Chapter 2

Effects of Climatic Variability and Change

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Contributors

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Introduction

Climate profoundly shapes forests. Forest species composition, productivity, availability of goods and services, disturbance regimes, and location on the landscape are all regulated by climate. Much research attention has focused on the problem of projecting the response of forests to changing climate, elevated atmospheric carbon dioxide (CO_2) concentrations, and nitrogen deposition, deepening our understanding since the publication of the last forest sector assessment (Ryan et al. 2008). We have many new examples of how changes in climate over the period 1971–2000 have affected forest ecosystems, including long-term monitoring data on forest change, multifactor experiments that document the potential interactions between temperature and elevated CO_2, and new modeling approaches that project the effect of projected changes in climate on forest ecosystems, their goods and services, and their disturbance regimes. Climate projections are being done on a finer spatial scale, and global climate models include more detail and feedbacks with terrestrial processes. Downscaled estimates from these models are more readily available and have been used for more regional and local assessments. Despite the large amount of new research, this new information has not substantially altered the primary projections made in the last assessment (Ryan et al. 2008). In this assessment, we have added more detail about the effects covered in the last assessment (especially altered disturbance regimes and potential effects on hydrologic processes), provided more information about regional effects, and covered additional topics.

Climate change, higher CO_2 concentrations, and increased nitrogen (N) deposition have already significantly affected the Nation’s forests. These effects are projected to get even larger in the future as the climate warms throughout this century and moves further from the historical climate. Although projecting the response of forest ecosystems to global change is difficult and complex, we have a high degree of confidence in many of the projections made for larger scales and for the next few decades. Our confidence comes from the observed changes that have occurred in response to the relatively small changes in climate over the past 30 years. Predicted future climate will likely bring even

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more dramatic effects, because temperatures are expected to be 2.5 to 5.3 °C warmer than in 1971 to 2000, and large effects have been seen with less than 1 °C warming over the past 30 years. For example, snowpack is melting earlier in the spring, forest fires are becoming larger, bark beetles are moving higher in elevation and attacking species that were climatically protected in the past, bark beetle and other insect outbreaks have become larger and more frequent without very cold winters to stop them, and drought has killed trees in the drier regions of tree species' ranges. For many factors, the aggregate ecosystem response over large areas is well understood, perhaps even better understood than the projections of future climate, which differ from model to model, are less certain about precipitation than temperature, and have less certain regional and local projections.

Sometimes, we do not know enough about the science to make good projections. For example, how do increased temperature and drought interact to affect tree mortality? Will mature trees respond to elevated CO_2? For these problems, further research will improve our projections. In addition, many outcomes rely on complex interactions and contingencies, making projections difficult, highly uncertain, or sometimes, impossible. Some of the projected climates will be novel, with no historical analog and hence, we have limited experience or data on how ecosystems might respond. Trees are long-lived organisms and individuals of some species may remain in place long after an altered climate would favor the establishment of different species. This is because seeds for replacement species may not move into an altered environment, so the best-adapted species to the new climate may not be available. The interaction of climate and disturbance will substantially alter forest ecosystems. As a result, species and forest ecosystem processes may not have time to adapt to a rapidly changing climate, and multiple disturbance and stressor interactions will make it even more difficult to understand and project responses to climate change.

Predicting outcomes for a particular location is very uncertain, because in general, projections of future climate and ecosystem response for a given area are very uncertain. Over a very large area, patterns that are obscured by interannual variability at an individual location begin to emerge. For example, in the Western United States, the annual area burned by fire has increased and snowmelt has occurred earlier as temperatures have warmed. However, projecting how fire or snowmelt will be affected at the local or forest-stand scale is much more subject to contingencies and local factors that were not assessed in developing regional relationships, making the projections very uncertain at smaller spatial scales.

Predictions for the long term are also uncertain. Projections of future climate differ among both the global climate models used and the different emission scenarios. For ecosystems, longer time periods allow more time for contingencies and unanticipated factors to shape the future, adding additional uncertainty.

In this chapter, we review studies that were either published after the last forest assessment (Ryan et al. 2008) or not previously covered. We summarize the state-of-knowledge on projected changes in future climate. Next, we discuss the potential effects of climate change on disturbance regimes and forest processes and their interactions. Finally, connections between biophysical responses and socioeconomic responses are discussed in the context of ecosystem services.

Projected Changes in Future Climate

Scenarios for Projecting Future Climate

Projected changes in future climate are based on output from 15 global climate models (GCMs) (box 2.1). All model runs used future scenarios of economic growth, population growth, and greenhouse gas emissions scenarios that were intended to represent the high (A2) and low (B1) ends of future emissions. A2 describes a world with continuous high population growth, slow economic development, slow technological change, and independently operating, self-reliant nations. B1 describes an environmentally friendly world with an emphasis on global solutions to economic, social, and environmental stability; a global population that peaks in mid-century and then declines, and with rapid changes in the economy toward a service and information economy, reductions in material intensity, and the introduction of clean and resource-efficient technologies. For models of effects, some additional scenarios and GCMs were used in this report and are noted where appropriate.

Trends in temperature and precipitation from weather stations show that the United States has warmed over the past 100 years, but the trends differ by region (Backlund et al. 2008). The southeastern United States has cooled slightly (<0.7 °C), and Alaska has warmed the most (~4.5 °C); other Northern and Western U.S. regions also show a warming trend (~1.5 °C). Much of the Eastern and Southern United States now receives more precipitation than 100 years ago, whereas other areas, especially in the Southwest, now receive less (Backlund et al. 2008).

Temperature Projections

Average annual air temperatures across the continental United States are likely to steadily increase over the next century under the two emission scenarios (fig. 2.1). Compared to 1971 through 2000, average annual air temperature will likely increase from 0.8 to 1.9 °C by 2050, from 1.4 to 3.1 °C by 2070, and from 2.5 to 5.3 °C by 2099. The range

Box 2.1—Global climate models and emission scenarios

Most of the climate projections used to describe future climatic conditions in this report are based on model ensembles, that is, syntheses of the output of various global climate models (GCMs).[3] The report includes output from four specific GCMs, as summarized below:

CCSM2 (Community Climate System Model, version 2)—U.S. National Center for Atmospheric Research (http://www.CESM.NCAR.edu).

CSIRO Mk3—Australian Commonwealth Scientific Industrial Research Organisation (Gordon et al. 2002).

Hadley (versions 1 to 3)—United Kingdom Hadley Center (Burke et al. 2006).

PCM (Parallel Climate Model)—U.S. National Center for Atmospheric Research (Washington et al. 2000).

This report also uses terminology that refers to standard greenhouse gas (GHG) emission scenarios as described by the Intergovernmental Panel on Climate Change (IPCC). Emission scenarios cited in the report are described below, in which A scenarios have higher GHG emissions and higher projected temperature increases than B scenarios.

A2—A2 scenarios represent a more divided world, characterized by independently operating, self-reliant nations; continuously increasing population, and regionally oriented economic development.

A1F1—A1 scenarios represent a more integrated world, characterized by rapid economic growth, a global population that reaches 9 billion in 2050 and then gradually declines, quick spread of new and efficient technologies, a world in which income and way of life converge between regions, and extensive social and cultural interactions worldwide. A1F1 emphasizes the use of fossil fuels.

A1B—Same as A1F1, except it emphasizes a balance of energy sources.

B1—B1 scenarios represent a more integrated, ecologically friendly world, characterized by rapid economic growth as in A1, but with rapid changes toward a service and information economy, population rising to 9 billion in 2050 and then declining as in A1, reductions in material intensity and the introduction of clean and resource efficient technologies, and an emphasis on global solutions to economic, social, and environmental instability.

B2—B2 scenarios represent a more divided but more ecologically friendly world, characterized by continuously increasing population but at a slower rate than in A2; emphasis on local rather than global solutions to economic, social, and environmental instability; intermediate levels of economic development; and less rapid and more fragmented technological change than in A1 and B1.

The forthcoming Fifth IPCC Assessment, scheduled for publication in 2014, will use representative concentration pathways (RCP) rather than the emission scenarios that were used in the Fourth Assessment (Solomon et al. 2007). The RCPs are four GHG concentrations (not emissions), named after a possible range of radiative forcing (increased irradiance caused by GHGs) values at the Earth's surface in the year 2100: RCP2.6, RCP4.5, RCP6, and RCP8.5, which represent 2.6, 4.5, 6.0, and 8.5 $W \cdot m^{-2}$, respectively (Moss et al. 2008). Current radiative forcing is approximately 1.6 $W \cdot m^{-2}$, which is equivalent to a global-scale warming effect of 800 terawatts.

