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

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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

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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.

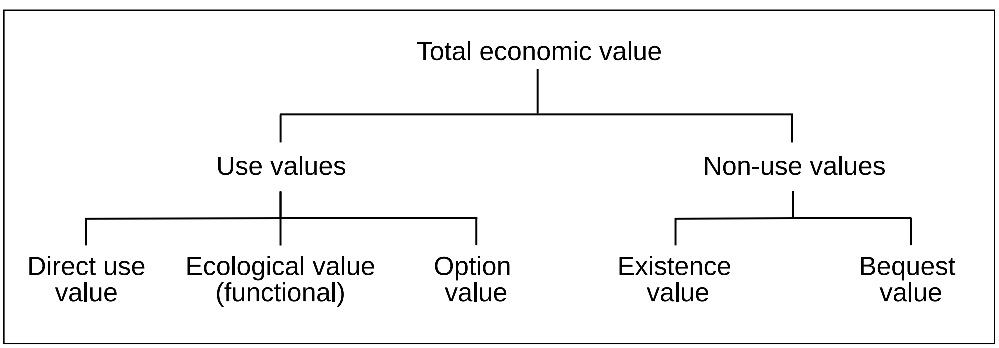

Figure 9.1—Total economic value typology. Source: Plottu and Plottu (2007).

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).

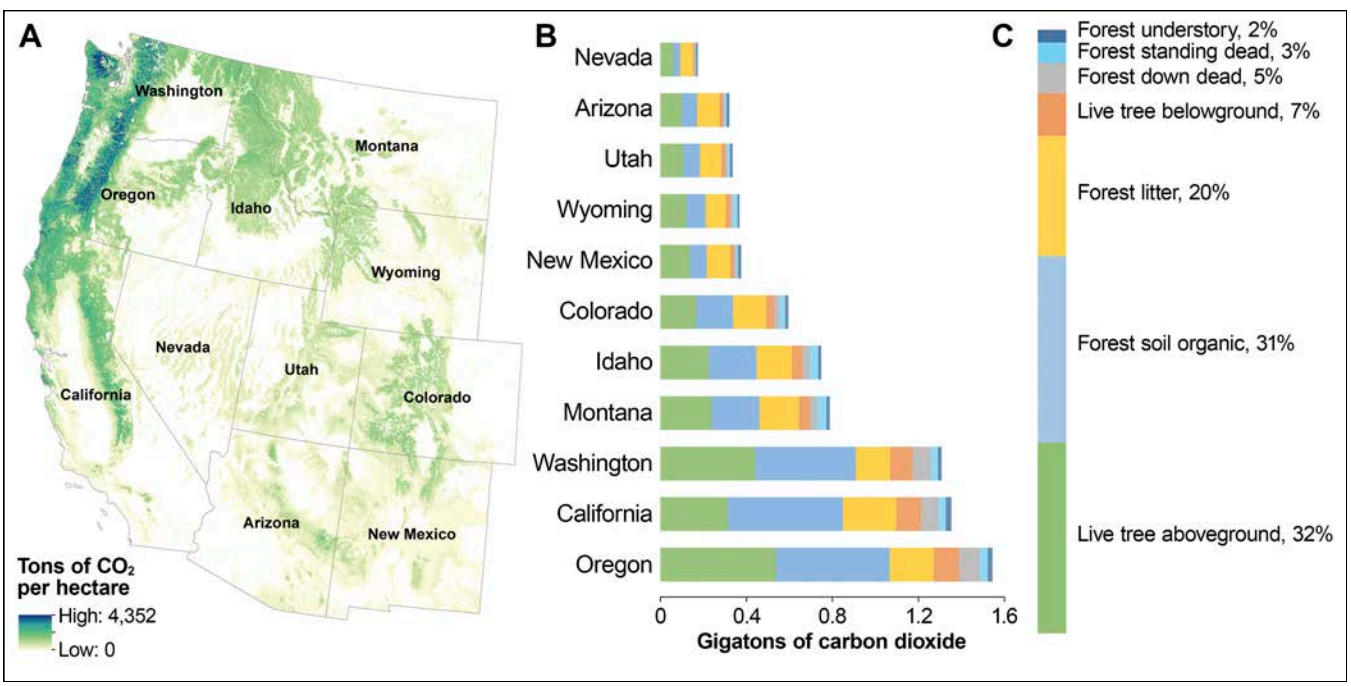

Figure 9.2—Forest carbon dioxide stock on public Western U.S. forest land (based on U.S. Forest Service Forest Inventory and Analysis data collected from 2000 to 2009): (A) Spatial distribution of forest carbon stock; (B) state-level carbon stocks across forest components; (C) proportion of carbon stored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

