Chapter 5

Disturbances to Forests and Rangelands

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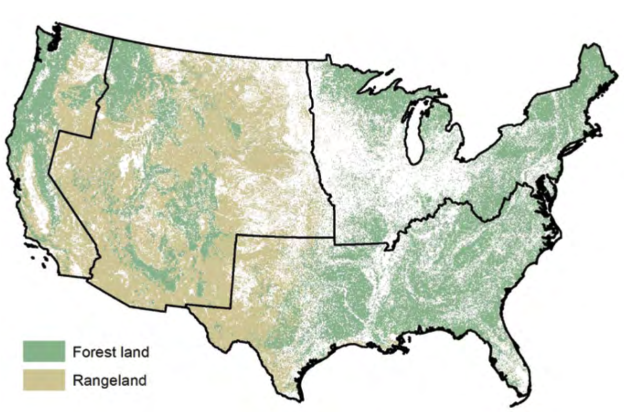
Disturbances including fire, insect and disease outbreaks, and drought are ubiquitous in forests and rangelands, and many disturbance events are parts of the natural dynamics of forest and rangeland ecosystems. This chapter is a new addition to the Resources Planning Act (RPA) Assessment and summarizes disturbance trends in the recent past and projected future trends within forests and rangelands across the conterminous United States. We assess status and trends of abiotic and biotic disturbance agents, including fire, drought, insects and disease, and nonnative invasive plants. Along with those agents, we summarize some forest management actions—prescribed burning and removals—that can be classified as disturbances because they alter environmental conditions and lead to changes in forest structure or community composition (White and Jentsch 2001), even though they can lead to forest resilience in the long term. The chapter is organized into sections focused on individual disturbance agents, most of which are summarized for forests and rangelands, with the exceptions of insect and disease agents and removals which are summarized only for forests. At the end of the chapter, we present a look at recent exposure of forests to multiple disturbances: removal, stress, and fire. Quantitative summaries emphasize *exposure* to disturbances: that is, trends or changes in the extent, severity, frequency, or duration of a disturbance from historical conditions, or expected future change from recent trends (Glick et al. 2011, Thorne et al. 2018). Where possible, we examine disturbance exposure alongside information about the sensitivity and adaptive capacity of forests and rangelands to disturbances and changing disturbance regimes. We conclude with general management considerations for incorporating information on changing disturbance regimes into planning and actions that can increase resilience of forest and rangeland ecosystems to global change.

Key Findings

- ❖ The annual area of fire in forests and rangelands has increased since 1984, and the average annual area burned from 2000 to 2017 was more than double the pre-2000 average.
- ❖ The two western RPA regions have generally had higher exposure to fire and drought than the eastern regions, as well as the greatest rates of tree mortality caused by insects and diseases. In contrast, forests in the RPA South Region have experienced the highest rates of removals.
- ❖ The highest rates of invasion by nonnative plants occur near agricultural or developed land uses, primarily in forests in the RPA South Region and portions of the North Region, as well as rangelands in the Pacific Coast Region.
- ❖ Fire-caused tree mortality in forests is expected to increase by 2070. The highest rates of fire mortality are expected if climate follows the hot or dry climate futures under any of the high warming RPA scenarios.
- ❖ Drought exposure for forests and rangelands is expected to increase by 2070, and forest and rangeland ecosystems in the Southwest are expected to experience the most substantial increases.

A disturbance can be defined as an event that changes environmental conditions within an ecosystem. Disturbances combine with other biotic, abiotic, and biophysical factors to affect forests, rangelands, and the services and resources derived from those ecosystems (Kelly et al. 2020, Seidl et al. 2016). As climate, other biophysical factors, and management regimes change, disturbance regimes are being altered (Bowman et al. 2020, Donovan et al. 2017, Pureswaran et al. 2018, Sommerfeld et al. 2018), with the possibility of some disturbance types becoming more frequent, severe, or longer in duration (Cook et al. 2015, Dale et al. 2001, Seidl et al. 2017). At the same time, some disturbance types have become less frequent in certain ecosystems (for example, Nowacki and Abrams 2014, Steel et al. 2015). These alterations to disturbance regimes have the potential to drive changes in the distribution, structure, species composition, or function of forest and rangeland ecosystems, putting those ecosystems at risk and presenting challenges for management (Anderson-Teixeira et al. 2013, Clark et al. 2016, Coop et al. 2020, Vose et al. 2018). There is mounting evidence that management actions such as thinning or prescribed fire may play key roles in mitigating or ameliorating the impacts of disturbances like drought in some ecosystems (Bradford and Bell 2017, Knapp et al. 2021, Krofcheck et al. 2018, Vose et al. 2019). Identifying trends in, and attributing causes of disturbances on forests and rangelands enables examination of effects on forest and rangeland resources and can inform regional and national management and policy. In this chapter we summarize trends within the conterminous United States (except where otherwise stated) and within RPA regions (figure 5-1). The time periods for summaries of recent past trends vary by disturbance agent, but most include data beginning in at least the 1990s, while future projections are for the period 2020 to 2070.

Figure 5-1. Distribution of forest land and rangeland in the four RPA regions.



Sources: The distribution of forest land is from Brooks et al. forest land use map (see Land Resources Chapter); the distribution of rangeland is from Reeves and Mitchell (2011).

Fire in Forests and Rangelands

- ❖ The annual area of large fires has increased in both forests and rangelands over the 1984 to 2017 period. The average annual area burned by large wildfires since 2000 is more than double the pre-2000 average.
- ❖ In forests, prescribed fires conducted for management have been most prevalent in the South Region.
- ❖ Increases in the volume of trees killed by fire in forests are expected by 2070, with the greatest increases associated with the hot and dry climate futures under the higher warming scenarios.
- ❖ In forests, increases in the annual area of moderate-severity fires are expected in all RPA regions by 2070 under all RPA scenarios. In the Pacific Coast and South Regions, the area of high-severity fires is also expected to increase, while in the Rocky Mountain and North Regions, the area of high-severity fires is projected to either increase or decrease, depending on the warming scenario.
- ❖ Extreme droughts lead to increased wildfire activity in rangelands where annual vegetation production is consistently high. Where average productivity is low but interannual variability in productivity is high, increased wildfire activity occurs following wet periods.

Forests

Fire is a dominant disturbance agent in many types of forests in terms of area affected, the extent of tree damage and mortality, and resulting effects on forest resources and ecosystem services (Pausas and Keeley 2019, Thom and Seidl 2016). At the same time, fire is a natural and integral feature of forest ecosystems, many of which are adapted to particular regimes of fire frequency, intensity, severity, and seasonality (Greenberg and Collins 2021). Beginning in the first half of the 20th century and until the 1950s, the average annual area of forest burned by all fires in the United States decreased, although year-to-year variability in burned area remained (Littell et al. 2009, Parisien et al. 2016, van Wagtendonk 2007). This decrease in average burned area disrupted natural fire regimes in many parts of the country, leading to accumulation of potential fire fuels and leaving some forest ecosystems vulnerable to larger and more severe future fires (Abatzoglou et al. 2017, Calkin et al. 2015, Parisien et al. 2016). The expansion in many forested regions of the wildland-urban interface (WUI), where human development and natural lands meet or intermix, has increased chances of human-caused fire ignitions and resulted in greater economic impacts (e.g., property damage and loss) and loss of human life (Calkin et al. 2014, Radeloff et al. 2018) (see the Land Resources Chapter). A warming climate is expected to magnify wildfire activity, including more extreme wildfire events as droughts become more likely (Abatzoglou and Williams 2016, Barbero et al. 2015, Littell et al. 2016).

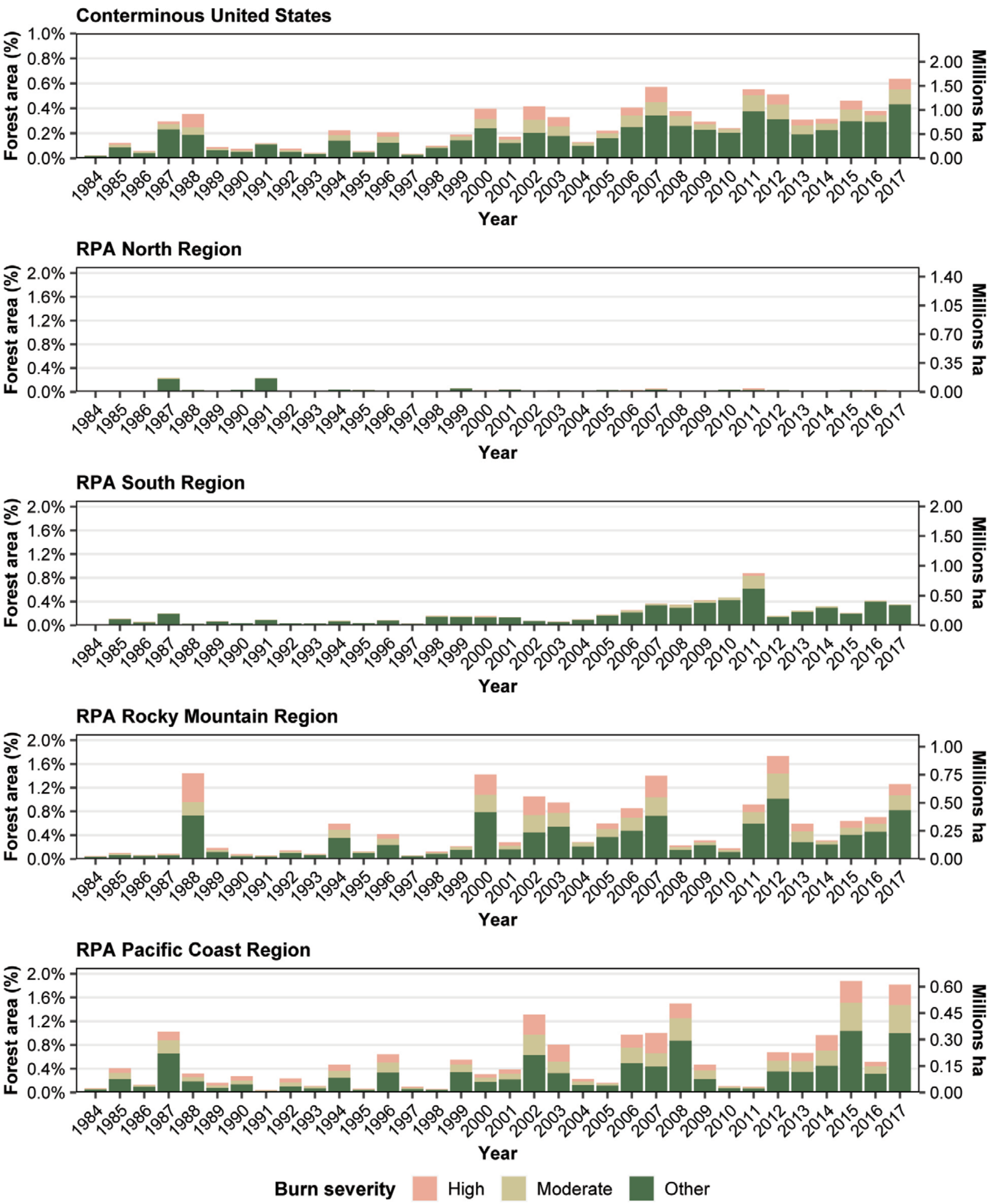
Trends in total forest area burned by large fires (defined as fires at least 405 ha in size in the Western United States and 202 ha in the East) and burn severity show notable differences over time and by region (figure 5-2). Across the conterminous United States, the annual forest area burned by large fires has shown an increasing trend. Between 1984 and 2000, burned forest area in the United States averaged 334,000 ha per year (about 0.13 percent of total forest area). Since 2000, the burned forest area averaged 965,000 ha per year (about 0.37 percent of total forest area), representing a 189-percent increase, or nearly triple the pre-2000 average. This same trend is seen at the regional scale, except for the RPA North Region, but burned area also varies widely for each region from year to year. Over the entire time period, the greatest area of large fires occurred in the two western RPA regions (Pacific Coast and Rocky Mountain). Since 2000, burned area averaged 259,000 ha per year in the Pacific Coast and 403,000 ha per year in the Rocky Mountain Region, representing increases of 165 percent and 219 percent over the pre-2000 average, respectively. Those two regions also had the greatest areas of moderate- and high-severity fires in all years. The RPA South Region experienced a 271-percent increase, to an average of 286,000 ha per year burned since 2000—a larger proportional increase than the two western regions—however moderate- and high-severity fires were rare. In contrast, there has been relatively little large-fire activity on forest lands in the North Region during the period of record. Many of the fires in the North Region are relatively small prescribed fires conducted by management agencies and thus not included here (see the following paragraphs). On average, the area of high-severity fires has increased across the United States since 2000, with 141,000 ha of high-severity fires burning annually since 2000, compared with 48,000 ha annually prior to 2000. The share of the total area of large fires classified as high severity remained approximately unchanged between the two periods, averaging 14.4 percent prior to 2000 and 14.6 percent since 2000. This increase in area but not in proportion of the total corroborates other assessments (e.g., Vose et al. 2018).

Since 2017, the United States, and especially the Rocky Mountain and Pacific Coast Regions, have set several records for areas burned. In 2020, more than 4.1 million ha burned on all lands (not just forest) in the United States—the largest burned area in a single year, and most of that area occurred in the Rocky Mountain and Pacific Coast Regions (Hoover and Hanson 2021). The large burned area in 2020 has been linked to dry atmospheric conditions and a higher vapor pressure deficit, which led to drier fuels that could ignite more easily; climate change was a substantial contributor to those conditions (Higuera and Abatzoglou 2020).

While the large fires summarized above can include some prescribed fires, many prescribed fires are smaller in extent than the cutoff for large fires, and are thus largely excluded from the analysis of large fires (Nowell et al. 2018). In addition, prescribed fires conducted by State agencies are explicitly excluded from the large-fire dataset (Picotte et al. 2020). Prescribed fire is the practice of using fire for management purposes, including maintaining or restoring ecological conditions, helping forests adapt to changing biophysical and climatic conditions, and reducing the risk of wildfires in fire-prone forests (Hunter and Robles 2020, Krofcheck et al. 2018, Ryan et al. 2013). In some forest ecosystems, it is therefore the absence of fire, rather than prescribed fire, that disrupts an ecosystem's dynamics and can be considered a "disturbance" to the ecosystem (e.g., Fill et al. 2015). Because prescribed fires are important to the dynamics of forests across the country, we summarize prescribed fire use by region to complement the summary of large-fire areas.

Nationally consistent, comprehensive data on the locations and severities of prescribed fires in forests are difficult to obtain (Nowell et al. 2018; but see Hawbaker et al. 2017, 2020). However, results from a recent State survey on prescribed burning activities show that approximately 3.68 million ha of prescribed fires were conducted in 2017 for forestry objectives nationwide (Melvin 2018). The treated area increased slightly from 3.37 million ha in the original 2011 survey conducted, and continued to increase in 2018 and 2019 (Melvin 2021). Most of the 2017 area (2.35 million ha, 64 percent of the total) occurred in the RPA South Region (Melvin 2018), supporting other recent studies that highlighted the general importance and widespread nature of prescribed burning in forests in the Southeastern United States (Mitchell et al. 2014, Nowell et al. 2018). See the sidebar COVID-19 as a Constraint on Prescribed Burning in the Southeastern United States for discussion of some recent challenges in applying prescribed fire in the Southeast. Importantly, the areas reportedly treated by prescribed fire exceed the area of forest affected by large wildfires in any single year of the wildfire data summarized here, for the country as a whole and for the South Region.

Figure 5-2. Percent and area of forest burned by large fires (at least 405 ha in the Western United States and 202 ha in the Eastern United States) over time by burn severity category. The "other" category combines the severity categories of underburned to low severity, low severity, and increased post-fire greenness/vegetation response.



ha = hectares.

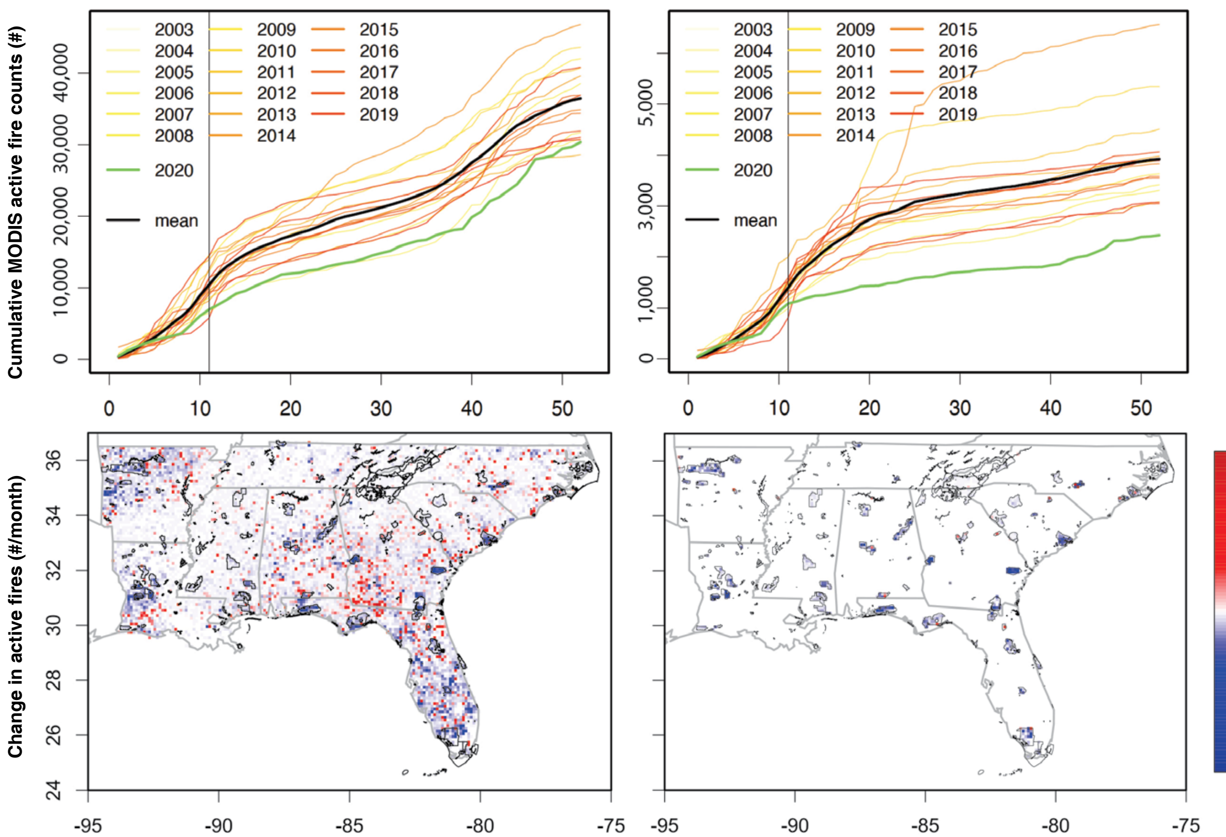
Source: Monitoring Trends in Burn Severity (MTBS, Eidenshink et al. 2007, Picotte et al. 2020).

COVID-19 as a Constraint on Prescribed Burning in the Southeastern United States

Prescribed fire is an essential management tool for many land management objectives and across a wide diversity of Southeastern ecosystems. There are diverse impediments to applying fire in the Southeast, including smoke management, limited resources, and public approval (Kobziar et al. 2015). Beginning in March 2020, the COVID-19 pandemic led to stay-at-home and shutdown orders across the world. Almost immediately, hypotheses emerged on how COVID-19 would affect all components of the Earth system (Diffenbaugh et al. 2020). To begin to determine the effects of COVID-19 on managed fire in the Southeast, we examined the record of active fires—that is, fires that were detected when NASA satellites passed overhead.

A decline in active fires was immediately observed as Federal and State agencies and private landowners adapted to work-from-home orders (Figure 5-3, Poulter et al. 2021). Following an exceptionally wet February, active fires increased for the first half of March, but then declined abruptly in mid-March and for the remainder of 2020. In some cases, land managers halted prescribed fire programs to avoid creating smoke conditions that might exacerbate health problems. In other cases, fire crews were unable to work because of COVID-19 safety regulations, or because of staff shortages as crew members were infected (Cahan 2020). In summer and fall 2020, a notable shift in the seasonal timing of prescribed fire application on all lands

Figure 5-3. Active fires detected by satellites in the Southeastern United States. The top two panels show cumulative weekly active fire counts by year (2003 to 2020) for all lands (left) and Federal lands only (right). The bottom two panels show the change in the number of active fires in April 2020 compared with the 18-year average for all lands (left) and Federal lands only (right), with fewer fires than average in blue and more fires than average in red. In the top panels, the vertical black line indicates March 15, the approximate date of COVID-19 stay-at-home orders in 2020. In the bottom panels, black outlines indicate Federal lands, which are those owned by the U.S. Departments of Interior, Defense, or Agriculture. Active fires are defined as places where a fire was burning when a satellite passed overhead.



Source: MODIS instrument on the NASA AQUA and TERRA satellites.

occurred in response to COVID-19, with increases in late-year burning to compensate for lost burned-acreage during the spring. By the end of 2020, the number of active fires was 21 percent below the 20-year average for all private and public lands, and 41 percent below the 20-year average for federally owned lands. This large reduction and seasonal shift in active fire detected in the satellite record was confirmed to come from a reduction in managed fires based on the Integrated Interagency Fuels Treatment Database (IIFT, https://iftdss.firenet.gov/).

The reduction in managed fires in 2020 follows a decline in early 2019 when the Federal government was shut down. Thus, the challenges in conducting burning due to COVID-19 added to an already expanding backlog of prescribed fire acreage in the Southeast as COVID-19 continued into 2021. In the near term, ecosystems and plant and animal species that are linked to frequent fire (including federally listed species) may suffer from the reduced habitat quality caused by reduced fire extent. Wildfire hazard reduction efforts on these lands have also been stalled, potentially exacerbating future wildfire threats. Moving forward, managers face the challenge of "catching up" on the backlog while confronting the need to maintain species, broader ecosystem processes, and fire hazard reduction targets across the region.

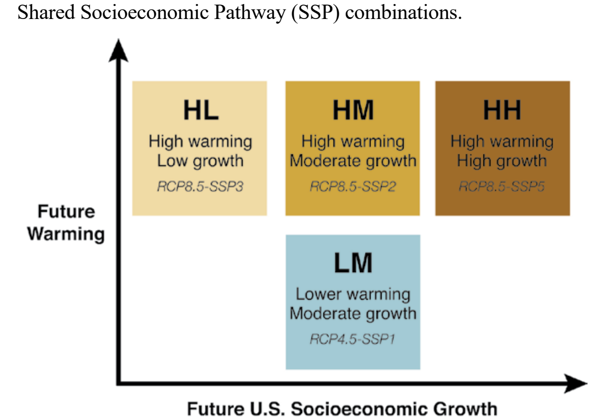
Future trends in volumes of tree mortality from wildfires were summarized from RPA Forest Dynamics Model results (see the Forest Resources Chapter) for the RPA scenarios (see the sidebar RPA Scenarios). The Forest Dynamics Model projects the future forest inventory, including volumes and areas of forest by RPA region and forest type group, forward in time for the 20 RPA scenario-climate futures (four RPA scenarios, five climate projections). A submodel projects the future fire occurrence and tree mortality resulting from fire based on Forest Inventory and Analysis (FIA) data and links to other submodels that modify forest characteristics over time, including basal area, down woody material that can act as fuels, stand age, species composition, and harvest probability. Because of the limited ability of FIA field crews to detect low-severity fires, fires that do not lead to tree mortality are omitted from the Forest Dynamics Model. Thus, the projections can be used to examine changes in annual mortality volume from fire and changes in areas burned by moderate- and high-severity fires, but they do not provide estimates of total burned areas. More information about the Forest Dynamics Model can be found in the Forest Resources Chapter and in Coulston et al. (in preparation).

RPA Scenarios

The RPA Assessment uses a set of scenarios of coordinated future climate, population, and socioeconomic change to project resource availability and condition over the next 50 years. These scenarios provide a framework for objectively evaluating a plausible range of future resource outcomes.

The 2020 RPA Assessment draws from the global scenarios developed by the Intergovernmental Panel on Climate Change to examine the 2020 to 2070 time period (IPCC 2014). The RPA scenarios pair two alternative climate futures (Representative Concentration Pathways or RCPs) with four alternative socioeconomic futures (Shared Socioeconomic Pathways or SSPs) in the following combinations: RCP 4.5 and SSP1 (lower warming-moderate U.S. growth, LM), RCP 8.5 and SSP3 (high warming-low U.S. growth, HL), RCP 8.5 and SSP2 (high warming-moderate U.S. growth, HM), and RCP 8.5 and SSP5 (high warming-high U.S. growth, HH) (figure 5-4). The four 2020 RPA Assessment scenarios encompass the projected range of climate change from the RCPs and projected quantitative and qualitative range of socioeconomic change from the SSPs, resulting in four distinct futures that vary across a multitude of characteristics (figure 5-5), and providing a unifying framework that organizes the RPA Assessment natural

Figure 5-4. Characterization of the 2020 RPA Assessment scenarios in terms of future changes in atmospheric warming and U.S. socioeconomic growth. These characteristics are associated with the four underlying Representative Concentration Pathway (RCP) – Shared Socioeconomic Pathway (SSP) combinations.



Source: Langner et al. 2020.

