	Tall oatgrass	0.10
Digitalis purpurea	Purple foxglove	0.08
Utah:		
Bromus tectorum	Cheatgrass	1.06
Lepidium latifolium[a]	Broadleaved pepperweed	0.10
Tragopogon dubius	Yellow salsify	0.07
Agropyron cristatum	Crested wheatgrass	0.06
Galinsoga parviflora[a]	Gallant soldier	0.06
Cirsium arvense	Canada thistle	0.05
Bromus rubens	Red brome	0.04
Tamarix chinensis[a]	Five-stamen tamarisk	0.03
Poa bulbosa	Bulbous bluegrass	0.03
Sisymbrium altissimum	Tall tumblemustard	0.02
Washington:		
Holcus lanatus	Common velvetgrass	0.88
Bromus tectorum	Cheatgrass	0.60
Mycelis muralis	Wall-lettuce	0.26
Hedera helix	English ivy	0.25
Cirsium arvense	Canada thistle	0.24
Rubus armeniacus	Himalayan blackberry	0.21
Hypochaeris radicata	Hairy cat's ear	0.21
Bromus racemosus[a]	Bald brome	0.21
Cirsium vulgare	Bull thistle	0.18
Rubus laciniatus	Cutleaf blackberry	0.17

[a] These species are based on large cover measurements on only one plot, so statewide abundance is less clear.

Discussion

The area of forest covered by nonnative plants is outside the range of natural variation almost by definition (Montréal Process criterion 3.15). Quantifying how far outside the natural range of variation invasive plants have moved forests, and which ecosystem functions are affected, is more challenging. The outcome of invasion by nonnative plants is highly dependent upon both the identity of the invader and the biotic and abiotic environmental context of the invaded site (Pyšek et al. 2012). Presumably the space taken up by invasive plants results in less space occupied by native species. Invasive plants can also alter the physical properties of ecosystems. For example, tamarisk trees might promote sediment deposition, resulting in narrower river channels and less frequent but more severe floods (Graf 1978). Invasive grasses that increase fire incidence or intensity can cause conversion of forest to savannah or grassland (D'Antonio and Vitousek 1992). This has been found in Hawaii, but the extent to which it occurs in the continental Western United States is not clear (although cheatgrass conversion of sagebrush ecosystems to annual grasslands is well documented). Invasive plants can change nutrient cycling in a variety of ways. In the case of the nitrogen-fixer Scotch broom, soil nutrients increase at some sites (Slesak et al. 2016), and its presence can transform the species composition of the plant community (Shaben and Myers 2009). Invasive plants can alter the pollination of native plants, affecting reproductive success and the composition of insect communities (Bartomeus et al. 2008). Invasive plant species can also affect the success of herbivores, seed predators, soil organisms, and higher trophic levels (e.g., Knapp 1996, Litt and Pearson 2013).

Once an invasive plant like cheatgrass becomes widespread, managers by necessity reprioritize their efforts from eradication to long-term management.

Although it is clear that invasive plants can affect ecosystems and communities they invade in many ways, this is not necessarily true for all introduced species, or even all invasive plants. Many are not expected to have any appreciable long-term impacts upon invaded sites. Even where invasive plants have been reliably shown to have significant impacts on native ecosystems and plant and animal communities, there is a lack of convincing evidence for extinctions attributable to invasives (Gurevitch and Padilla 2004). Once an invasive plant like cheatgrass becomes widespread, managers by necessity reprioritize their efforts from eradication to long-term management that attempts to reduce negative effects and take advantage of the positive ones (Kitchen 2014). On the other hand, preventative measures could be effective at maintaining the currently low abundance of nonnative species in boreal forests (Sanderson et al. 2012).

Much of the literature on invasive plants focuses on impacts in nonforested environments like rangeland, grasslands, or wetlands. The presence of nonnative plants in a forest dominated by native trees is less obvious, and the impacts perhaps more subtle, than occurs in nonforest environments, although examples like

kudzu in the Southeastern United States, English ivy in the Northwest, or strawberry guava in Hawaii are certainly available. The greater abundance of nonnative plants in forests of the Eastern United States could be a result of greater disturbance (including recovery from widespread agriculture in the 1900s), higher human population density, or longer exposure to introduced species. Given the propensity of many invasive plants to occupy disturbed sites, their impact on tree regeneration through competition or other effects (e.g., promoting fire) may be the most significant from an economic perspective (e.g., The Research Group 2014). Most state forestry regulations in the West require prompt tree seedling establishment after regeneration harvesting. In many forest types, site preparation in the form of prescribed burning or herbicide application (only on nonfederal lands for the latter) has been traditionally used after harvest. It is not clear whether the presence of nonnative plants has required greater use and expense of site preparation than would be undertaken to control native plants, or whether some plantations have failed because of the dominance of nonnative plants in western forests. Nonetheless, abundant evidence exists for negative impacts of invasive plants on tree recruitment, via either direct competition or their ability to promote higher rates of seed predation (e.g., Gorchov and Trisel 2003, Meiners 2007, Stinson 2006).

Mechanical and chemical control are usually employed in initial attempts to control or eradicate early and localized infestations. Biological control by introducing the predators of invasive plants from their regions of origin (e.g., nonnative insects) is an attractive option for widespread plant invasions because it can be relatively cheap and effective, as in the case of St. John's wort in western rangelands (Crompton et al. 1988). Despite years of research and rigorous testing in controlled environments, unintended consequences from introduction of biological control agents can occur, however. For example, the introduction of gall flies to control spotted knapweed increased deer mouse populations, which could result in increased exposure of humans to hantavirus (Pearson and Callaway 2008). A significant challenge with targeted control measures is that in many cases, secondary invaders respond to the newly available space better than the natives (Pearson et al. 2016).

In this chapter, we summarized the major impacts of invasive plants on indicators of sustainable forestry and quantified the abundance of those plants in western forests. Identified research priorities for sustainable forestry include (Chornesky et al. 2005):

- More taxonomists in the field, because accurate species identification is a key challenge in detection and monitoring

- Integration of invasive plant control considerations into all aspects of management, from transportation and travel corridors to silviculture, because the connections between disturbances, activities, and land types can either accelerate or retard invasions
- Landscape consideration and planning, with models that integrate different types of activities with the probabilities of invasion.

Better quantification of invasive plant impacts on sustainable forests could involve linking site-specific research on the impacts of particular species to broad-scale estimates of species' abundance in different habitat types.

The impacts of nonnative plants on forest lands will most likely increase in the future, as suggested by our change analysis. Although there are active efforts to prevent arrival of new invaders, increasing global trade makes it likely that new introductions will continue to occur (Moser et al. 2009). Many species do not appear to occupy all habitat that is currently available, or may become available with climate change (e.g., Gray 2005, Kerns et al. 2009). Given the importance of ruderal and graminoid invaders in western forest lands, every disturbance creates an opportunity for invasive plants to become established, become part of the local seed bank, and increase the propagule pressure on the landscape. Species that become abundant can have sufficient propagule pressure to colonize and affect otherwise marginal habitats (e.g., cheatgrass in closed forest). The increase of shade-tolerant invasive plants in our moister western forests will probably continue as well, as there appear to be large areas of climatically suitable land available (e.g., Gray 2005). Many of the shade-tolerant nonnative species that have become abundant in temperate forests of the Eastern United States are also being found in the West (e.g., garlic mustard) and are likely to become important here as well. The expected increase in the extent of plant invasions in Western U.S. forests suggests the utility of successful monitoring strategies in these areas.

Monitoring of Invasive Plants in Western U.S. Forests

Monitoring approaches can be split into early detection, tactical, and strategic categories. Early detection and rapid response is often viewed as the most effective means of controlling an invasion and minimizing its impacts (Sharov and Liebhold 1998, Simberloff 2003). Nevertheless, early detection of burgeoning invasions is uncommon owing to challenges inherent in detecting a relatively rare invasive within a native-dominated community (Hulme 2006). Early detection is often serendipitous but involves having people with the proper taxonomic skills observing likely locations for invasion (Reichard and Hamilton 1997).

Tactical monitoring may also be serendipitous but often involves identifying particular species in specific habitats of concern. Standardized protocols exist for documenting the location and extent of infestation (e.g., North American Weed Management Association), often with the objective of determining change over time or the response to eradication and control efforts. Although vital to land management, the targeted nature of tactical monitoring, tied to the landowner's particular objectives, makes comprehensive assessment of invasive plant impacts difficult.

Strategic monitoring is based on a statistical design that samples a defined area with consistent procedures such that the probabilities of detection are known (Bechtold and Patterson 2005). The FIA currently covers all the forests in the nation with a common set of procedures and a consistent remeasurement cycle. Yet the standard grid density and vegetation plot size samples only 1/36,000th of the landscape, making FIA ineffective for early detection and tactical monitoring. The Western States implement a P2 understory vegetation protocol that quantifies cover of the four most abundant vascular plant species per lifeform present at >3 percent cover on each of four subplots, and most states across the country record any observation on the plot (regardless of cover) from a targeted list of invasive plants. As seen in our comparison of P2 and P3 in the states where both are available, the P2 measurement underestimates the prevalence and abundance of nonnative plants. The P2 focus on dominant species, use of personnel with moderate botanical expertise, and long field seasons and varied phenological stages over which sampling occurs probably all contribute to the underrepresentation. However, for species that are readily identified and abundant, the large number of P2 plots (16 times the standard density of P3) make the P2 data useful for modeling abiotic and biotic attributes related to their distribution (Baer and Gray, n.d., Lovtang and Riegel 2012). Training to identify a targeted set of invasive list species, and their measurement on P2 plots, is useful even though the scope is often limited to the regions or states for which they are listed. The P3 all-species measurement with experienced botanists during the peak phenological period on a subset of plots can be logistically challenging to integrate in an annual inventory, but the data provide the best assessment of nonnative species composition and make it possible to assess both native and invasive species richness and a more refined view of community dynamics over time.