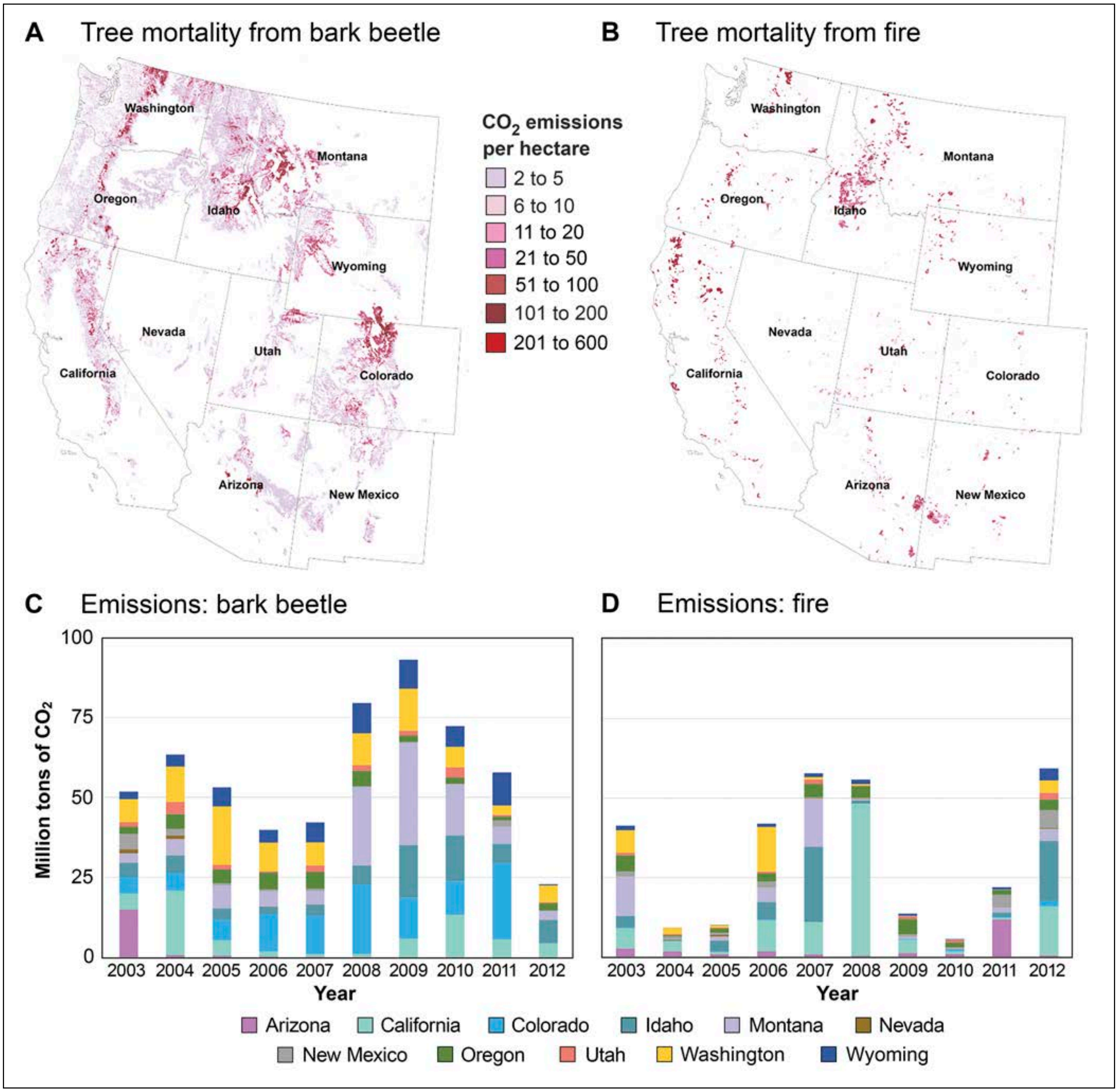

Figure 9.3—Carbon dioxide (CO_2) emissions resulting from tree mortality (2003–2012). Cumulative and annual state-level CO_2 emissions from aboveground tree biomass loss generated by bark beetle (A and C), and fire (B and D), respectively. These two disturbances and timber harvest constitute the main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