Figure 5-5. Characteristics differentiating the 2020 RPA Assessment scenarios. These characteristics are associated with the four underlying Representative Concentration Pathway (RCP) – Shared Socioeconomic Pathway (SSP) combinations.

RPA Scenario (RCP-SSP)	Global Temperature Rise	U.S. Population Growth	U.S. Economic Growth Rate	Bioenergy Demand	Energy Sector Focus	Global Energy Usage	International Trade Openness
LM Lower warming Moderate growth *RCP4.5-SSP1*	Lower	Medium	Medium-High	High	Renewables	Low	Medium
HL High warming Low growth *RCP8.5-SSP3*	High	Low	Low	Low	Fossil fuels	Medium	Low
HM High warming Moderate growth *RCP8.5-SSP2*	High	Medium	Medium	Medium	Mixed	Medium	Medium
HH High warming High growth *RCP8.5-SSP5*	High	High	High	High	Fossil fuels	High	High

resource sector analyses around a consistent set of possible world views. The Scenarios Chapter describes how these scenarios were selected and paired; more details are provided in Langner et al. (2020).

The 2020 RPA Assessment pairs these four RPA scenarios with five different climate models that capture the wide range of projected future temperature and precipitation across the conterminous United States. An ensemble climate projection that averages across the multiple model projections is not used because of the importance of preserving individual model variability for resource modeling efforts. The five climate models selected by RPA represent least warm, hot, dry, wet, and middle-of-the-road climate futures for the conterminous United States (table 5-1); however, characteristics can vary at finer spatial scales. Although the same models were selected to develop climate projections for both lower and high-warming futures, there are distinct climate projections for each model associated with RCP 4.5 and RCP 8.5. The Scenarios Chapter describes how these climate models were selected. Joyce and Coulson (2020) give a more extensive explanation.

Throughout the RPA Assessment, individual scenario-climate futures are referred to by pairing RPA scenarios with selected climate projections. For example, an analysis run under “HL-wet” assumes a future with high atmospheric warming and low U.S. population and economic growth (HL RPA scenario), as well as a wetter climate for the conterminous United States (wet climate projection).

Table 5-1. Five climate models selected to reflect the range of the full set of 20 climate models in the year 2070. Each model was run under RCP 4.5 and RCP 8.5, providing a range of different U.S. climate projections.

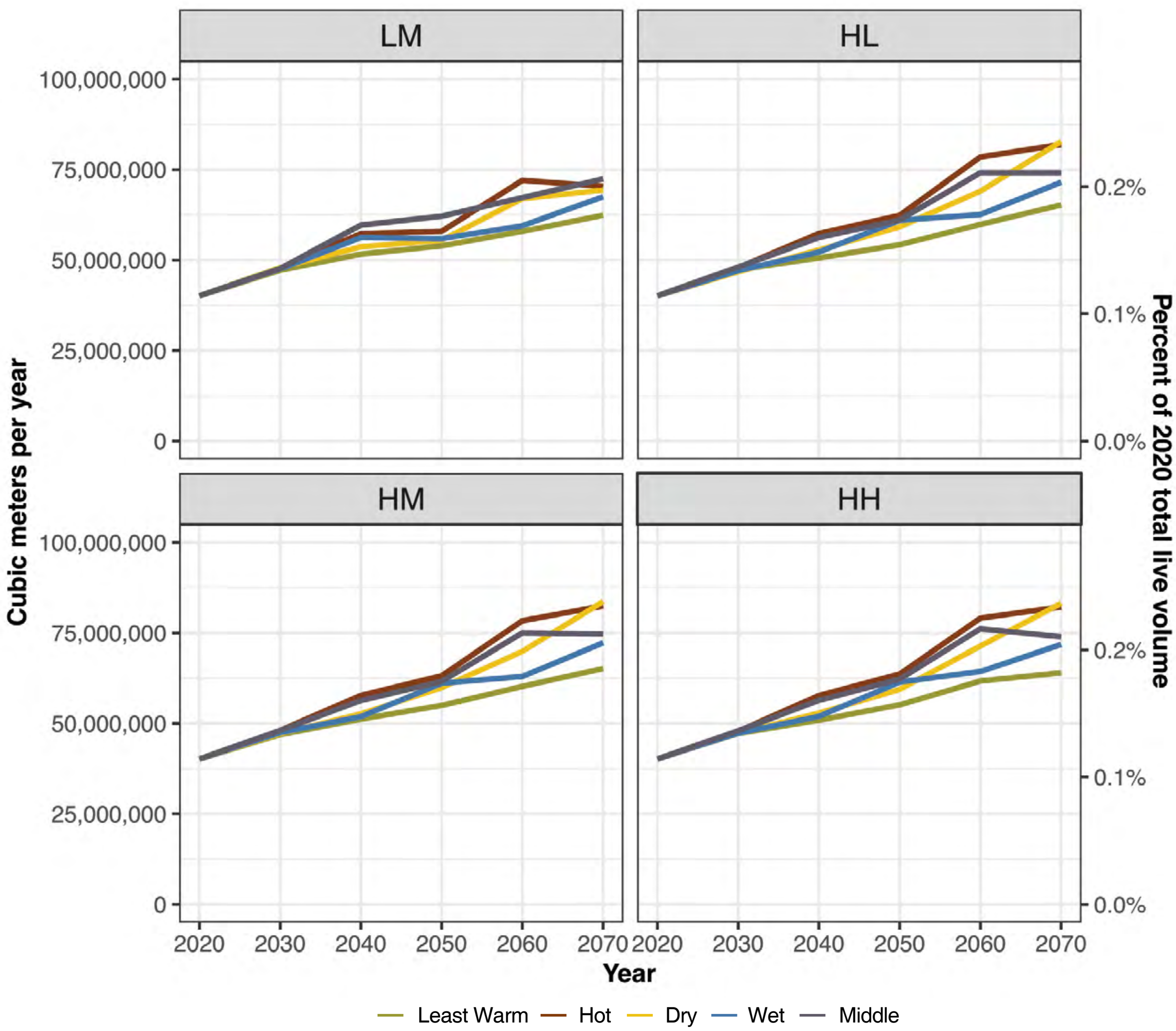
	Least warm	Hot	Dry	Wet	Middle
Climate model	MRI-CGCM3	HadGEM2-ES	IPSL-CM5A-MR	CNRM-CM5	NorESM1-M
Institution	Meteorological Research Institute, Japan	Met Office Hadley Centre, United Kingdom	Institut Pierre Simon Laplace, France	National Centre of Meteorological Research, France	Norwegian Climate Center, Norway

RCP = Representative Concentration Pathway.

Source: Joyce and Coulson 2020.

Annual fire mortality volume is projected to increase over time across the United States and in each RPA region under all 20 scenario-climate futures (figure 5-6)—from 40 million cubic meters in 2020 (0.10 percent of total live volume in all forests) to between 62 million cubic meters under LM-least warm (the LM scenario and least warm climate model) and 84 million cubic meters under HM-dry (the HM scenario and dry climate model) in 2070, representing an increase of between 55 and 108 percent relative to 2020 values. The result that all futures project the same directional change indicates relatively low uncertainty in the impact of future climate and socioeconomic change on fire mortality volume. Generally, the greatest increases in fire mortality volume by 2070 were projected for plausible futures that included the dry or hot climate projections under the three high-warming RPA scenarios (HL, HM, and HH). The smallest increases were projected for the least warm climate projection regardless of the RPA scenario. These projections generally agree with studies that point to expected increases in fire occurrence over much of the country, especially as climate becomes warmer and drier (Gao et al. 2021, Littell et al. 2016). While a substantial increase in fire mortality volume was projected, the combined average annual volume of removals for timber harvest in Florida, Georgia, North Carolina, and South Carolina totaled just over 86 million cubic meters in 2016 (Oswalt et al. 2019), slightly exceeding the most extreme fire mortality volume projection for the conterminous United States in 2070 (84 million cubic meters).

Figure 5-6. Projected annual fire mortality volume over time for all RPA scenarios. Results summarize output from Forest Dynamics Model simulations (see the Forest Resources Chapter for more details on the model). In each panel, the dark lines represent the median outcome of 100 simulations, and the shaded area represents the inter-quartile range of those simulations. The right-hand vertical axis shows the values in terms of percent of total live volume in 2020. Both vertical axes apply to all four panels. Because the total live volume of forests is expected to increase over time (see the Forest Resources Chapter), the volume killed by fire represents a lower proportion of the total volume in 2070 than is displayed.

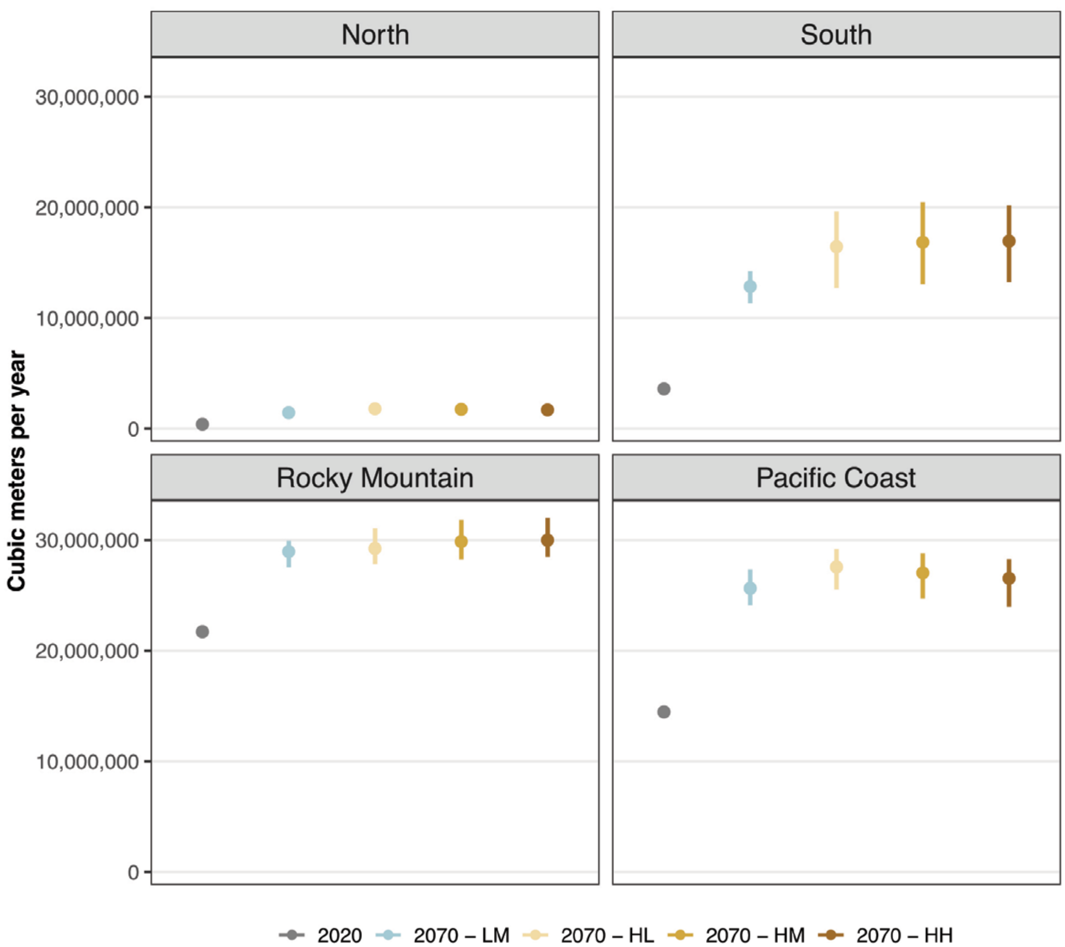


LM = lower warming-moderate U.S. growth; HL = high warming-low U.S. growth; HM = high warming-moderate U.S. growth; HH = high warming-high U.S. growth.

The expected trends in annual fire mortality volume within RPA regions mirror the nationwide trend, with increases projected in all regions. The relative magnitudes of increase differ by region, and the projected changes in forest and fire dynamics that result in increased volume differ slightly by region. In the Rocky Mountain Region, fire mortality volume is expected to increase between 20 and 55 percent, from 22 million cubic m in 2020 to between 26 and 34 million cubic m by 2070 (table 5-2, figure 5-7). In the Pacific Coast Region, annual fire mortality volume in 2020 was lower than in the Rocky Mountain Region, but is expected to increase to a level either slightly below or comparable to the Rocky Mountain Region by 2070—from approximately 14 million cubic m in 2020 to between 24 and 29 million cubic m in 2070, representing a 63- to 100-percent increase. In the South, while fire mortality volume is lower overall than in the two western regions currently and throughout the future period, an increase of 184 to 505 percent, to between 10 and 22 million cubic m, is projected by 2070. In the North Region, where there is very little fire activity, annual fire mortality volume is expected to increase as well, but remain lower than all three other regions. Increases to between 1.2 and 2.0 million cubic m are projected by 2070.

A projected increase in annual tree volume killed by fire in a region can be due to an increase in the area burned by fire, an increase in the proportion of live volume in burned forest stands that is killed by fire, or a combination of both factors. In the Rocky Mountain Region, the annual area of moderate-severity fires (between 30- and 70-percent mortality by volume) is expected to more than double from 2020 to 2070 (108- to 179-percent increase) (table 5-2), while projections

Figure 5-7. Annual fire mortality volume for RPA regions in 2020 and projected in 2070 for all RPA scenarios. Results summarize output from Forest Dynamics Model simulations (see the Forest Resources Chapter for more details on the model). For the values in 2070, dots represent the mean of the five RPA climate projections under each RPA scenario, while vertical bars indicate the range of values across those climate projections.



LM = lower warming-moderate U.S. growth; HL = high warming-low U.S. growth; HM = high warming-moderate U.S. growth; HH = high warming-high U.S. growth.

Table 5-2. Projected changes from 2020 to 2070 (value and percent change) in overall annual fire mortality volume, fire mortality volume as a percent of total volume in burned locations, and annual areas of moderate- and high-severity fires for each RPA region. Moderate-severity fires are defined as those that kill 30 to 70 percent of volume, while high-severity fires killed at least 70 percent of volume. The first column under each variable indicates the absolute change, and the second column indicates the percent change by 2070 over 2020 values.

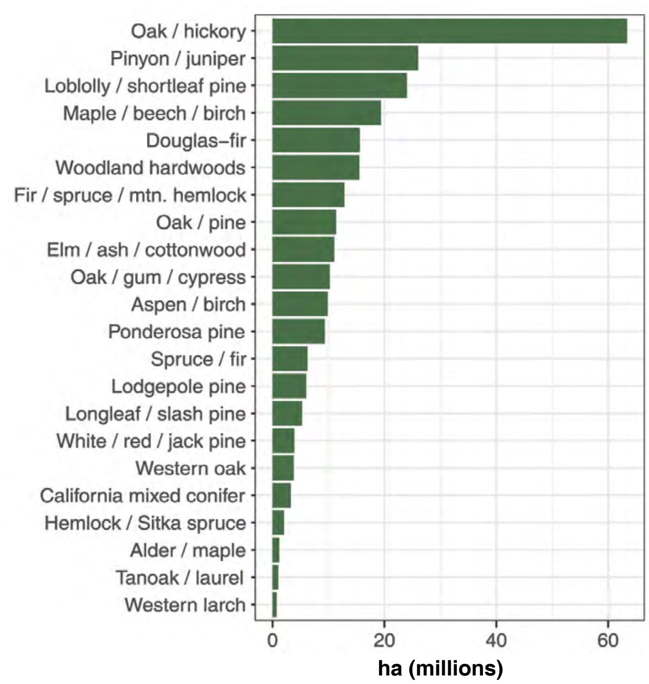
	Change in fire mortality volume		Change in area of moderate-severity fires		Change in area of high-severity fires		As percent of volume in burned locations	
	million m^3	percent	ha	percent	ha	percent	percentage points	percent
North	0.83-1.6	196-385	6,000-11,000	483-884	-1,300-4,800	-16-62	-3.4- -2.5	-19- -14
South	6.6-18.2	184-505	12,000-54,000	72-330	19,000-70,000	70-256	0.4-3.5	2-17
Rocky Mountain	4.4-12.0	20-55	46,000-76,000	108-179	-3,300-34,000	-2-24	-10.0- -7.1	-16- -12
Pacific Coast	9.1-14.4	63-100	40,000-53,000	141-185	36,000-49,000	69-95	2.9-3.9	6-8

ha = hectares; m^3 = cubic meters.

of high-severity fires (at least 70 percent mortality by volume) show either decreases or small increases in annual areas. In other words, under all scenarios, the annual area of moderate-severity fires in the Rocky Mountain Region is projected to increase more than the area of high-severity fires between 2020 and 2070. The overall average annual proportion of live volume killed by fire in locations that burned is expected to decrease 12 to 16 percent over that time in the region (table 5-2). In the Pacific Coast and South Regions, the projected annual areas of both moderate- and high-severity fires increase by 2070, along with the average proportion of volume killed (table 5-2). While few studies have examined projected trends in fire severity, most research has suggested the potential for higher fire severity as climate changes, including portions of the Western United States (Halofsky et al. 2020, Van Mantgem et al. 2016), and increases in the number of extreme fire events in portions of the South (Terando et al. 2017). That aligns with our results for the Pacific Coast and South, but our 2070 projection of either an increase or a decrease in area of high-severity fires for the Rocky Mountain Region highlights the uncertainty associated with projecting fire severity. Parks et al. (2016) modeled future fire severity for the Western States and projected the potential for lower fire severity for most of the West, including the Rocky Mountains, if vegetation changes occur that result in reduced fuels. However, future changes to fuel levels are highly uncertain and depend on many factors, including climate, forest productivity, management, and fire history.

Each RPA region is heterogeneous and contains forests characterized by more frequent, low-severity fires, as well as those characterized by less frequent, moderate- or high-severity fires (Greenberg and Collins 2021, Schoennagel et al. 2004). Understanding the projected dynamics of fire within each type of forest (figure 5-8) can provide insights into the potential effects of future fire on those forests. Most forest type groups are expected to have greater fire mortality volumes by 2070 compared with 2020, although the magnitude of increase is expected to vary by forest type group (figures 5-9, 5-10). Several of the western type groups that have high or moderate annual fire mortality volumes in 2020 are expected to experience large increases under all RPA scenarios, including Douglas-fir, ponderosa pine, woodland hardwoods, and pinyon/juniper, and the annual area of high-severity fires is also expected to increase in those groups (figure 5-9). The latter three of those groups each occur, at least in part, in relatively dry portions of the South Central and Southwestern United States, where dry conditions are expected to become more common in the future (see the section Drought in Forests

Figure 5-8. Area of forest for each forest type group in the FIA database, circa 2013. All analysis in this chapter that was based on FIA data excluded nonstocked, exotic, and tropical groups, and two others that were limited in extent: the western white pine and redwood type groups.

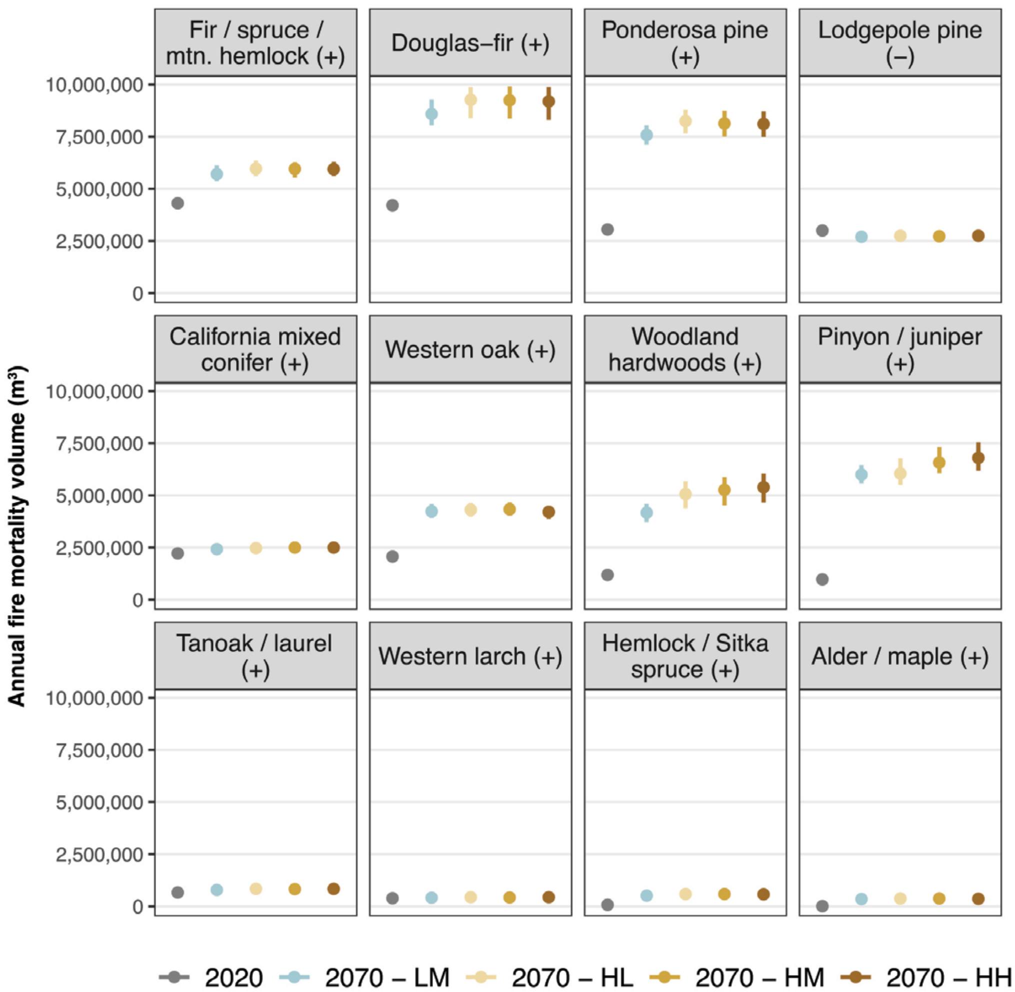


FIA = Forest Inventory and Analysis; ha = hectares.

and Rangelands). Much or all of the extents of those type groups are characterized by relatively low live volumes and frequent, low-severity fire regimes that kill few trees, but in many places those fire regimes have shifted toward higher-severity fires (Greenberg and Collins 2021). An increase in the area of high-severity fires could therefore further threaten those forest ecosystems. Douglas-fir forests are historically characterized by less frequent, higher severity fires, and the expected increase in fire mortality volume, along with increasing area of high-severity fires, could imply more frequent severe fires in that forest type. Lodgepole pine is one notable forest type group with lower projected fire mortality volume in 2070 than in 2020. The average annual area of high-severity fires in the lodgepole pine type group is also projected to decrease (figure 5-9), accounting for much of the decline in fire mortality volume.

Forest type groups found predominantly in the East are expected to see relatively modest changes in fire tree mortality volume (figure 5-10). One exception is the oak/hickory forest type group, whose fire mortality volume is projected to at least double by 2070 and whose annual

Figure 5-9. Annual fire mortality volume for western forest type groups in 2020 and projected in 2070 for all RPA scenarios. Results summarize output from RPA Forest Dynamics Model simulations (see the Forest Resources Chapter for more details on the model). For the values in 2070, dots represent the mean of the five RPA climate projections under each RPA scenario, while vertical bars indicate the range of values across those climate projections. Forest type groups are arranged according to their 2020 observed annual fire mortality volume (highest at the top left to lowest at the bottom right). Pluses and minuses in parentheses after each forest type group name indicate an increase (+) or decrease (-) in annual area of high-severity fire projected by 2070, defined as fires that result in at least 70 percent of live volume killed, or whether an increase was projected for some futures and a decrease was projected for others (-/+).