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[3] Kunkel, K.E.; Stevens, L.E.; Sun, L. [et al.]. [N.d.]. Climate of the contiguous United States. Tech. Memo. National Oceanographic and Atmospheric Administration. On file with: North Carolina State University, 151 Patton Avenue, Asheville, NC 28801.

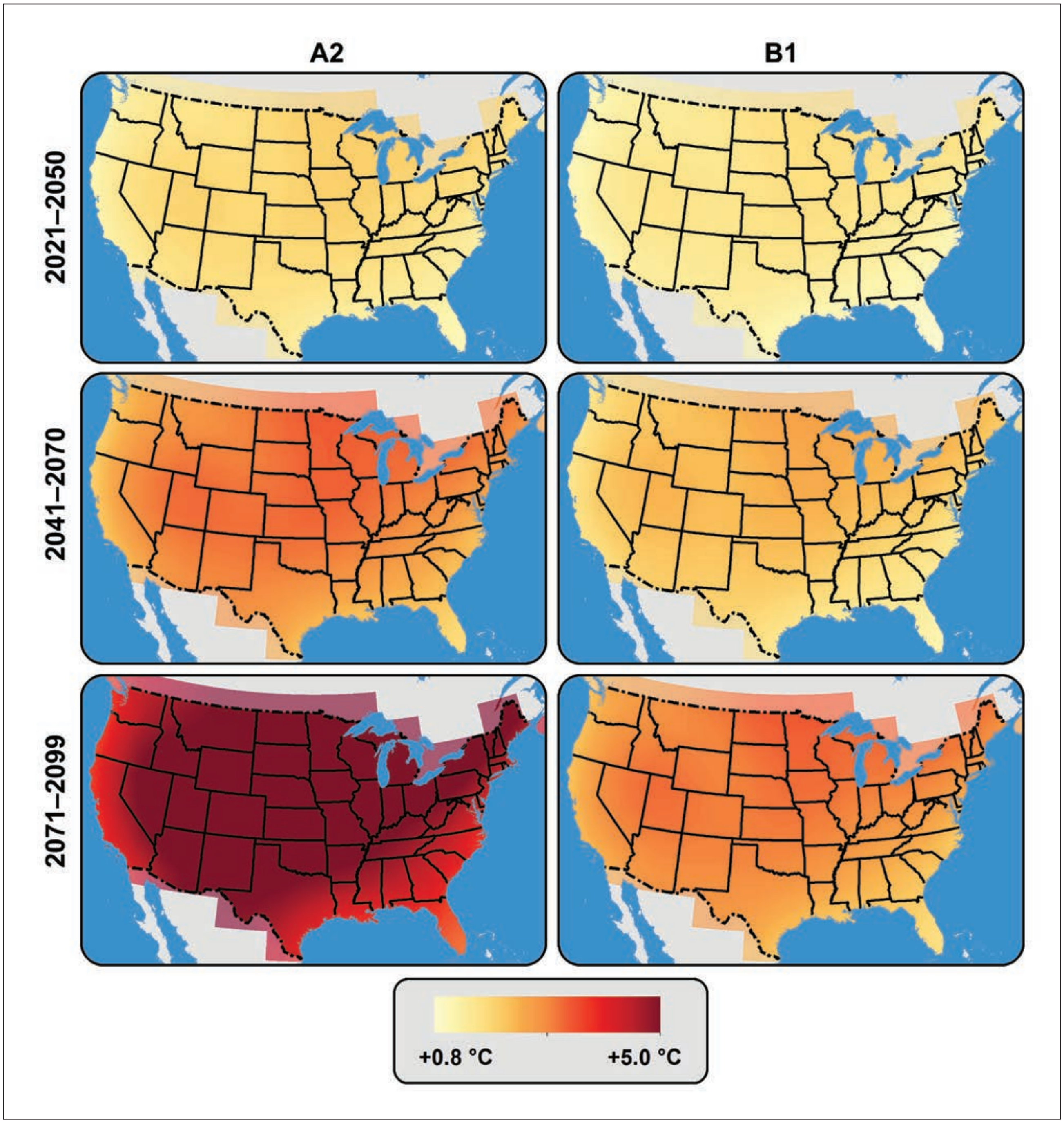

Figure 2.1—Multimodel mean annual differences in temperature between the three future periods compared to 1971 to 2000, from 15 global climate models using two greenhouse gas emission scenarios (A2 and B1). The A2 scenario is for higher greenhouse gas emissions than for B1 (see text). For most interior states, models project a 1.4 to 1.9 °C temperature increase, rising to 2.5 to 3.6 °C for 2051 to 2071, and to > 4.2 °C for 2071 to 2099, depending on the emission scenario. (Kunkel, K.E.; Stevens, L.E.; Sun, L. [et al.]. [N.d.]. Climate of the contiguous United States. Manuscript in preparation. On file with: NOAA's National Climate Data Center, 151 Patton Avenue, Asheville, NC 28801.)

of these estimated temperatures is bounded by the B1 and A2 emission scenarios. Within each scenario, the magnitude of increase depends on both latitude and proximity to coastal areas. Greater warming is projected in more northern and interior locations. For example, the largest temperature increases are projected for the upper Midwest, and the smallest temperature increases are projected for peninsular Florida. Seasonally, these two constraints on the magnitude of warming are also apparent. For the higher emission scenario, the least amount of warming is expected for autumn (1.9 to 3.1 °C) and spring seasons (1.4 to 2.5 °C). Winter season shows the most pronounced warming across the United States, with little change across the South and increases up to 3.6 °C in the North. During the summer, greater warming is projected for more interior locations (up to 3.6 °C warming across the central United States from Kentucky to Nevada).

In addition to overall warming over the next century, both the number of days when maximum temperatures exceed 35 °C and when heat waves occur (defined as the number of consecutive days with maximum temperatures