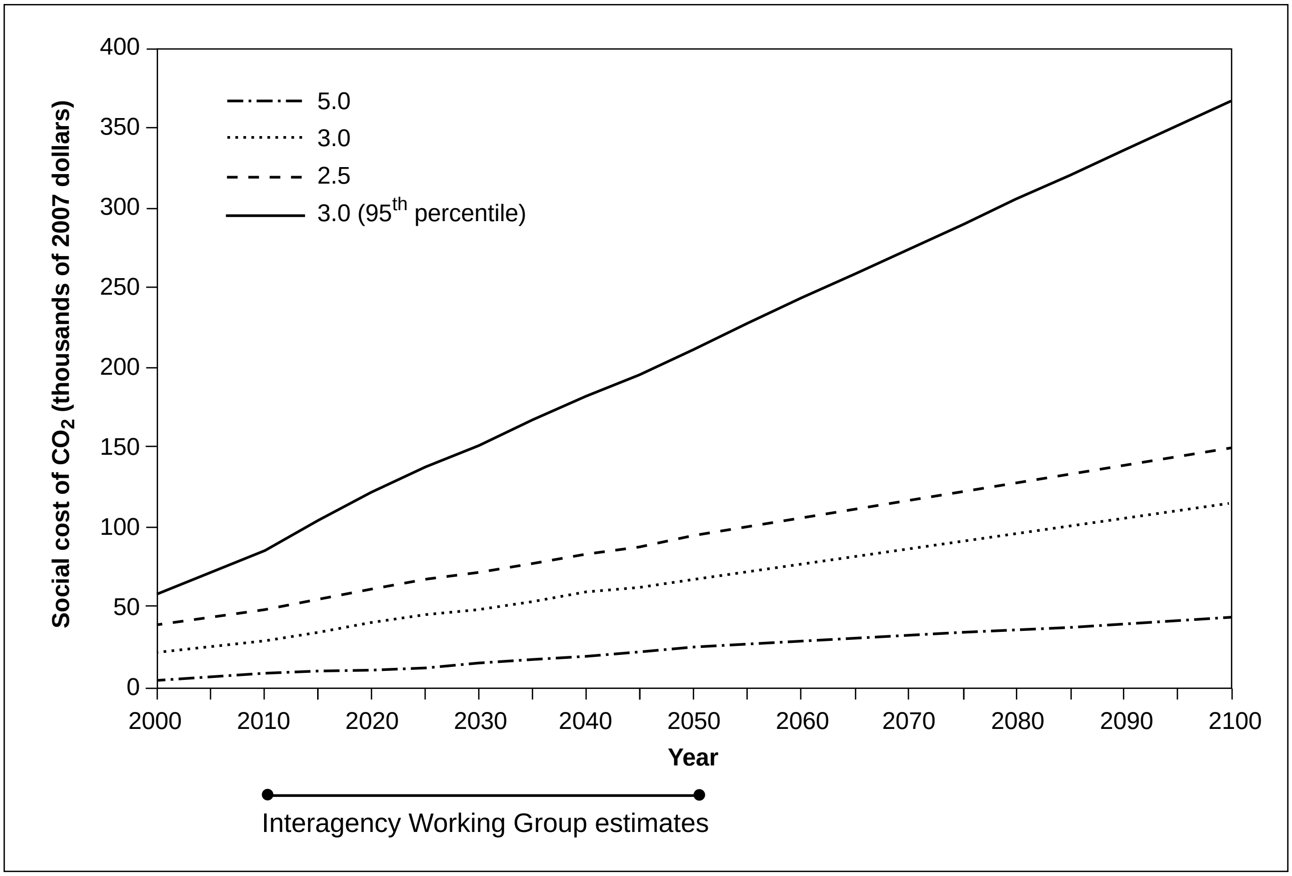

Figure 9.4—Social cost of carbon dioxide (CO_2) from 2000 to 2100 (2007 U.S. dollars per metric ton of CO_2) at different discount rates. Social costs estimated by the Interagency Working Group (2016) for the period 2010–2050 were used to generate linear estimates for 2000–2009 and 2051–2100. Tipping 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).