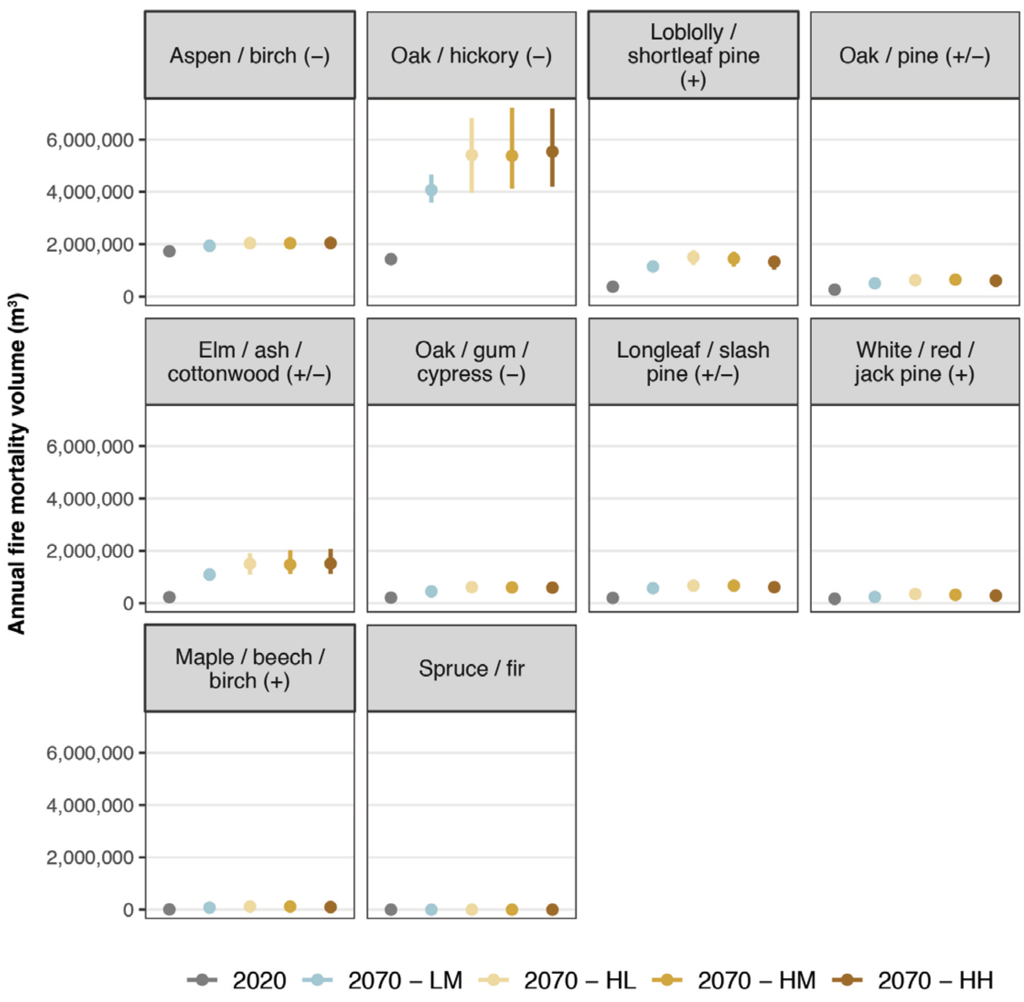


LM = lower warming-moderate U.S. growth; HL = high warming-low U.S. growth; HM = high warming-moderate U.S. growth; HH = high warming-high U.S. growth.

area of high-severity fires is projected to increase in all futures. Oak/hickory forests, like many forest types in the Eastern United States, have been experiencing reduced frequency and increased severity of fire relative to historical conditions, when fires burned frequently and resulted in low tree mortality (Nowacki and Abrams 2008). As a result, oak/hickory forests have recently declined. While the specific local ecological effects of fire depend on many factors, an increase in fire mortality volume could be beneficial to oak/hickory forests in the East if it signals more fire overall in that forest type. However, an increase in the area of high-severity fires could further alter the oak/hickory forest ecosystems.

The projected changes in fire mortality volumes of trees provide some insights into the changing dynamics of fire in U.S. forests. In addition to direct effects on forests themselves, increases in fire mortality volume and high-severity fires also have implications for human health and property in the wildland-urban interface (WUI) and

Figure 5-10. Annual fire mortality volume for eastern forest type groups in 2020, and projected in 2070 for all RPA scenarios. Results summarize output from RPA Forest Dynamics Model simulations (see the Forest Resources Chapter for more details on the model). For the values in 2070, dots represent the mean of the five RPA climate projections under each RPA scenario, while vertical bars indicate the range of values across those climate projections. Forest type groups are arranged according to their 2020 observed annual fire mortality volume (highest at the top left to lowest at the bottom right). Pluses and minuses in parentheses after each forest type group name indicate an increase (+) or decrease (-) in annual area of high-severity fire projected by 2070, defined as fires that result in at least 70 percent of live volume killed, or whether an increase was projected for some futures and a decrease was projected for others (-/+). The spruce/fir and longleaf/slash pine forest type groups had no high-severity fire projected in 2020 or 2070.



LM = lower warming-moderate U.S. growth; HL = high warming-low U.S. growth; HM = high warming-moderate U.S. growth; HH = high warming-high U.S. growth.

beyond. Expansion of the WUI and increasing fire activity are already contributing to loss of human life and property from fire, presenting challenges for fire suppression and increasing costs associated with suppression (Abt et al. 2009, Radeloff et al. 2018). The increases in high-severity fires projected in most regions and forest types could add to those already-substantial challenges and costs of fire management. Any substantial increase in fuel treatments, such as thinning or prescribed burning, across large landscapes or regions could result in reduced fire severity and reduced risk of large, difficult-to-manage fires in some forests. Forest types such as ponderosa pine forests, which are adapted to frequent, low-severity fires and have experienced a build-up of fuels resulting from fire suppression, could especially benefit from such treatments (Halofsky et al. 2020, Moritz et al. 2014, Schoennagel et al. 2004). Furthermore, these projections do not include any changes to fire ignitions, such as increased numbers of human-caused ignitions during periods with high fire hazard (Balch et al. 2017, Fusco et al. 2016) that could occur in the future. Additional ignitions could increase fire occurrence and severity in some forest ecosystems (Pausas and Keeley 2021). Further work could incorporate increased treatment levels or changes in ignitions and fuel availability into the RPA Forest Dynamics Model and examine the effects of those on projected fire mortality volume and fire severity.

Rangelands

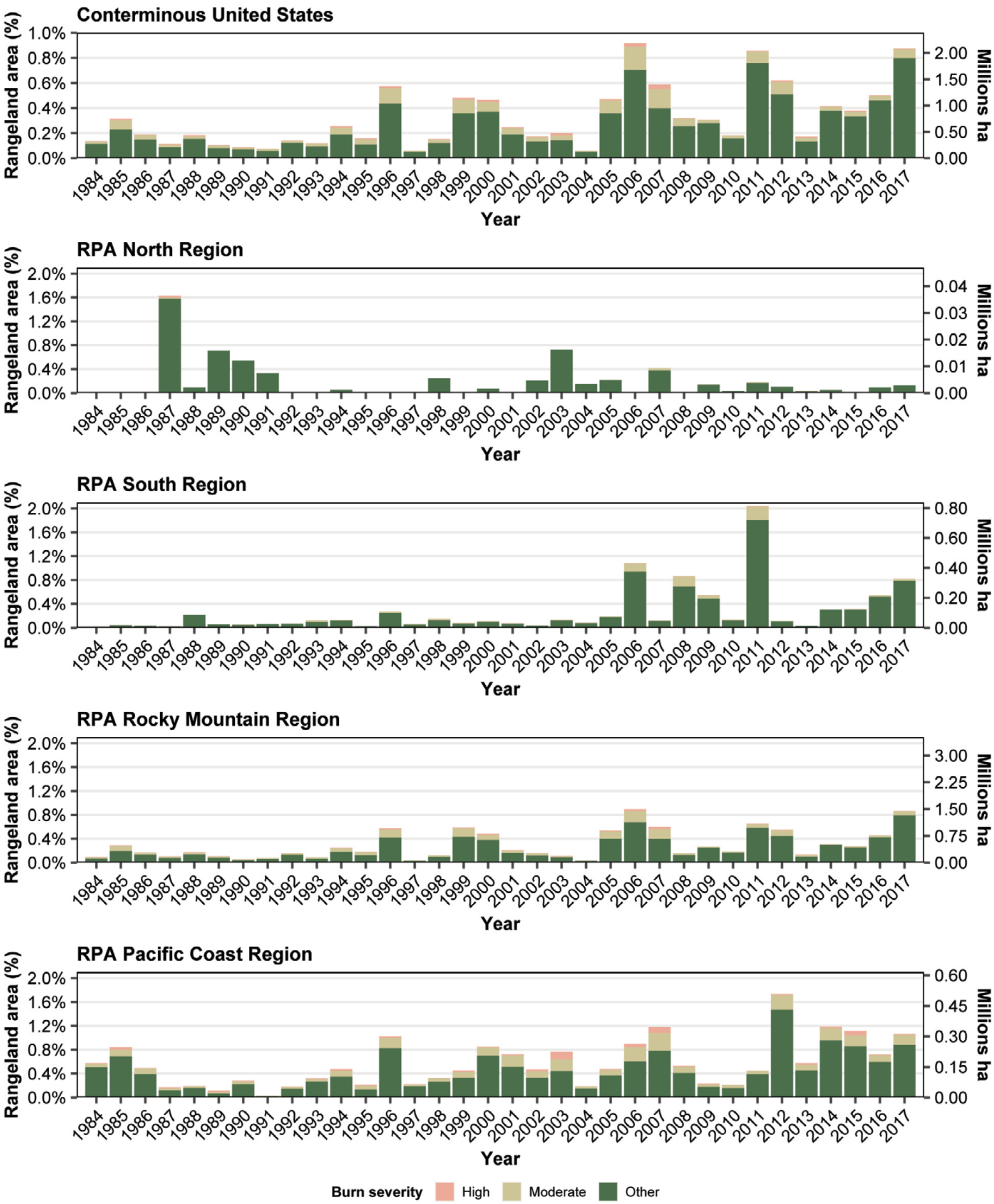
Fire plays an important role in maintaining vegetation and ensuring forage for livestock in rangelands (Fuhlendorf et al. 2012, Limb et al. 2016). While fires are part of the natural dynamics of rangelands, invasive grasses and drought have led to more frequent and larger fires in some rangeland systems (Abatzoglou and Kolden 2011, Coates et al. 2016). An analysis of the rangeland areas burned by large wildfires (again defined as fires at least 405 ha in size in the Western United States and 202 ha in the Eastern United States) indicates an increase in burned rangeland area from 1984 to 2017, distributed asymmetrically across the RPA regions. Before 2000, burned area averaged about 470,000 ha per year (figure 5-11; about 0.19 percent of rangeland area). Since 2000, the total rangeland area burned per year increased substantially to an average of about 1 million ha per year (about 0.45 percent of rangeland area), an increase of 119 percent over the pre-2000 average. The 2006 fire season produced the highest annual area burned at 2.3 million ha (about 0.9 percent of the rangeland area). The RPA Rocky Mountain Region had the highest average annual rangeland area burned since 2000 (approximately 638,000 ha per year), followed by the Pacific Coast Region (218,000 ha burned per year). In both regions the average areas burned increased 100 percent over the pre-2000 averages. In the South Region, average area burned increased over 300 percent from the pre-2000 amounts to 168,000 ha per year, and the 2011 fire season produced the largest burned area in the record for the region, with over 800,000 ha burned that year (over 2.0 percent of the South's rangeland area). Only the North Region, which has a relatively small amount of rangeland area (approximately 6.1 million ha), exhibited a decreasing trend in the area burned per year.

The national and regional nature of this analysis obscures the fine-scale patterns of wildfires occurring in rangelands. The relationships between climate, fuels, and fire in rangeland ecosystems are complex. The annual area burned is linked with drought patterns, but the relationship is not linear, is sometimes counterintuitive, varies by ecosystem, and fires can occur months after drought has occurred (Krawchuk and Moritz 2011). Droughts can lead to larger fires and a greater number of fires, but only if sufficient fuels are present (Abatzoglou and Kolden 2013, Littell et al. 2018). Some rangeland areas consistently have high levels of vegetation productivity (figure 5-12) and thus fuels are consistently available. In those areas during drought years, relatively continuous fuels combined with low fuel moisture lead to extreme fire behavior and large areas burned. For example, in Texas and Oklahoma where annual vegetation production is consistently high, widespread extreme droughts occurred in 2011, contributing to the large rangeland area burned in the RPA South Region that year (figure 5-11; also see the section Drought in Forests and Rangelands).

In the northern Great Plains in 2011, fire activity was relatively low because of comparatively cool and mesic conditions. This low fire activity contributed to a moderate area burned in the Rocky Mountain Region in 2011 (figure 5-11). The high precipitation and high resultant annual production of 2011, however, led to large amounts of standing dead material by the end of the year. When drought occurred in the region the next year (2012), this high amount of standing dead material increased ignition potential and fire behavior (Reeves et al. 2020). While the total area burned in the Rocky Mountain Region during 2012 in our analysis is lower than for other years (figure 5-11), some of the largest individual wildfires on record occurred in 2012 during record-setting heat and drought (e.g., the Ash Creek Complex in Montana) (Karl et al. 2012, Reeves et al. 2020).

In contrast to the Great Plains, much of the rangeland area of the Western United States typically has relatively low production, which leads to small amounts of fuel available in an average year. However, some areas with relatively low production on average tend to exhibit the greatest interannual variability in production, and thus high variability in fuels, especially fine fuels less than 6.35 mm in diameter (e.g.,

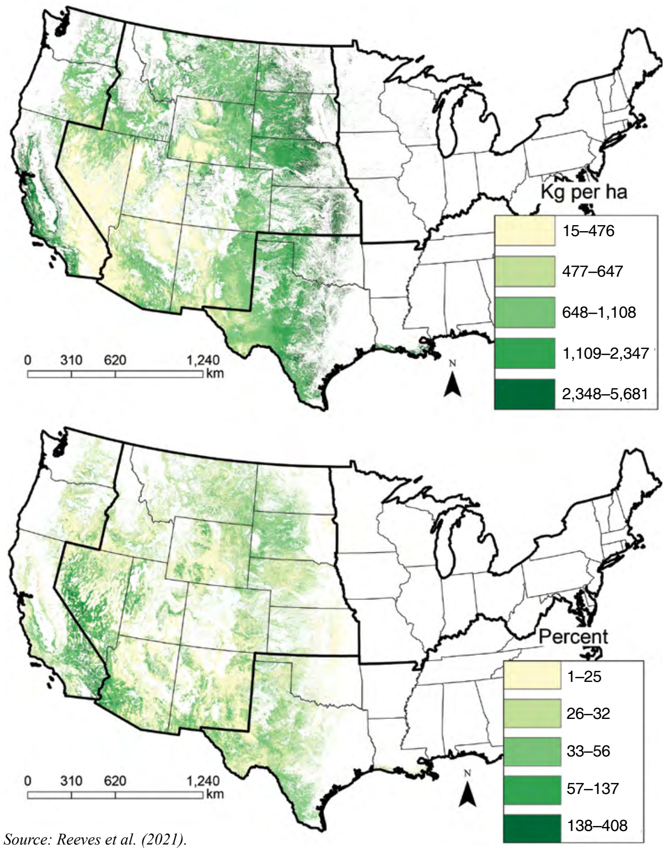
Figure 5-11. Percent and area of rangelands burned by large fires (at least 405 ha in the Western United States and 202 ha in the Eastern United States) over time by burn severity category. The "other" category combines the severity categories of underburned to low severity, low severity, and increased post-fire greenness/ vegetation response.



ha = hectares.

Source: Monitoring Trends in Burn Severity (MTBS, Eidenshink et al. 2007, Picotte et al. 2020).

Figure 5-12. Average annual production (top) and average interannual variability (bottom) in U.S. rangelands from 1984 to 2020.



Source: Reeves et al. (2021).

grasses and forbs). These areas are subject to heat and dryness in most years. The ecosystems that meet these criteria, including the Sonoran and Mojave Deserts (figure 5-12), can experience substantial areas burned in some years when annual production exceeds normal.

The complex relationships between climate, fuels, and fire in rangeland ecosystems ensure a complex future of fire in those systems. While we do not include fire projections for rangelands here, existing literature and knowledge of these relationships allow some general statements about possible future fire trends. Areas toward the eastern edge of the rangeland domain that produce fuels continuously but typically have surplus moisture may have larger annual burned areas, as dry conditions become more common (Littell et al. 2018). On the other hand, areas that are fuel-limited and require wet years to produce fire, are more likely to have variation in fire activity from year to year because interannual variability of herbaceous vegetation production is expected to increase in the future (Klemm et al. 2020, Reeves et al. 2017).

Drought in Forests and Rangelands

- ❖ Forests in the RPA Pacific Coast Region have had higher exposure to drought than other regions since 2005.
- ❖ Rangelands in the RPA Pacific Coast Region have similarly experienced high drought exposure since 2005, and rangeland exposure was also high in the South and Rocky Mountain Regions from 2011 to 2012.
- ❖ Forest and rangeland exposure to drought is expected to intensify over this century, particularly if the climate tends toward the hot, dry, or middle climate futures.
- ❖ Forest and rangeland vegetation types in the Southwest are projected to have the greatest drought exposure in the future, specifically the pinyon/juniper woodlands forest type group, and the grassland and creosotebush desert scrub rangeland vegetation types.

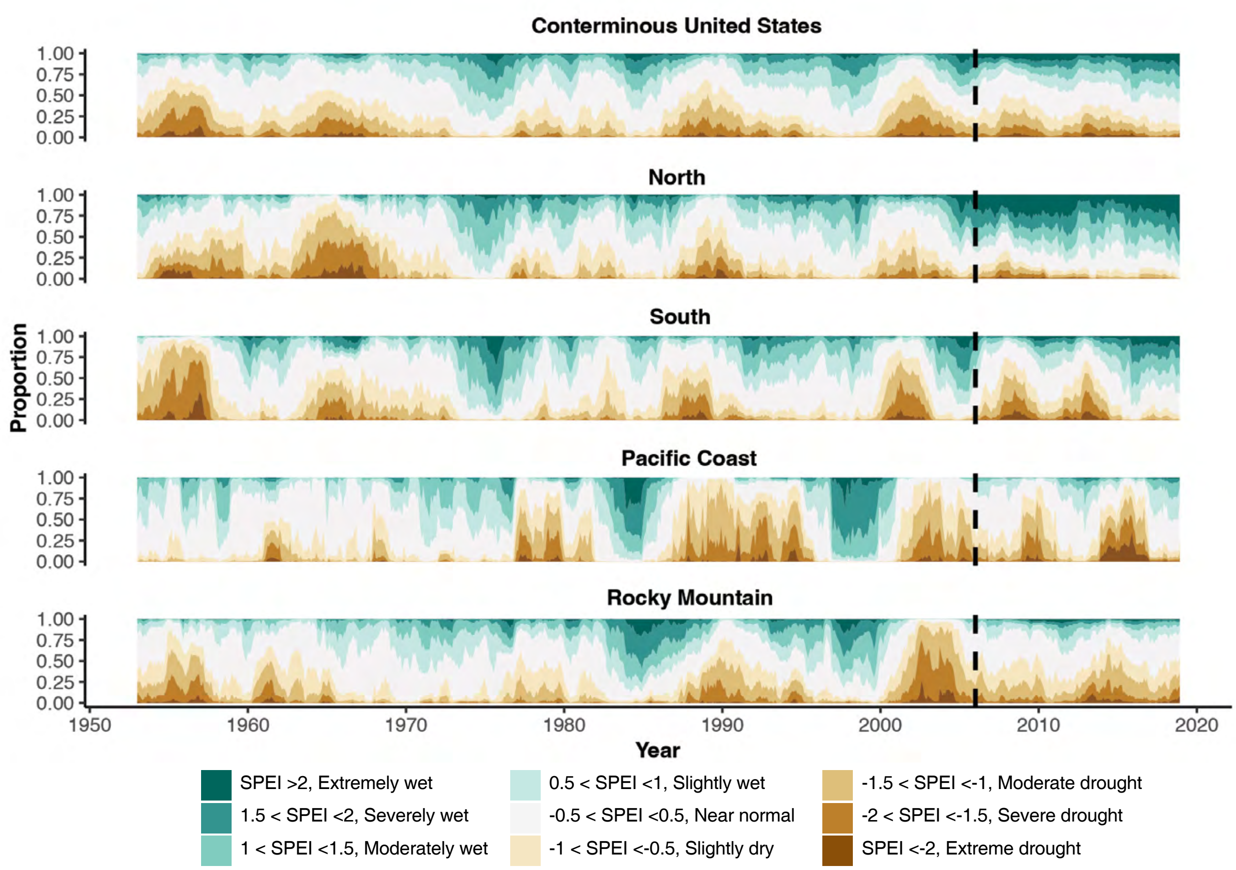
Forests

Drought, an important stressor affecting forests, is commonly defined as a period of moisture deficit resulting from below-average precipitation, high temperatures, or both (Clark et al. 2016). Alone or in combination with other disturbances, drought can reduce forest productivity, cause shifts in forest types, affect the ability of forests to regenerate, and diminish the capacity of forests to provide ecosystem services (Anderegg et al. 2013, Desprez-Loustau et al. 2006, Jactel et al. 2012, Peters et al. 2015, Trouet et al. 2010, Vose et al. 2016). As climate warms and many parts of the world become drier, droughts are expected to become more widespread, frequent, and severe (Ahmadalipour et al. 2017, Cook et al. 2015, Dai 2011, 2013, Prudhomme et al. 2014, Swain and Hayhoe 2015). While the effects of drought on trees and individual forest stands have been demonstrated for local areas, it is difficult to both measure moisture conditions in situ and determine the direct effects of drought on forests across broad geographic regions (Bennett et al. 2015, Clark et al. 2016, Gazol et al. 2018). Many scientists therefore use meteorological drought indices, which track relative departure from normal climate conditions and can be correlated with resulting effects on forests (Druckenbrod et al. 2019). Meteorological drought indices are distinct from other measures of drought, including hydrologic drought, which tracks reductions in water supply to rivers and lakes. Information on where and when forests have been exposed to meteorological drought in the past or are likely to be exposed in the future can be used to inform where management action or further research is warranted.

We use the Standardized Precipitation Evapotranspiration Index (SPEI) to summarize recent and future trends in drought exposure for forest land in the conterminous United States (Costanza et al. 2022a, 2022b; for details on SPEI, see Beguería et al. 2014, Vicente-Serrano et al. 2010). The SPEI allows for comparisons among locations for historical as well as future conditions, and can be computed over multiple time scales, making it useful for monitoring drought in different ecological contexts (Ault 2020, Slette et al. 2019, Vicente-Serrano et al. 2010). We used the 36-month SPEI, which assigns values for a given month by comparing the cumulative climatic water balance (precipitation minus potential evapotranspiration, or PET) for the previous 36 months to the same cumulative 36-month water balance for all months in a reference period (defined here as 1950 to 2005). Prolonged droughts that persist for multiple years are more likely to cause lasting impacts to forests than shorter-term droughts of equal magnitude (Berdanier and Clark 2016, Bigler et al. 2006, Guarín and Taylor 2005, Jenkins and Pallardy 1995, Millar et al. 2007). For most of the results shown here, PET was calculated using the standard method recommended by world organizations (Abatzoglou 2013, Allen et al. 1998). However, for summaries of observed SPEI (figure 5-13), calculation of PET via the preferred method was not possible because of data limitations, and we used an alternative method that has been recommended in such circumstances but may overestimate dry conditions in places with seasonally humid climate (Beguería et al. 2014, Hargreaves 1994).

The major trends in observed SPEI values (figure 5-13) corroborate known incidence of past drought, including drought periods in the 1950s across much of the RPA South and Rocky Mountain Regions (Andreadis et al. 2005, Heim 2017) and in the 1960s across much of the North Region (Barlow et al. 2001, Namias 1966). Since 2005, the Nation's forests have experienced relatively even proportions of dry and wet conditions, although regionally there has been more variation from year to year. For example, the Pacific Coast Region was exceptionally dry on forest lands during the mid-2010s, a period that has been shown to be drier than any

Figure 5-13. Proportion of forest land area in categories of observed 36-month SPEI over time, based on PRISM climate data, 1953 to 2018, for the United States and RPA regions. The period to the left of the dashed line in each graph indicates the reference period that was used to calibrate SPEI values.



SPEI = Standardized Precipitation Evapotranspiration Index.
Source: Costanza et al. 2022b.

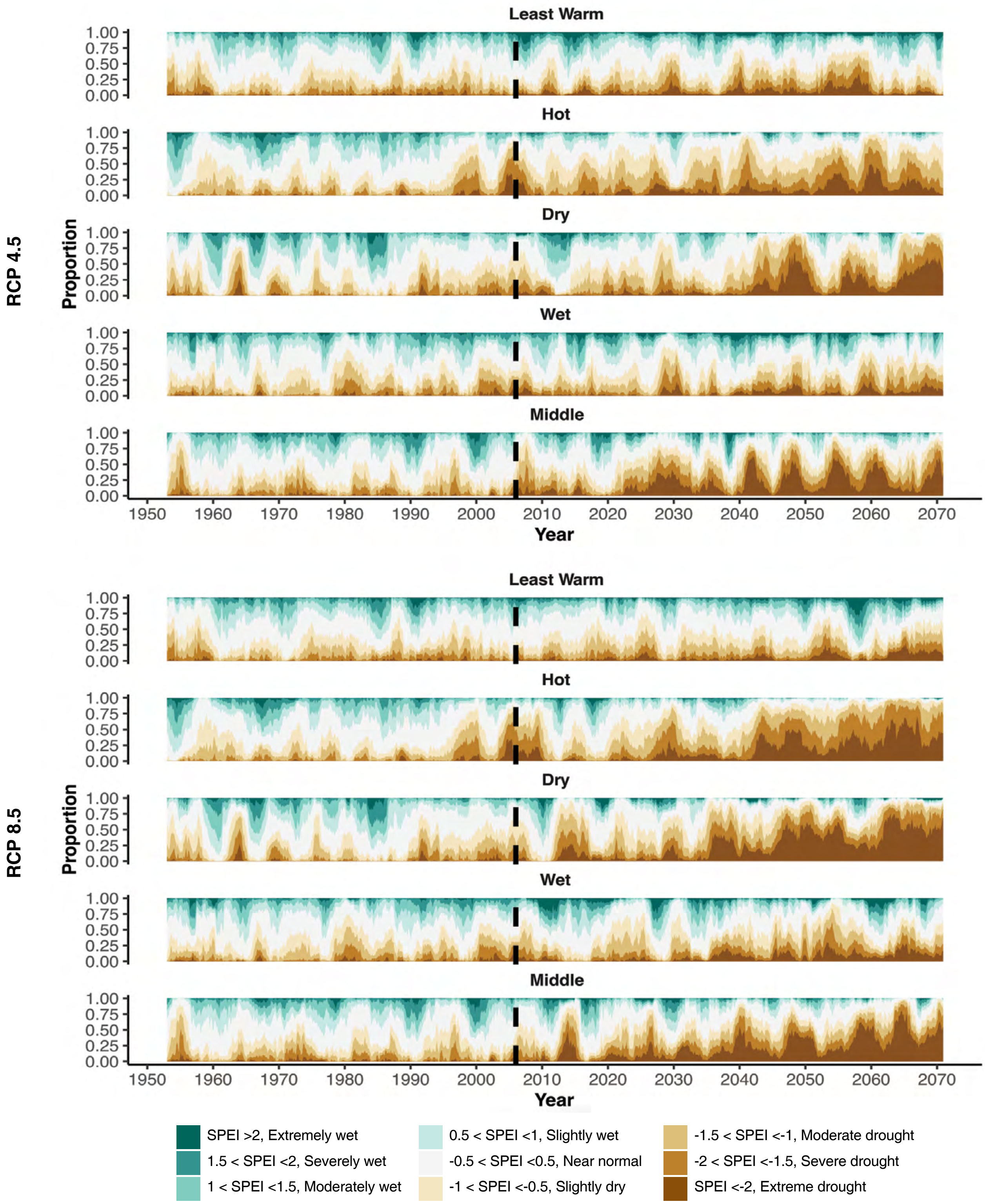
historical precedent in California (Robeson 2015), and which corresponded with high wildfire activity and insect outbreaks in the region (Fettig et al. 2019, Halofsky et al. 2020, Marlier et al. 2017, Pile et al. 2019). In contrast, the North Region was relatively wet nearly every month since 2005 (figure 5-13). Obscured in these regional trends are localized drought events that were smaller in geographic extent but had substantial forest impacts, including high rates of tree mortality and growth declines (see the sidebar Vulnerability to Drought for an example).