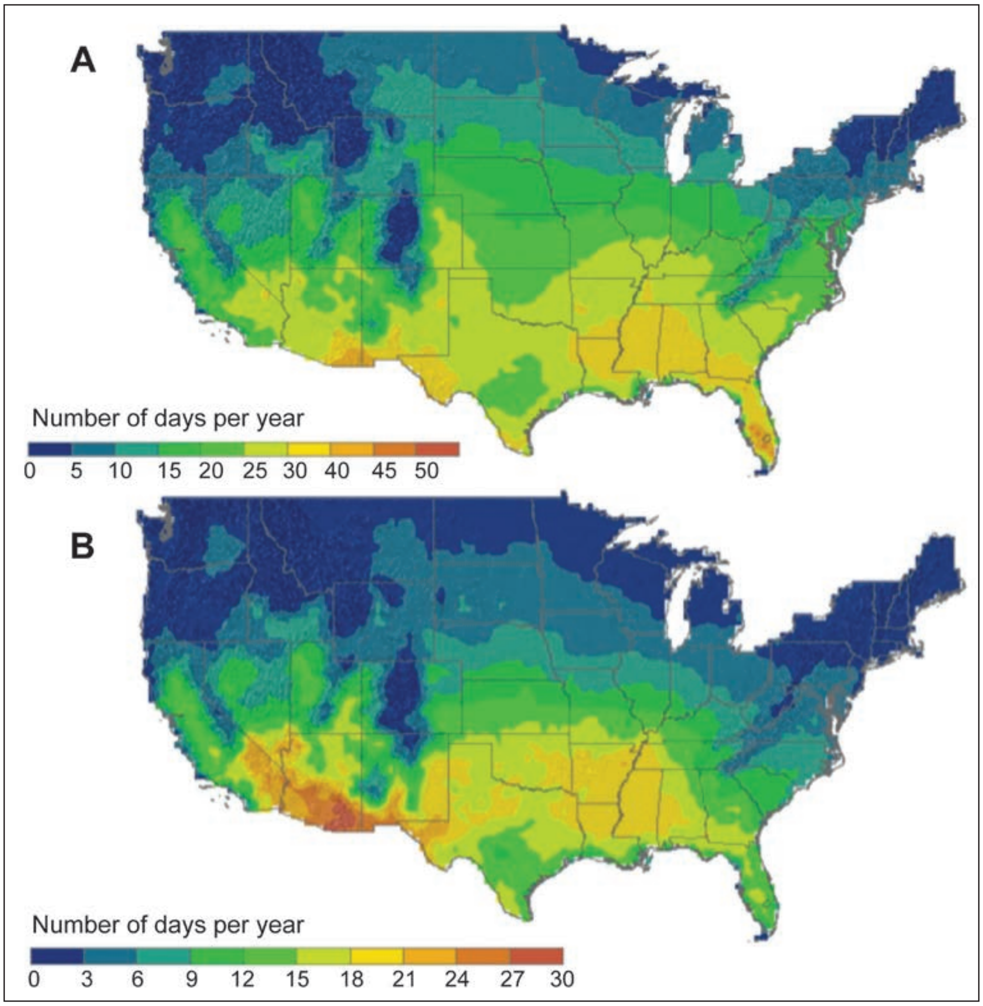

Figure 2.2—Spatial distribution of the mean change in the annual number of days with a maximum temperature above 35 °C (A), and in the annual number of consecutive days with a maximum temperature greater than 35 °C (B) between 1971 to 2000 and 2041 to 2070. Models project that much of the Southeastern and Southwestern United States will experience more days with maximum temperature exceeding 35 °C, and longer runs of those days. Results are for the high (A2) emission scenario only, from the North American Regional Climate Change Assessment Program multimodel means (n = 9 GCMs). (Kunkel, K.E; Stevens, L.E.; Sun, L. [et al.]. [N.d.]. Climate of the contiguous United States. Manuscript in preparation. On file with: NOAA's National Climate Data Center, 151 Patton Avenue, Asheville, NC 28801.)

exceeding 35 °C) are likely to increase over the next century (fig. 2.2). Under the higher greenhouse gas (GHG) emission scenario, the southeast will likely experience an additional month of days with maximum temperatures exceeding 35 °C, and the Pacific Northwest and Northeast regions will likely experience 10 more of these days per year. Under future GHG emission scenarios, the United States will likely experience longer heat waves. In the Southwest, the average length of the annual longest heat wave will likely increase by 20 days or more. Little or no change is predicted for heat waves in the northwest, northeast and northern parts of the Great Plains and Midwest regions. Most other areas will likely see longer heat waves of 2 to 20 additional days.

Precipitation projections—

Precipitation differs even more than temperature across the United States and through seasons and years. Any long-term trends in precipitation are less apparent within the high variation across years and decades. Observed data from the past century across the United States show that mean annual precipitation has significant interannual variability, with two particularly dry decades (1930s and 1950s) followed by a few relatively wet decades (1970–99); the overall result is a century-long increase in precipitation (Groisman et al. 2004).

Over the next century, multimodel mean projections of precipitation across the entire United States generally predict little or no net change in precipitation, although the variance among models is high (fig. 2.3). Some models predict a significantly drier future (at least in some regions), and others a significantly wetter future. The agreement among models in the future forecasts for precipitation is high for some models (Solomon et al. 2007). For example, there is general consensus among GCMs that annual precipitation in the Southwest will decrease by 6 to 12 percent (fig. 2.4), whereas precipitation in the northern states will increase by 6 to 10 percent (Easterling et al. 2000a, 2000b; Groisman et al. 2004; Huntington 2006; Pachuri and Reisinger 2007; Solomon et al. 2007).

Many regions of the United States have experienced increases in precipitation extremes, droughts, and floods over the last 50 years (Easterling et al. 2000a, 2000b; Groisman et al. 2004; Huntington 2006; Pachuri and Reisinger 2007; Solomon et al. 2007). In most GCMs, as the climate warms, the frequency of extreme precipitation events increases across the globe, resulting in an intensification of the hydrologic cycle (Huntington 2006). For example, the upper 99th percentile of the precipitation distribution is projected to increase by 25 percent with a doubling of CO_2 concentration (Allen and Ingram 2002). The timing and spatial distribution of extreme precipitation events are among the most uncertain aspects of future climate scenarios (Allen and Ingram 2002, Karl et al. 1995).

Drought projections—

As the climate warms from increasing GHGs, both the proportion of land experiencing drought and the duration of drought events will likely increase (Burke et al. 2006). The spatial distribution of changes in drought over the 21st century using the A2 scenario predicts significant

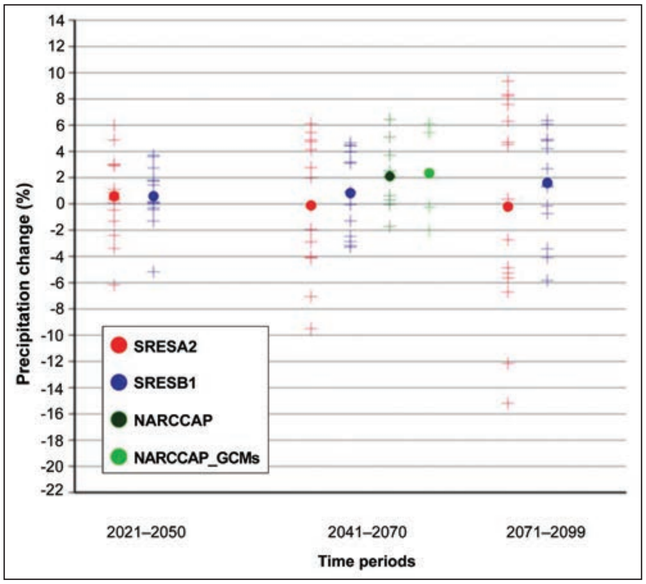

Figure 2.3—Mean annual percentage of precipitation change for three future time periods, relative to a 1971 to 2000 reference period. Little change in annual precipitation is projected for the continental United States as a whole, but individual model projections differ widely. Model projections for the high (A2) and low (B1) emission scenarios for all three time periods used 15 GCMs. Also shown are results for the North American Regional Climate Change Assessment Program simulations for 2041–2070 and the four GCMs used in the NARCCAP experiment (A2 only). Plus signs are values for each individual model; circles show overall means. (Kunkel, K.E.; Stevens, L.E.; Sun, L. [et al.]. [N.d.]. Climate of the contiguous United States. Manuscript in preparation. On file with: NOAA's National Climate Data Center, 151 Patton Avenue, Asheville, NC 28801.)