Table 8.11—Scientific and common of plants identified in this chapter

Scientific name	Common name
Achillea millefolium L.	Common yarrow
Agropyron cristatum (L.) Gaertn.	Crested wheatgrass
Alliaria petiolate (M. Bieb.) Cavara & Grande	Garlic mustard
Ardisia elliptica Thunb.	Shoebutton
Arrhenatherum elatius (L.) P. Beauv. ex J. Presl & C. Presl	Tall oatgrass
Avena fatua L.	Wild oat
Brachypodium distachyon (L.) P. Beauv.	Purple false brome
Bromus arvensis L.	Field brome
Bromus diandrus Roth	Ripgut brome
Bromus hordeaceus L.	Soft brome
Bromus racemosus L.	Bald brome
Bromus rubens L.	Red brome
Bromus tectorum L.	Cheatgrass
Centaurea stoebe L. ssp. *micranthos* (Gugler) Hayek	Spotted knapweed
Cirsium arvense (L.) Scop.	Canada thistle
Cirsium vulgare (Savi) Ten.	Bull thistle
Clidemia hirta (L.) D. Don	Soapbush
Cynoglossum officinale L.	Gypsyflower
Cynosurus echinatus L.	Bristly dogstail grass
Cytisus scoparius (L.) Link	Scotch broom
Dactylis glomerata L.	Orchardgrass
Digitalis purpurea L.	Purple foxglove
Dodonaea viscosa (L.) Jacq.	Florida hobbush
Elymus repens (L.) Gould	Quackgrass
Eragrostis lehmanniana Nees	Lehmann lovegrass
Erodium botrys (Cav.) Bertol.	Longbeak stork's bill
Erodium cicutarium (L.) L'Hér. ex Aiton	Redstem stork's bill
Erysimum cheiranthoides L.	Wormseed wallflower
Galinsoga parviflora Cav.	Gallant soldier
Geranium lucidum L.	Shining geranium
Hedera helix L.	English ivy
Holcus lanatus L.	Common velvetgrass
Hypericum perforatum L.	Common St. John's wort

Table 8.11—Scientific and common of plants identified in this chapter (continued)

Scientific name	Common name
Hypochaeris radicata L.	Hairy cat's ear
Lepidium latifolium L.	Broadleaved pepperweed
Matricaria discoidea DC.	Disc mayweed
Melastoma candidum D. Don	Asian melastome
Melilotus officinalis (L.) Lam.	Yellow sweetclover
Microlaena stipoides (Labill.) R. Br.	Weeping grass
Mycelis muralis (L.) Dumort.	Wall-lettuce
Pennisetum clandestinum Hochst. ex Chiov.	Kikuyugrass
Phalaris arundinacea L.	Reed canarygrass
Phleum pratense L.	Timothy
Plantago major L.	Common plantain
Poa bulbosa L.	Bulbous bluegrass
Psidium cattleianum Sabine	Strawberry guava
Psidium guajava L.	Guava
Pueraria montana (Lour.) Merr.	Kudzu
Rubus armeniacus Focke	Himalayan blackberry
Rubus laciniatus Willd.	Cutleaf blackberry
Rumex crispus L.	Curly dock
Schedonorus phoenix (Schreb.) Dumort., nom. cons.	Tall fescue
Schinus terebinthifolius Raddi	Brazilian peppertree
Sisymbrium altissimum L.	Tall tumblemustard
Taeniatherum caput-medusae (L.) Nevski	Medusahead
Tamarix chinensis Lour.	Five-stamen tamarisk
Tamarix ramosissima Ledeb.	Saltcedar
Taraxacum officinale F.H. Wigg.	Common dandelion
Thinopyrum intermedium (Host) Barkworth & D.R. Dewey	Intermediate wheatgrass
Torilis arvensis (Huds.) Link	Spreading hedgeparsley
Tragopogon dubius Scop.	Yellow salsify
Trifolium spp.	Clover
Urochloa maxima (Jacq.) R. Webster	Guinea grass
Ventenata dubia (Leers) Coss.	North Africa grass
Vicia cracca L. ssp. *cracca*	Bird vetch

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Chapter 9: Economic Value of Ecosystem Service Losses Resulting From Disturbances in Western U.S. Forests

José J. Sánchez, Lorie Srivastava, Raymundo Marcos-Martinez, and Dominique Bachelet[1]

Introduction

Forest disturbances are events that disrupt the structure and composition of forest ecosystems as well as their protective, productive, and social functions (Führer 2000, Holmes et al. 2008). The increasing severity and frequency of environmental and anthropogenic forest disturbances challenge the sustainable management and protection of forests in the United States and the benefits that they provide to individuals and communities. About 1.42 percent (3.6 million ha) of U.S. forest land was disturbed annually between 1985 and 2015 (Masek et al. 2013). Tree mortality stemming from disturbance regimes has been increasing since the 1980s (Williams et al. 2016), resulting in forest cover loss at rates comparable to global deforestation hotspots (Hansen et al. 2013, 2010). The effect of climate change on disturbance agents is predicted to amplify the frequency, intensity, and damage from forest disturbance regimes (Seidl et al. 2017).

The increasing severity and frequency of forest disturbances challenge the sustainable management and protection of forests and the benefits that they provide.

In the Western United States, wildfires, droughts, timber harvesting, and bark beetle outbreaks are the main drivers of tree mortality (Masek et al. 2013, Woodall et al. 2015). Such events can affect the composition and function of forests and their provision of use and nonuse ecosystem services such as climate change mitigation, biodiversity conservation, aesthetic values, and timber production (Ferraro et al. 2015, Sánchez et al. 2016). Additionally, the interplay between disturbance regimes and the slow process of forest regrowth and expansion could result in long-lasting social, economic, and environmental effects (Dale et al. 2000, Marcos-Martinez et al. 2019). Information on the economic consequences of forest disturbances could potentially help forest managers in identifying planning, resourcing, and budgeting priorities for their managed lands.

The Millennium Ecosystem Assessment provides a framework to classify the benefits that people obtain from ecosystems into four groupings: (1) provisioning

[1] **José J. Sánchez** is a research economist, U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, 4955 Canyon Crest Drive, Riverside, CA 92507; **Raymundo Marcos-Martinez** is a research scientist, Commonwealth Scientific and Industrial Research Organisation, Black Mountain, ACT 2601, Australia; **Lorie Srivastava** is a research associate, University of California–Davis, Department of Environmental Science and Policy, One Shields Avenue, Davis, CA 95616; **Dominique Bachelet** is an associate professor, Oregon State University, Department of Biological and Ecological Engineering, 116 Gilmore Hall, Corvallis, OR 97331.

services such as food, water, timber, and fiber; (2) regulating services that affect climate, floods, disease, wastes, and water quality; (3) cultural services that provide recreational, aesthetic, and spiritual benefits; and (4) supporting services such as soil formation, photosynthesis, and nutrient cycling (Millennium Ecosystem Assessment 2005). This framework facilitates the identification of how changes in forest ecosystems affect human well-being and can help to identify tradeoffs and benefits from protecting specific services (e.g., timber extraction versus biodiversity conservation).

Each disturbance event will have a unique impact on forest conditions and related ecosystem services.

Each disturbance event will have a unique impact on forest conditions and related ecosystem services, depending upon a range of factors. In cases in which the damage is extreme, extensive, or marks a substantial departure from previous disturbance patterns, the impact on ecosystem services can be severe, spanning a broad range of valued services and outputs. Some of these impacts can be readily measured in quantitative terms, but others, particularly those associated with less tangible values such as aesthetics or existence values,[2] are much more difficult to measure or predict. Also, though we tend to view the effects resulting from forest disturbances as predominantly negative, this is not always the case. Many disturbance events have both positive and negative effects depending upon the particular ecosystem service being considered. As a result, a full and quantitative accounting of the social impact of a major disturbance event or process is extremely difficult, if not impossible.

It is possible, however, to provide a description of the general types of impacts arising from different disturbances as well as examples of economic analyses applied to specific ecosystem services—this is the strategy followed here. Accordingly, this chapter provides monetized estimates of forest disturbances (fire, drought, or disease) for two critical areas of value: recreation and carbon storage.[3] In addition, we provide an overview of potential effects over a broader range of ecosystem services such as biodiversity, air pollution mitigation, timber value, and aesthetic benefits. This information may improve stakeholders' understanding of how forests disturbances may affect economic and environmental processes at different temporal and spatial scales. The results could also inform the design of spatially targeted forest conservation, pest control, or fire suppression strategies to minimize adverse impacts and generate greater benefits to society.

[2] Existence value is defined to be the value people place on a natural or environmental good without using it; such value often reflects a sense of well-being from knowing the good exists.

[3] Hereafter, we used the term carbon emissions to indicate the release of carbon dioxide (CO_2) from forest biomass loss and carbon sequestration to refer to the storage of atmospheric CO_2 in forest biomass. The terms carbon pools and carbon stocks are used interchangeably to indicate carbon stored within forest ecosystems.

Approaches to Valuing Environmental Goods and Services

Natural resources and environmental amenities provide benefits to individuals and communities ranging from marketable products to recreational use, biodiversity conservation, and aesthetic values. Some of these amenities are public environmental goods for which there are no markets. Therefore, these goods and services are not traded, and demand is not directly observable from price-quantity variations. Nevertheless, environmental and natural resource economists have developed several valuation methods to help estimate the value of nonmarket goods and services. It is useful to describe these valuation methods briefly because valuing the change in human welfare is a key consideration in determining the implications for forest disturbances with respect to social and economic realms.

These approaches stem from the concept of total economic value, which may be considered the economic foundation of decisions made concerning environmental and natural resources (Plottu and Plottu 2007). A generally accepted typology is given in figure 9.1, which distinguishes between use and nonuse values. Use values refer to benefits that people derive from the direct use of a good or service. Non-use values reflect the value that people attribute to goods and services that they do not or may never use. In this chapter, the focus is on both direct use and indirect non-use values. Nonmarket goods can embody use or non-use values.

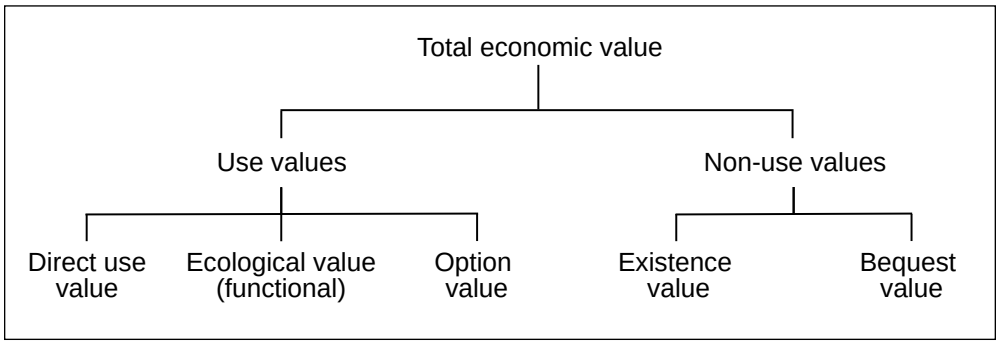

Figure 9.1—Total economic value typology. Source: Plottu and Plottu (2007).

There are three approaches to valuing nonmarket goods and services: revealed preference, stated preference, and benefit transfer methods (Champ et al. 2017). Revealed preference methods are used to estimate nonmarket values from observations of behavior in markets for related goods and services. The travel cost and hedonic pricing methods are two of the most commonly used revealed preference methods. The **travel cost method** uses individual trip costs to reach a site as a

proxy for the value of the recreation site. Trip costs include direct costs (e.g., equipment, food, fuel) incurred by the individual to visit the resource being valued along with the opportunity cost of time, which is the cost associated with the time of traveling to and from the visited area (Hagerty and Moeltner 2005). Recreationists who have higher trip costs would be expected to take fewer trips to the recreation site than individuals with lower travel costs. This information, along with travel costs and visitation rates, helps us derive the travel demand curve for the recreation site of interest. Individual willingness-to-pay (WTP) to access the recreation site can be obtained directly using this method, allowing specification of the demand function, and computation of consumer surplus (the amount of value received over and above the costs incurred). See Parson (2017) for more detailed information on the travel cost model.

The **hedonic pricing method** relies on market transactions to indirectly value environmental characteristics by comparing two closely related goods, with one having extra environmental attributes. The method has been applied to different markets such as houses, automobiles, and computers. The hedonic method is commonly used in housing markets to value nonmarket goods and services or disservices. For instance, hedonic models allow the estimation of price premiums that individuals are willing to pay to purchase a home near an environmental amenity, or the price discount received for locating near a disamenity (e.g., Mueller et al. 2009, Price et al. 2010). For more information on the hedonic method, see Taylor (2017).