Forest SPEI projections provide an outlook on forest drought exposure under 10 different plausible climate futures across the United States. The integrated RPA scenarios were not used for these projections due to an inability to apply the socioeconomic factors, but we did apply the climate futures and climate projections selected by RPA (two RCPs, five climate projections; see the sidebar RPA Scenarios). The amount of forest land projected to experience drought increases under both RCPs (figure 5-14). By 2050, the hot, dry, and middle climate projections produce marked increases over the historical period in both the extent and frequency of drought across the United States under both RCPs. Under RCP 8.5 and the hot and dry climate projections, more than 50 percent of the Nation's forests are exposed to moderate, severe, or extreme drought in most years after 2040. Wetter conditions and less warming result in lower percentages of forest area exposed to drought relative to the hot and dry projections. While the middle climate projection represents moderate changes in temperature and precipitation compared with the other projections, it still projects more frequent and widespread drought conditions, similar to results from the hot and dry projections. This is likely the result of high interannual variation in precipitation under RCP 4.5 and warm temperatures under RCP 8.5 projected by the middle model.

Analysis of forest exposure to drought by FIA forest type group (figure 5-15) provides insights into geographic patterns of forest exposure. We focus on exposure to severe or extreme drought conditions (SPEI <-1.5) for a 30-year period in the future (2041 to 2070, "mid-century") and compare that exposure to a period in the modeled data during the recent past (1991 to 2020, "recent past"). The future drought exposure for several forest type groups, including three smaller type groups that occur in California—western oak, California mixed conifer, and tanoak/laurel—may be similar to the past (figure 5-15). However, projections under both RCPs using some climate projections indicate levels of drought exposure that far exceed recent exposure for many forest type groups. By mid-century, the median projected exposure to severe or extreme drought for the climate projections under RCP 8.5 in the pinyon/juniper, woodland hardwoods, aspen/birch, and ponderosa pine type groups was at least 60 percent, far exceeding the historical exposures for those type groups. For the former three of those type groups, exposure was projected at more than 75 percent, using at least one climate projection under RCP 8.5. Several of the type groups having the highest projected future exposures, including pinyon/juniper and ponderosa pine, occur in the already-arid Southwestern United States; our results agree with other assessments showing the potential for unprecedented drought and resulting ecological impacts to forests in the Southwest toward the latter half of this century (Cayan et al. 2010, Cook et al. 2015, Jiang et al. 2013, Seager et al. 2007, Thorne et al. 2018, Williams et al. 2013, 2020). By mid-century, the projected range of drought exposure for each forest type group reflects not only the wide selection of RPA climate projections—least warm, hot, dry, wet, middle—but also the geographic range of the forest type group. Planning for a dry or a hot future at the local scale may be important to address the potential risk to the resources in these forest types. However, it is important to note that the SPEI index of exposure does not capture the actual water use efficiency of different forest vegetation types in local conditions, nor any changes in that water use efficiency that could result from shifts in vegetation over time. Therefore, actual exposure could vary in ways that are not captured in this analysis.

A high level of drought exposure does not necessarily translate to significant ecological impacts for a forest type group or forested area. Information on exposure can be used in conjunction with research on the drought sensitivities of forest type groups and associated tree species to determine the degree of likely ecological effects from drought and guide management efforts to ameliorate these impacts (see the sidebar Vulnerability to Drought for an example using these SPEI data). For example, recent severe droughts in combination with other stressors including herbivores, parasites, and wildfires, have played a role in widespread tree mortality and growth declines in pinyon-juniper forests (Flake and Weisberg 2019a, 2019b, Shaw et al. 2005), with higher mortality occurring on the driest sites as well as sites with deeper soils and higher stand density (Flake and Weisberg 2019a). This suggests that management actions, such as stand thinning to reduce tree density, might be necessary to increase the adaptive capacity of pinyon-juniper forests in response to these impacts (Bradford and Bell 2017). On the other hand, the longleaf/slash pine type group that occurs in the Southeastern United States is projected to face low to moderate drought exposure, and at least one of its dominant species (longleaf pine, *Pinus palustris*) is likely more drought-tolerant than other tree species (Samuelson et al. 2012, 2019). This type group may therefore be relatively resilient to future drought exposure, despite a projected increase in exposure by mid-century (figure 5-15). The likely drought resilience of longleaf pines is one reason why restoration of forests in the Southeast has recently begun to emphasize creating or maintaining a prominent longleaf pine component as a strategy for climate adaptation (Clark et al. 2018b).

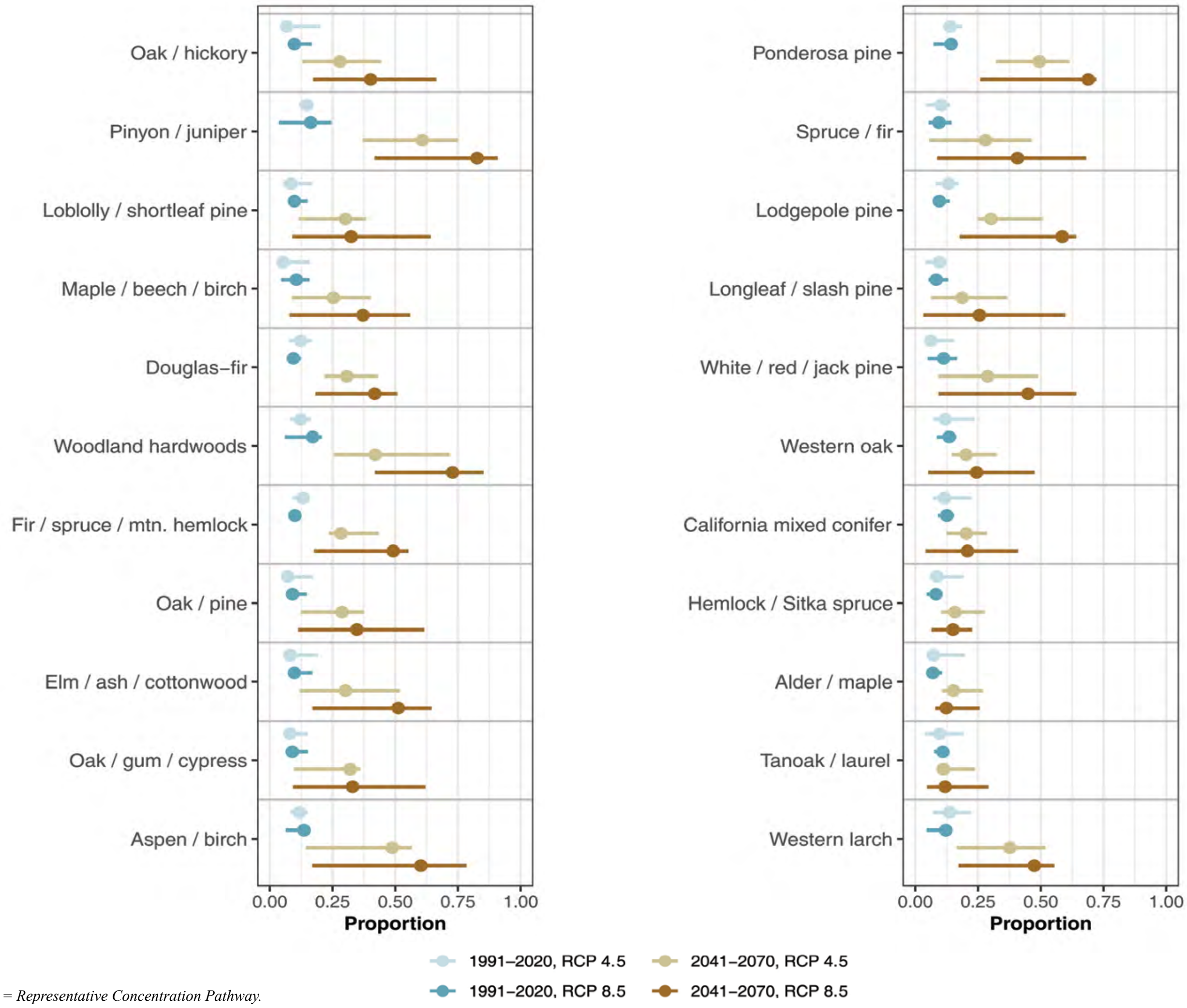
Figure 5-14. Proportion of forest land area in categories of 36-month SPEI for historical (1953 to 2005) and future (2006 to 2070) periods using the RPA climate projections under RCP 4.5 (top) and RCP 8.5 (bottom). The period to the left of the dashed line in each graph indicates the reference period that was used to calibrate SPEI values.



RCP = Representative Concentration Pathway; SPEI = Standardized Precipitation Evapotranspiration Index.

Source: Costanza et al. 2022a.

Figure 5-15. Comparison of monthly proportion of forest type groups in severe or extreme drought for each of the RCPs at mid-century (2041 to 2070) with the same metric during the recent past (1991 to 2020). Dots represent the median of the five RPA climate projections for the given time period, and horizontal bars indicate the range of values across those climate projections. Forest type groups are arranged according to their area (largest at the top left to smallest at the bottom right; see figure 5-8 for areas of forest type groups).



RCP = Representative Concentration Pathway.

Exposure and sensitivity of forests to drought are only one set of factors in determining ecological effects and resulting impacts on goods and services. Drought impacts to forests depend on a number of factors, including landscape characteristics such as the extent and configuration of forest and other land uses, and patterns of human activities related to water supply and demand, as well as management (Crausbay et al. 2017; also see the Water Resources Chapter). For example, evidence from the 2011 drought in east Texas shows that pines, and especially those in managed pine stands that had been thinned, had lower drought mortality rates than other genera (Klockow et al. 2020), suggesting that tree species and management both affected forest drought impacts. Recent emerging frameworks of ecological drought aim to integrate across these ecological and socioeconomic factors to characterize water deficits that result in substantial impacts to ecosystems and ecosystem services. Integrated metrics of ecological drought that incorporate both exposure to drought and measures of impact to forests, rangelands, and the ecosystem services they provide (as in the sidebar Vulnerability to Drought) can be expanded nationwide. Approaches that account for expected human population and land use shifts within and among U.S. regions can help mitigate future drought impacts (Warziniack and Brown 2019). Human adaptations to drought such as groundwater mining can help ameliorate impacts in the short term, but are ineffective in the long term (Brown et al. 2019, USDA Forest Service 2016). Additional research is needed regarding ways to meet the water demands of cities and agriculture while ensuring that forests are sufficiently drought-resilient in the face of climate change.

Vulnerability to Drought: The Case Study of Tree Mortality and Rangeland Productivity in Texas

The vulnerability of forests and rangelands to drought depends on their degree of exposure, sensitivity to drought conditions, and capacity to adapt to those conditions (Crausbay et al. 2017). While individual species and the ecosystems to which they belong can have different levels of drought tolerance (Archaux and Wolters 2006, Berdanier and Clark 2016, Brodrick et al. 2019, Peters et al. 2015), the impact of an event that approaches or exceeds historical extremes in duration or magnitude can be substantial, particularly if it occurs over a large geographic area (Clifford et al. 2013, Schwantes et al. 2017). We illustrate this with a case study of a period of exceptional drought in Texas.

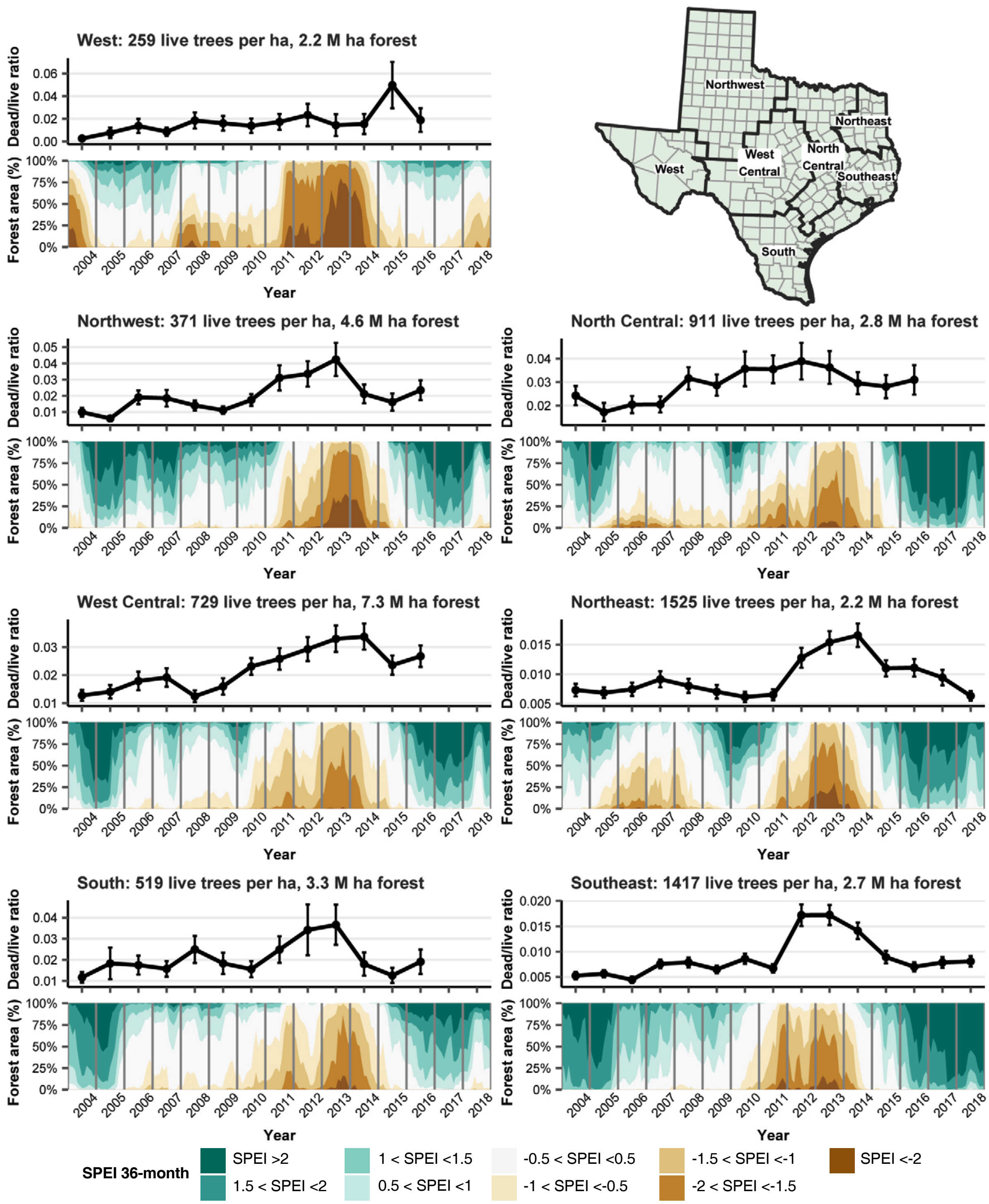
Texas and other parts of the Central United States experienced one of the worst droughts on record in 2011 (Fernando et al. 2016, Grigg 2014, Moore et al. 2016, Nielsen-Gammon 2012). After a relatively dry winter, extreme drought conditions extended throughout Texas during the spring and summer of 2011, persisting in some parts of the State through the end of the year (Fernando et al. 2016). A heat wave during the summer of 2011 exacerbated the drought (Hoerling et al. 2013) and was a secondary contributor to widespread forest mortality. Similar compound extreme events could become more common in the future, highlighting the importance of understanding the impact of this compound event on forests and rangelands. According to FIA data, an estimated 301 million trees, more than 6 percent of trees statewide, were killed by a combination of drought and historically high temperatures (Hoerling et al. 2013, Moore et al. 2016). Rainfall during early 2012 improved moisture conditions across much of Texas, but extreme drought lasted throughout 2012 and into 2013 in some locations elsewhere in the Central United States (Fernando et al. 2016, Tadesse et al. 2015). In Texas alone, agricultural losses from the drought were estimated at $7.6 billion (Fannin 2012), exceeding the previous record of $4.1 billion in 2006. Of this $7.6 billion, livestock losses were estimated at $3.2 billion, reflecting increased feeding costs and market losses. Rangeland impacts were felt beyond these economic effects. The drought resulted in forage yields far below any levels recorded since 1984, the first year of annual production measures from the Rangeland Production Monitoring Service (Reeves et al. 2020, 2021). We show how two metrics of drought sensitivity—forest tree mortality and rangeland production—and their relationships with meteorological drought measured via SPEI changed over space and time for forests and rangelands in Texas.

Distinct signatures of the drought can be seen in each of the seven regions of Texas (figure 5-16). Darker brown areas reveal drier conditions, both in magnitude (taller on the Y axis) and duration (wider range on the X axis). Because of the 36-month window used when computing SPEI, the signatures of the 2011 drought are evident until 2014, even though moisture conditions in Texas generally followed long-term trends from early 2012 until early 2014 (Fernando et al. 2016). At certain points during the signature period, severe or extreme drought conditions (SPEI <-1.5) extended across at least 70 percent of the forested areas in every region. Most importantly, the plots suggest a consistent relationship between the SPEI, a metric of drought exposure, and forest mortality (as depicted by the standing dead tree/live tree ratio), a metric of drought sensitivity. The relationship appears strongest in the northeast and southeast regions of Texas, which have the highest forest density, and weakest in the west region, where forest is sparsely distributed. Differences between the regions in terms of forest mortality, such as when the ratios of standing dead/live trees reached their peak values, may be partly explained by differences in the regions' predominant tree species, which can exhibit varied mortality rates based on their capacity to survive drought stress or associated disturbances, such as drought-triggered pest outbreaks (Klockow et al. 2018).

Figure 5-17 shows the temporal and spatial relationships between meteorological drought measured via 6-month SPEI and rangeland production, another metric of drought sensitivity, on about 69 million ha of rangeland in regions of Texas for the 1984 to 2018 period. There is a notable relationship between the SPEI and production data over time and by region. During drier periods, a corresponding decrease in annual production can be seen in the rangeland production trend. In most regions, 2011 and 2012 show the longest and most far-reaching sustained period of extreme drought (SPEI <-2) in the record. During that time, forage conditions were the second worst since 1984, except for the northwest region of the State, where forage conditions were by far the worst on record.

These figures suggest that the SPEI can be a useful metric for examining forest and rangeland health. The SPEI can also inform management actions to increase adaptive capacity of forests and rangelands to drought, including thinning and prescribed burning in forests and removal of

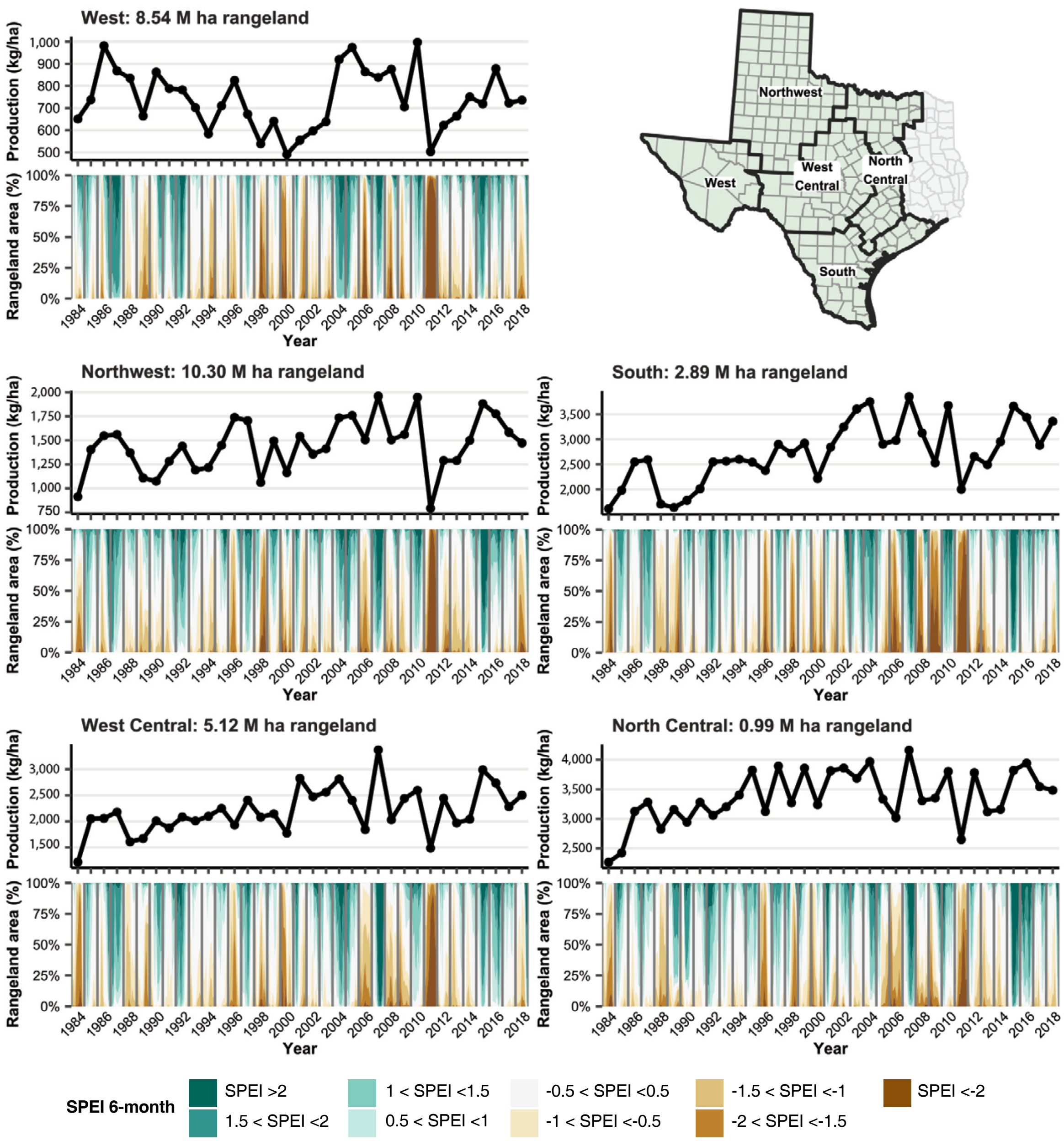
Figure 5-16. SPEI and the ratio of dead/live trees by region in Texas, 2004 to 2018. For each region, the line chart shows the annual ratio of standing dead trees to live trees, estimated from FIA data and representing forest mortality. The plot below the line chart shows meteorological drought as the monthly proportion of the region's forest area in each of the SPEI categories. The number of live trees per hectare and area of forest (FIA data circa 2016) are listed for each region because the regions differ in forest area and density.



FIA = Forest Inventory and Analysis; SPEI = Standardized Precipitation Evapotranspiration Index.
Source: Costanza et al. 2022b.

trees or large shrubs where encroachment has occurred on rangelands. The incidence of droughts of this magnitude and duration are projected to increase in the future (figures 5-14, 5-21), suggesting that substantial tree mortality and decreases in rangeland productivity, along with associated economic losses, will become more frequent in these regions of Texas and elsewhere. Similar analyses are needed for other U.S. forest and rangeland ecosystems to further explore relationships between exposure and sensitivity to drought.

Figure 5-17. SPEI and rangeland production by region in Texas, 1984 to 2018. For each region, the line graph shows annual production obtained from the Rangeland Production Monitoring Service. The plot below the line chart shows meteorological drought as the monthly proportion of the region's rangeland area (circa 2011) in each of the SPEI categories.



Sources: Rangeland Production Monitoring Service (Reeves et al. 2020, 2021); Costanza et al. 2022b; Reeves and Mitchell (2011).