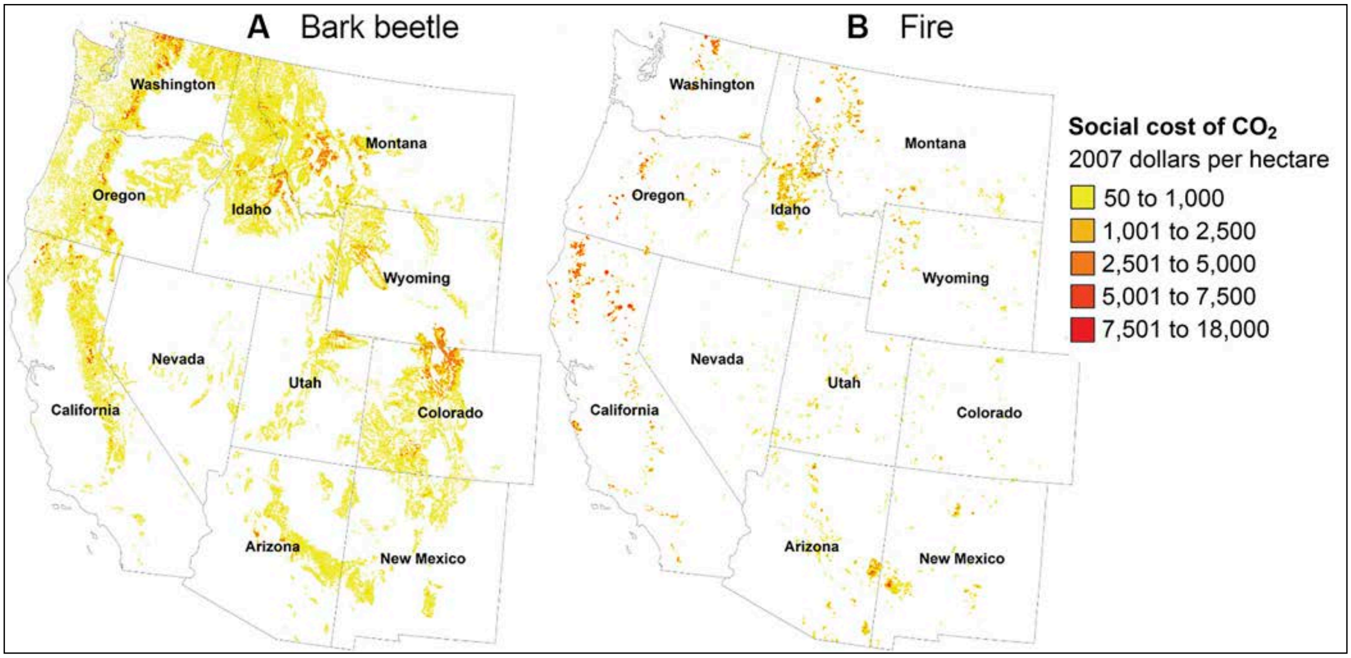

Figure 9.5—Cumulative social cost of carbon dioxide (CO_2) emissions from tree mortality caused by (A) bark beetle and (B) fire disturbances (2003–2012). Estimates using social cost of CO_2 are discounted at 3 percent discount rate.