In contrast to revealed preference methods, stated preference methods derive nonmarket values from hypothetical questions. The **contingent valuation method** (CVM) is a survey-based method that derives monetary values (WTP) for goods and services not bought and sold in actual markets. CVM typically involves describing precise changes in environmental amenities through various hypothetical scenarios. The respondents are presented with information about those scenarios and answer questions about how much they would be willing to pay for the improved goods or services. Values for environmental services can be calculated by using the answers to these questions.

Although CVM is now widely accepted by the economic discipline, this method has been criticized in the past for valuating passive use and existence values. Partly for this reason, the National Oceanic and Atmospheric Administration (NOAA) created a “blue-ribbon” panel, which concluded that CVM studies, if done correctly, “can produce estimates reliable enough to be the starting point of a judicial process of damage assessment, including lost passive-use values” (Arrow et al. 1993). The NOAA panel provided recommendations as to how CVM studies should be conducted appropriately. Other studies have provided further details in conducting reliable surveys to achieve better welfare estimates (Carson 2000, Carson et al. 2001, Hanemann 1994). See Carson (2000) for a guide on CVM.

The **benefit transfer** method relies on preexisting analyses at one or more sites or policy contexts to predict welfare estimates or related information (e.g., WTP) for other, typically unstudied sites or policy contexts (Johnston et al. 2015). Two types of benefit transfers approaches are common: unit value transfers and benefit function transfer. The **unit value transfer** uses a single number or set of numbers from previous primary studies to transfer to the new study area. For **benefit function transfers**, a single pre-estimated function from a single primary study—called a single-site benefit function transfer—can be used, or a set of previously calibrated functions for a meta-analysis—called meta-regression models—can be employed. It is generally accepted that benefit function transfers provide more accurate estimates than unit value transfers (e.g., Kaul et al. 2013). More information can be found in Johnston et al. (2015) and Rosenberger and Loomis (2001).

Case Studies: The Economic Impact of Forest Disturbances on Recreation and Carbon

Case Study 1: Recreation

Outdoor recreation is one of the most recognized ecosystem services provided by national forests and other public lands. About 49 percent of the U.S. population participated in an outdoor activity at least once in 2017; this figure has remained relatively stable for the past decade (Outdoor Industry Association 2018). The most popular activity is running (including jogging and trail running), followed by fishing, bicycling, hiking, and camping (Outdoor Industry Association 2018). Higher visitation and engagement in outdoor recreation activities are expected as population growth is projected to be the primary driver of growth for outdoor recreation (Cordell 2012). Overall, participating in outdoor recreation has many benefits to society and individuals, such as income to local communities. For recreationists, there is a potential to improve individual health and well-being.

Outdoor recreation is an important economic driver for many rural communities in the United States. In 2016, more than 889 million visits were made to federal lands, with recreationists spending $49 billion and supporting around 826,000 jobs (Cline and Crowley 2018). The Outdoor Industry Association (2017) reported that American consumers spend more on outdoor recreation than on pharmaceutical products and fuel combined. In this chapter, we focus on the economic value of recreation, i.e., the benefits that recreationists received from enjoying a recreation activity. This differs from the economic impact of recreation, which measures how spending by recreationists affects economies in a geographical area (Rosenberger et al. 2017).

Aside from economic benefits, outdoor recreation contributes to improvements in individual health and well-being. A literature review by Gladwell et al. (2013) found that outdoor natural environments might help reduce stress and mental

fatigue as well as improve mood, self-esteem, and perceived health. Similar results were found by Thomsen et al. (2018) when reviewing literature that focused solely on the relationship between wildland recreation and physical and mental health found similar results. There is even a push for doctors to start prescribing hiking doses to improve mental and physical health (ParkRx 2018). The Park Prescription Program started in 2013 and has increased significantly throughout the nation.[4]

Forest disturbances such as drought, wildfire, insects (e.g., bark beetles) and disease alter forest health and landscape characteristics, often in dramatic ways. Significant changes to forest ecosystems will affect the benefits that individuals and communities receive from engaging in outdoor recreation activities, e.g., through reduced participation rates, number of days available for outdoor recreation, and changes in the quality of outdoor amenities. Drought conditions are expected to continue to affect most of the West and may alter forest recreational services. Severe drought conditions tend to affect water/snow-based forest recreation activities such as fishing, boating, rafting, kayaking, snowmobiling, and skiing (Hand and Lawson 2018). Snow-based activities are affected the most as diminished quantity and quality of snow may result in fewer ski days.

Severe drought conditions tend to affect water and snow-based forest recreation activities.

Additionally, shorter seasons and fewer sites with an adequate snow level may result in crowding issues and a decline in participation rates. White et al. (2016) found that under climate change, the greatest decreases in future projected total participation rates for outdoor recreation activities will be in snowmobiling and undeveloped skiing (e.g., cross-country skiing and snowshoeing). Conversely, the shorter winter season will affect summer recreation activities as sites will open sooner and close later in the season. Hewer et al. (2019) found that warmer temperatures increase park visitation during the fall season. Similarly, other studies have found that warm weather days expand visitation season for national parks (Albano et al. 2013, Fisichelli et al. 2015).

For water-based recreation activities, future total participation projections will decrease for canoeing, kayaking, and rafting (White et al. 2016). Hand and Lawson (2018) suggested that recreationists might be able to adapt by selecting different sites at higher elevations or changing their recreation activity such as from motorized to nonmotorized boating. Nevertheless, if streamflow is severely affected, this can have a negative effect on recreationists' participation behavior. For example, Loomis and Crespi (1999) found that streamflow is positively associated with the number of days spent on rafting, kayaking, and canoeing.

[4] More information on Park Prescription programs are available at the ParkRx website (http://www.parkrx.org).

A large body of literature has explored the effect of wildfire on forest recreation demand. For example, Duffield et al. (2013) estimated that more than 35,000 annual visits to Yellowstone National Park were lost because of wildfires. But wildfires appear to affect recreation values differently, depending on wildfire type, time since the disturbance, and type of recreation activity. Early studies found that intense fires are likely to negatively affect recreation (Flowers et al. 1985, Vaux et al. 1984). For mountain biking, several studies (Hesseln et al. 2003, 2004; Loomis et al. 2001) found that recreation benefits decrease in value as a result of crown fires (i.e., fires that spreads along treetops). Sánchez et al. (2016) surveyed backcountry visitors of the San Jacinto Wilderness in southern California. They found that fires causing trail closures resulted in a total recreation loss of $3.7 million per hiking season. On the other hand, hiking visitation rates actually increased and provided an additional annual benefit of $1.2 million for recent low-intensity fires. Similarly, Starbuck et al. (2006) suggested that low-intensity fire slightly increases visitation. Rausch et al. (2010) found that recent fires initially decreased annual trips for camping activity, and that visits will recover once stands return to "old-growth" levels.

Wildfires appear to affect recreation values differently depending on wildfire type, time since the disturbance, and the type of recreation activity.

Other studies have analyzed the effect of fire on recreation over time. For example, Hilger and Englin (2009) studied two hiking trails in the Cascade Mountains that were affected by a large-scale forest fire that burned more than 16 200 ha. In the short term, the number of visits increased, and trip values were the same as before the fire. Englin et al. (2001) examined how recreation value changed in the long term following forest fires in three different states: Colorado, Idaho, and Wyoming. They found that visitation increased for areas that experienced a fire recently, then decreased for the next 17 years, finally followed by a rebound in visits after 8 more years. A similar study by Boxall and Englin (2008) developed the intertemporal time path for recreational canoeing and found that the decrease in value of benefits from recreation occurred during the early years following a fire, but after 35 years of regrowth, the amenity values provided by the forests returned to their prefire levels.

Large biological invasions such as nonnative forest insects and disease threaten the benefits that individuals and communities receive from forest ecosystems. In the United States, insects and pathogen outbreaks are estimated to affect 20.4 million ha, with an average annual economic cost of $1.5 billion (Dale et al. 2001). Outdoor recreation aesthetics are reduced by outbreak of insects and disease as forest landscapes change. For example, Sheppard and Picard (2006) reviewed published studies on the effect of bark beetle on visual aesthetics and found that reduced visual quality is a concern among recreationists. Similarly, Arnberger et al. (2018) found that recreation sites that are heavily affected by forest insects would be less desirable, so visitors may avoid these affected sites. This reduction in aesthetic values may lead to fewer visits as recreationists will no longer visit or shift to substitute sites.

Data

The Forest Service tracks the recreation activities and number of participants on national forests using the National Visitor Use Monitoring (NVUM) program (English et al. 2002). In this chapter, we use the NVUM dataset to examine the activities individuals engage in during their visit, and the number of annual visitors to each of the national forests in the 11 Western States.[5] We focus on 28 primary recreation activities (see tabulation below) for annual visitation rates, and estimate their economic benefits.

Primary recreation activities:

Backpacking	Nature study
Bicycling	Nonmotorized water
Cross-country skiing	Off-highway vehicle use
Developed camping	Other motorized activity
Downhill skiing	Other activities[6]
Driving for pleasure	Other nonmotorized
Fishing	Picnicking
Gathering forest products	Primitive camping
Hiking/walking	Relaxing
Horseback Riding	Resort use
Hunting	Snowmobiling
Motorized trail activity	Viewing natural features
Motorized water activities	Viewing wildlife
Nature center activities	Visiting historic sites
Nature study	

Methods: Economic Value and Benefits

Rosenberger et al. (2017) estimated recreation economic values by using the benefit transfer method based on the updated recreation use visitor database (Loomis 2005, Rosenberger and Loomis 2001) and annual visitation estimates from the NVUM survey. We used the same procedure to estimate the annual economic benefits to individuals' recreating in the national forests in the western United States. Equation 1 uses the NVUM dataset to determine total annual recreation visits per national forest, and the reported percentage of primary activities that visitors engage in, to obtain the estimated number of visits per activity.

[5] Data downloaded from NVUM website: https://apps.fs.usda.gov/nvum/results/A05001.aspx/FY2016.

[6] In the National Visitor Use Monitoring survey, this category is labelled "Some Other Activity."

Number of visits per activity = NVUM total annual visits × main activity percentage (1)

Next, the number of visits is multiplied by the conversion coefficient and average economic value (equation 2) that was derived by Rosenberger et al. (2017) to estimate the aggregate recreation benefit value. The conversion coefficient translates visits into primary activity days, and the average economic value is based on a metaregression model that estimates the average value for each recreation activity (Rosenberger et al. 2017).

Aggregate recreation benefit value = number of visits per activity × conversion coefficient × economic value (2)

For example, using the NVUM dataset from southern California's Angeles National Forest we multiplied the percentage of main activity (table 9.1, column D) with the annual number of visits (2,879,953) to obtain the number of visits per activity (table 9.1, column E). To estimate the economic benefit of each recreation activity, we multiplied the conversion coefficient (table 9.1, column B) by the average economic value (table 9.1, column C) and the number of visits (table 9.1, column E). The total economic benefit from recreating on the Angeles National Forest is about $231.7 million. The same procedure was undertaken for all other national forests in the Western States. The annual aggregate economic benefit ranges from $11.1 million (Modoc National Forest in California) to $1.03 billion (White River National Forest in Colorado).