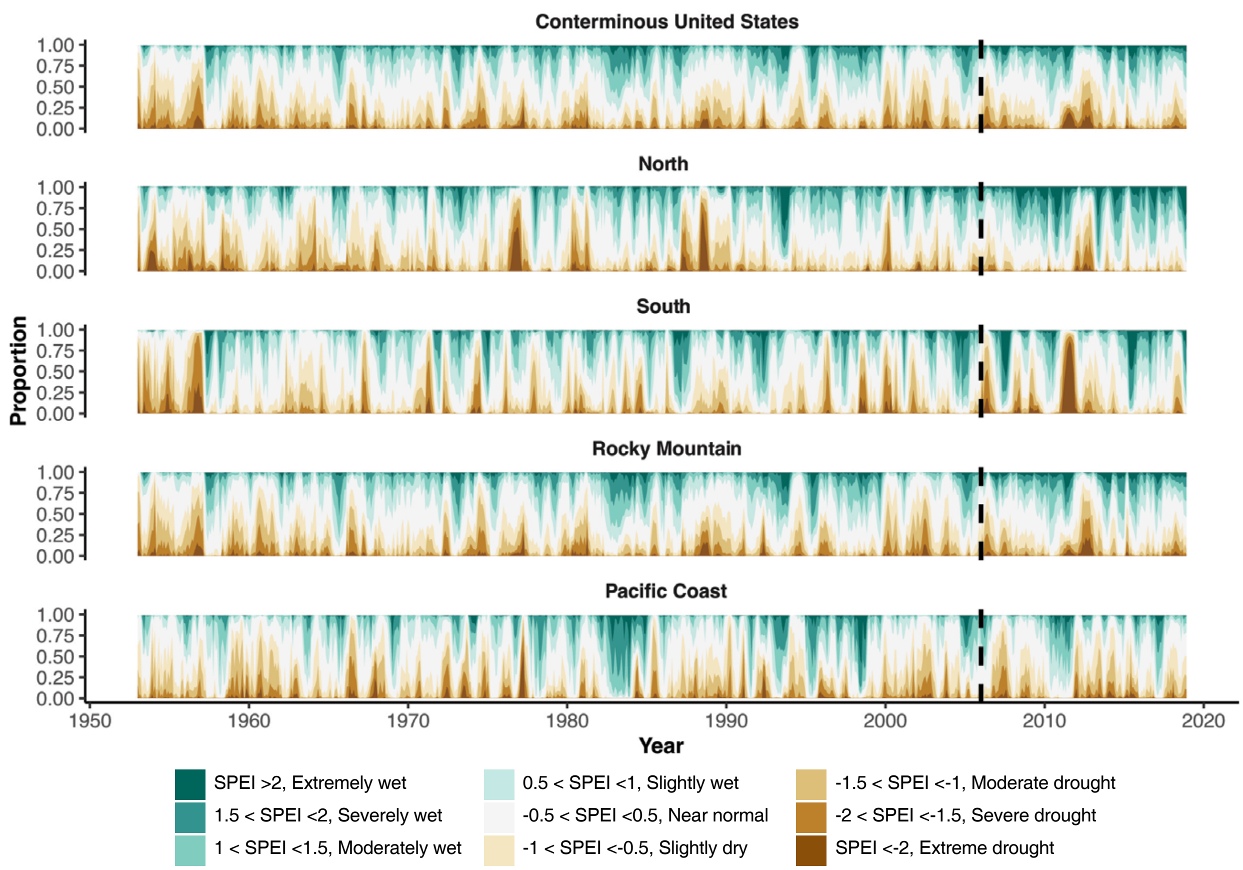
Rangelands

Rangeland drought effects are similar to forest drought effects. Ecologically, drought results in reduced growth rates, defoliation, and increased stress on rangeland vegetation. From a range management perspective, drought generally reduces the supply of water and associated forage vegetation, which can lead to reduced livestock production, and in some cases substantial economic losses (Kelley et al. 2016). Additionally, because many rangeland droughts are driven by warm temperatures that lengthen the growing season, the vegetation that remains during droughts can exhibit increasing demand for water through increased evapotranspiration (Udall and Overpeck 2017). Rangeland droughts have been increasing in frequency and severity over the last 50 years, particularly in the central Great Plains and Southwest, and the trend is expected to continue (Cook et al. 2015).

To assess current and future exposure of rangelands to drought, we used the 6-month SPEI, rather than the 36-month SPEI employed for forests. This shorter period reflects the fact that rangelands are dominated by herbaceous or shrub vegetation, which respond more quickly to drought than forests in terms of both effects and recovery (Finch et al. 2016).

Results from SPEI analysis for the observed historical period generally confirm known intervals of drought and relatively wet conditions, both across the U.S. and within RPA regions (figure 5-18). Major recent rangeland drought events occurred in 2002 in the Rocky Mountain Region, 2011 and 2012 in the South Region, and 2012 through 2016 in the Pacific Coast Region. Of these, the droughts of 2011 and 2012 produced the greatest economic impacts in the rangeland sector (see the sidebar Vulnerability to Drought). Evaluating drought trends at national and regional levels may obscure highly significant events occurring at sub-regional levels. For example, although the summary of SPEI across the Rocky Mountain Region does not show a marked drought signal in 2018, Coconino, Navajo, and Apache counties in Arizona had such severe drought conditions at the time that they were designated as natural disaster areas by the U.S. Secretary of Agriculture (https://www.fsa.usda.gov/state-offices/Arizona/news-releases/2019/stnr_az_20190328_rel_01). Coupling national and regional

Figure 5-18. Proportion of rangeland area in categories of observed 6-month SPEI over time, based on PRISM climate data, 1953 to 2018. The period to the left of the dashed line in each graph indicates the reference period that was used to calibrate SPEI values.



SPEI = Standardized Precipitation Evapotranspiration Index.

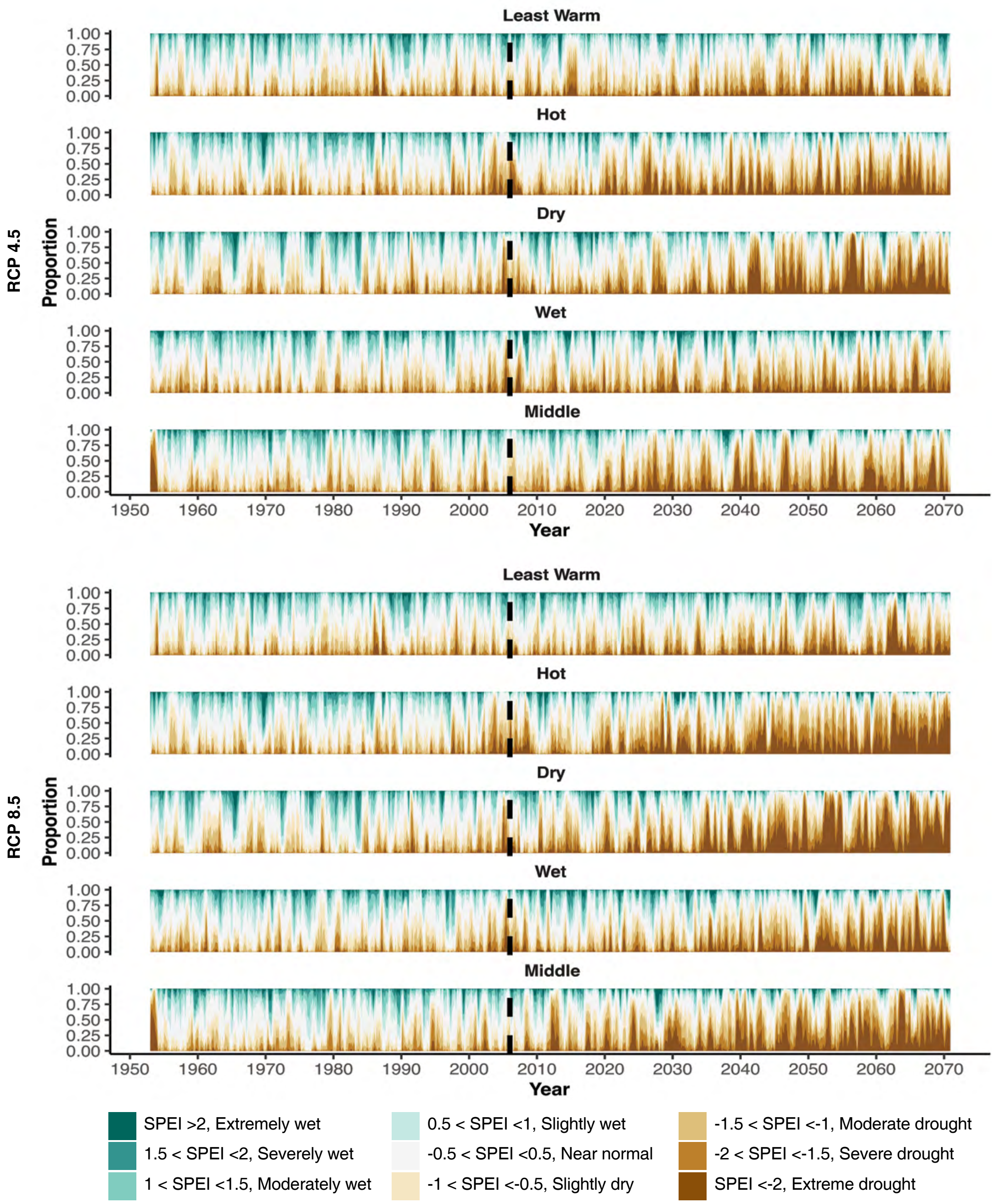
Source: Costanza et al. 2022b.

analyses with analysis and monitoring of local drought conditions is critical for determining drought extent and for more accurate accounting of impacts.

Future projections of drought show that the frequency of drought exposure is expected to increase for rangelands across the United States, under both RCPs and all RPA climate projections (figure 5-19), especially by mid-century (2041 to 2070). The hot and dry futures projected the most frequent, widespread, and severe drought across U.S. rangelands, particularly during the period approaching 2070 under both RCPs. A substantial increase in drought was also projected under RCP 8.5 using the middle climate projection.

We assessed the projected future drought exposure of dominant rangeland vegetation types (figure 5-20). We summarized the monthly proportion of each vegetation type in severe or extreme drought (SPEI <-1.5) for the same time periods assessed in the forest type group analysis (recent past, mid-century). Overall, the analysis shows the potential

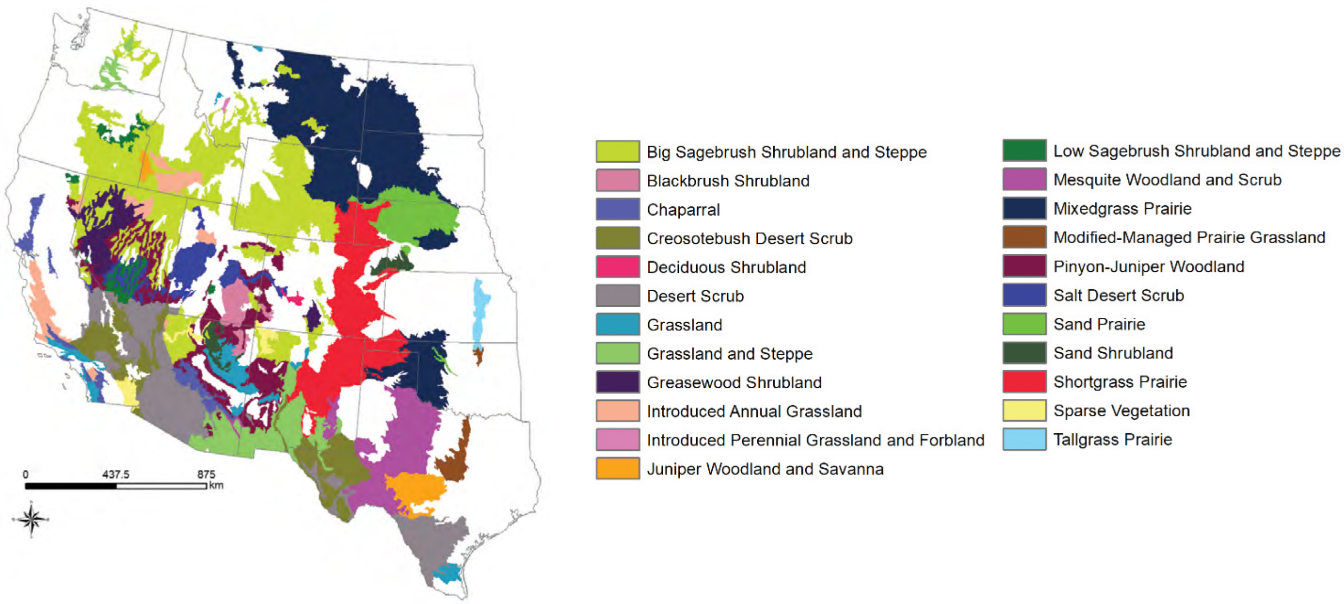
Figure 5-19. Proportion of rangeland area in categories of 6-month SPEI for historical (1953 to 2005) and future (2006 to 2070) periods using the RPA climate projections under RCP 4.5 (top) and RCP 8.5 (bottom). The period to the left of the dashed line in each graph indicates the reference period that was used to calibrate SPEI values.



RCP = Representative Concentration Pathway; SPEI = Standardized Precipitation Evapotranspiration Index.

Source: Costanza et al. 2022a.

Figure 5-20. Ecological subsections and their associated dominant vegetation types for summarizing SPEI projections.



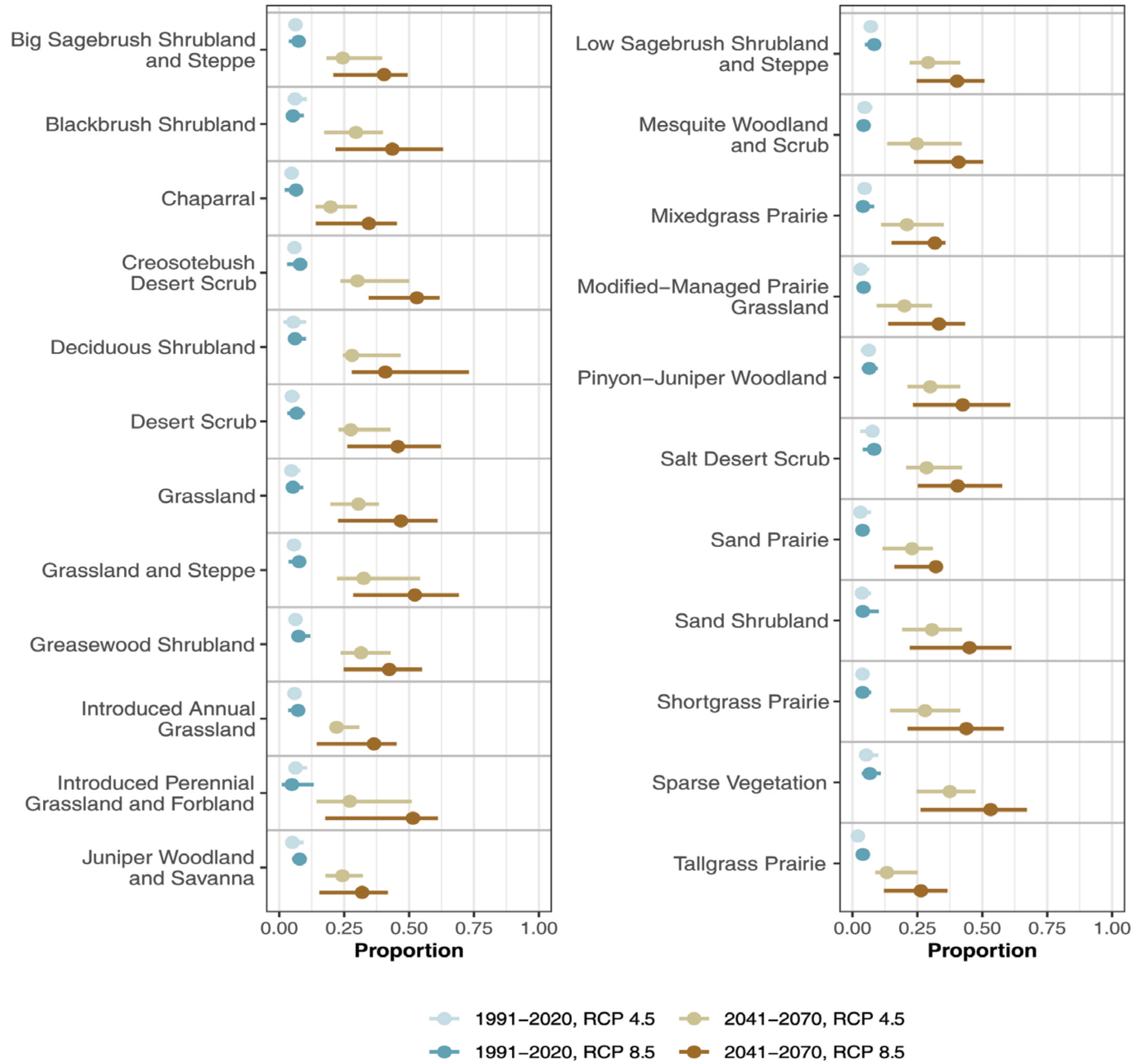
SPEI = Standardized Precipitation Evapotranspiration Index.
Sources: Ecological subsections are from Cleland et al. (2007). Vegetation types are Ecological Systems (Comer et al. 2003) that were mapped in 2012 LANDFIRE Existing Vegetation Type data (LANDFIRE 2012).

for much higher exposure to drought nearly everywhere by mid-century, with differing amounts of exposure by vegetation type, and higher exposure generally under RCP 8.5 (figure 5-21). By mid-century, the vegetation types with the highest level of exposure projected under RCP 8.5 using at least one climate projection include those located in the Southwestern United States, such as creosotebush desert scrub, grassland, and grassland and steppe. Each of those types is common in Arizona and New Mexico, and the former two are also present in southern California (figure 5-21). A comparison of the median exposures for the two time periods indicates that these and other vegetation types occupying the arid regions of the Southwest are expected to experience a four- to five-fold (RCP 4.5) or six- to eight-fold (RCP 8.5) increase in exposure to severe or extreme drought conditions by mid-century (figure 5-21). The increase seen here is similar to results from other recent research showing the potential for unprecedented drought in the Southwestern United States toward the latter half of this century (Cook et al. 2015), and a general agreement among climate models that drought exposure will increase in already-dry regions of the West (Bradford et al. 2020). In addition to those three southwestern types, median projections for other vegetation types that have had moderate drought exposure in the recent past, shortgrass prairie and sand shrubland indicate even greater changes in exposure rates by mid-century. Six- to seven-fold (RCP 4.5) or 10-fold (RCP 8.5) increases in exposure to severe or extreme drought are projected for those types by mid-century (figure 5-21).

By mid-century, the projected range of drought exposure for each rangeland type reflects not only the wide selection of RPA climate projections (least warm, hot, dry, wet, middle) but also the geographic distribution and extent of the rangeland system. Planning for a dry or a hot future may be important to address the potential risk to the resources in these rangeland types at the local scale.

Higher future exposure to severe or extreme drought nearly everywhere, especially in the arid Southwestern United States suggests that the water resources already scarce in that region could be further strained by the end of the projection period, having impacts on ecosystem goods and services (see the Water Resources Chapter). Altered timing of peak flows and shifts from perennial to more intermittent flow, especially in streams in the Southwest (Gutzler and Robbins 2011, Zipper et al. 2021) may further complicate the timing and amount of water availability. Forage resources would likely become sparse under these conditions, suggesting that significant reductions in the density of native and domestic ungulates may be necessary (Ford et al. 2019, Reeves et al. 2017). In addition, the expansion of invasive species such as red brome (*Bromus rubens*) and Lehmans lovegrass (*Eragrostis lehmanniana*) may be enhanced if native perennials and annuals undergo more stress related to soil moisture deficits (Curtis and Bradley 2015). Projection results for all rangeland vegetation types show the possibility of worsening exposure to severe or extreme drought under both RCPs by mid-century compared with the early century time period, suggesting the importance of timely implementation of management or mitigation actions to enable adaptation that is robust to worsening drought (see the Water Resources Chapter for examples).

Figure 5-21. Comparison of monthly proportion of rangeland ecosystems in severe or extreme drought for each of the RCPs at mid-century (2041 to 2070) with the same metric during the recent past (1991 to 2020). Dots represent the median of the five RPA climate projections for the given time period, and horizontal bars indicate the range of values across those climate projections. See figure 5-20 for a map of these rangeland systems.



RCP = Representative Concentration Pathway.

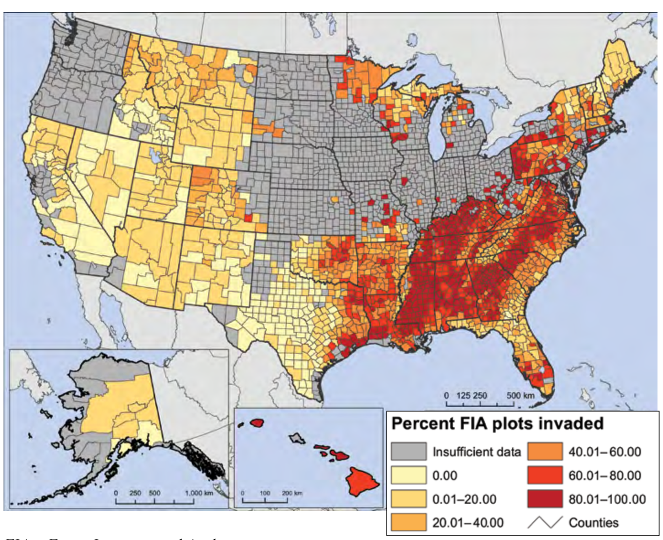
Nonnative Invasive Plants in Forests and Rangelands

- ❖ The highest rates of forest invasion have occurred throughout the RPA South Region as well as in metropolitan areas and agriculture-dominated counties in the RPA North Region.
- ❖ Forest type groups in those regions had the highest rates of invasion, especially where forest was privately owned.
- ❖ Future increases in developed or agricultural land use in the Eastern United States could lead to higher forest invasion rates.
- ❖ Counties in the RPA Pacific Coast Region had the highest rates of rangeland invasion, specifically in coastal and southern California.

Forests

Nonnative invasive plant species cause long-term detrimental effects on forest ecosystems, including declines in biological diversity, alterations to forest succession, and changes in nutrient, carbon, and water cycles (Liebhold et al. 2017, Mack et al. 2000, Martin et al. 2009). The damage caused by these invasive species, and the efforts to control them, are costly (Pimentel et al. 2005), even before accounting for the impacts to nonmarket economic services such as recreation and landscape aesthetics (Holmes et al. 2009). The Forest Inventory and Analysis (FIA) program collects invasive plant data based on expert-derived lists of problematic invasive plant species (Oswalt et al. 2015), defined as those of any growth form likely to cause economic or environmental harm (Ries et al. 2004). A national analysis of FIA plot data across the United States (including Alaska and Hawaii) from 2005 to 2018 revealed a strong differentiation in the percent of invaded plots between counties in the East and West (figure 5-22). Counties throughout much of the RPA South Region and the mid-Atlantic and Midwestern States of the RPA North Region had the highest percent of invaded plots, with lower levels of invasion in parts of the southern Appalachians, the southeastern Coastal Plain, northern Florida, and the Great Lakes States. These results likely underestimate the overall presence of invasive plant species because field crews only record species that have been identified previously as problematic. The geographic patterns are consistent with recent work that also detected the highest prevalence of forest plant invasion in the Southeast, in the agriculturally-dominated Midwest, and near metropolitan areas (Iannone et al. 2015). These results further underscore the finding that eastern FIA plots are most likely to be invaded in relatively more productive, fragmented forest in interface landscapes containing more than 10 percent agriculture or developed land cover (Riitters et al. 2017; also see the Land Resources Chapter).

Figure 5-22. Percent of FIA forest plots invaded by county. Counties with fewer than five plots that were surveyed for invasive plants were omitted and are shown in gray.



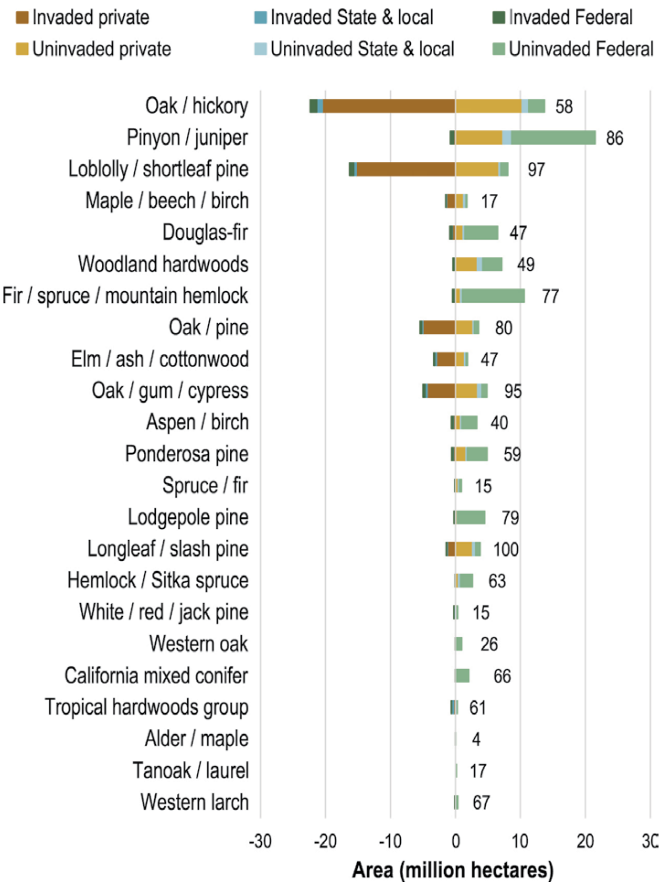
FIA = Forest Inventory and Analysis.
Source: Potter and Riitters 2023.