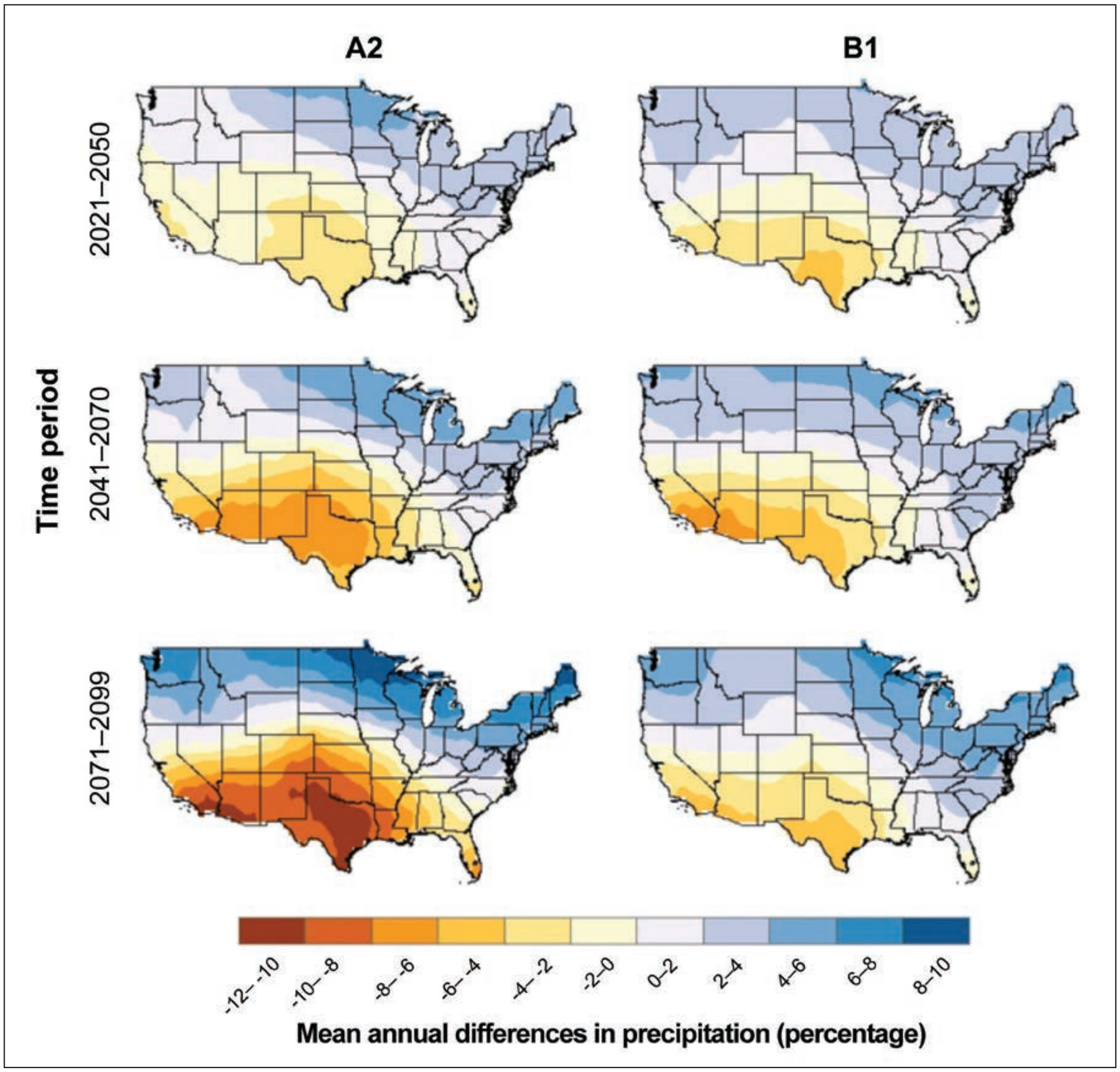

Figure 2.4—Mean percentage of annual differences in U.S. precipitation between three future periods relative to a 1971 to 2000 reference period. The Northeast, northern Midwest and Northwest are projected to have slightly more precipitation, and the Southwest is projected to have 2 to 12 percent less precipitation, depending on the emission scenario, location, and time period. Means are for all 15 GCMs. (Kunkel, K.E.; Stevens, L.E.; Sun, L. [et al.]. [N.d.]. Climate of the contiguous United States. Manuscript in preparation. On file with: NOAA's National Climate Data Center, 151 Patton Avenue, Asheville, NC 28801.)

drying over the United States (fig. 2.5). Globally, the Palmer Drought Severity Index is predicted to decrease by 0.3 per decade (indicating increased drought) for the first half of the 21st century. Relative to historical figures, the percentage of the land surface in drought annually is predicted to increase in 2010–2020 from 1 to 3 percent for the extreme droughts, from 5 to 10 percent for the severe droughts, and from 20 to 28 percent for the moderate droughts (fig. 2.6). This drying trend continues throughout the 21st century. By the 2090s, the percentage of the land area in drought is predicted to increase for extreme, severe, and moderate droughts to 30 percent, 40 percent, and 50 percent, respectively. For extreme and severe droughts, the number of drought events is projected to double; for moderate drought the number of events remains stable. The duration of all forms of drought events also increases.

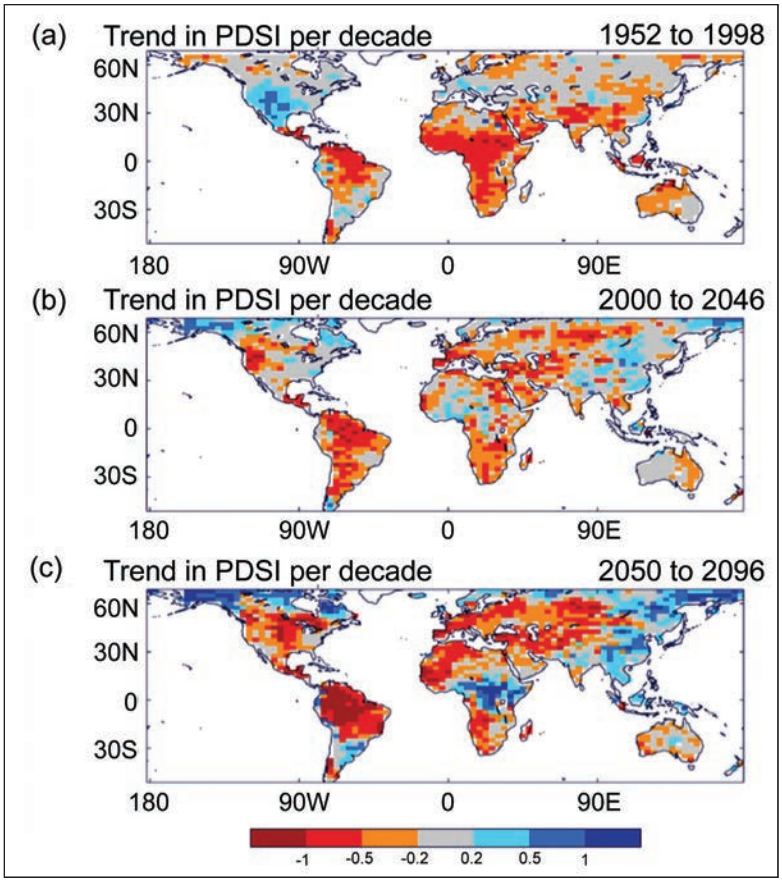

Figure 2.5—The trend in the Palmer Drought Severity Index (PDSI) per decade for (a) observed data and the mean of (b) the first half and (c) the second half of the 21st century. The PDSI is projected to decrease by 0.5 to 1 unit per decade for the period 2050–2096. For the PDSI, -1.9 to 1.9 is near normal, -2 to -2.9 is moderate drought, -3 to -3.9 is severe drought, and less than -4 is extreme drought. Projections are made by the third version of the Hadley Centre coupled ocean–atmosphere global climate model (HadCM3) with the A2 emission scenario. Figure from Burke et al. (2006). © British Crown Copyright 2006, Met Office.

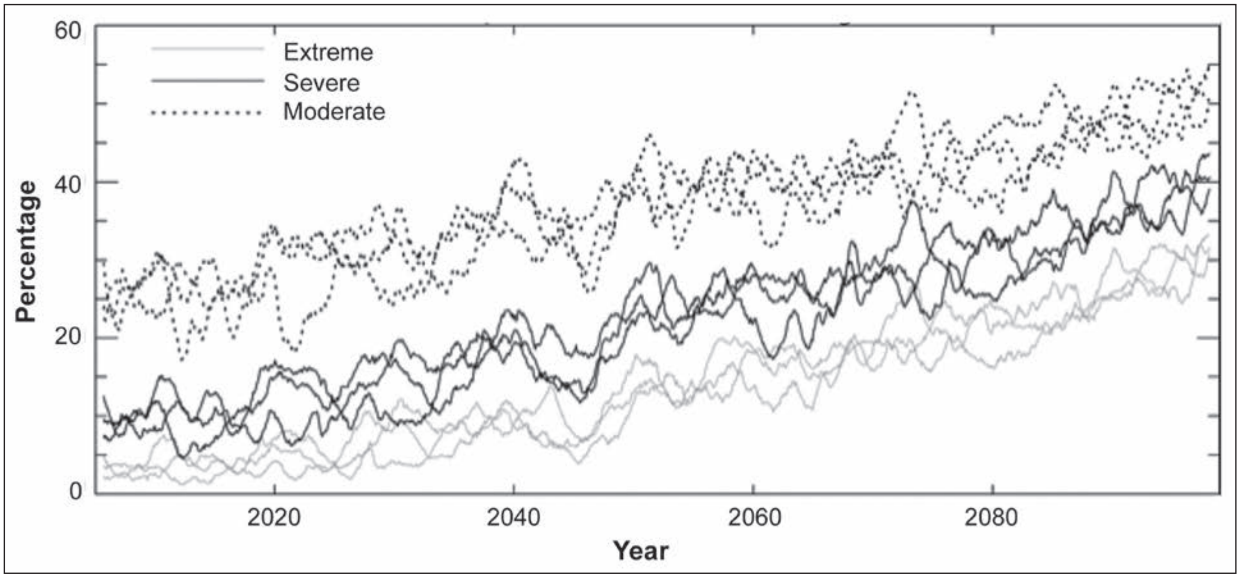

Figure 2.6—The projected average annual proportion of the global land surface in drought each month shows drought increasing over the current century. Drought is defined as extreme, severe, or moderate, which represents 1 percent, 5 percent, and 20 percent, respectively, of the land surface in drought under present-day conditions. Results from the three simulations are from the third version of the Hadley Centre coupled ocean–atmosphere GCM (HadCM3) with the A2 emission scenario. Figure from Burke et al. (2006). © British Crown Copyright 2006, Met Office.