Effects of Drought Conditions on Inyo National Forest

Having developed a methodology for estimating total and per-visit recreational values (Rosenberger et al. 2017), we can begin to analyze the effects of anticipated changes in recreation activity associated with different forest disturbance types, such as drought in this specific example. California experienced drought conditions from 2007 to 2009 (CDWR 2010) and from 2011 to 2017 (NOAA NIDIS 2018). Some recreation activities that depend upon water and winter conditions—such as downhill skiing—were significantly affected. These recent droughts have notably affected national forests in the Sierra Nevada.

In the Inyo National Forest, downhill skiing is the most popular outdoor recreation activity. Prior to the 2007–2009 California drought, 2006 NVUM data show more than 1.19 million annual visits for downhill skiing. Data from the following two NVUM samples (2011 and 2016) indicate that the number of annual visits fell by 19 and 37 percent, respectively, despite population increases in California's Inyo County and the United States as a whole during the same period. Applying equations 1 and 2 shows the pre-drought (2006) estimated

Table 9.1—Estimate of annual aggregate economic benefits to individuals recreating on the Angeles National Forest

(A) Activity participation	(B) Conversion coefficient	(C) Average economic use value	(D) Percentage of main activity	(E) Number of visits	(F) Economic benefit
					Dollars
Backpacking	2.8	26.64	0.60	17,191	1,282,330
Bicycling	1.1	80.23	3.92	112,845	9,958,941
Cross-country skiing	1.1	50.01	0.09	2,502	137,647
Developed camping	2.8	29.11	3.15	90,700	7,392,789
Downhill skiing	1.1	75.72	9.35	269,140	22,417,234
Driving for pleasure	1.1	58.49	1.34	38,516	2,478,070
Fishing	1.3	65.01	2.56	73,748	6,232,664
Gathering forest products	1.1	58.49	0.05	1,305	83,959
Hiking/walking	1.1	77.95	55.63	1,602,193	137,380,022
Horseback riding	1.2	58.49	0	—	—
Hunting	1.5	70.9	0.29	8,415	894,913
Motorized trail activity	1.3	43.94	0.44	12,696	725,226
Motorized water activities	1.3	51.87	0.07	1,909	128,712
Nature center activities	1.1	58.49	0.22	6,351	408,601
Nature study	1.1	58.49	0.10	2,989	196,670
No activity reported	1.0	58.49	0.22	6,298	368,370
Nonmotorized water	1.4	102.42	0	—	—
Off-highway vehicle use	1.2	43.94	3.52	101,257	5,339,072
Other motorized activity	1.2	43.94	0.12	3,413	179,969
Other nonmotorized	1.2	58.49	0.71	20,541	1,441,760
Picnicking	1.2	42.67	4.64	133,562	6,838,917
Primitive camping	2.3	26.64	0.28	8,170	500,593
Relaxing	1.5	58.49	2.65	76,177	6,683,428
Resort use	3.2	58.49	0	—	—
Snowmobiling	1.2	43.94	0.06	1,707	89,984
Some other activity	1.1	58.49	6.16	177,440	11,416,342
Viewing natural features	1.2	53.62	4.39	126,435	8,135,361
Viewing wildlife	1.1	53.62	0.45	13,095	772,353
Visiting historic sites	1.1	53.62	0.11	3,155	186,068
Total					231,669,997

Notes: The values in columns B and C were obtained from Rosenberger et al. (2017); those in D were obtained from the National Visitor Use Monitoring survey. Applying formula 1 and 2 to the data in this table results in slight differences because of rounding.

economic benefit for downhill skiing to be $99 million (based on 2016 dollars). Fewer snow days because of drought result in fewer visits, causing a decline in the economic benefit for downhill skiing, calculated to be $80 million in 2011 and $62 million in 2016. The actual economic loss may be larger than these values, as these estimates do not include any other reduction in revenue streams such as loss of special use fees. Similar results can be found on other national forests that are highly dependent on winter conditions and that support numerous water-based recreation activities.

Key Insights

Outdoor recreation provides enormous benefits to both communities and individuals; e.g., it generates billions of dollars each year for local communities across the country (Cline and Crowley 2018). Consequently, the flow and value of benefits can be reduced by forest disturbances such as drought and wildfires. During drought conditions, the length of the season and the number of sites for water/winter-based recreation activities are reduced. The recreation activities most likely to be negatively affected by drought conditions are canoeing, kayaking, rafting, snowmobiling, and cross-country skiing as they depend on adequate levels of water or snow. As a result, recreation enthusiasts may have to adapt by traveling to different sites, tolerating crowded sites, or taking fewer trips per season. Conversely, drought conditions may also benefit those pursuing summer recreation activities as sites may open earlier and close later in the season. For wildfires, the effect depends upon the type of activity, fire intensity and severity, and time of occurrence of the wildfire. For low-intensity fires, the benefits to recreationists appear to increase as there is usually an increase in visitation numbers. In contrast, during catastrophic wildfires, economic benefits decline as recreationalists are not allowed to enter recreation sites for safety reasons. Forest and land managers can potentially use this information to reduce the negative effects of forest disturbances on outdoor recreation and flow of economic benefits.

Case Study 2: Social Costs of Carbon Fluxes From Forest Disturbances

The contribution from forests to national carbon dioxide (CO_2) emission reduction targets is significant. Yet, past natural and anthropogenic disturbances—such as timber harvest, land clearing, insect outbreaks, and fires—on U.S. forest lands have reduced forest carbon stocks to about half their maximum storage potential

(Williams et al. 2016). Ensuing forest regeneration, specifically biomass growth in primary forests, and forest regrowth on previously cleared land, resulted in the sequestration of about 1,600 million metric tons of CO_2 (Mt CO_2) between 1990 and 2016 (Woodall et al. 2015). During this same period, fire and timber harvesting generated reductions in forest carbon stock of about 791 Mt CO_2. On balance, these carbon fluxes resulted in the net sequestration of about 800 Mt of CO_2, or around 30 Mt CO_2 per year on average (Woodall et al. 2015). Forest carbon fluxes offset around 15 percent of domestic carbon emissions from fossil fuel combustion per year (Woodall et al. 2015). Increasing the incidence and severity of disturbance events, however, could compromise forest regeneration and ecosystem services fluxes, including climate change mitigation via carbon sequestration.

Western U.S. forests accounted for almost one-third of the national net forest CO_2 sequestration from 1990 to 2016. California, Idaho, Oregon, and Washington contain about 63 percent of the forest carbon stock within the region (fig. 9.1). In terms of carbon pools, almost two-thirds of this sequestered carbon in Western U.S. forests is stored in aboveground biomass and soils, while the rest is accumulated in forest litter such as fallen leaves and branches (20 percent of the CO_2 stock), dead trees (8 percent), and non-tree understory vegetation (2 percent) (Wilson et al. 2013).

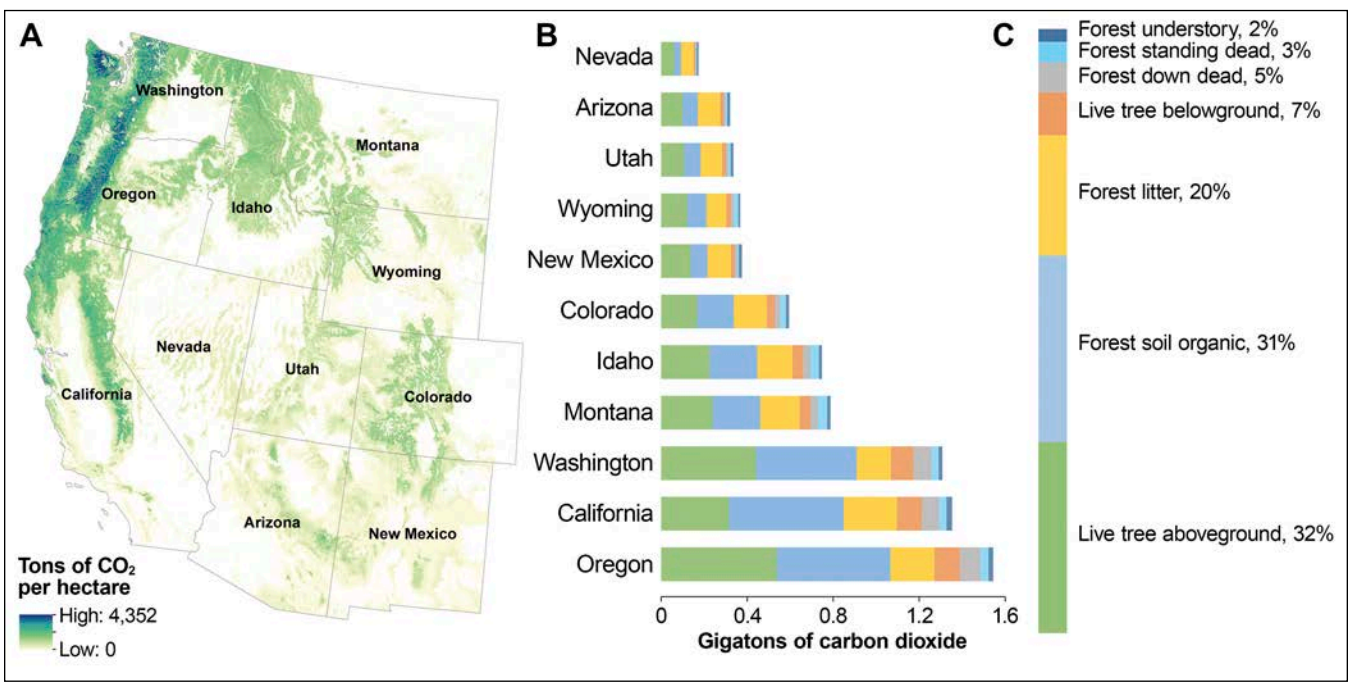

Figure 9.2—Forest carbon dioxide stock on public Western U.S. forest land (based on U.S. Forest Service Forest Inventory and Analysis data collected from 2000 to 2009): (A) Spatial distribution of forest carbon stock; (B) state-level carbon stocks across forest components; (C) proportion of carbon stored across forest components. Generated with data from Wilson et al. (2013).

Forest Disturbances and Carbon Fluxes

Anthropogenic and natural forest disturbances have a large effect on carbon fluxes, and consequently also on climate change mitigation and adaptation strategies (Marcos-Martinez et al. 2019, Sleeter et al. 2018). Concurrently, changes in temperature and precipitation affect the frequency, extent, and intensity of disturbance events that can result in increased rates of tree mortality or forest die-off, and additional releases of forest carbon. The interplay between forest disturbances and climate change can have cascading effects on ecological and socioeconomic systems across temporal and spatial scales (Huo et al. 2019). Additionally, interactions between disturbance events can amplify their effect on forest processes and services. For instance, tree insect outbreaks have the potential to alter the probability, extent, or severity of wildfires, as well as postfire tree regeneration (Hicke et al. 2012). Drought in combination with other physiographic parameters, insect types, and time since insect outbreaks are key determinants of fuel conditions and fire frequency and impact (Andrus et al. 2016, Hicke et al. 2012).