We used FIA data to estimate the forest area that has been invaded by nonnative plant species nationally, within RPA regions, and by ownership within major forest type groups (Riitters and Potter 2019). Nationally, approximately 62.7 million ha of forest were invaded (36.2 percent of the forest inventoried for invasive plants, figure 5-23). Forest land in the South Region had the highest proportion of invaded forest area (57.7 percent of inventoried area, 52.7 million ha), followed by the North Region (54.5 percent). The forest area in the two western regions was considerably less invaded (7.5 percent in the Rocky Mountain Region and 5.0 percent in the Pacific Coast Region). These proportions and areas of invaded forest are likely substantial underestimates because only 61 percent of all forest was inventoried for invasive plants, with much smaller percentages inventoried in the North and Pacific Coast Regions.

For the most invaded, commonly occurring forest type groups, such as oak/hickory, loblolly/shortleaf pine, oak/pine, and oak/gum/cypress, the large majority of invaded forest was in private ownership (figure 5-23). The large proportion of invaded forest in private ownership agrees with previous research showing that privately owned forest lands in the Eastern United States had the highest rates of invasion (Riitters et al. 2018), likely because they are closer to human land uses, which contribute seed sources that are responsible for plant invasions.

As land use changes, future projected increases in forest area contained within the WUI (see the sidebar Wildland-Urban Interface in the Land Resources Chapter) or exposed

Figure 5-23. Area of forest invaded and not invaded, by ownership within FIA forest type groups. The numbers at the end of each bar indicate the percent of forest within each type group that was surveyed for invasive plants. Bars to the left of the 0 line indicate invaded; bars to the right indicate not invaded.



FIA = Forest Inventory and Analysis.

to nearby agriculture and development (see the section Projected Forest Fragmentation and Landscape Context in the Land Resources Chapter) will likely increase seed sources and thus increase invasion rates in forest land. Road construction is similarly expected to increase rates of forest plant invasions in nearby forests (Forman and Alexander 1998). While privately owned forest land had higher rates of invasion than public land, the proximity of private land to human land uses, rather than ownership per se, is likely the underlying factor responsible for the difference. Therefore, changes in ownership or protection status alone are unlikely to prevent future invasions (Riitters et al. 2018). In addition to land use change, widespread intercontinental movement of plants for ornamental purposes is almost certain to ensure future introductions of new nonnative invasive plants into forests (Theoharides and Dukes 2007). Once forest land is invaded, it is unlikely to become un-invaded in most future circumstances, given that management of invasive plant species in forests often results in their replacement by other nonnatives (Reid et al. 2009). These results add up to a future in which invasion rates are likely to increase on forest land.

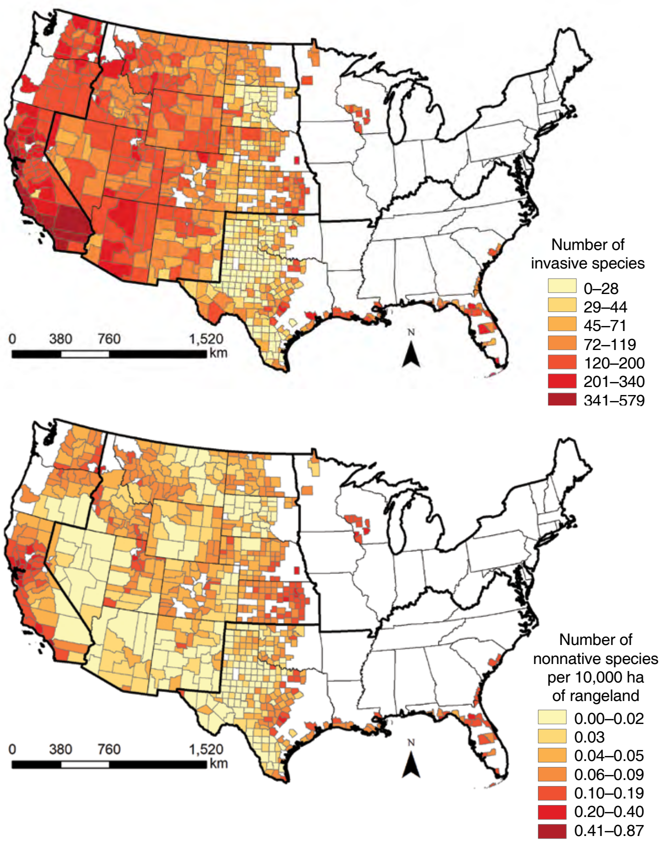
While these summaries of invaded forest areas do not directly address the ecological or economic impacts to forests, some impacts to forests are likely because the invasive species surveyed by FIA are considered problematic (Oswalt et al. 2015). Information about forest invasion rates and impacts is likely to improve as a temporal record of data from invasive plant surveys at broad scales is accumulated and if FIA expands invasive plant inventories to include forest land that has not yet been surveyed for invasive plants (Oswalt et al. 2021).

Rangelands

Nonnative invasive plant species can cause wholesale changes to the ecological and economic health of rangeland ecosystems. Many rangelands that were dominated by perennial bunchgrasses have been invaded by nonnative annual grasses, which increase water demand; cause more frequent, higher severity, larger fires; lower livestock yields and forage quantity; and lead to substantial economic losses (DiTomaso 2000, Rottler et al. 2015). No consistent national invasive species rangeland inventory is available that covers all public and private lands. Hence, we used data from the Center for Invasive Species and Ecosystem Health at the University of Georgia (the Bugwood Program, www.Bugwood.org) to investigate nonnative plants in counties containing substantial rangeland area (exceeding 60,703 ha, based on Reeves and Mitchell 2011; see figure 5-1 for the distribution of rangeland). Data for the Bugwood Program is usually collected by volunteers recording locations of nonnative species and thus may be biased toward higher counts in populous areas or counties with more public land (Wallace 2020).

The number of nonnative plant species in rangeland counties generally increased from east to west, peaking in coastal California (figure 5-24). San Diego, Los Angeles, and Marin counties are reported to host 579, 566, and 494 nonnative species, respectively. Counties in the RPA Pacific Coast Region contained the highest numbers of nonnative species, followed by counties in the western portion of the Rocky Mountain Region. The lowest numbers of nonnative species were exhibited by grassland areas of the Great Plains, including the eastern portion of the Rocky Mountain Region as well as parts of Oklahoma and Texas in the South Region. A few counties in the North Region had enough rangeland area to be included in this analysis but were insufficient for discerning a geographic pattern. When the number of nonnative species in each county was standardized by the area of rangeland in the county ("density" of nonnative species; figure 5-24), the geographic pattern was slightly different. Similar to the result for the overall number of nonnative species, the RPA Pacific Coast Region had the

Figure 5-24. Total number (top) and density (bottom) of nonnative plant species in rangeland counties. Rangeland counties are defined as those that contain more than 60,703 ha of rangeland (Reeves and Mitchell 2011). See figure 5-1 for distribution of rangeland.



ha = hectares.

Source: Center for Invasive Species and Ecosystem Health at the University of Georgia (the Bugwood Program, www.Bugwood.org).

greatest density of nonnative plant species, with the highest densities in counties in and around the California bay area and along the California coast. Unlike the overall geographic pattern for number of nonnative species per county, the geographic pattern of nonnative species density did not increase generally from east to west. Scattered counties in central Utah, the upper Snake River Plain, and in eastern Kansas also had high densities of nonnative species.

The large numbers of nonnative plant species in many western counties may suggest that rangelands have a relative lack of resistance to invasion. Research in many rangeland ecosystems has demonstrated an invasive grass-fire cycle, wherein longer, more favorable growing conditions, inappropriate grazing regimes, and altered fire regimes can allow nonnative annual grasses to survive (D'Antonio and Vitousek 1992, Fusco et al. 2019). Those grasses subsequently alter the moisture and fire regimes, creating new environments that favor even greater richness and abundance of nonnative annual grass species (Roundy et al. 2018). On the other hand, the low numbers of nonnative plant species in parts of the Great Plains could reflect greater resistance to invasion in some rangeland ecosystems. An emerging framework that summarizes the rangeland ecosystem attributes and landscape characteristics that affect resilience to plant invasion and resulting wildfire (Chambers et al. 2014, 2019) could be incorporated in future RPA Assessments to provide further insights into invasion patterns.

Given the potential biases in the data toward higher counts on public lands, caution is recommended for interpretation of these results. For example, many counties in Texas show relatively low numbers of nonnative species, but rangeland counties in the State exhibit approximately 98 percent private land ownership, and some private landowners might be reluctant to make data about their land widely accessible. In addition, because these data document even individual occurrences of a nonnative plant species in a given county, they do not necessarily represent geographic patterns of ecological or economic impact. While data collection efforts in several agencies do cover such occurrences, including the National Resources Inventory of the USDA Natural Resources Conservation Service and the Assessment, Inventory, and Monitoring Strategy of the U.S. Bureau of Land Management, obtaining those data in rangeland counties is challenging due to privacy concerns. Nonetheless, using those datasets in tandem could improve the assessment of invasive plant distributions in rangelands, improve understanding of their impacts, and enable future projections of their spread.

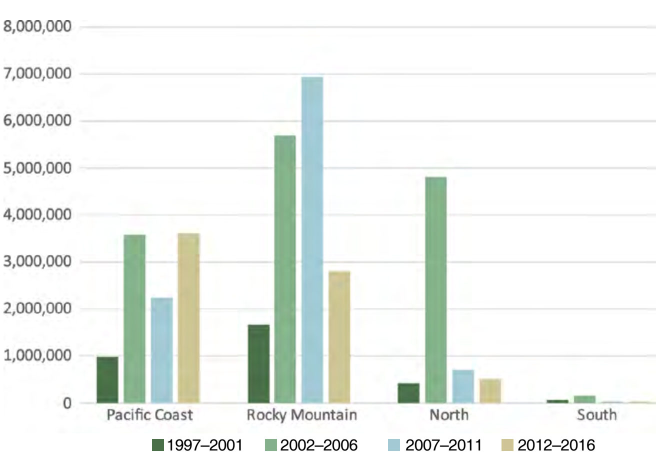
Insect and Disease Disturbances in Forests

- ❖ The overall area of forest tree canopy mortality caused by insects and diseases was usually higher in the RPA Rocky Mountain and Pacific Coast Regions than in the South and North Regions.
- ❖ Nonnative insects and diseases had a larger effect on forest mortality in the North Region than in other regions.
- ❖ Defoliation was more widespread in the North and South Regions than in the two western regions.
- ❖ The future effects of insects and diseases in forests are uncertain, but most factors associated with a warmer climate contribute to a greater potential for outbreaks.

Insects and diseases, especially nonnative invasive agents, have the capacity to cause ecological and economic damage to forests (Lovett et al. 2016, Tobin 2015). Individual insects and diseases have extirpated entire tree species or genera and fundamentally altered forests across broad regions. For example, chestnut blight, a canker disease caused by the introduced fungus *Cryphonectria parasitica*, functionally eliminated the American chestnut from its range across the Eastern United States (Loo 2009). This elimination process is now being repeated for several ash species in the United States and Canada by the emerald ash borer (*Agrilus planipennis*), an insect introduced from northeastern Asia (Klooster et al. 2018). Tracking insect and disease infestations over time is necessary to understand the extent and duration of their impacts on forest ecosystem structure, function, and dynamics. Twenty years of Insect and Disease Survey (IDS) data, collected annually by the Forest Health Protection program of the U.S. Department of Agriculture, Forest Service (FHP 2019), enable trend detection over time for insect and disease damage (Potter et al. 2020). We summarized the forest area in which tree canopy was affected by insects or diseases nationally (including Alaska and Hawaii) and within RPA regions in four 5-year time windows (1997 to 2001, 2002 to 2006, 2007 to 2011, and 2012 to 2016) to highlight places where forests were impacted by insect or disease agents.

The tree canopy area affected by native and nonnative mortality-causing agents has been consistently large across the three most recent 5-year assessment periods. The RPA North Region experienced its greatest affected area in 2002 to 2006, the Pacific Coast Region (which here includes Alaska and Hawaii) in 2002 to 2006 and 2012 to 2016, and the Rocky Mountain Region in 2007 to 2011 and 2002 to 2006, while the South had comparatively limited area with mortality (figure 5-25). Forest mortality from insects and diseases may be underrepresented in the South Region because of the more intense management cycles including rapid removal of affected trees, and higher growth and decay rates leading to more rapid forest recovery after disturbance. Forest mortality is likely overrepresented in

Figure 5-25. Area of mortality attributed to both insect and disease agents in 5-year intervals, by RPA region (Alaska and Hawaii are included in the Pacific Coast Region).



Source: Insect and Disease Survey (IDS) data (FHP 2019).

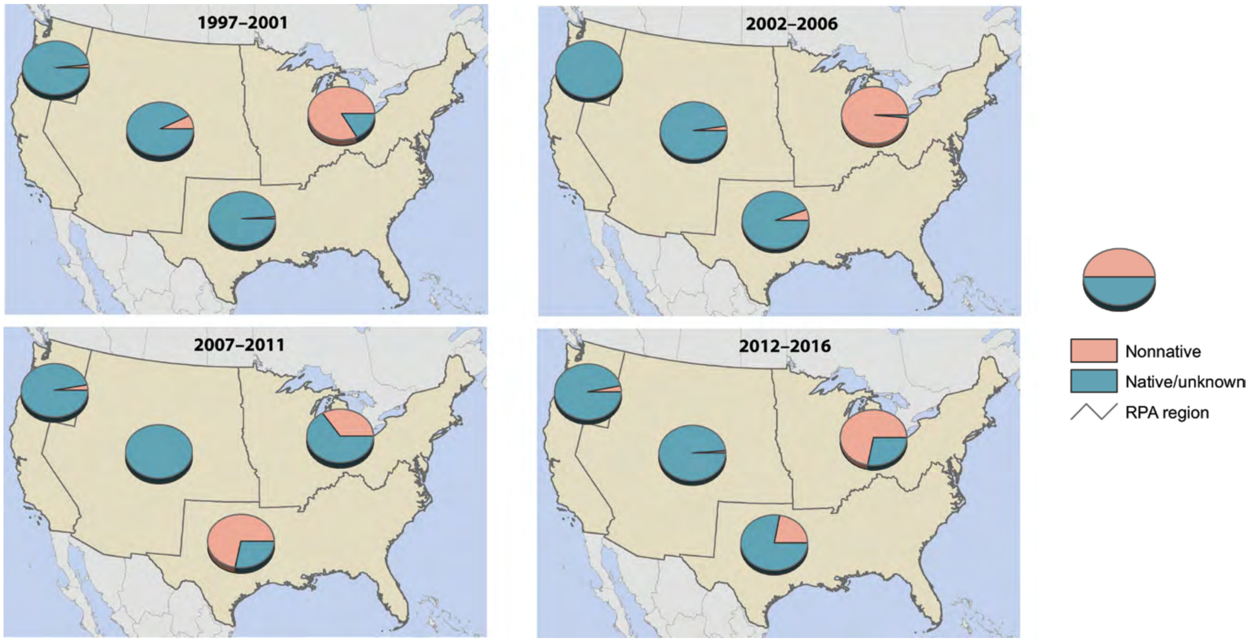
the North Region during the 2002 to 2006 period because surveyors drew polygons to encompass large areas affected by dispersed emerald ash borer and balsam woolly adelgid (*Adelges piceae*) infestations, rather than defining only the affected areas as was done in other regions. Documented mortality has generally been much more widespread from insects than diseases, with bark beetles consistently reported as the most important mortality agents across all regions and over time, particularly in the West (Potter et al. 2020). Mountain pine beetle (*Dendroctonus ponderosae*) was responsible for a mortality peak in the Rocky Mountain Region from 2007 to 2011, while fir engraver (*Scolytus ventralis*) and western pine beetle (*Dendroctonus brevicomis*) caused increased mortality in the Pacific Coast Region from 2012 to 2016.

Nonnative invasive insects and diseases had a larger relative contribution to forest mortality in the North Region than elsewhere in the United States (figure 5-26). The list of such species in the North Region is lengthy, including emerald ash borer, hemlock woolly adelgid (*Adelges tsugae*), balsam woolly adelgid, beech bark disease (caused by the insect *Cryptococcus fagisuga* and associated *Neonectria* fungus), and oak wilt (caused by the fungus *Bretziella fagacearum*). Nonnative invasive agents had substantial impacts elsewhere as well, including Hawaii, where rapid ‘ōhi‘a death, a fungal disease caused by *Ceratocystis huliohia* and *C. lukuohia*, is causing considerable mortality to one of the State's most ecologically and culturally important tree species (Fortini et al. 2019). Elsewhere, and especially in the West, native agents including the western pine beetle mentioned above have been consistently important causes of mortality.

While tree canopy mortality is one critical effect of insects and diseases, some agents also cause substantial damage via defoliation. The tree canopy area affected by defoliation agents has remained relatively consistent over time and has usually equaled or exceeded the area affected by mortality agents, with nonnative defoliators more significant in the RPA North Region (including European gypsy moth, *Lymantria dispar*; larch casebearer, *Coleophora laricella*; and winter moth, *Operophtera brumata*) and South Region (European gypsy moth) compared to the western regions (Potter et al. 2020). This evaluation of recent mortality and defoliation from insects and diseases provides context for managers about the implications and scope of current forest health threats at a national scale.

Knowing how these trends will change in the future can provide critical information for land management planning and decision making. The future impacts of forest insects and diseases are highly uncertain, compounding uncertainty about climate change with uncertainty about the effects of climatic conditions on insects and diseases, as well as on the distribution of tree host species, and about what new

Figure 5-26. The proportion of mortality attributed to nonnative invasive agents versus native agents and those with unknown origin in 5-year intervals, by RPA region (Alaska and Hawaii are included in the Pacific Coast Region).



Source: Insect and Disease Survey (IDS) data (FHP 2019).

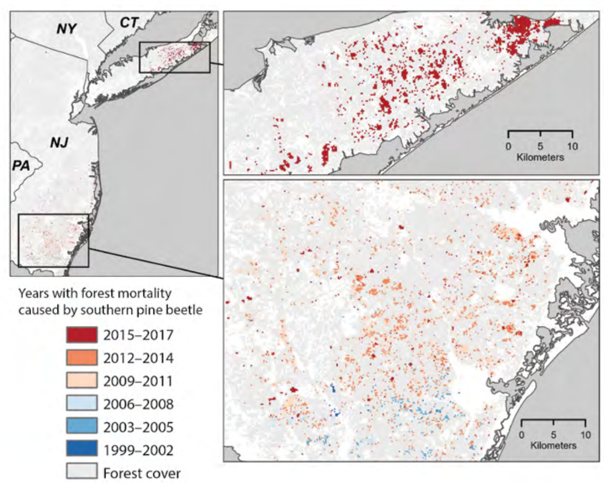
invasive agents will be introduced into the United States. Specifically, predicting the consequences of climate change on the forest health impacts of pests is difficult given the complex relationships among abiotic stressors, host trees, insect herbivores, and the natural predators and parasitoids of those insects (Jactel et al. 2019). Several factors suggest an increased potential for insect and disease outbreaks in the future. For example, it is possible that warmer temperatures may result in higher numbers of broods within a year for some insects, resulting in population outbreaks (Bentz et al. 2019), and allow insect herbivores to expand their ranges into areas that were previously too cold (Dukes et al. 2009). The local expansion of the ranges of some insects and diseases due to climate change has already caused forest mortality and presents challenges for management (see the sidebar Southern Pine Beetle Recent Range Expansion for a summary and example). In addition, climate model projections point to more drought under some plausible futures (see the section Drought in Forests and Rangelands). Droughts may benefit forest insect pests by reducing tree resistance, with bark beetles, sap feeders, and leaf chewers more likely than other insect guilds to benefit from drier conditions (Jactel et al. 2012), although the degree of drought stress affects how well trees resist bark beetles (Raffa et al. 2008). Finally, changing climate conditions may increase the frequency and severity of storms that result in fallen or broken trees that trigger bark beetle outbreaks (Marini et al. 2017, Raffa et al. 2015). At the same time, other factors related to changing climatic conditions may counteract the potential for increased future pest outbreaks. For example, forest insect developmental rates decrease rapidly between an optimal temperature and a hot lethal threshold (Davídková and Doležal 2019), so warming conditions could result in increased insect mortality (Mech et al. 2018). Higher temperatures may also result in smaller size and lower dispersal capacity of newly emerged adult insects (Pineau et al. 2017), while variability in temperatures could reduce forest insect survival (David et al. 2017). Increased CO_2 may also negatively impact forest insect performance, although this could be offset by elevated temperatures (Zvereva and Kozlov 2006). Climate change may also affect relationships between forest insects and their predator and parasitoid enemies, although how these relationships change is likely to be complicated by several factors (Jeffs and Lewis 2013). Changing climate conditions are generally expected to benefit forest pests, but negative effects of warming may mitigate their impacts on forest health in some circumstances (Jactel et al. 2019) while interactions among disturbances could produce feedbacks that prevent worst-case outcomes (Lucash et al. 2018).

Southern Pine Beetle Recent Range Expansion into New Jersey and New York

Climate change has already enabled the spread of some native forest insects and diseases into areas outside their historical ranges (Dodds et al. 2018, Heuss et al. 2019, Weed et al. 2013). In many of these instances, warmer winter temperatures have reduced or removed cold-temperature restrictions that previously kept populations in check (Kolb et al. 2016, Lesk et al. 2017). Such range shifts give pests access to novel, nonadapted host species or areas that previously were only marginally suitable for a pest, and can therefore have notable ecological and economic consequences for forests. Ecological consequences can include direct impacts to trees in terms of mortality or stress, as well as disruption of existing disturbance regimes and increased susceptibility to related forest health threats such as wildfires and drought (Anderegg et al. 2015, Pureswaran et al. 2018). Economic consequences include mitigation costs as well as direct economic losses from tree mortality (Heuss et al. 2019, Kolb et al. 2016, Weed et al. 2013).

The southern pine beetle (*Dendroctonus frontalis*) is the most economically significant forest pest in the Southeastern United States. Prior to the 2000s, most outbreaks of the beetle occurred in a region extending from Texas to Virginia, although infestations were infrequently reported as far north as Pennsylvania and southern New Jersey (Dodds et al. 2018). Outbreaks were historically most common in forests dominated by loblolly (*Pinus taeda*) and shortleaf (*P. echinata*) pines. Since 2001, southern pine beetle outbreaks have followed a steady northward progression into forests dominated instead by pitch pine (*P. rigida*); this expansion coincides with a documented warming trend (Dodds et al. 2018, Lesk et al. 2017). Insect and Disease Survey (IDS) data show areas of forest mortality caused by the southern pine beetle in New Jersey and New York from 1999 to 2017 (figure 5-27). Gradual northward advancement of mortality is evident in southern New Jersey, and by the 2015 to 2017 period, the beetle was widespread in the pitch pine barrens of Long Island, an area where it had not been previously recorded (Heuss et al. 2019). Pitch pine has been nearly eliminated from affected sites, which have shifted toward hardwood dominance as a result. Efforts to suppress the infestations have also led to accumulation of downed woody debris, increasing fire risk (Heuss et al. 2019). The beetle has since been captured in traps in Connecticut, Rhode Island, and Massachusetts (Dodds et al. 2018), raising concerns that climate-driven range expansion could allow it to exploit other potential hosts such as red pine (*P. resinosa*) and jack pine (*P. banksiana*). This expansion of southern pine beetle, and similar range expansions by other forest insects and diseases, presents a challenge to managers, who may have to adapt their methods to a possibly unfamiliar pest based on knowledge acquired in other geographic settings, which may not translate well to their circumstances (Weed et al. 2013).

Figure 5-27. Forest mortality caused by southern pine beetle in New York and New Jersey from 1999 to 2017.



Source: Insect and Disease Survey data (FHP 2019).

Forest Removal Areas

- ❖ While removals have wide-ranging effects on forests, removals are an important forest management tool for preventing or mitigating impacts from natural disturbances.
- ❖ The annual area of forest canopy loss from removals in the United States averaged 2.44 million ha between 1986 and 2010, with 65 percent of the total occurring in the RPA South Region.

Removals are trees taken out of forests during timber harvesting or other cultural treatments, or due to land-use change. Like other types of disturbances, removals can have wide-ranging effects on forests and their associated goods and services. Removals can negatively affect forest community assembly, structure and function, and productivity (Duncker et al. 2012, Fall et al. 2004, Jactel et al. 2009); carbon storage (Birdsey et al. 2006); water and soil quantity and quality (Birdsey and Lewis 2002, Nave et al. 2010, Yanai et al. 2003); and wildlife habitat and biodiversity (Verschuyl et al. 2011). Removals to decrease forest stand densities, however, can serve to prevent or mitigate impacts from other disturbances such as fire or insect and disease outbreaks (Fettig et al. 2014, Leverkus et al. 2018, Lindenmayer and Noss 2006, Mason et al. 2006), help some forests adapt to increasing water stress (Bottero et al. 2017, Bradford and Bell 2017), increase productivity for timber management (D'Amato et al. 2011, Fox 2000), and provide critical early-succession habitat for wildlife species in the absence of other disturbances (King and Schlossberg 2014). Removals can be directly and immediately influenced by policy, economic incentives, and management goals (Cubbage and Newman 2006, Ellefson et al. 2006, Legaard et al. 2015), unlike many other disturbance processes (but see the sidebar Effects of Air Pollution on Forest Ecosystems for an exception in which the Clean Air Act has had substantial effects on acid deposition). Characterizing the spatial and temporal patterns of removal regimes is an important component of understanding sustainability in light of disturbance interactions and climate change (Kurz et al. 1998, Leverkus et al. 2018, Seidl et al. 2008).