Sea Level Rise

Global sea level rise results from changing the ocean's water volume because of changes in temperature, salinity, ice melting, and land surface runoff. Global sea level responds to climate cycles of alternating glacial and interglacial conditions over millions of years (Kawamura et al. 2007). Mean sea level rose by 120 m since the most recent ice age, at a rate of about 1 m per century. For the last 6,000 years, sea level has remained relatively stable, with observed data indicating a global mean level increase of 0.17 m per century (Grinsted et al. 2010). As the temperature rises in GHG emission scenarios, a combination of factors (e.g., polar ice sheet melting) contributes to sea level rise. Four scenarios of projections of sea level rise are shown in fig. 2.7 (Parris et al. 2011). The low scenario is a linear extrapolation of historical trends (1.7 mm·yr $^{-1}$) in sea level rise over the entire period of tidal observations (1880 through 2009); the two intermediate scenarios (A2 and B1 simulations) are quadratic extrapolations of four semiempirical studies based on average sea level rise for 2100; and the high scenario is a quadratic extrapolation based on analysis of plausible glaciological conditions required for large sea level rise (2 m) to occur by 2100. Depending on the scenario, global sea level is projected to rise 0.2 to 2.0 m by 2100.

Satellite altimetry records show that the mean sea level rise since the middle of the 19th century is not uniform (fig. 2.8). The Pacific Coast of the United States showed little sea level rise, consistent with tide gage records (see discussion in Parris et al. 2011). In contrast, sea level rise in the Gulf of Mexico has averaged 3.2 mm·yr $^{-1}$ since 1992. Whether the observed spatially explicit trends will continue in the future is a topic of active research. For example, the spatial trend in the Pacific is thought to be a combination of wind stress patterns associated with the short-term climatic factors of the Pacific Decadal Oscillation (PDO) and El Niño-Southern Oscillation (ENSO). Because PDO and ENSO regularly shift phases, the likelihood is low that the observed sea level rise trends will continue with the same magnitude and direction.

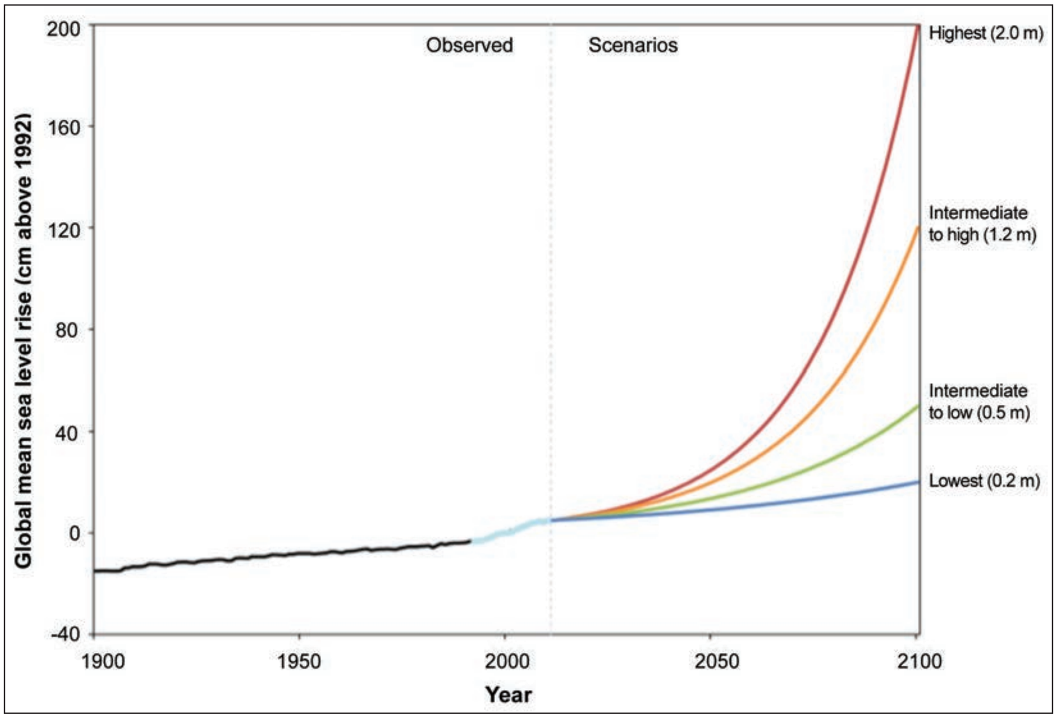

Figure 2.7—Four scenarios of projections of sea level rise from Parris et al. (2011) show sea level increasing from 0.2 to 2.0 m by 2100. The low scenario (dark blue line) is a quadratic extrapolation to the period 1990 to 2100 of historical trends in sea level rise over the entire period of observations (1880 to 2009). The two intermediate scenarios (high and low) are based on averages of the A2 simulation (orange line) and B1 simulation (green line), respectively, of four semiempirical studies. The high scenario (red line) is based on analysis of plausible ice melting required for a large sea level rise (2 m) to occur by 2100.

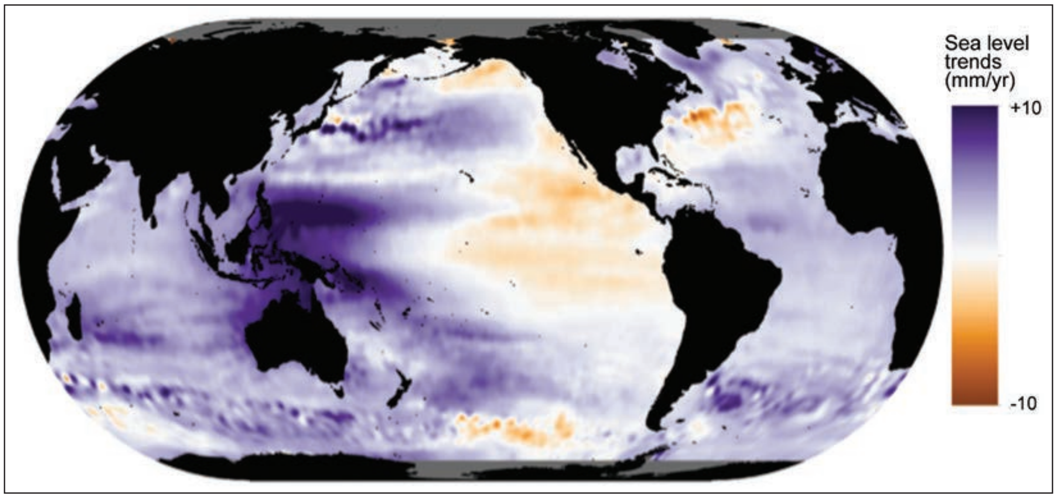

Figure 2.8—Geographic variability in the rate of global sea level change (1992–2010) based on three satellite records (TOPEX, Jason 1 and Jason 2) shows that little sea level rise occurred for the coastal United States during that period. Figure from NOAA Laboratory for Satellite Altimetry – Accessed November 2, 2011.

Key Findings

- Using the A1 and B2 emission scenarios, average annual temperatures will likely increase from 2.5 to 5.3 °C by 2100 relative to 1971 to 2000, and the highest temperature increases will likely be in the northern and interior United States; days with temperature higher than 35 °C will also likely increase.
- Average annual precipitation in the Southwest will likely decrease 6 to 12 percent by 2100 and increase for northern states by 6 to 10 percent.
- Drought will likely increase and the increase will likely intensify as temperature increases.
- Global sea level will likely rise between 0.2 and 2.0 m by 2100.

Key Information Needs

- Improved projections of the timing, spatial distribution, and severity of extreme precipitation events.
- Expanded and more coordinated monitoring networks and data accessibility to enable detection and evaluation of changes in meso- and small-scale microclimatic conditions.

Effects of Climate Change on Disturbance Regimes

Disturbances such as fire, insect outbreaks, disease, drought, invasive species, and storms are part of the ecological history of most forest ecosystems, influencing vegetation age and structure, plant species composition, productivity, carbon (C) storage, water yield, nutrient retention, and wildlife habitat. Climate influences the timing, frequency, and magnitude of disturbances (Dale et al. 2001). As the climate continues to change, we should expect increased disturbance through more frequent extreme weather events, including severe storms, drought, tornadoes, hurricanes, and ice storms. Indirect effects may amplify these changes, with conditions that favor fire, insect and pathogen outbreaks, and invasive species. In this section, we focus primarily on indirect effects of climate change on important forest disturbances across the United States.