Tree removal for harvest or land use changes, fire, and other tree stressors (e.g., insects, disease, and drought) accounted for 84, 11, and 5 percent, respectively, of the forest cover loss observed in conterminous U.S. land between 2003 and 2011 (Huo et al. 2019). Woodall et al. (2015) reported that the main drivers of U.S. forest carbon stock change between 2006 and 2010 were timber harvest (69 percent), fires (10 percent), land use change (6 percent), wind (7 percent), insect outbreaks (7 percent), and droughts and other natural disturbances (1 percent).

Although timber harvest is the main driver of forest cover and forest carbon stock reductions in the United States, carbon embedded in some wood products could remain stored for decades (Loeffler et al. 2014, Williams et al. 2014). The large range of carbon decay rates across wood product types and final use complicates the estimation of the net carbon flux effect of timber harvesting through the value chain. In addition, selective logging and postharvest reforestation, where it occurs, can set up conditions for gradual recovery of forest carbon stocks. The average residence time for carbon in forest biomass is more than twice that of wood products (Law et al. 2018); once forests are disturbed, their ability to reach preexisting levels of carbon storage and other ecosystem service provisions could be significantly compromised (Marcos-Martinez et al. 2018).

In the Western United States, the rate of forest carbon loss resulting from timber harvest, fire, and bark beetle outbreaks has been estimated at around 1.1 percent per year from 2003 to 2012 (Berner et al. 2017). Timber harvest accounts for about half the forest carbon removal in the region, followed by bark beetles (~32 percent) and fires (~18 percent) (Berner et al. 2017). Owing to the complexities associated with estimating

net carbon fluxes from timber harvest through the value chain, we focus instead on the analysis of carbon emissions resulting from tree mortality from fires and bark beetles. Spatially explicit changes in forest carbon from these two disturbances were quantified using tree mortality data at 1 km resolution for the period 2003–2012 (fig. 9.3) (Berner et al. 2017). In this dataset, tree mortality is quantified as the amount of aboveground forest carbon that is lost to fire and bark beetle disturbances.

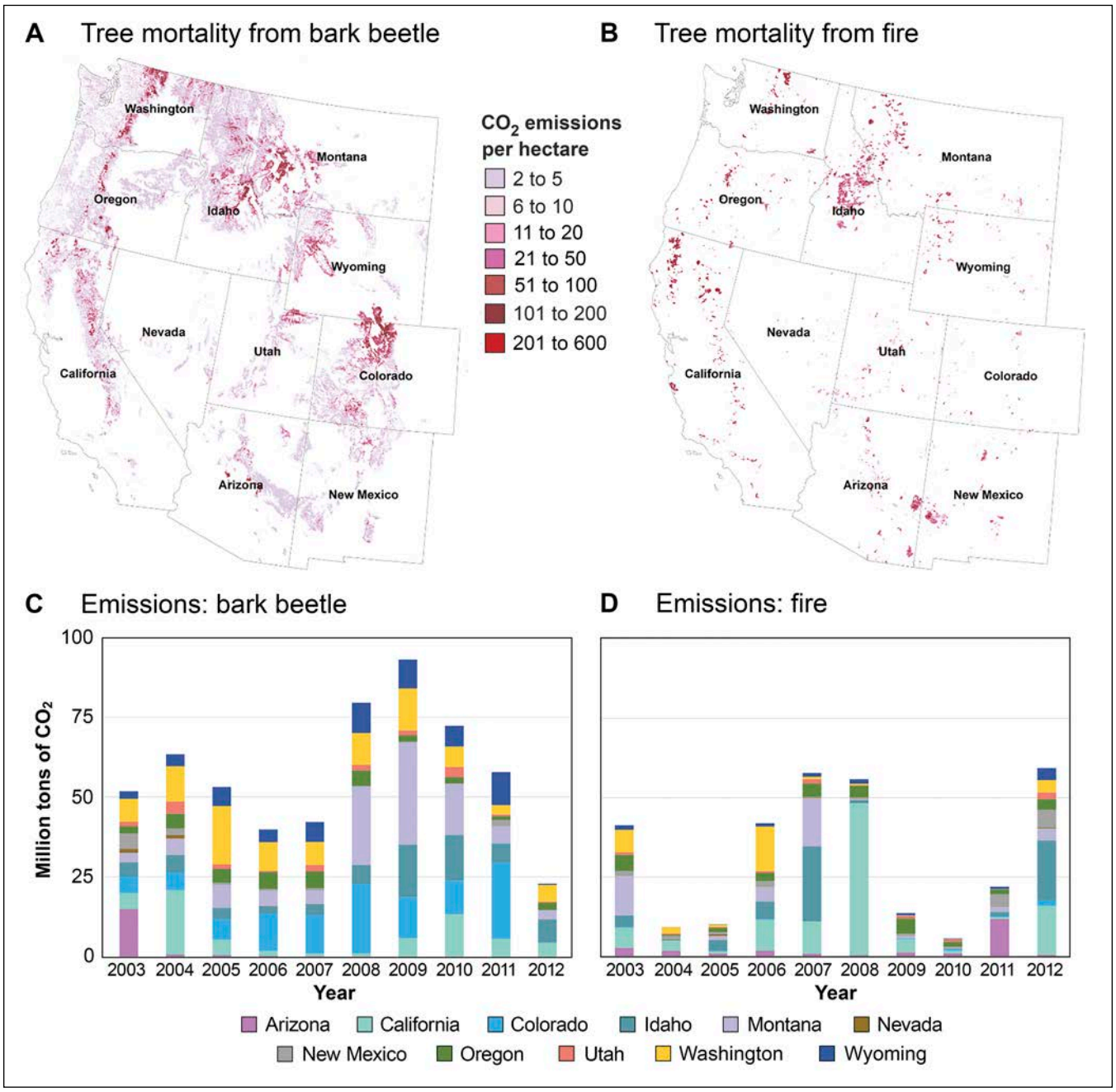

Figure 9.3—Carbon dioxide (CO_2) emissions resulting from tree mortality (2003–2012). Cumulative and annual state-level CO_2 emissions from aboveground tree biomass loss generated by bark beetle (A and C), and fire (B and D), respectively. These two disturbances and timber harvest constitute the main drivers of change in forest cover loss and forest carbon stock in the Western United States.

The magnitude and spread of forest CO_2 emissions are larger for bark beetles than for fire, with some spatial clusters of high emissions (fig. 9.3a). State-level emissions from beetle-related tree mortality were significant in Colorado (97 percent of the total fire- and beetle-related emissions), Wyoming (84 percent), Washington (76 percent), and Montana (73 percent). Fire accounted for around 62 percent of the state-level forest carbon emissions in Arizona, California, and New Mexico (fig. 9.3b). Carbon dioxide emissions ranged from 73 to 136 Mt CO_2 per year (mean of 89 Mt CO_2 per year), with the largest level of emissions observed between 2008 and 2010 (figs. 9.3c and 9.3d). Estimates of CO_2 emissions from 2002 to 2012 (893 Mt CO_2) differ from estimates reported by Woodall et al. (2015) because of methodological differences in the estimation of carbon emissions.

Valuing the Cost of Forest Carbon Fluxes From Disturbances

Once released into the atmosphere, around 60 to 80 percent of the CO_2 will be absorbed by the ocean or the terrestrial biosphere within 200 to 2,000 years, while the remaining CO_2 may persist for several thousands of years (Archer et al. 2009). This fact highlights the substantial effect that CO_2 emissions may have on global climate long into the future. As carbon emissions accumulate in the atmosphere and exhaust the carbon budget associated with certain levels of climate change, the marginal damage caused per additional ton of CO_2 emission increases. The Interagency Working Group (IWG) on the Social Cost of Carbon (IWG 2016) relied on three global integrated assessment models (IAMs) of human and ecological systems to estimate the global net monetary impact per additional CO_2 emitted between 2010 to 2050 (fig. 9.4). This social cost of CO_2 (SC-CO_2) estimate accounts for potential climate change impacts—both benefits and costs—of small increases in regional temperatures across multiple economic sectors, such as changes in agricultural productivity, human health consequences, and increased risks to property.

Given the long-term permanence of CO_2 emissions in the atmosphere, the IAMs used by the IWG assess the net economic costs from the year of the emission to 2300. Different discount rates of social time preference were applied (2.5, 3, and 5 percent)[7] to allow a comparison of the stream of benefits and costs estimated to 2300, which will affect the welfare across generations. The SC-CO_2 is also estimated for a scenario that approximates extreme climate effects by using the 95th percentile of the distribution estimates at the 3 percent discount rate. SC-CO_2 estimates reported by the IWG for the period 2010–2050 were used to approximate,

[7] Discount rates are a key determinant of the social costs of carbon. High discount rates (e.g., around 7 percent) place a greater value on the welfare of current generations than on the well-being of future generations. Discount rates between 0 to 7 percent are standard in the literature related to the estimation of social costs of carbon emissions.

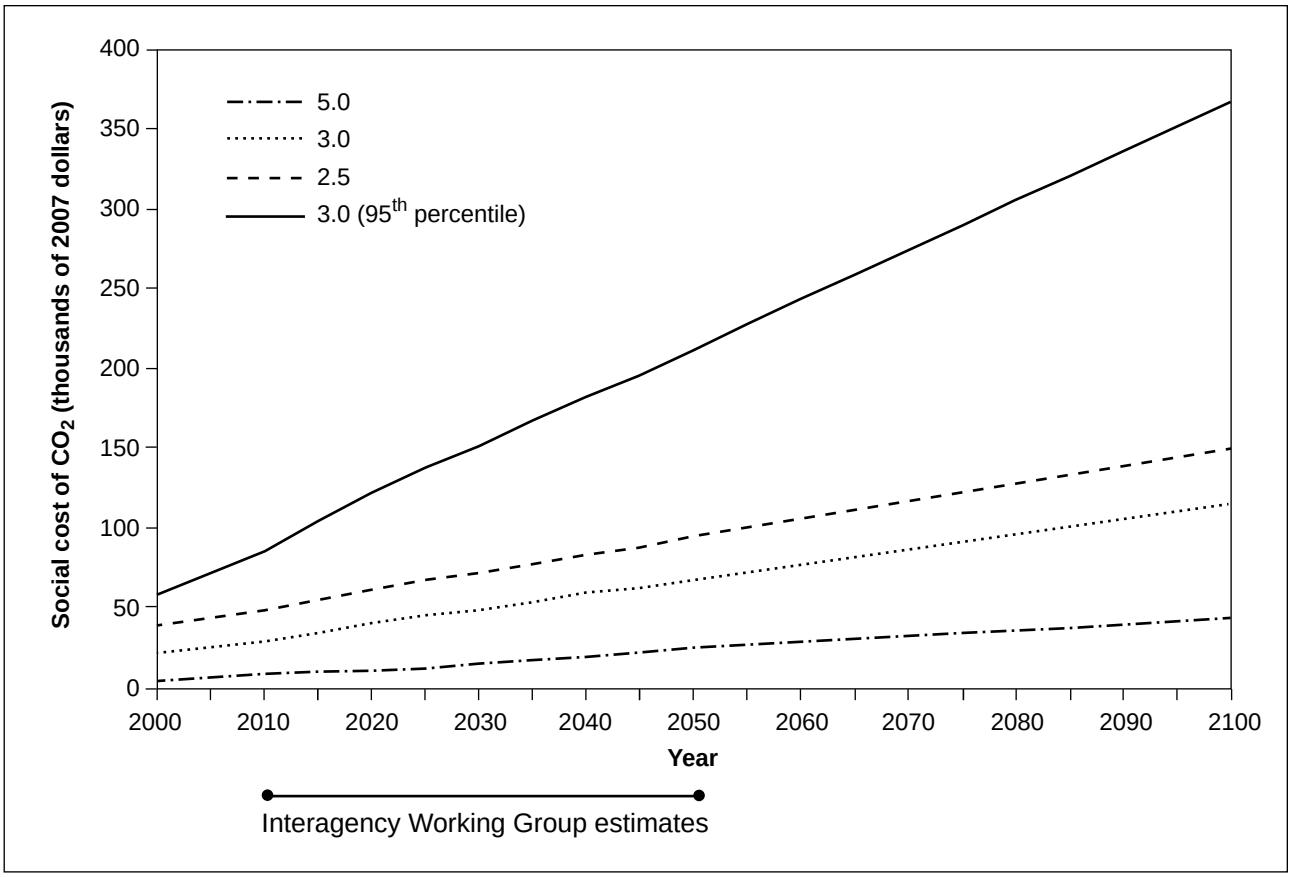

Figure 9.4—Social cost of carbon dioxide (CO_2) from 2000 to 2100 (2007 U.S. dollars per metric ton of CO_2) at different discount rates. Social costs estimated by the Interagency Working Group (2016) for the period 2010–2050 were used to generate linear estimates for 2000–2009 and 2051–2100. Tipping points in the climate system may lead to large changes in the social costs after 2050, though this effect is not captured in the social costs estimates.

using statistical regressions, the cost of annual emissions from disturbance events from outside this range, i.e., from 2003 to 2012 and from 2051 to 2100.