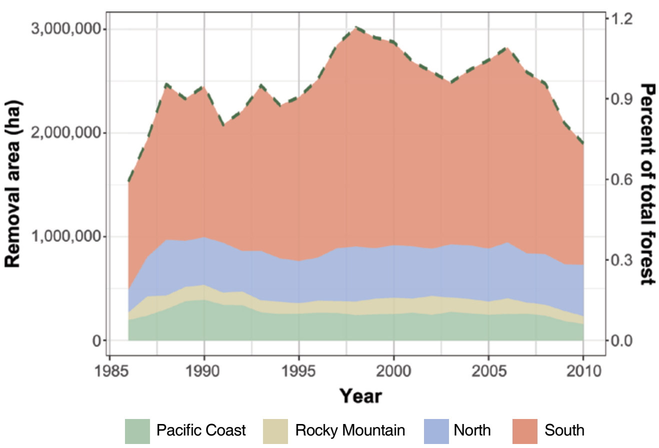
Annual areas of forest removal, measured here in terms of the area of forest canopy loss from removals each year, were derived from a time series of Landsat satellite imagery for the period 1986 to 2010 (Schleeweis et al. 2020) (figure 5-28). Nationally, removals occurred at a mean annual rate of 2.44 million ha (roughly 1 percent of total forest per year) and ranged between 1.53 million ha and 3.01 million ha (dashed line in figure 5-28). The RPA South Region had the highest removal rate in all years, accounting for more than 65 percent of all removals each year, and the most variability from year to year. Although substantially lower than the South Region, the North Region had the next highest annual removal rate on average, followed by the Pacific Coast and Rocky Mountain Regions.

It is important to benchmark these results against the area of annual removals estimated from ground-based forest inventories for similar periods. Reports based on FIA data show consistent national average removal rates of 4.5 million ha yr^{-1}, across multiple decades (although this estimate includes 0.35 million ha reported in Alaska) (Birdsey and Lewis 2002, Oswalt et al. 2014, Smith et al. 2009). While forest inventory data can have a more inclusive definition of removals, optical satellite imagers like Landsat can only detect removals that result in overstory tree canopy loss, and are less accurate when less than 20 percent of canopy cover has been removed (Cohen et al. 2016, Zhao et al. 2018).

The observed trends in removal areas correspond with known trends in policy and markets. First, the peak removal rate and subsequent decrease observed from 1988 to 1990 in the RPA Pacific Coast Region corresponds to documented shifts of regional timber sales due to endangered spotted owl habitat restrictions (Huang et al. 2012, Wear and Murray 2004). Second, record lumber consumption from 2003 to 2005, high levels of housing starts in 2005, and the subsequent crash in housing prices and lumber markets during the global financial crisis of 2007 to 2009 correspond to the timing and directions of removal trends across all regions (Ince and Nepal 2012, Woodall et al. 2012). Third, the timing of the peak removal rate in the South Region occurring around 1997 to 1998 corresponds to regional trends in volume removal for roundwood production (Smith et al. 2009, Wear and Greis 2013). Fourth, all regions show steep increases in removal rates at the beginning of the record. Data from FIA also show a steep increase in the South's annual volume removal rate over the period 1986 to 1997 (Smith et al. 2009), and all regions had an increase in lumber volume supply during that time (Wear and Murray 2004).

We report summaries of removals in terms of area because the remote sensing products we used focus on area estimates. Other sources, including reporting based on FIA, have summarized removals in terms of volume estimates (Smith et al. 2009; see the Forest Resources Chapter for volume-based reporting). It is therefore useful to understand the relationship between volume and area of removals, which depends on three factors: (1) the harvest intensity (i.e., volume per unit area harvested); (2) the natural or managed timber productivity of the land (volume available per unit area); and (3) how variable the harvest intensity is across time and space. While total regional productivity is relatively stable over time, FIA data have shown that harvest intensity varies considerably across and within regions (Masek et al. 2011, Schleeweis et al. 2013). In lower productivity areas, where it takes more forest area to reach a certain volume of removal, a decrease in low-intensity harvesting can have a substantial effect on area-based metrics, even if total volume removed only changes slightly. For example, the Pacific Northwest's highly productive forests report an average extraction intensity roughly twice as high as in the Southeast's forests (200 m^3/ha versus 100 m^3/ha) (Masek et al. 2011). For every 1 m^3 decrease in total annual volume harvested in the South, there is a 0.5 ha decrease in harvested area, whereas the same decrease in volume harvested (1 m^3) leads to a 1 ha decrease in harvest area in the Pacific Northwest. While volume metrics remain steady or show only slight trends, area-based summaries of removals may be more variable through time. The disconnect between volume and area-based metrics may be greater especially in locations with lower productivity and/or more variable harvest intensities, such as the South (figure 5-28).

Figure 5-28. Annual areas of forest canopy loss events attributed to removals and percent of total forest that was lost to these removal events, 1986 to 2010, by RPA region. Regional areas are stacked on top of one another, so that the dotted line represents the total area for the conterminous United States. See text for a discussion of removal areas compared with removal volumes.



Source: Schleeweis et al. (2020).

Discussing removals in terms of both area and volume from traditional inventories and remote sensing gives a more robust understanding of the disturbance. Information from remote sensing, like that reported here, can include higher temporal detail than tree volume information from forest inventories, while forest inventory data can include more detail on the size, age, or species of the trees removed and the management objectives of the removal. Recent studies have shown that in some areas, such as the Southern States, intensity and ratio of partial to clear-cut harvest can vary dramatically on an annual time step (Huang et al. 2015, Tao et al. 2019). In the future, combining information from satellite image time series with plot-based data can provide additional information and allow a wider range of removal intensities to be detected and mapped (Tao et al. 2019). Additionally, outcome-based metrics, such as those related to the effectiveness of removals at reducing fuels on forest land with high fire risk but low volume and acreage, could be a good addition to area- and volume-based metrics in national reporting and assessment.

Multiple Forest Disturbances: A Neighborhood Perspective

- Ninety-four percent of places where forest was lost between 2001 and 2010 had at least one identifiable disturbance process occurring nearby, and 15 percent of forest loss locations experienced cumulative pressures from more than one change process.
- During the same time, nearly half of all forest area was exposed to forest removals occurring nearby, with smaller proportions exposed to stress or fire, and even smaller areas exposed to land conversion.
- Most forest type groups in the Eastern United States had higher exposure to removals and lower exposure to stress and fire. In contrast, most forest type groups in the Western United States had higher rates of exposure to stress and fire and relatively lower exposure to removals.

Multiple Disturbances Near Recent Forest Loss

Earlier sections in this chapter focused on individual disturbances occurring in isolation. Many disturbance processes occur in close proximity to one another, and can together put cumulative pressure on forests and their resources (Drummond et al. 2017, Drummond and Loveland 2010). By assessing the extent to which multiple disturbances have occurred in or near forests, we can gain insight into those cumulative pressures.

Regional trends and rates of forest cover change have varied since 2001 across the conterminous United States (see the Land Resources Chapter). From 2001 to 2010, the total gross forest loss was approximately 140,000 km^2 (14 million ha, 6 percent of the 2001 forest area). To gain insights about which disturbance processes have occurred near forest loss, we summarized the co-occurrence of multiple disturbances nearby. We evaluated disturbances occurring within a 4.41-ha neighborhood of forest cover loss from 2001 to 2010, with forest cover loss defined as pixels that changed from forest to nonforest over this time period in the National Land Cover Database (USGS 2019a, 2019b). Although co-occurrences of common forest disturbance processes are rarely mapped over large spatial extents, there have been recent strides in creating the datasets needed for such analyses in the United States (Huo et al. 2019; Schleeweis et al. 2013, 2020; Vogelmann et al. 2011). These new disturbance attribution data allow novel insights about the likely causes of change (Riitters et al. 2020). The data described in the section Forest Removal Areas use a consistent methodology to map forest

Effects of Air Pollution on Forest Ecosystems

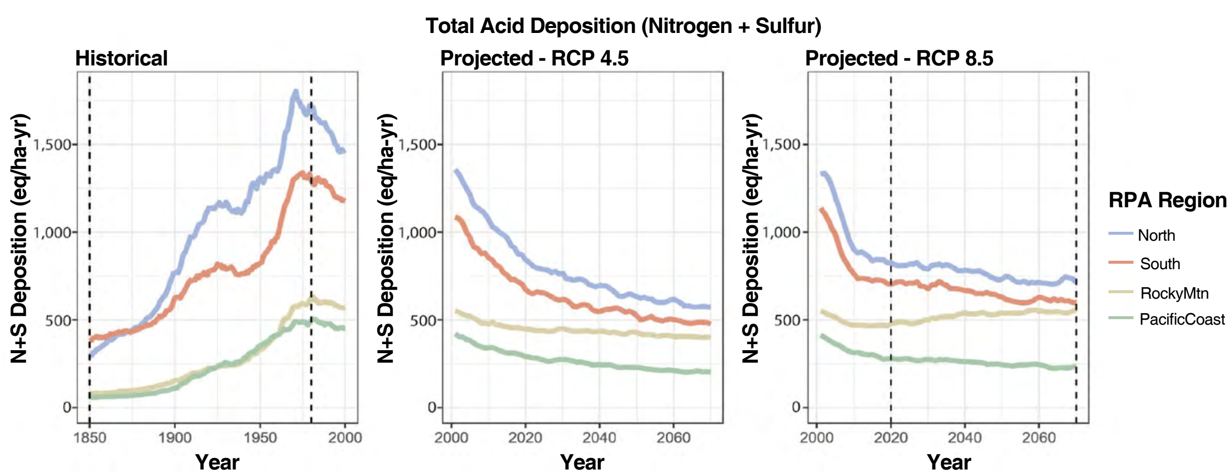
Impaired air quality stresses forest and rangeland ecosystems, leading to altered species composition, modified ecological function, and impacts to ecosystem goods and services (for example, Agathokleous et al. 2020, Pardo et al. 2011, Sams 2007). Air quality trends in the United States are therefore relevant and important to the management of forests and rangelands. Some air quality effects are already incorporated into the RPA water quality assessment (see the Water Resources Chapter) and forest productivity modeling (see the Forest Resources Chapter). Here we provide an overview of specific types of air pollutants, recent and future trends in the deposition of air pollution, and potential effects on forest and rangeland ecosystems and resources.

Emissions from a variety of sources, including agriculture, oil and gas development, fossil fuel combustion, and natural sources such as wildfire, contribute to impaired air quality (US EPA 2020). Deposition of emitted pollutants from the air to the ground leads to effects on forest and rangeland ecosystems that vary by pollutant (Davidson et al. 2012, Fenn et al. 2011). For example, sulfur and nitrogen deposition have been shown to significantly impact forest resources through the acidification of soils and surface waters, leading to decreased growth of certain tree species, reduced species richness, and diminished nutrient availability (Fenn et al. 2011, Pardo et al. 2011).

Critical loads are deposition levels above which components of forests or rangeland ecosystems experience harmful ecological effects; deposition levels greater than the critical load result in a critical load exceedance for a given ecosystem component (Porter et al. 2005). We can identify where ecosystems are likely impacted by air pollution by comparing maps of past or future deposition with maps of critical load thresholds.

Historical and recent trends in exceedances of surface water critical loads can serve as a case study to highlight the effect of air pollution on renewable resources. Surface waters in the United States, especially in the Northeast and along the Appalachian Mountains, have been impacted by deposition of sulfur and nitrogen in the form of "acid rain," predominantly from industrial and fossil fuel sources (Aber et al. 1989, Driscoll et al. 2001, Greaver et al. 2012). As emissions and acid rain increased throughout the 20th century (Galloway et al. 2004) (figures 5-29, 5-30a, 5-30b), surface water critical loads were exceeded at many locations in the RPA North and South Regions (figure 5-30b). Resulting acidification degraded soils, which affected water chemistry and reduced the presence of aquatic organisms, from macroinvertebrates to game species of fish. These effects on habitats and wildlife ultimately impacted ecosystem services such as drinking water and recreation (Beier et al. 2017)

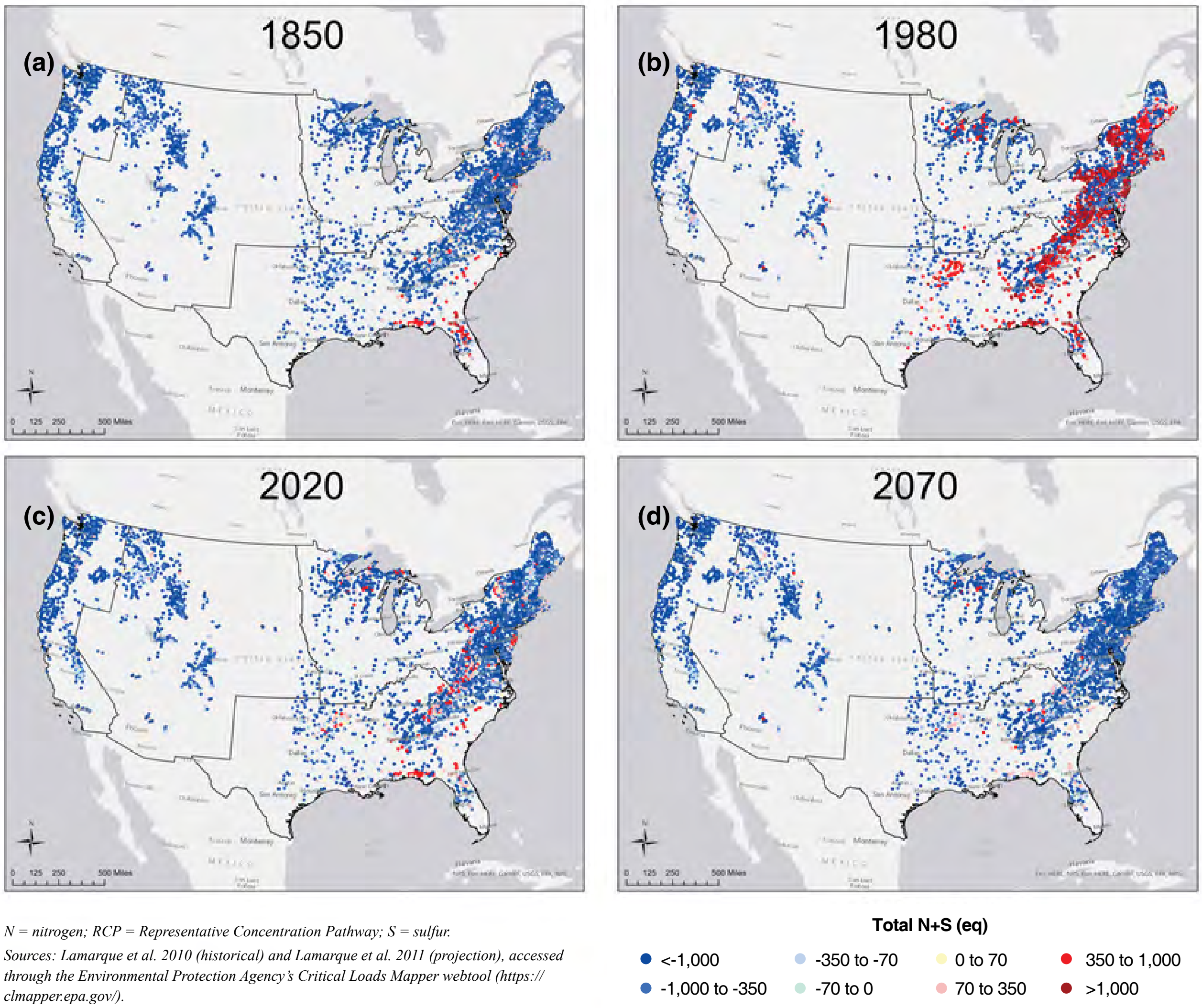
Figure 5-29. Historical (1850 to 2000) and projected (2000 to 2070) average annual acid deposition for each RPA region. Projections are shown for RCPs 4.5 and 8.5. Acid deposition is the total deposition of sulfur and nitrogen compounds. Dashed lines represent time points where deposition values are used to map critical load exceedances in figure 5-30.



ha = hectares; N= nitrogen; RCP = Representative Concentration Pathway; S = sulfur.

Sources: Lamarque et al. 2010 (historical) and Lamarque et al. 2011 (projection), accessed through the Environmental Protection Agency's Critical Loads Mapper webtool (https://clmapper.epa.gov/).

Figure 5-30. Maps of critical load exceedances for surface water acidification for four periods from 1850 to 2070: (a) 1850, before intense industrialization and accompanying increases in emissions and acid deposition; (b) 1980 at peak of emissions and acid deposition in most areas of the U.S.; (c) 2020; and (d) 2070. Negative critical load exceedance values (shades of blue) indicate that acid deposition levels are below the critical load, while positive critical load exceedance values (shades of red) mean that acid deposition is above the critical load and indicate that that area is likely experiencing ecological impacts. For 2020 and 2070, maps are depicting deposition levels from projections based on RCP 8.5.



N = nitrogen; RCP = Representative Concentration Pathway; S = sulfur.

Sources: Lamarque et al. 2010 (historical) and Lamarque et al. 2011 (projection), accessed through the Environmental Protection Agency's Critical Loads Mapper webtool (https://clmapper.epa.gov/).

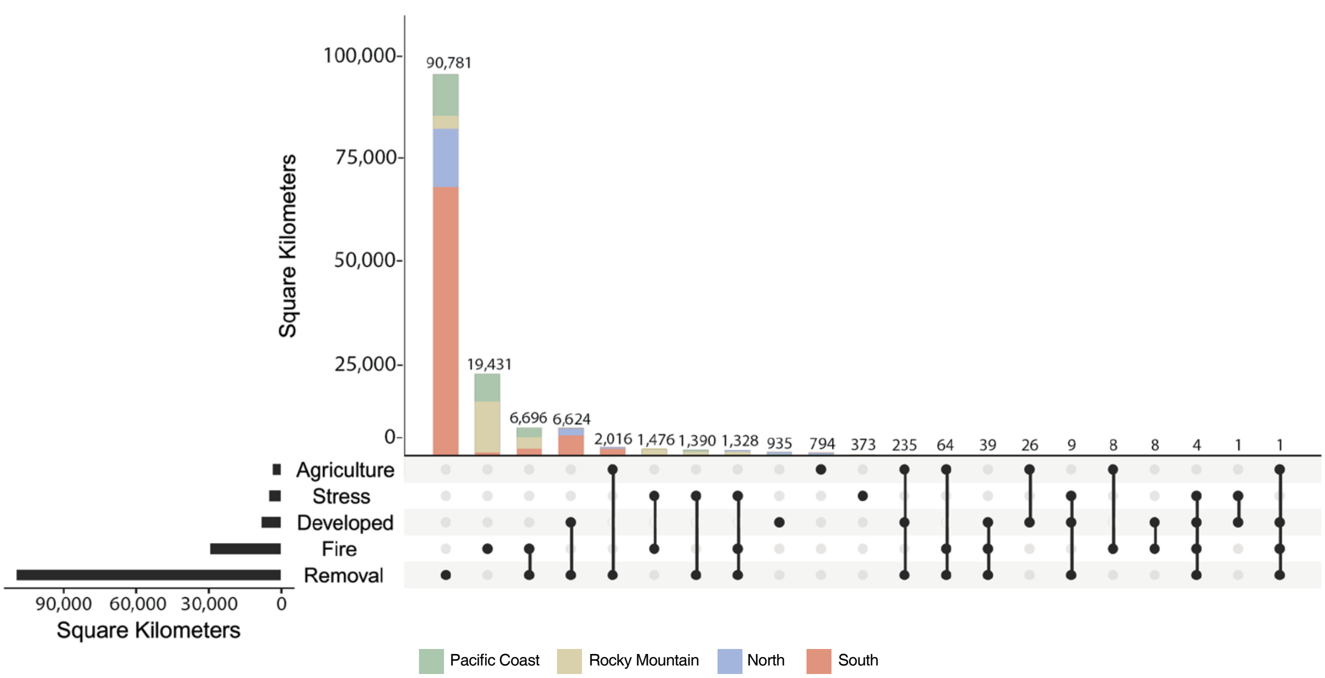
Congress passed the Clean Air Act Amendments of 1990 to reduce the impacts of acid rain by targeting sulfur and, to a lesser degree, nitrogen emissions (Greaver et al. 2012). Subsequent emissions reductions have decreased acid deposition substantially in all regions, from a nearly 25-percent reduction in the Rocky Mountain Region to an over 50-percent reduction in the North Region (figure 5-29). In numerous places, these reductions have eliminated critical load exceedances and allowed ecosystems to recover, some to the point of allowing the reintroduction of previously extirpated fish species (Sullivan et al. 2018, Sutherland et al. 2015) (figure 5-30c). In some locations, however, the severity of acid deposition and/or the sensitivity of the ecosystem created long-lasting effects that could continue to impact ecosystems into the future (Burns et al. 2020, Sullivan et al. 2018).

Future projections of acid deposition and its impacts have been made for both selected RPA climate futures: RCPs 4.5 and 8.5 (Clark et al. 2018a, Lamarque et al. 2010, 2011). Acid deposition is projected to continue to decrease under RCP 4.5 and, to a lesser extent, RCP 8.5, except for the Rocky Mountain Region under RCP 8.5 (figure 5-29). Projected increases in the Rocky Mountain Region are primarily driven by nitrogen deposition, which is more complicated than sulfur deposition with a broader suite of chemical compounds, sources, and effects (Galloway et al. 2004, Gruber and Galloway 2008). Although the Clean Air Act Amendments of 1990 decreased emissions of nitrogen compounds that contribute to acidification, emissions of other nitrogen compounds have continued to increase, complicating ecosystem recovery (Butler et al. 2001, Davidson et al. 2012, Sullivan et al. 2018). Projected decreases in acid deposition are expected to continue to decrease critical load exceedances and further reduce impacts to surface waters (figure 5-30d); however, the changing chemical composition of deposition means some ecosystems may experience additional impacts and a disrupted recovery. Research on air pollution impacts and the development of critical loads have enabled mapping impacts to ecosystem goods and services and developing projections of future impacts.

canopy cover loss attributed not only to removal, but also to fire and "stress" (drought, insects, diseases) (Schleeweis et al. 2020). Our analysis also included two types of disturbance from land-use change: increased agriculture and development from the National Land Cover Database (Homer et al. 2020; USGS 2019a, 2019b). Our estimates of the area with combined pressures in a 4.41-ha neighborhood are different from the disturbance areas reported elsewhere in this document. Here, we consider a disturbance process to have affected a particular forested location if that process was observed at that location or on forest nearby. We summarize disturbance occurrence only for areas where forest loss was observed, not for all forest land.

Ninety-four percent of pixels where forest cover was lost had at least one disturbance identified nearby, while two or more disturbance processes were identified near 15 percent of all forest loss locations. Removal was the most common disturbance process, occurring near a total of 109,187 km^2 (10.9 million ha) of forest cover loss (black horizontal bar in figure 5-31). Fire was next most common, occurring near 29,060 km^2 (2.9 million ha) of forest cover loss.

Figure 5-31. Summary of forest disturbance processes for locations with forest cover loss, 2001 to 2010. The figure depicts the occurrence of each process alone or in combination with one or more others. The horizontal black bars indicate the total area of forest cover loss that had each process in its local neighborhood, whether alone or in combination with another process. The vertical bars indicate the area of forest cover loss that had a unique combination of processes. The combinations captured in each vertical bar are depicted by black dots beneath the vertical bar, with a connecting line if two or more are included in the set.



Stress, conversion to developed land use, and conversion to agricultural land use were less common (<10,000 km^2 or <1 million ha each).

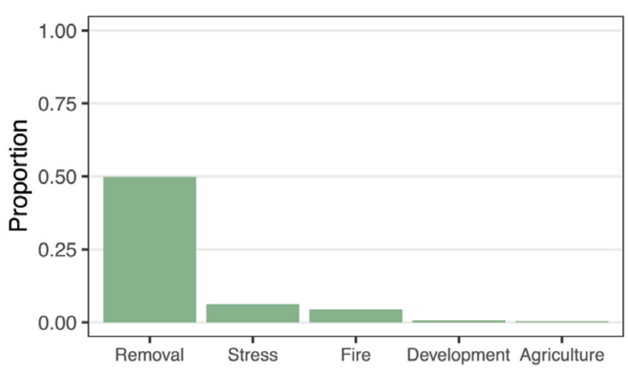
Removal occurred alone in 83 percent (90,781 km^2 or 9.1 million ha) of the places where it occurred (figure 5-31). Sixty-six percent (72,417 km^2 or 7.2 million ha) of the removal that occurred near forest cover loss occurred in the RPA South Region. Where removal co-occurred with other processes, it was found most often with either fire or increases in developed land use.

After removal alone, the next most common process near forest cover loss was fire alone, which occurred twice as often alone as with other processes (19,431 km^2 or 1.9 million ha versus 9,629 km^2 or 1.0 million ha). Sixty-two percent (11,988 km^2 or 1.2 million ha) of the places where fire events occurred alone near forest cover loss were in the RPA Rocky Mountain Region, with an additional one-third (6,510 km^2 or 651,000 ha) occurring in the Pacific Coast Region. When fire was observed with another process, it was found most often with removal.