Fire

Climate and fuels are the two most important factors controlling patterns of fire within forest ecosystems. Climate controls the frequency of weather conditions that promote fire, whereas the amount and arrangement of fuels influences fire intensity and spread. Climate influences fuels on longer time scales by shaping species composition and productivity (Marlon et al. 2008, Power et al. 2008) and large-scale climatic patterns, such as the ENSO, PDO, Atlantic Multidecadal Oscillation, and Arctic Oscillation (Kitzberger et al. 2007) (interior West: Collins et al. 2006; Alaska: Duffy et al. 2005, Fauria and Johnson 2006) are important drivers of forest productivity and susceptibility to disturbance.

Current and past land use, including timber harvest, forest clearing, fire suppression, and fire exclusion through grazing (Allen et al. 2002, Swetnam and Betancourt 1998) have affected the amount and structure of fuels in the United States. For example, in the montane forests in the Southwest (Allen et al. 2002) and other drier forest types in the interior West, removal of fine fuels by grazing and fire suppression has increased the number of trees and fuels; these changed forest conditions have increased fire size and intensified fire behavior. In colder or wetter forests in the Western United States, such as subalpine forests in Yellowstone National Park and forests in the maritime Northwest, grazing and fire suppression have not altered fire regimes as extensively. Forests in the Northeasten United States (Foster et al. 2002) and the upper Midwest developed after widespread timber harvest, land clearing, and forest regrowth after land abandonment. These forests burn less often and with smaller fires than forests in other regions of the United States. Forests in the Southeastern United States are often managed for timber, and prescribed fire is generally more prevalent than uncontrolled ignitions (National Interagency Coordination Center 2011). Prescribed fire occurs every 2 to 4 years in some fire-dependent ecosystems in the southeast (Mitchell et al. 2006). Fire suppression and deer herbivory in the central hardwoods section of the Eastern United States has pushed the composition towards more mesic and fire-intolerant species (e.g., oak-dominated to maple-dominated) (Nowacki and Abrams 2008).

Weather remains the best predictor of how much area will burn, despite the changes in land use and the resulting effects on fuels. Correlations between weather and either the area burned by fire or the number of large fires are similar for both presettlement fires and fires of the last few decades. These syntheses of fire-weather relationships for both presettlement and modern records exist in several subregions of the West (Northwest: Hessl et al. 2004; Heyerdahl et al. 2002, 2008a; Southwest: Grissino-Mayer and Swetnam 2000, Swetnam and Betancourt 1998; Northern Rockies: Heyerdahl et al. 2008b; Westwide: Littell et al. 2009; Westerling et al. 2003, 2006) and East (Hutchinson et al. 2008). Presettlement fire-weather relationships are derived from trees scarred by fires or age classes of trees established after fire and independently reconstructed climate, and modern fire-weather comparisons are derived from observed fire events and observed weather occurring in the seasons leading up to and during the fire. These studies agree that drought and increased temperature are the basic mechanisms that promote large fires, but the effects differ by forest and region (Littell et al. 2009, Westerling et al. 2003). Weather can also influence fire through higher precipitation, increasing understory vegetation growth, which later becomes fuel (Littell et al. 2009, Swetnam and Betancourt 1998). Fire in some forests responds to drought and to precipitation enhancement of fine fuels (Littell et al. 2009). Increased temperature and altered precipitation also affect fuel moisture during the fire season and the length of time during which wildfires can burn during a given year.

The potential effects of climate change on forest fire area have been assessed using statistical models that project area burned from climatic variables, and by using global climate models to predict future climatic variables (Westwide: McKenzie et al. 2004, Spracklen et al. 2009, Littell et al. 2010; Northwest: Littell et al. 2010; Yellowstone region: Westerling et al. 2011). Estimated future increases in annual area burned range from less than 100 percent to greater than 500 percent, depending on the region, timeframe, methods, and future emissions and climatic scenario. Dynamic vegetation models have also been used to project future fire activity. Based on climate projections derived from global climate models over the West, these projections suggest a wide range of changes in biomass area burned (from declines of 80 percent to increases of 500 percent, depending on region, climate model, and emissions scenario) (Bachelet et al. 2001). Future fire potential is expected to increase in summer and autumn from low to moderate in eastern regions of the South, and from moderate to high levels in western regions of the South (Liu et al 2010). Models have not yet estimated the effects of future climate on fire severity (i.e., the proportion of overstory mortality). These effects are less certain because severity may be more sensitive than area burned to arrangement and availability of fuels

The risk posed by future fire activity in a changing climate can be assessed by its likely effects on human and ecological systems. At the wildland-urban interface, higher population and forest density have created forest conditions that are likely to experience more area burned and possibly higher fire severity than in the historical record. Fire risk is likely to increase in a warmer climate because of the longer duration of the fire season, and the greater availability of fuels if temperature increases and precipitation does not sufficiently increase to offset summer water balance deficit. Where fuels management is common, forest fuel reduction and restoration to presettlement tree density and ground fire regimes help to mitigate fire hazard under current and future climatic conditions. However, with current resources, only a small portion of the landscape can be treated. Finally, future fire risk may depend on whether extreme fire weather conditions will change in step with monthly to seasonal climate changes. Even if fire weather and ignitions do not change, it is likely that risk driven only by seasonal climate changes will increase—particularly in the wildland-urban interface and managed forests, where fire has been historically rare or fully suppressed and climate has not been as strong an influence as in wildland fires. The current increase in annual area burned may be partially related to increased fuels in frequent-fire forest types, in addition to more frequent weather conditions conducive to fire. The effects of climate change intersecting with these increased fuel loads in frequent-fire forests will be an exceptional management challenge.

Key Findings

- Annual area burned and length of the fire season will likely increase throughout the United States, altering the structure, function, and potentially the species composition of forest ecosystems.
- Increased fire in the wildland-urban interface will likely create social and economic challenges, including higher fire-suppression costs.
- Hazardous fuel treatments and forest restoration will likely reduce fire severity at the local scale, but it is unlikely that treatments can be applied widely enough to modify fuels across large landscapes.
- Concentrating precipitation into more intense storms may increase fire risk through development of fine fuels and longer drought periods.

Key Information Needs

- Quantifiable effects of increased fire occurrence on natural resource conditions and ecosystem services, including wildlife, water, fisheries, and C dynamics.
- Improved accuracy and spatial and temporal resolution of models that project extreme fire events.
- Additional empirical data on and models for interactions among seasonal hydrology, fuels, and fire occurrence in mountain environments.

Insects and Pathogens

Biotic disturbances are natural features of forests that play key roles in ecosystem processes (Adams et al. 2010, Boon 2012, Hicke et al. 2012a). Epidemics by forest insects and pathogens affect more area and result in greater economic costs than other forest disturbances in the United States (Dale et al. 2001). By causing local to widespread tree mortality or reductions in forest productivity, insect and pathogen outbreaks have broad ecological and socioeconomic effects (Pfeifer et al. 2011, Tkacz et al. 2010).

The first National Climate Assessment (Melillo et al. 2000) projected increased disturbance in forests, especially from insects, and especially from bark beetles, because of their high physiological sensitivity to climate, short generation times, high mobility, and explosive reproductive potential. These projections have been upheld, and current observations suggest that disturbances are occurring more rapidly and dramatically than imagined a decade ago (boxes 2.2 and 2.3). Understanding how these disturbances are influenced by climate change is therefore critical for quantifying and projecting effects.

General Concepts

The powerful general effect of temperature on insects and pathogens is among the best known facts of biology (Gillooly et al. 2002), and recognition of climate change

Box 2.2—Mountain pine beetle and five-needle pines

Five-needle pines, including whitebark (*Pinus albicaulis* Engelm.), limber (*P. flexilis* James), and bristlecone (*P. aristata* Engelm.) pines, play key roles in forest ecosystems of the West. They provide food resources for wildlife, affect snow distribution and melt, stabilize the soil, provide cover for other vegetation (Jewett et al. 2011, Logan et al. 2010), and are valued by the public for these services (Meldrum et al. 2011). However, these conifers are currently subjected to a climatically induced increase in biotic disturbance that is expected to continue in the coming decades. Mountain pine beetles (*Dendrotonus ponderosae* Hopkins) are attacking five-needle pines across the West; aerial surveys indicate that 1 million ha were affected by five-needle pine mortality during 1997 through 2010. Research has identified higher temperatures and drier conditions as important climate drivers (Jewett et al. 2011, Logan et al. 2010, Perkins and Swetnam 1996). These factors influence winter survival and development rate and population synchronization of beetles (Logan et al. 2010) as well as susceptibility of host trees (Perkins and Swetnam 1996).