Annual SC-CO_2 estimates from 2003 to 2012 (in 2007 dollars) were first multiplied by the corresponding annual forest carbon emission resulting from fire- and beetle-induced tree mortality, then the results were aggregated to generate a cumulative estimate. With a 3 percent discount rate, the cumulative costs of forest carbon emissions range from around $50 to $18,000 per hectare (2.47 ac) of forest disturbed in any year between 2003 and 2012 (fig. 9.5). The range indicates a large difference in the impact per unit area of disturbed forest, i.e., in some regions, tree mortality resulting in CO_2 emissions was very low, while in others almost all the tree biomass was lost. The total cost during this period is around $9.2 billion for fire and $16.6 billion for bark beetles. Average annual SC-CO_2 ranges from $11 million in Nevada to $469 million in California (fig. 9.6). The total SC-CO_2 at 2.5 and 5 percent discount rates are $42.4 billion and $7.2 billion, respectively (table 9.2), but increase to $71.7 billion under the high-impact scenario (i.e., 3 percent discount rate and 95th percentile of climate impact estimates).

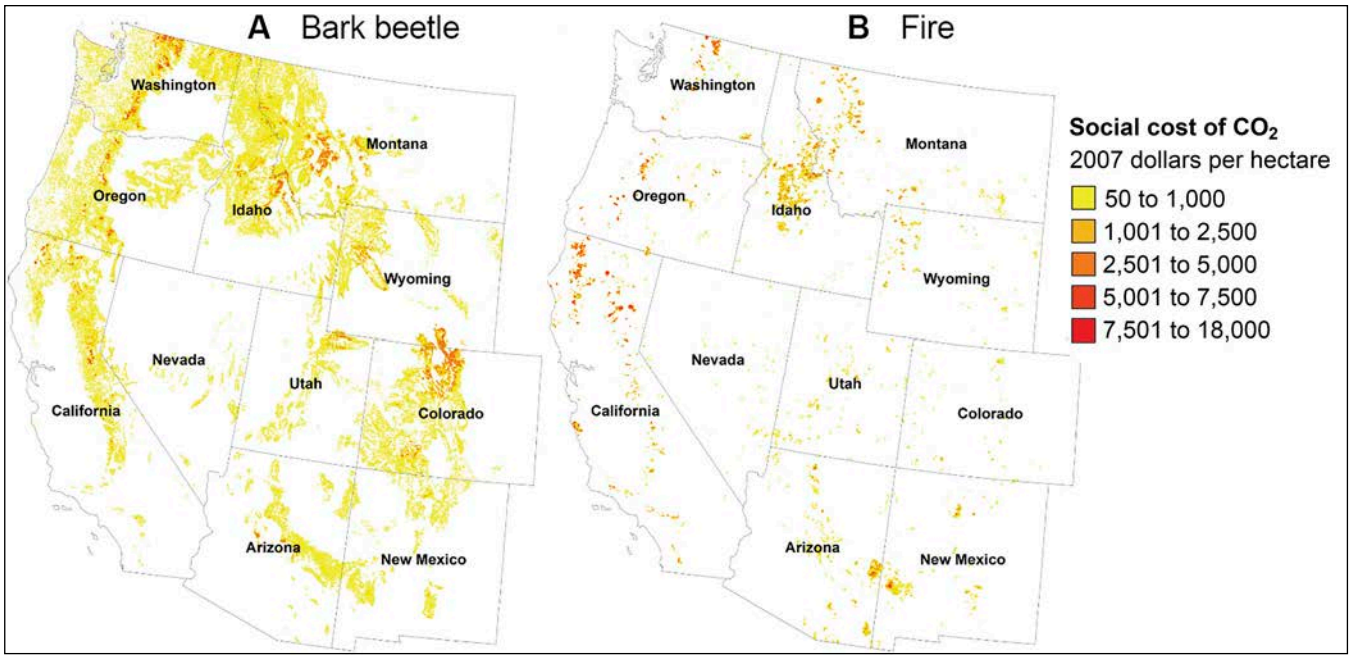

Figure 9.5—Cumulative social cost of carbon dioxide (CO_2) emissions from tree mortality caused by (A) bark beetle and (B) fire disturbances (2003–2012). Estimates using social cost of CO_2 are discounted at 3 percent discount rate.

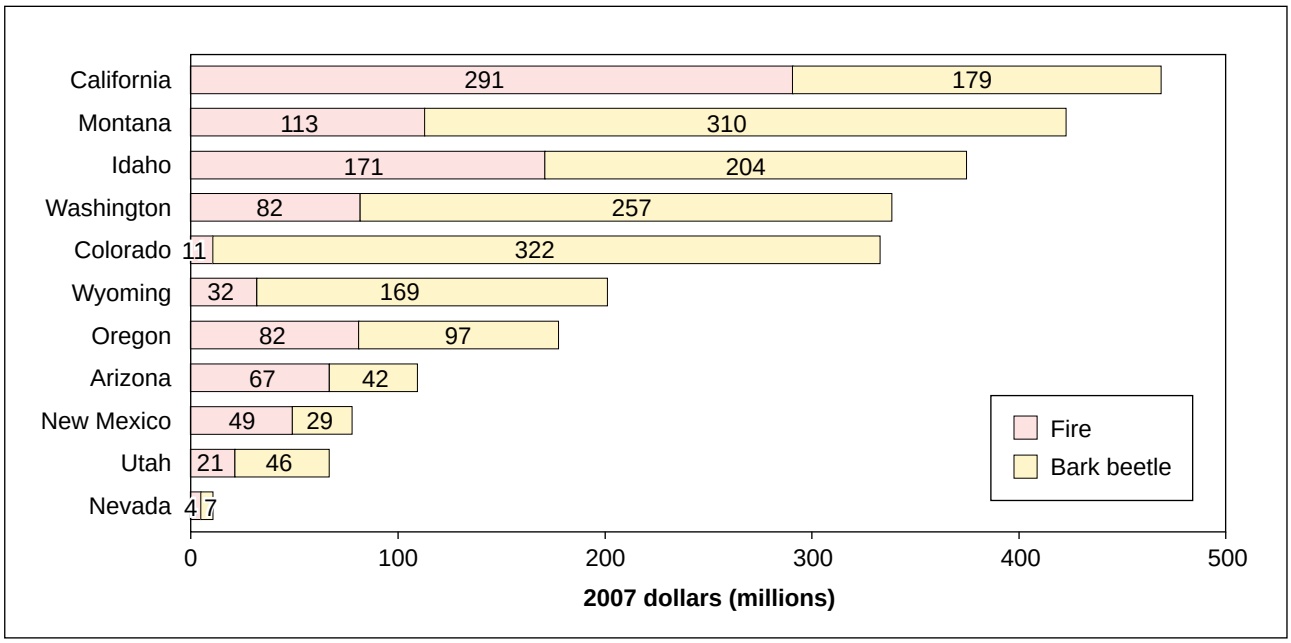

Figure 9.6—Average annual social cost of carbon dioxide emissions from fire and bark beetle forest disturbances at 3 percent discount rate (2003 to 2012).

Table 9-2—State-level cumulative social costs of carbon dioxide emissions from forests resulting from fire and beetle disturbances at different discount rates (2003–2012)

	Discount rate (percent)			
	2.5	3	5	3 (95th percentile)
	Billions of 2007 dollars			
California	7.70	4.69	1.31	13.01
Montana	6.94	4.23	1.19	11.72
Idaho	6.11	3.75	1.08	10.43
Colorado	5.44	3.33	0.95	9.24
Washington	5.60	3.39	0.91	9.37
Wyoming	3.29	2.01	0.58	5.58
Oregon	2.93	1.78	0.49	4.94
Arizona	1.82	1.10	0.30	3.02
New Mexico	1.28	0.78	0.22	2.17
Utah	1.10	0.67	0.19	1.85
Nevada	0.18	0.11	0.03	0.30
Total	42.41	25.85	7.24	71.65

Projected Climate Change Impacts on Forest Carbon Stocks

Recent carbon emissions from forest disturbances have been offset by carbon storage in the form of forest biomass growth and forest cover expansion. To assess if such trends are expected to continue under climate change projections, we used the dynamic global vegetation model MC2 (Bachelet et al. 2015). MC2 simulates vegetation dynamics under climate projections from 20 global climate models and for the representative concentration pathways (RCPs) 4.5 and 8.5.[8] MC2 accounts for spatially heterogeneous differences in soil and vegetation productivity, as well as wildfire suppression.

Fire effects are simulated to account for tree mortality and combustion,[9] and for tree mortality without combustion where carbon is transferred to dead pools and undergoes gradual decomposition.[10] The effect of fire suppression on forest emissions is modeled under a set of thresholds. Large fires are allowed to run in the model, assuming that once conditions are perfect for spread, suppression efforts will fail. The model also assumes that, under high CO_2 concentrations, trees will be less stressed as they will be more water-use efficient, thus there will be higher water content in the live carbon pools; consequently, these trees will be less likely to

[8] RCP 4.5 projects temperatures to be 1.8 °C warmer than today by 2100, whereas under RCP 8.5, it will be 3.7 °C warmer by 2100.

[9] Combustion implies that emissions are released immediately.

[10] Decomposition implies that emissions are released gradually over time.

combust. In addition, for forests damaged by fire, MC2 specifies vegetation adapted to new (warmer/drier) conditions to regrow that may be more resilient to drought, though with lower potential carbon stocks (i.e., potentially with lower biomass production and consequently fewer fuels).