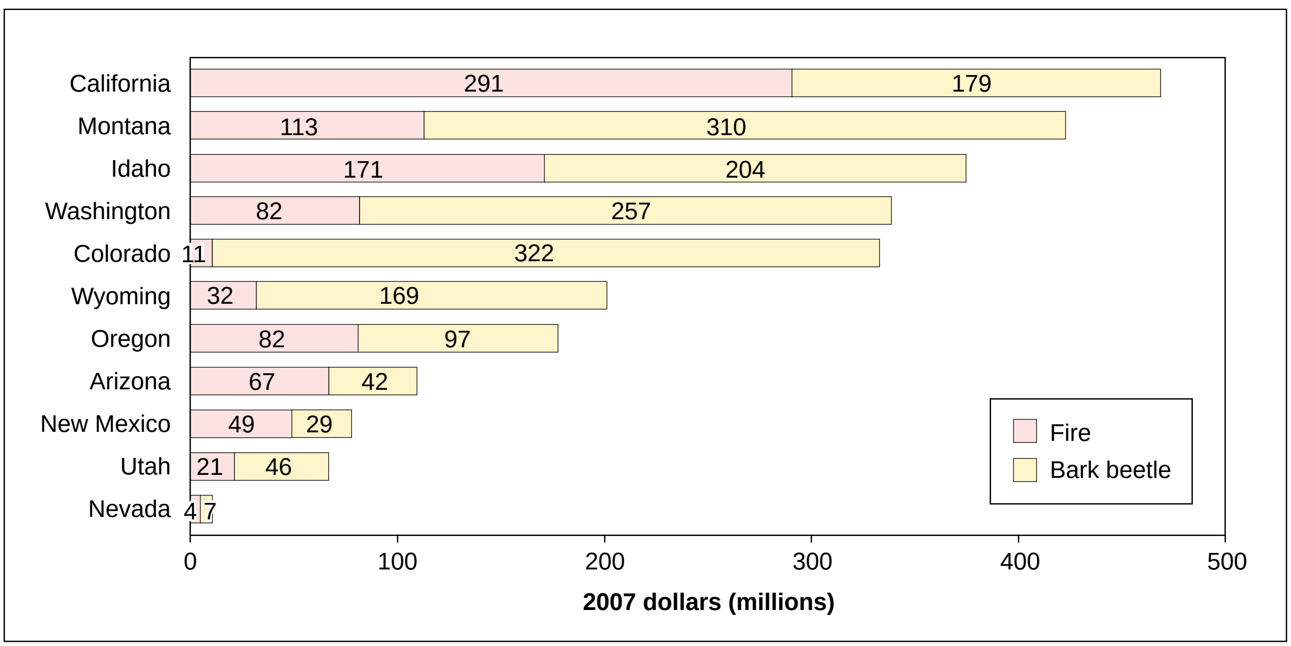

Figure 9.6—Average annual social cost of carbon dioxide emissions from fire and bark beetle forest disturbances at 3 percent discount rate (2003 to 2012).

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.

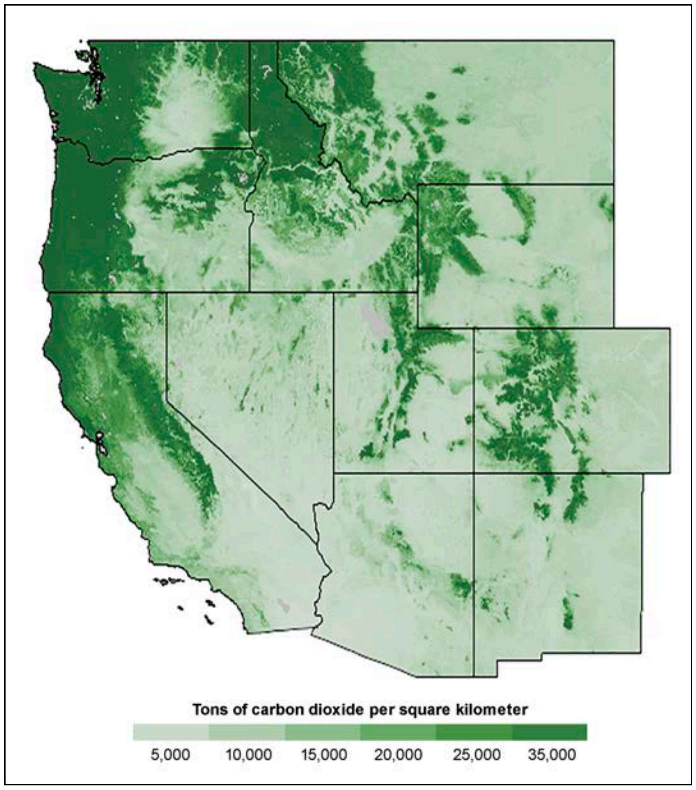

Figure 9.7—Projected average forest carbon stocks under RCP 4.5, 2070–2099 (tons of carbon dioxide stored in forest biomass per square kilometer).

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