Similar epidemics occurred in the 1930s (Perkins and Swetnam 1996), also associated with a period of warmer years, but several differences exist between the mortality then and today. Most importantly, a cooler period followed the 1930s that was less suitable for the beetle (Logan and Powell 2001). In contrast, the current warming trend which has persisted for several decades, with resultant increases in climate suitability (Logan et al. 2010) for mountain pine beetle, is expected to continue for decades to come (Littell et al. 2010, Logan et al. 2010). The recent beetle epidemics in five-needle pine stands are already more extensive than in the 1930s and are killing very old trees that survived previous outbreaks (Logan et al. 2010). Finally, white pine blister rust is predisposing whitebark pines to lethal attacks by mountain pine beetle (Six and Adams 2007).

What is the future of these five-needle pine ecosystems? Given the trajectory of future warming, strong ties between temperature and beetle epidemics, and extensive mortality that has already occurred in some areas such as the Greater Yellowstone Ecosystem, significant consequences are expected for these forests and the ecosystem services that they provide (Logan et al. 2010). The recent decision of the U.S. Court of Appeals to re-list grizzly bears (*Ursus arctos*) as an endangered species in the Greater Yellowstone area cited the expectation of reduced food for bears because of climatic release of mountain pine beetle into whitebark pine forests.[1]

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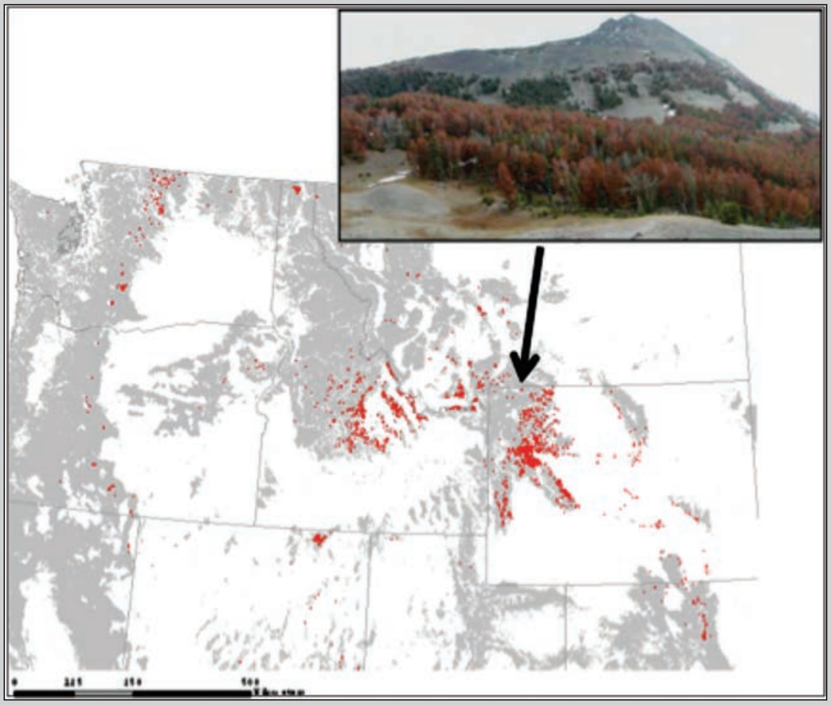
[1] *Greater Yellowstone Coalition v. State of Wyoming.* No. 09-361000, 10-35043, 10-35052, 10-35053, 10-35054. 16 U.S.C. 1533(a)(1)(D). (2011).

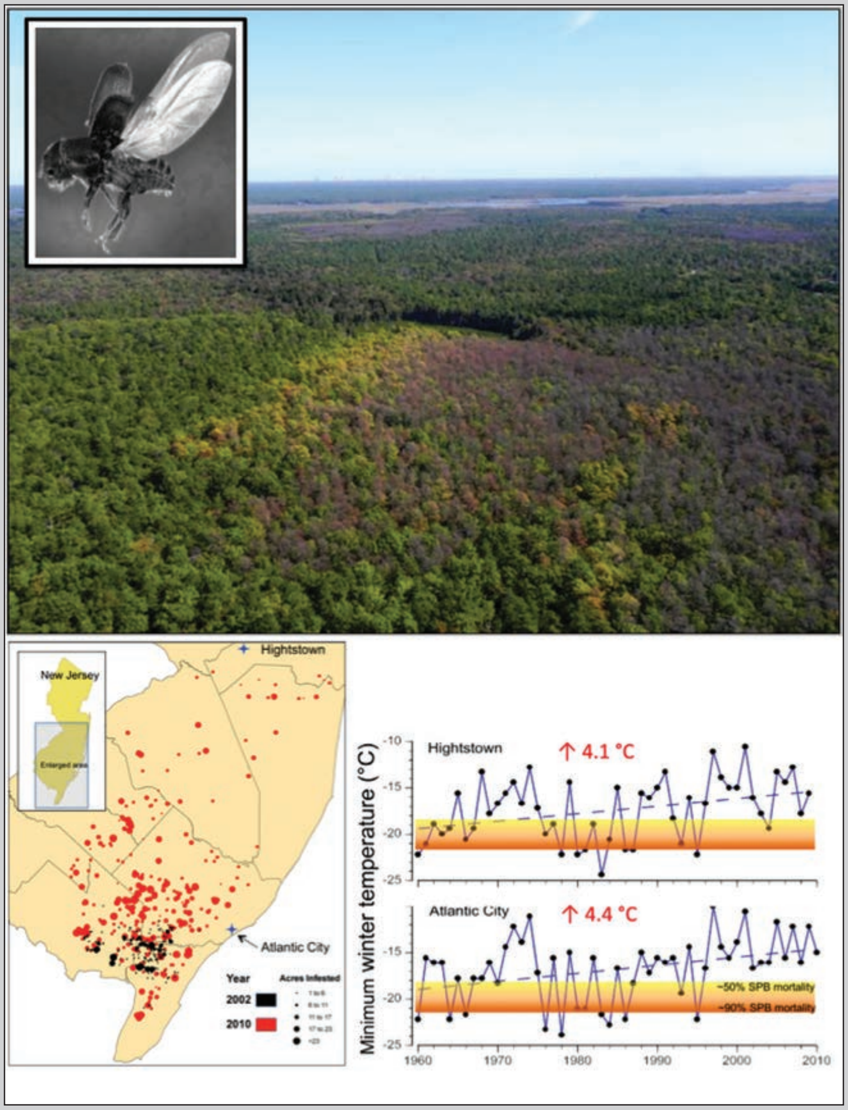


Area affected by mortality in stands of whitebark, limber, and bristlecone pine in 1997–2010 as detected by aerial surveys conducted by the USDA Forest Service. Affected area includes live and dead trees. Gray shading indicates locations of forest. Inset shows whitebark pine mortality in 2004 in Yellowstone National Park. Credits: Polly Buotte, University of Idaho (map), Jeffrey Hicke (photo).

Box 2.3—The southern pine beetle reaches New Jersey Pinelands

The southern pine beetle (*Dendroctonus frontalis* Zimmermann) is the most destructive herbivore in the most productive forests of the United States (Pye et al. 2011). Like the closely related mountain pine beetle (*D. ponderosae* Hopkins), it uses aggregation pheromones to coordinate mass attacks that overwhelm the resin defenses of otherwise healthy trees; virtually every attacked tree dies within weeks. It has multiple generations per year (at least four to five in the warm Gulf Coast region), so the aggregations that typically form in spring can expand throughout the year as growing "spots" of tree mortality within forest landscapes. Effective suppression of these epidemics involves locating the spots and cutting the infested trees (Billings 2011). Effective prevention involves silvicultural thinning to reduce the occurrence of stands with high basal area (overstocked) that are especially suitable for beetle population growth. Monitoring, suppression, and prevention of southern pine beetle are integral to the management of pine ecosystems in the southeastern United States.