Carbon stocks are expected to increase by approximately 5 and 7 percent under RCP 4.5 and 8.5, respectively, by 2091 relative to 2011 levels (table 9.3). Owing to the increasing marginal costs of carbon emissions, the social benefit from the increase in net carbon stocks by 2071 are around four times the level in 2011, and six times by 2091 for the modeled RCPs at various discount rates. Carbon stocks are projected to increase in areas of high net primary productivity (fig. 9.7).

Table 9-3—Projected social benefits of net carbon dioxide (CO_2) sequestration resulting from vegetation dynamics in Western U.S. forests

	2011	2021	2031	2041	2051	2061	2071	2081	2091
	Gigatonnes of CO_2								
Total ecosystem carbon with wildfire suppression:									
RCP 4.5	168.73	170.37	172.07	173.26	173.98	174.83	175.02	175.77	176.58
RCP 8.5	168.73	170.14	172.35	173.41	174.55	175.17	176.09	178.11	180.43
	Billions of 2007 dollars								
SC-CO_2:									
RCP 4.5—3 percent	6,074	8,178	10,152	12,128	14,788	18,183	22,228	27,244	33,373
RCP 4.5—5 percent	1,856	2,385	3,097	4,158	5,045	6,294	7,701	9,316	11,478
RCP 8.5—3 percent	6,074	8,167	10,169	12,139	14,836	18,218	22,364	27,607	34,101
RCP 8.5—5 percent	1,856	2,382	3,102	4,162	5,062	6,306	7,748	9,440	11,728

SC-CO_2 = social costs of CO_2. CO_2 emissions have a social cost through their effect on climate change impacts; CO_2 sequestration has a social benefit by reducing climate change impacts.

There are multiple interactions that make RCP 4.5 and 8.5 similar. A warmer and drier climate (RCP 8.5) causes a shift in vegetation type, better adapted to cope with the new conditions, but often with less biomass production, resulting in lower fuel loads and ultimately fewer fires. Moderate climate conditions under RCP 4.5 foster more forest production than under the more severe RCP 8.5 scenario, but they also generate more fuel for the fires that dry years will trigger; ultimately, these relatively moderate conditions may lead to lower carbon stocks. Overall warm[11] and moist conditions make trees grow more (i.e., produce more biomass), but this extra

[11] Although temperatures under RCP 4.5 are warmer than today, it is still cooler than under RCP 8.5.

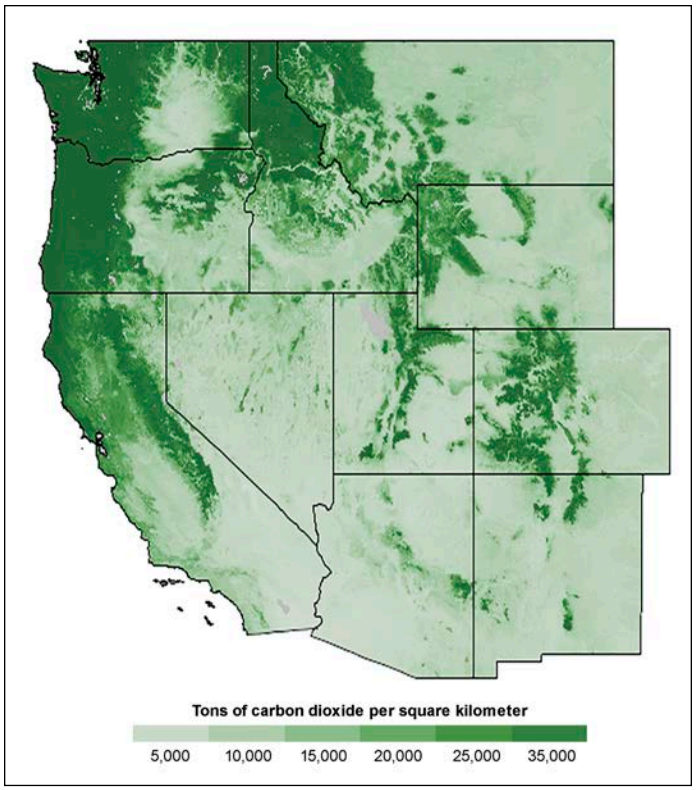

Figure 9.7—Projected average forest carbon stocks under RCP 4.5, 2070–2099 (tons of carbon dioxide stored in forest biomass per square kilometer).

growth will fuel fires that may occur during projected very dry warm years. In such conditions, simulated fire suppression causes woody vegetation to expand in areas dominated by shrubs or grasses, causing a general reduction in areas dominated by herbaceous vegetation. Such an expansion is expected to be low where agriculture has already reduced the extent of prairies and grasslands. Fire suppression has a significant effect on the modeled carbon stock change, particularly for undisturbed land with high biomass production potential.

Key Insights

The recent effect of disturbances on forest carbon stocks and the corresponding social costs are significant.

In Western U.S. forests, forest regrowth in previously disturbed land and tree biomass growth within forested areas have generated net increases in carbon stocks. Vegetation dynamics under climate change could result in further net increases in such stocks, generating valuable social benefits by offsetting CO_2 emissions from other industrial sectors. Nevertheless, the recent effect of disturbances on forest carbon stocks and the corresponding social costs are significant. By the end of the 21st century, around half of the Western United States could experience climate profiles unrelated to contemporary climate (Bentz et al. 2010). This change in climate could trigger changes in vegetation types and extent, net primary productivity, wildfire frequency, expansion of the range of tree-damaging insects, and disease vectors.

Although the MC2 model used for simulating forest carbon dynamics accounts for some of these changes, unmodeled disturbances could increase tree mortality, significantly reducing biomass production. For instance, as insect outbreaks become more prevalent, the need to properly model their long-term effect on forest ecosystems and services becomes more relevant. Despite the limitations and uncertainties around the possible future of U.S. forests, models indicate that climate change will amplify negative effects of forest stressors. Estimates of the social impact of Western U.S. forests disturbances can guide the assessment of strategies to preserve or enhance forest health in this region, and to protect their flow of ecosystem services for future generations.

Other Ecosystem Services Affected by Forest Disturbances

Biodiversity

Society benefits from direct, indirect, and potential (i.e., unidentified or undiscovered) goods and services generated by biodiversity. Forests with significant biodiversity richness provide food, fuel, medicine, and fiber to local communities; harbor raw materials for commercial drug development; and provide tourism benefits (National Research Council 1999). In addition to these more concrete benefits, biodiversity and its conservation **for its own sake** is a central, if less tangible, value for many people and a powerful motivator for political engagement; economists label this **existence value**.

Nevertheless, methods to estimate use and non-use values of biodiversity face significant challenges. For instance, willingness to pay estimates for biodiversity protection generated through contingent valuation methods may be sensitive to informational asymmetries about the role of biodiversity in the health and functions of ecosystems (Hanley et al. 1995, Rambonilaza and Brahic 2016); bequest values for biodiversity preservation may be biased toward charismatic species (Trimble and Van Aarde 2010). These challenges in measurement notwithstanding, the values associated with biodiversity are substantial, as evidenced by the billions of dollars that Americans contribute to conservation and related charities.

In general, forest ecosystem services are negatively affected by disturbance events (Thom and Seidl 2016); however, biodiversity appears to be positively affected as long as the frequency and intensity of disturbance regimes are within the tolerance of relevant species (Kulakowski et al. 2017). The estimation of the effects of forest change on biodiversity is complicated by nonlinear responses to ecological or anthropogenic disturbances, threshold and cascade effects, and different resilience profiles across species. For instance, large-scale outbreaks of bark

beetle in German forests have been associated with biodiversity increases (Beudert et al. 2015). This highlights the relevance of assessing the impacts that forests disturbances have across multiple parameters to effectively manage their tradeoffs and benefits.

Cultural

Aesthetic experiences are considered to be part of cultural services, a class of non-material benefits people obtain from ecosystems (Millennium Ecosystem Assessment 2005). Forest landscapes and other natural environments are an important source of aesthetic pleasure for some people. People living in the wildland-urban interface (WUI) enjoy aesthetic values—or scenic quality—from their views of the forest landscape. Outdoor enthusiasts also derive benefits from individuals' scenic viewshed access while recreating in natural settings.

Insects and diseases have a negative effect on aesthetic pleasure because they reduce the level of satisfaction that people receive from scenic viewsheds.

Insect and diseases have a negative effect on aesthetic pleasure because they reduce the level of satisfaction that people receive from scenic viewsheds. For example, Sheppard and Picard (2006) conducted a literature review of the perceived visual quality of insect infestations, primarily beetles. The authors found that dying trees reduced the aesthetic value of forested landscapes. Similar effects have been found for WUI residents, as studies have identified declines in property values because of dead trees (Holmes et al. 2010, Price et al. 2010).

Wildfires also affect aesthetic values or scenic quality for recreation enthusiasts and WUI residents. Studies have found that property values of homes in proximity to and in view of areas burned by wildfire decrease (Huggett et al. 2008, Stetler et al. 2010). There are mixed results for wildfire effects on scenic quality that recreationists derive from recreation activities (e.g., hiking, photography, nature viewing, etc.). An early study (Ribe et al. 1989) found evidence that wildfire detracts from beauty. A more recent study by Sánchez et al. (2016), however, found that scenic quality might not be completely degraded by certain types of fires, specifically recent low-intensity fires. Starbuck et al. (2006) suggested that low-intensity fires open and enhance the viewshed, while others have found that wilderness visitors perceived burned areas to be interesting landscape features worthy of exploration (Schroeder and Schneider 2010).

Timber

Wildfires, insect and disease outbreaks, and other disturbances can reduce the supply and value of trees for commercial logging. The effect of forest disturbances on timber markets depends upon the extent, intensity, and frequency of the damage, and on postdisturbance responses by timber companies. If loggers need to salvage a significant amount of damaged timber, local timber prices may decrease in the short

term. On the other hand, if forest disturbances damage a large proportion of the tree inventory, the effect on local timber markets, industries, and livelihoods may last much longer (Prestemon and Holmes 2008). At regional or national scales, the effect of fires on the timber industry may be limited, though this observation applies mainly to the case of relatively stable fire regimes. For instance, the 2017 fires in the Western United States and Canada resulted in a short-term increase of around 6 percent in benchmark western spruce-pine-fir lumber prices in Canada (Sagan 2017). If future fire regimes involve significantly more acres burned on an annual basis, then regionwide impacts may well emerge.

Timber salvage from public lands following natural disturbances may reduce financial losses; yet damaged trees provide significant ecological services, and their removal could alter ecosystem functions, affect biodiversity indicators, and increase restoration costs (Castro et al. 2011, Leverkus et al. 2012). Forest managers need to weigh the potential benefits of commercial extraction of damaged trees versus the social welfare implications of such removal in light of their potential effect on ecosystem services.

Human Health

The World Health Organization defines health as "…a state of physical, mental and social well-being and not merely the absence of disease or infirmity" (WHO 2020). According to the Millennium Assessment Ecosystem, ecosystem services "are indispensable to the well-being and health of people everywhere" (Corvalan et al. 2005). Ecosystem services provide basic needs for life such as food, water, air, and relative climatic constancy. Nascent research into understanding the linkages between the environment, ecosystem services, and human health may be critical in guiding policy efforts (Sandifer et al. 2015) that attempt to improve human health and well-being.