Stress was observed near forest cover loss much less frequently than removal or fire, and co-occurred with removal, fire, or both processes 11 times more often than it occurred alone. The co-occurrence of stress with fire and removals reinforces other research that has found coincidence between insect outbreaks, drought, fire, and removal (Hood et al. 2017, Rhoades et al. 2018).

Like stress, increases in developed and agricultural land uses also occurred near other processes more frequently than by themselves. Conversion toward both of these land

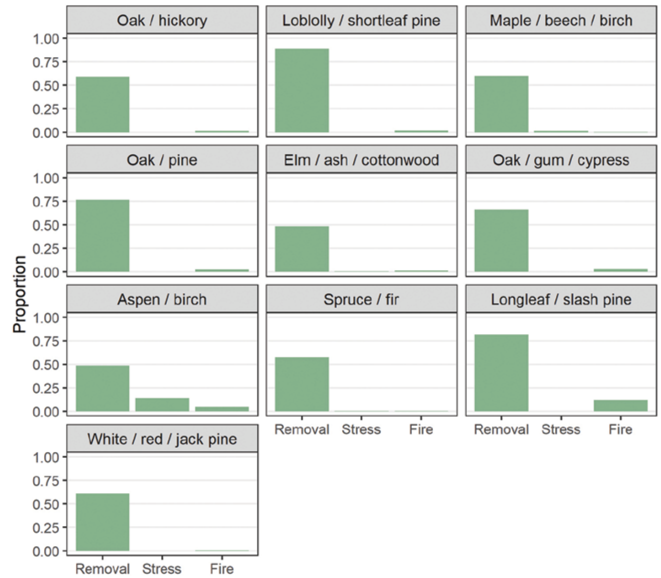
Figure 5-32. Proportion of FIA forest land exposed to removal, stress, fire, increase in developed land, or increase in agriculture observed within a 4.41-ha neighborhood from 2001 to 2010.



FIA = Forest Inventory and Analysis.

Sources: Removals, fire, and stress came from canopy disturbance attribution data for 2001 to 2010 and represent the proportion exposed to at least one event over that period (Schleeweis et al. 2020), while increase in agriculture and/or developed land uses came from NLCD data for 2001 to 2011 and represent the proportion exposed to at least one event over that period (Homer et al. 2020; U.S. Geological Survey 2019a, 2019b).

Figure 5-33. Proportion of FIA forest land in each FIA forest type group in the Eastern United States that was exposed to removal, stress, and fire events, 2001 to 2010. Exposure is defined as an observed loss of forest canopy within a 4.41-ha neighborhood surrounding FIA plot locations. Forest type groups are arranged by decreasing area from top left to bottom right (see figure 5-8 for areas). Some of the aspen/birch group occurs in the Western United States.



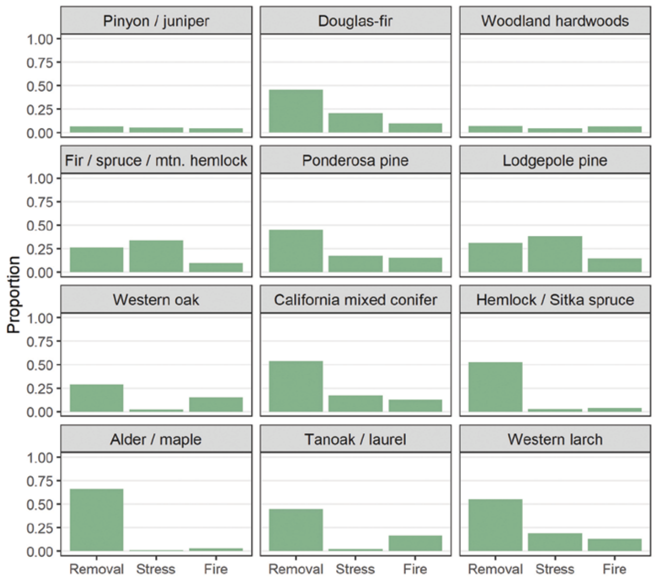
FIA = Forest Inventory and Analysis; ha = hectares.

Source: Schleeweis et al. 2020.

uses co-occurred most frequently with removal. While this analysis summarizes events occurring nearby one another during a 10-year period and not in sequence with one another at the same forested location, the co-occurrence of the two suggests that those removal events may be related to land use conversion. An increase in developed land use (alone or combined) was 2.5 times more common than increased agriculture (alone or combined) near places where forest was lost, suggesting that forest cover was more often lost for development than for agriculture.

The differences in the frequencies of these processes by region have important implications for forest loss and change. In the RPA South Region, removal alone was by far the most common process observed near forest cover loss, demonstrating forest management. While co-occurrence of removal and increased development was rare nationally, it occurred most often in the South Region, reflecting the fact that housing development is a comparatively frequent phenomenon in the region's forests (Radeloff et al. 2018). Similarly, removal alone and the co-occurrence of removal with increased development were the top two types of processes occurring near forest loss in the North Region. These results suggest that forests in the North and South Regions face similar pressures. However, the areas of forest

Figure 5-34. Proportion of FIA forest land in each FIA forest type group in the Western United States that was exposed to removal, stress, and fire events, 2001 to 2010. Exposure is defined as an observed loss of forest canopy within a 4.41-ha neighborhood surrounding FIA plot locations. Forest type groups are arranged by decreasing area from top left to bottom right (see figure 5-8 for areas).



FIA = Forest Inventory and Analysis; ha = hectares.

cover loss associated with these events were smaller in the North than in the South Region (figure 5-31), suggesting that forests in the South face these pressures more often. In the Pacific Coast Region, removal alone was the top process occurring near forest loss, but fire alone was a close second, followed by fire and removal together. The Rocky Mountain Region was the only region where the most common process was fire alone, rather than removal alone. This region has less merchantable timberland than other regions (Oswalt et al. 2019), a higher proportion of forest that is public or protected (Nelson et al. 2020), and more area burned during the period of observation (see the section Fire in Forests and Rangelands). The Rocky Mountain Region also contained the most observations of stress, alone and in combination with other processes, which reflects the high rates of insect and disease activity as well as drought in that region.

Exposure of All Forest Lands to Disturbance Processes

To gain insights about the degree to which all current forest land in the conterminous United States was exposed to disturbances occurring nearby, we applied a similar approach to existing FIA forest land (as opposed to forest loss areas). We summarized the proportion of FIA forest land area with each of the five forest canopy cover disturbance processes occurring within a 4.41-ha neighborhood from 2001 to 2010. This summary, reported by forest type group, is supplemented by a parallel analysis of "core" forest cover loss in the Land Resources Chapter. Exposure of forest land to removal during the period 2001 to 2010 was substantially higher than any other process: nearly half (49 percent) of forest land was exposed to at least one removal event from 2001 to 2010 (figure 5-32). By contrast, only 6.2 percent and 5.2 percent of forest land, respectively, was exposed to stress and fire. Even smaller portions of forest land were exposed to increases in developed and agricultural land uses (0.7 percent and 0.4 percent of forest land, respectively) (figure 5-32). This result highlights the common occurrence of removal events in forest land across the country (Cohen et al. 2016), whether for silvicultural or other purposes, and confirms the highly dynamic nature of forest cover documented in earlier RPA reports (Nelson et al. 2020). While locally important, increases in agriculture and developed land are relatively rare near FIA forest land overall (figure 5-32), and therefore excluded from further analyses.

The forest canopy disturbances described above occur in some forest types more often than others. Like the results for all forest land, many individual FIA forest type groups (figure 5-8) had a higher exposure to removal events than to any other process (figures 5-33, 5-34). Specifically, the forest type groups that are relatively widespread in the Eastern United States were among those with a high proportion exposed to removal and little or no exposure to fire and stress events (figure 5-33). Examples include the oak/hickory, loblolly/shortleaf pine, and maple/beech/birch groups, as well as the white/red/jack pine group, which has a smaller range (figures 5-8, 5-33). This result further underscores the relatively large areal footprint of removal in the Eastern United States. Eighty-nine percent of the commercially important loblolly/shortleaf group was exposed to removal nearby over the 10-year period. Relatively high exposure to removal is not unexpected in this group, and removal for timber harvest is usually quickly followed by replanting and intensive management (Drummond et al. 2017). While fire may occur relatively frequently in some of those eastern forest types, it generally is of low enough severity not to disturb the forest canopy and therefore largely does not appear in the eastern type groups. The longleaf/slash pine group is a notable exception, having 12 percent of total area exposed to fire over the 10-year period, likely because frequent fire is important for maintaining ecosystem function and biodiversity (Peet et al. 2018). The aspen/birch type group was the only eastern group with notable exposure to stress (14 percent), but some of that type group also occurs in the Western United States.

Forest type groups occurring primarily in the Western United States tended to have greater exposure to fire and stress events than those occurring primarily in the Eastern United States (figure 5-34). This result is consistent with the high rates of large, high-severity wildfires, drought, and insect disturbances shown for the western regions in the earlier sections of this chapter. The Douglas-fir, ponderosa pine, and California mixed conifer type groups had higher exposure to stress and fire than any of the eastern type groups, while still having relatively high exposure to removal, underscoring the multiple pressures those forests face. The fir/spruce/mountain hemlock and lodgepole pine type groups were also exposed to all three forest canopy threats, with exposure to stress being highest for both groups during the 10-year period. The hemlock/Sitka spruce and alder/maple groups had relatively low exposure to both stress and fire, as expected given the distributions of those forest type groups in relatively moist sites. The pinyon/juniper and woodland hardwoods type groups had low exposure to all three canopy disturbance types; however, we know that these forests are subject to disturbance events including drought, as shown in the section Drought in Forests and Rangelands. Given that the forest canopy is often relatively sparse in these forest types, disturbance events may not always lead to measurable loss of the forest canopy, meaning that those disturbance events are likely not well captured in this exposure analysis for these forest type groups.

While this analysis focused on exposure of forests and forest type groups to disturbance, the results can be used in conjunction with information on the sensitivities of these forests to the disturbance processes to determine the ecological or economic impacts of these disturbances. One example of demonstrated high sensitivity to multiple disturbance processes occurs in dry portions of Douglas-fir and ponderosa pine forests of the Western United States, where high-severity wildfires combined with warm and dry climate can cause tree regeneration failure and subsequent conversion to nonforest (Coop et al. 2020, Davis et al. 2019, 2020). Forest type groups represent assemblages of tree species, each with its own disturbance sensitivities to consider. As a result, shifts in forest species composition may be likely because of differential responses of tree species to these disturbance processes. Summaries of these disturbance processes at a finer level of forest classification, such as by species, or within more restricted areas, would likely allow for more insight about how these disturbances affect forests. In addition, summaries of exposure of FIA forest land to additional disturbances not included here, such as hurricanes and other storms and sea level rise (see the sidebar Sea Level Rise Effects on Forests for a synthesis of forest impacts) would provide a more holistic picture of the disturbances and stressors facing our forests.

Management Implications

Disturbance is relevant to both management and policy, especially as climate changes, human populations increase, and developed land use expands. Management actions, policies, and initiatives can help restore natural disturbance regimes, where appropriate, and increase the capacity of forests and rangelands to adapt to changing regimes or recover following disturbance. In those ways, management can reduce the vulnerability of forests and rangelands to disturbances themselves and help increase the resilience of those ecosystems to climate change and other global change drivers. As in the case of removals, however, management actions can themselves be considered disturbances. While some management implications of single disturbance types in forests or rangelands have been mentioned throughout this chapter, a few cross-cutting ideas apply.

In some places, management of forests and rangelands to mitigate multiple disturbances may be desirable. Our analysis shows that forests in the RPA Pacific Coast Region may be particularly exposed to multiple co-occurring disturbances. Dry forests of California have experienced recent tree mortality due to interactions of drought, wildfires, and bark beetles (Fettig et al. 2019). Forest thinning and prescribed fire together have reduced the effects of those interacting disturbances (Knapp et al. 2021). Similarly, fuel treatments like thinning in forests of the Pacific Northwest may help increase resilience to fire, insects, and drought, and facilitate post-disturbance recovery (Halofsky et al. 2020).

As the characteristics of disturbances and disturbance regimes change—becoming more severe, more frequent, longer in duration, or spreading to previously unaffected ecosystems—they could challenge the effectiveness of existing management techniques and paradigms, and may force changes or adjustments. For example, management actions that include accepting a range of fire severities when and where they are safe, reducing wildfire occurrence in the wildland urban interface (WUI), and improved planning of residential communities to avoid or withstand wildfires may be appropriate in the Western United States, where climate and land-use change are increasing both the total area burned by wildfires and the area burned in the WUI (Calkin et al. 2014, Kelly et al. 2020, Radeloff et al. 2018, Schoennagel et al. 2017). In rangelands, managers are searching for novel approaches to curb the spread of nonnative annual plants, especially cheatgrass (*Bromus tectorum*) and red brome (*Bromus rubens*), to break the annual grass-fire cycle. Incorporating more flexible grazing strategies, specifically targeted grazing that aims to reduce the cover of these species, shows promise, and the USDA Forest Service and U.S. Bureau of Land Management are increasingly looking for ways promote and expand targeted grazing. Doing so faces several challenges, including increased flexibility in grazing allotment administration. New technologies such as the Rangeland Production Monitoring System (Reeves et al. 2020, 2021) are part of a strategic support system that may help managers detect nonnative grasses and identify targeted grazing opportunities.

In addition to changing disturbance regimes, the ability for professionals to conduct management to mitigate larger or more severe disturbances and increase ecosystem resilience may also be affected by global change drivers. As the area and severity of wildfires increases and the WUI expands in the Western United States, wildfire management is becoming more challenging. Prescribed burning is already becoming more difficult in some places, at least in part due to climate and land use change, and increased challenges are projected in the future. Reductions in the number of days with suitable meteorological conditions for prescribed burning are projected in the future in the South Region (Kupfer et al. 2020), suggesting that decreases in the area burned are likely, especially as the expanding WUI places additional challenges on burning (see the sidebar COVID-19 as a Constraint on Prescribed Burning in the Southeastern United States for more information on recent challenges). Such reductions in wildfire management, prescribed burning, or any other management, can result in forests and rangelands that are less resilient through time, having concomitant effects on the resulting resources and services.

Partnerships and collaborations among scientists, managers, and public and private landowners can help address the increasing need for management, growing challenges associated with management, and uncertainties in future conditions (Glick et al. 2021). Adaptive silviculture for climate change is an effort among scientists and managers to identify the management actions that are likely to increase the adaptive capacity of forests to the effects of changing climate, including disturbance (Nagel et al. 2017). Several recent initiatives involving the USDA Forest Service have aimed to create partnerships among agencies to identify treatments and other management actions to meet multiple objectives, including reducing risk of wildfire and other disturbances (USDA Forest Service 2018). These initiatives include the Collaborative Forest Landscape Restoration Program, the Wildfire Crisis Strategy, and the Shared Stewardship Strategy. In rangelands, the ecological and economic threat that invasive grasses pose to local communities has inspired an unprecedented level of cooperation among land managers, nonprofits, government agencies, and the business community. One example of a cooperative model is the Southern Arizona Buffelgrass Coordination Center, which uses cross-jurisdiction coordination and community engagement to help control buffelgrass (*Pennisetum ciliare*), an invasive perennial threatening several rangeland ecosystems. Fostering more cooperation and coordination throughout U.S. rangelands may be beneficial in the future, as increased frequency and duration of drought combine with invasive species to exacerbate changes in fire regimes in many places. Partnerships, especially when conducted at large scales or when replicated in different regions, could benefit future management of a wide variety of disturbances in forests and rangelands.

Conclusions

Disturbance is a constant presence in many forest and rangeland ecosystems. For the first time in an RPA Assessment, the analysis in this chapter provides a comprehensive look at the recent, and in a few cases, future disturbances in both forests and rangelands across the United States. Our results highlight that many of these disturbances are becoming more frequent, widespread, or severe over time, and that regional variability exists in the type, amount, and intensity of disturbances that occur in forests and rangelands.

In terms of recent historical trends, the average annual area burned by fire in both forests and rangelands has increased nationwide and in all RPA regions except the North Region. Drought exposure has been high in forests and rangelands in the West, particularly the Pacific Coast Region. Nonnative invasive plants have been most prevalent in forests near agricultural and developed areas in the East, and in rangelands within counties in California. In addition to the direct exposure of forests to disturbances, many forests exist in dynamic landscapes that experience multiple disturbance pressures, including combinations of removals, stress, and fire, as well as conversion of land use to agriculture or development.

Looking ahead to 2070, the disturbance types discussed in this chapter have the potential to become more frequent, widespread, or severe in many locations (with the notable exception of acid deposition in forests, see the sidebar Effects of Air Quality on Forest Ecosystems). Forest mortality from fire is expected nationwide and within each RPA region. Increases in the area of moderate- and high-severity fire are also projected in many locations, especially in the RPA Pacific Coast and South Regions. Forest and rangeland exposure to drought is projected to increase as well, particularly for ecosystems in the Southwest. While not explicitly projected, literature summarized in this chapter suggests potential for increasing threats from insects and disease and nonnative invasive plants.

The Nation's forests and rangelands face pressures from these disturbances against a backdrop of changing climate, socioeconomic conditions, and land use. These disturbances, alone and in concert, are affecting forests and rangelands and the goods and services they provide. For example, fire and drought together are already transforming some dry forests to grasslands in the Western United States, and the co-occurrence of drought with extreme heat preceded forest mortality and reduced rangeland production in

Texas. The magnitude of disturbance impact on ecosystems, however, can vary with a number of factors, including species composition and landscape characteristics. Not all fires are threats to forests or rangelands, and not all forests or rangelands have the same vulnerability to drought. These additional factors are relevant to comprehensive assessment of effects on forests and rangelands. The impacts from disturbance can also be affected by management, as increasing evidence is pointing to the importance of actions like prescribed fire and thinning for improving the resilience of forests to disturbance and other global change drivers.

Disturbances are integral parts of forest and rangeland ecosystems that affect the goods and services those ecosystems provide. Disturbances are likely to continue to increase in many locations, especially as climate changes, human population increases, and developed land use expands. Information about status and trends in these disturbances over time informs forest and rangeland management that can better facilitate adaptation of the Nation's forests and rangelands to global change.

Sea Level Rise Effects on Forests

Past and Future Sea Level Rise

Thermal expansion of ocean waters and glacial and ice sheet melting, both consequences of global warming, have contributed to sea level rise (SLR) over the past 200 years. Studies indicate that the pace of global mean SLR has accelerated in the recent past, from about 0.05 inches per year during 1901 to 1990 to 0.12 to 0.14 inches per year during the period 1993 to 2010 (Dangendorf et al. 2017, Hay et al. 2015). While the rate of future SLR depends on global temperature change, current projections are for global mean sea level to rise by 0.4 to 2.5 m by 2100 (Oppenheimer et al. 2019).

Coastal forest retreats, replacement of coastal forests by saltmarsh, and the appearance of ghost forests (dead trees adjacent to marshes) due to SLR have already been observed on low-lying coastal and estuarine landscapes (Kirwan and Gedan 2019). Future SLR could lead to permanent inundation, increased frequency and intensity of flooding from storm surges, increased coastal erosion, and expanded saltwater intrusion into the soil, groundwater, and freshwater systems. This, in turn, will result in loss, alteration, and degradation of coastal ecosystems and natural resources, including forests and wetlands (Kirwan and Gedan 2019, Schuerch et al. 2018), which can have indirect effects on the forest sector, including altered supply and demand conditions in markets for ecosystem services and forest goods.

Direct Effects of SLR on Coastal Forests

Direct effects of SLR on forests include: (1) loss of coastal forests due to flooding and extreme sea level events such as storm surges and tidal waves, and (2) altered structure, composition, growth, regeneration, and productivity of coastal forests due to saltwater intrusion, impeded drainage, and flooding. The availability of current and future space for coastal forest retreat is a critical factor determining future gain or loss of such ecosystems and is affected by many factors, such as the economic factors driving coastal land use changes (Kirwan and Gedan 2019, Schuerch et al. 2018).

Two types of coastal forests can be distinguished for the purpose of describing SLR effects: estuarine coastal forests that are adapted to saltwater (e.g., mangrove, beach, and peat swamp forests), and freshwater coastal forests that cannot tolerate salt. The effects on and likelihood of losing coastal forest differs between these two types of forest.

The effects of SLR on coastal forests that are adapted to saltwater are projected to be minimal at the current and projected mid-century SLR, although several studies suggest that mangrove forests are threatened in many parts of the world and are not keeping pace with local SLR rates (Friess et al. 2019). For example, in the tropics under the high-warming scenario (RCP 8.5), relative SLR is expected to exceed the tolerance of mangroves because rates of SLR in the tropics are expected to be higher than the global average (Saintilan et al. 2020). The likelihood of losing coastal forests to SLR depends on many factors, such as the local rates of SLR and the rate of sediment accretion for these ecosystems.

Limited research is available on the effects of SLR on freshwater coastal forests, and most of our understanding is based on research conducted in the United States. Increasing saline and frequent flooding are thought to cause declines in forest health and productivity, basal area and tree density, species diversity, seed germination and regeneration, and increased tree mortality (Grieger et al. 2020). Ghost forests are also reported primarily along the Atlantic coast of North America, where SLR is currently occurring at a rate greater than the global average (Kirwan and Gedan 2019, Smart et al. 2020). The likelihood of losing these coastal forests to SLR will depend on local rates of SLR, rate of saltwater intrusion into the groundwater, species composition, and tolerance to saltwater especially for regeneration.

Indirect Effects of SLR on the Forest Sector

The indirect effects of SLR on the forest sector include dynamics that are tied to changes in supply and demand for forest goods and ecosystem services. SLR-induced losses in forest area are likely to affect forest product markets by reducing the overall availability of timber, leading to a combination of reduced timber product output and higher timber prices. At the same time, about 350 to 480 million people globally are projected to be exposed to SLR by 2100 (Kulp and Strauss 2019), requiring replacement of their present dwelling. As a result, demand for wood products for new housing is likely to increase (Desmet et al. 2018, Hauer et al. 2020, Nepal et al. 2022).

Increased demands for wood to rebuild could affect not only coastal regions but also noncoastal timber-growing regions through altered harvesting activity, changing local market conditions, and altered international flows of traded forest products (Nepal et al. 2022). Higher product demands by the construction sector can lead to increased forest product prices, which can affect the competitive advantage of a country or a region to harvest timber and to produce, consume, and trade in forest products. Price increases also provide an economic incentive to keep forests as forests or to invest in intensified forest management activities such as thinning or fertilization (e.g., Daigneault and Favero 2021, Nepal et al. 2019). Changes in timber harvests, forest management, and wood products manufacturing activities, indirectly induced by SLR through increased prices, may have additional consequences for net carbon emissions mitigation by the forest sector. Mitigation potential would be affected through changes in the total quantities of carbon stored in forests and in harvested wood products. Likewise, mitigation potential would also be affected by avoided fossil carbon emissions resulting from substitution of wood for more carbon-emissions-intensive nonwood materials in construction, such as steel or concrete (Leskinen et al. 2018, Nepal et al. 2016, Nepal et al. 2022, Sathre and O’Connor 2010). As shown by Nepal et al. (2022), increased global harvests to accommodate higher wood product demand for rebuilding SLR-destroyed residential structures would shrink global forest carbon by up to 2.0 percent. However, policies favoring rebuilding destroyed residential structures with wood construction materials worldwide could reduce global CO_2 equivalent emissions by 0.47 to 2.13 tons per ton of CO_2 equivalent carbon contained in those additional wood construction materials. This emissions reduction was connected most directly to the replacement of fossil fuel-intensive building products with wood.

Assessing the Future Effects of Sea Level Rise on Coastal Forests: Critical Needs

Coastal forests provide a wide variety of ecosystem services globally, including provisioning (fisheries, fuel, water supply, tourism, and cultural resources), regulating (coastal protection, carbon sequestration, sustaining biodiversity) and supporting (soil, sediment and sand formation, nutrient cycling, habitat). In addition to altering existing coastal forests, future SLR could disrupt local economies and even result in humanitarian crises around the world. Advancing science on the effects of SLR on coastal forests is critical for assessing the effects and designing adaptation strategies.

Improved understanding and representation of coastal processes and feedbacks in global forest sector models would provide better information on sea level rise from local to global extents, and on its interactions with projected loss or gain of coastal ecosystems (Ward et al. 2020). On the local level, better understanding of how SLR affects groundwater salinity and the gradual losses of coastal forests is needed. Scientific evidence on the linkages between SLR-related coastal forest health and other forest disturbances (e.g., cyclones, insects and diseases, invasive species, and wildfires) is limited yet critical for assessing the full set of potential impacts of SLR. Establishing the effects of sea level rise on habitat for aquatic and terrestrial wildlife is also a critical need.

Coastal forest conservation efforts could benefit from additional research on the potential feasibility and outcomes of alternative coastal forest conservation strategies, including protection and expansion of open spaces to enable coastal ecosystem migration, engineering approaches that might include the creation of physical structures, and assisted migration of coastal ecosystem species. Research could additionally explore how such strategies could be implemented through possible incentives. Furthermore, because the effects of SLR are not restricted to coastal areas, scientific analysis could focus on how the losses of residential and other structures could affect forest land in locations away from coasts.

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