Disturbances such as wildfires, insect outbreaks, and drought negatively affect the provision of these services from forests, thereby reducing concomitant public health benefits and well-being. For example, in 2017, the Oregon Folk Festival was canceled for the first time in 22 years because wildfire smoke had degraded local air quality (McDonald 2017). Kochi et al. (2010) highlighted the importance of estimating the health-related cost of wildfire smoke exposure when developing wildfire management policy, though at the time of their study they found only six studies worldwide that estimated health-related economic costs of wildfire smoke. In California, wildfires have made breathing air so hazardous that air quality indices have compared breathing the polluted air to smoking 14 cigarettes (Resnick 2018). Richardson et al. (2012) estimated the economic cost of health effect from large fire

to be $84.42 per exposed person per day for California's Station Fire of 2009. More recently, Kochi et al. (2016) estimated the medical costs for southern California wildfires in 2007 to be more than $3.4 million. Despite the intuitive importance of natural systems to people, the empirical evidence to support these claims has been sparse, spurring researchers to expand efforts in such disciplines as disease ecology (Myers et al. 2013). Nevertheless, much more research needs to be done to clearly understand the connection between forest disturbances, human health, and well-being.

Water

Forests provide a range of water provisioning services, including hydrological regulation, flood control, groundwater recharge, and water quality enhancement (Lele 2009). Researchers have recognized this and have attempted to value water quality-related ecosystem services from forested watersheds (e.g. Elias et al. 2013, Lele 2009, Ojea and Martin-Ortega 2015), some specifically focusing on valuing water from public forests in the Western United States (Aguilar et al. 2018). In the 19th century, public concern about adequate supplies of clean water resulted in the establishment of federally protected forest reserves in 1891 (USDA FS 2000). The vast majority of the nation's freshwater originates from forests (about 80 percent), of which about 14 percent is from national forests (USDA FS 2000).

In the West, national forests provide around one-third of freshwater runoff (Brown et al. 2016) because they encompass most of the forest land in the region and include the headwaters of major rivers and numerous mountain ranges. Through natural filtration, forests and the watersheds they encompass have been and continue to be a low-cost supplier of raw clean water, making them extremely valuable natural assets. Stressors like wildfires affect forests and their ability to generate water provisioning and water quality services. For example, researchers have recently discovered that large wildfires can cause an increase in streamflow that can last for years or even decades. Although enhanced river flows can be beneficial in times of scarce water, the increased flows also result in an increase in contaminants, sediments, and nutrients (Hallema et al. 2018).

Although natural assets such as forests and the watersheds they encompass perform fundamental life support services, these services are discounted or implicitly valued at zero. Although these services are not paid for, their loss in terms of water treatment facility costs are high. For example, in southern California, the Angeles National Forest plays a vital role in preventing mudslides. Sediment naturally comes off the San Gabriel Mountains and collects in debris basins, obliging public agencies such as the Los Angeles County Department of Public Works to constantly clear them at an average cost of almost $12 per cubic meter (Wohlgemuth et al. 2018). If there is heavy precipitation immediately following a bad wildfire season,

these debris basins can quickly overflow, resulting in mudslides to the surrounding communities from San Fernando to San Dimas. The interwoven relationship between ecosystems and human well-being is insufficiently recognized in the wider philosophical, social, and economic well-being literature (Summers et al. 2012). The importance of forests as a source of water in the Western United States warrants more research to understand how both volume and quality will be affected as climate conditions change.

Fire Impacts on Structures

Fire suppression costs have exponentially increased in recent decades. From 2010 to 2017, the Forest Service and agencies within the U.S. Department of the Interior have spent annually an average of over $1.8 billion to suppress fires, triple what had been spent 30 years ago (Peterson and Barrett 2020). Fire suppression costs surpassed the $2 billion threshold for the first time in 2017 (NIFC 2019a). Since fiscal year 2015, more than half of the Forest Service budget is spent annually on fire suppression, and it is projected to grow to 67 percent by 2025 (USDA FS 2015). This has been a problem in previous years as funds are transferred in a process known as "fire borrowing" from other Forest Service programs to help subsidize fire suppression costs (USDA FS2015). This is no longer expected to be an issue starting in fiscal year 2020, however, as the Forest Service will now have access to more than $2 billion a year outside of its regular fire suppression budget to contend with wildfire expenditures.

Higher suppression costs are due partly to fires becoming more frequent, larger, and more devastating as they expand to the human communities adjacent to forested areas.

Part of the reason for higher suppression cost is that fires are becoming more frequent, larger, and more devastating as more urban development expands to the WUI, resulting in communities being adjacent to forested areas. Nationwide in 2005, more than 44 million homes were located in the WUI, with California having the highest number—5.1 million housing units (Radeloff et al. 2005). Furthermore, from 1990 to 2010, more than 43 million new houses were built in WUI areas (Radeloff et al. 2018). Peterson and Barrett (2020) reported that the annual area burned increased to 2.7 million ha in the past decade. In 2018, wildfires destroyed 25,790 structures; with 23,647 structures burned, California accounted for the highest number of structures lost (NIFC 2019b). In 2018, California experienced the largest wildfire (Mendocino Complex) in the state's history. The wildfire burned 459,123 ac (~186,000 ha) and destroyed 280 structures (Inciweb 2018). Late in the same year, the Camp Fire (Paradise, California) became one of the most destructive and deadliest fires in California's history, destroying more than 18,800 structures and causing 88 fatalities (CAL FIRE 2018). The Camp Fire temporarily created the worst air quality in the world for the affected region (Resnick 2018). Similarly, other Western States in recent years have experienced some of their largest wildfires.

Conclusion

Disturbance processes have played an integral role in shaping the forests of the Western United States, including influencing the benefits that people derive from these same forests. Forest disturbances are also creating costs; for example, fire and bark beetle outbreaks have had significant effects on tree mortality, carbon loss, and property values. People are adapting to **current** disturbance regimes in a myriad of ways in which they interact with forested landscapes; actions include both public and private efforts, such as government initiatives, budget priorities, public and private investments, and recreation activities. Future disturbance regimes are likely to increase in frequency and severity. These may include disturbances relating to insect and disease outbreaks, wildfires linked to growing forest fuel loads across the region, continuing encroachment of human settlement, and, crucially, future effects of climate change. As this chapter shows, these disturbances will result in both benefits and costs, often in surprising ways. Nevertheless, on balance, the effect on human welfare is likely to be negative, and commensurate with the resilience of the forest landscape and human adaptability.

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Tree and Insect Species Identified in This Report

Scientific name	Common name
Trees:	
Abies amabilis (Douglas ex Louden) Douglas ex Forbes	Pacific silver fir
Abies balsamea (L.) Mill.	Balsam fir
Abies concolor (Gord. & Glend.) Lindl. ex Hildebr	White fir
Abies grandis (Douglas ex D. Don) Lindl.	Grand fir
Abies lasiocarpa (Hook) Nutt.	Subalpine fir
Abies magnifica A. Murray bis	California red fir
Abies procera Rehder	Noble fir
Alnus rubra Bong.	Red alder
Callitropsis nootkatensis(D. Don) Oerst. ex D.P. Little	Yellow-cedar
Notholithocarpus densiflorus (Hook. & Arn.) P.S. Manos, C.H. Cannon, & S.H. Oh	Tanoak
Larix occidentalis Nutt.	Western larch
Picea engelmannii Carr.	Engelmann spruce
Picea glauca (Moench) Voss	White spruce
Picea × *lutzii* Little	Lutz spruce
Picea sitchensis (Bong.) Carr.	Sitka spruce
Picea mariana (Mill.) B.S.P.	Black spruce
Pinus albicaulis Engelm.	Whitebark pine
Pinus contorta var. *latifolia*	Lodgepole pine
Pinus coulteri D. Don	Coulter pine
Pinus edulis Engelm.	Pinyon pine
Pinus flexilis E. James	Limber pine
Pinus jeffreyi Grev. & Balf.	Jeffrey pine
Pinus lambertiana Dougl.	Sugar pine
Pinus monophylla Torr. & Frem.	Pinyon pine
Pinus monticola Dougl. ex D. Don	Western white pine
Pinus ponderosa Dougl. ex Laws	Ponderosa pine
Populus tremuloides Michx.	Quaking aspen
Pseudotsuga menziesii (Mirb.) Franco)	Douglas-fir
Quercus agrifolia Née	Coast live oak
Quercus chrysolepsis Liebm.	Canyon live oak
Quercus garryana Douglas ex Hook.	Oregon white oak
Quercus wislizeni A. DC.	Interior live oak
Rhododendron L.	Rhododendron
Sequoia sempervirens (Lamb. Ex D. Don)	Redwood
Tsuga heterophylla (Raf.) Sarg.	Western hemlock
Umbellularia californica (Hook. & Arn.) Nutt.	California bay laurel

Scientific name	Common name
Insects:	
Adelges piceae (Ratzeburg)	Balsam woolly adelgid
Agrilus coxalis	Goldspotted oak borer
Agrilus liragus	Bronze poplar borer
Agrilus panipennis	Emerald ash borer
Choristoneura freemani Razowski	Western spruce budworm
Choristoneura fumiferana Clemens	Eastern spruce budworm
Coleophora laricella Hubner	Larch casebearer
Coloradia pandora (Blake)	Pandora moth
Dendroctonus brevicomis LeConte	Western pine beetle
Dendroctonus jeffreyi Hopkins	Jeffrey pine beetle
Dendroctonus ponderosae Hopkins	Mountain pine beetle
Dendroctonus pseudotsugae Hopkins	Douglas-fir beetle
Dendroctonus rufipennis Kirby	Spruce beetle
Dryocoetes confuses Swaine	Western balsam bark beetle
Elatobium abietinum (Walker)	Spruce aphid
Ips confusus LeConte	Pinyon ips
Ips paraconfusus LeConte	California fivespined ips
Ips perturbatus Eichhoff	Northern spruce engraver
Ips pini Say	Pine engraver
Lambdina fiscellaria lugubrosa (Hulst)	Western hemlock looper
Malacosoma californicum (Packard)	Western tent caterpillar
Malacosoma disstria Hubner	Forest tent caterpillar
Neodiprion spp.	Pine sawflies
Neodiprion autumnalis Smith & Wagner	Conifer sawfly
Neophasia menapia Felder & Felder	Pine butterfly
Orgyia pseudotsugata McDunnough	Douglas-fir tussock moth
Phaenops californica Van Dyke	California flatheaded borer
Phaenops drummondi Kirby	Flatheaded fir borer
Procryphalus mucronatus LeConte	Aspen bark beetle
Scolytus ventralis LeConte	Fir engraver
Trypophloeus populi Hopkins	Aspen bark beetle

U.S. Equivalents

When you know:	Multiply by:	To find:
Millimeters (mm)	0.0394	Inches
Centimeters (cm)	0.394	Inches
Meters (m)	3.28	Feet
Kilometers (km)	1.61	Miles
Hectares (ha)	2.47	Acres
Square meters (m^2)	10.76	Square feet
Square kilometers (km^2)	0.386	Square miles
Cubic meters (m^3)	35.3	Cubic feet
Degrees Celsius (°C)	1.8 °C + 32	Degrees Fahrenheit
Megatonnes (Mt)	1.102	Million tons
Gigatonnes (Gt)	1,102	Million tons

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1220 SW 3rd Ave., Suite 1400
P.O. Box 3890
Portland, OR 97208–3890

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