The northern distribution of southern pine beetle is constrained by the occurrence of lethal winter temperatures (Ungerer et al. 1999). As part of the first National Climate Assessment (Ayres and Lombardero 2000), it was estimated that an increase of 3 °C in minimum annual temperature would permit a northern expansion of about 180 km for this beetle. In fact, there was a regional increase of just over 3 °C from 1960 through 2005, and beetle populations are now epidemic in the New Jersey Pinelands, about 200 km north of forests with a long history of such epidemics (Trần et al. 2007). Warming winters did not cause the current epidemic but may have permitted it. Given the natural population dynamics of southern pine beetle and the continued absence of lethal winter temperatures (which should be expected), the New Jersey Pinelands has entered a new phase where southern pine beetle will be influencing all aspects of forest ecology and management, as they have throughout the Southeastern United States.



A view in October 2011 of one of many infestations of southern pine beetle in the New Jersey Pinelands. Aerial photo by Bob Williams, Land Dimensions. Close-up of southern pine beetle by Erich Vallery, USDA Forest Service. (Bottom)—Southern pine beetles die when winter air temperatures drop below about -17.7 °C. A subcontinental pattern of warmer winters has eliminated a climatic barrier to occupancy of the New Jersey Pinelands by the beetle and permitted an epidemic that is presently growing and expanding northward.

has motivated scientific inquiry into climatic effects on the extent and severity of forest disturbances by insects and diseases. Clear examples exist of climatic effects on insects (boxes 2.2 through 2.4), yet the most important insects and pathogens of American forests remain poorly studied with respect to the interaction with climate and resulting effects on forests (tables 2.1 and 2.2).

Climate and atmospheric changes associated with increasing GHGs can influence biotic disturbances of forests through effects on (1) the physiology of insects and pathogens that cause changes in their abundance and distribution, (2) tree defenses and tolerance, and (3) interactions between disturbance agents and their own enemies, competitors, and mutualists (fig. 2.9). Current and projected increases in temperature can enhance forest disturbance by reducing winter mortality of insects and increasing their range northward (Paradis et al. 2008, Safranyik et al. 2010, Trần et al. 2007), and by increasing the development rate of insects and pathogens during the growing season (Bentz et al. 2010, Gillooly et al. 2002). Temperature increases can also alter phenology, such as bringing leaf maturation into synchrony with insect feeding (Jepsen et al. 2011) or changing the life cycle synchrony of bark beetles, which depend on mass attack to overwhelm tree defenses (Bentz et al. 2010, Friedenberg et al. 2007).

A broader set of atmospheric drivers affect tree defenses against, and tolerance to, herbivores and pathogens (Bidart-Bouzat and Kliebenstein 2008, Lindroth 2010, Sturrock et al. 2011). Deficiencies of water or mineral nutrients can both increase and decrease tree defenses, depending on the severity of the deficiency, biochemical pathways, and the type of defense (Breshears et al. 2005, Herms and Mattson 1992, Lombardero et al. 2000, Worrall et al. 2008a). In addition, tree mortality from severe drought may permit an increase in bark beetles, which then become abundant enough to successfully attack healthy trees (Greenwood and Weisberg 2008, Raffa et al. 2008). Limited understanding exists on the effects of climate on tree-pathogen interactions, despite a theoretical expectation for strong effects from temperature and moisture (Sturrock et al. 2011). Climatic sensitivity related to the joint phenology of plants, their pathogens, and the environment is not well studied (Grulke 2011, Rohrs-Richey et al. 2011). Outbreak dynamics of forest insects respond to interactions between herbivores and their enemies (Dwyer et al. 2004), and these interactions should be sensitive to temperature (Berggren et al. 2009, Klapwijk et al. 2012), but empirical studies are rare (Siegert et al. 2009). Similarly, for the many forest insects that involve mutualisms with fungi, it is logical that outbreak dynamics will be sensitive to climatic effects on the mutualism, but studies are limited (Evans et al. 2011, Hofstetter et al. 2007, Lombardero et al. 2000, Six and Bentz 2007). Such interactions may not be predictable because of complexity and contingency.

Recent climatic patterns are likely affecting forest disturbance by insects and pathogens in North America (Raffa et al. 2008, Trần et al. 2007). Given the range of mechanisms (most still poorly studied) by which climate change can affect forest disturbance, existing scientific knowledge almost certainly captures only some of the current effects.

Climate and Biotic Disturbances

Bark beetles—

Multiple species of indigenous bark beetles affect millions of hectares of coniferous forests in North America (table 2.1). Major species include mountain pine beetle (*Dendroctonus ponderosae* Hopkins), the most important disturbance agent of pines in the Western United States (box 2.2), southern pine beetle (*D. frontalis* Zimmermann), the analog in the productive pine forests of the southeastern United States (box 2.3), and spruce beetle (*D. rufipennis* Kirby). In the early 2000s, severe drought, coupled with several species of bark beetles, killed trees of several conifer species in the Southwest (Ganey and Vojta 2011), most notably pinyon pines (*Pinus edulis* Engelm.) attacked by pinyon ips (*Ips confusus* LeConte) across 1.2 million ha (Breshears et al. 2005).

All of these bark beetles are native to North America, have population dynamics that are innately explosive, and have been exerting powerful effects on American forests for millennia. However, their outbreak tendencies are sensitive to climatic variation, and the massive extent and expanding distribution of recent outbreaks have been permitted or exacerbated by increasing temperatures during recent

Box 2.4—Hemlock woolly adelgid

Invasive insects and pathogens are an important class of biotic disturbance to American forests. A subset of invasives causes extensive tree mortality owing to lack of genetic resistance in host trees and the absence of natural enemies. Thus, nonindigenous insects and pathogens may be especially likely to cause the loss of native tree species and produce other substantial effects on forests, wildlife, biodiversity, and the many services provided by forest ecosystems. Climate change can exacerbate the effects of established invasives by permitting their expansion into previously unsuitable climatic regions (as with the expansion of the hemlock woolly adelgid [*Adelges tsugae* Amand] into New England) and by producing mismatches between mature trees and their new climate. Perhaps most importantly, warming is increasing the ports of entry where new potential invasives can become established in American forests.

The hemlock woolly adelgid, accidentally introduced from Japan sometime before 1951, is a major biotic disturbance within American forests that has been killing eastern hemlock (*Tsuga canadensis* [L.] Carrière) and Carolina hemlock (*T. caroliniana* Engelm.) in advancing waves from its point of establishment in Virginia (Orwig et al. 2002). Hemlock woolly adelgid is an aphid-like insect that kills its American host trees slowly but inevitably. Since establishment, this insect has largely eliminated hemlocks from a large swath of eastern forests, including national icons such as the Shenandoah and Great Smoky Mountains National Parks. Consequences include lost value to property owners (Holmes et al. 2010) and persistent alterations to hydrological regimes, soil biogeochemistry, carbon stores, biodiversity, and forest composition, including promoting the establishment of undesirable invasive plants (Knoepp et al. 2011, Orwig et al. 2008, Peltzer et al. 2010, Stadler et al. 2006).

Hemlocks north of the infested regions have thus far been spared by winter temperatures that are lethal to hemlock woolly adelgid (Parker et al. 1998). However, these conditions are changing with the amelioration of extreme winter temperatures in the Eastern United States (see also box 2.3), and projections under even conservative climate scenarios predict the loss of hemlock forests through most of the current range of hemlock (Dukes et al. 2009, Fitzpatrick et al. 2012, Paradis et al. 2008).

Dead mature eastern hemlocks killed by hemlock woolly adelgid in western North Carolina (photo: Forest Health Management International. Bugwood.org. http://www.invasive.org/browse/detail.cfm?imgnum=2167012. (4 December 2012). (photo: http://upload.wikimedia.org/wikipedia/commons/a/a0/Adelges_tsugae_3225077.jpg).

Table 2.1—Insects that are notable agents of biological disturbance in North American forests and therefore candidates for consequential changes to disturbance regimes as a result of climate change

			References	
Syndrome	**Herbivore**	**Hosts**	**General references**	**Studies related to climate**[a]
Defoliation by autumnal moth	*Epirrita autumnata*[b]	Many broadleaved trees and conifers	Selas et al. 2001, Tenow et al. 2007	Jepsen et al. 2008, Peterson and Nilssen 1996 (T), Virtanen et al. 1998 (T)
Defoliation by gypsy moths and tussock moths	*Lymantria dispar,*[b] *Orgyia* spp.	*Quercus* spp., many other broadleaved trees and conifers	Mason 1996	Lindroth et al. 1993 (CO_2), Williams and Liebhold 1995 (P,T)
Defoliation by budworms	*Choristoneura fumiferana, C. occidentalis, C. pinus*	*Abies* spp., *Pseudotsuga* spp., *Picea* spp., *Pinus* spp.	Royama 1984	Fleming 1996 (T), Rauchfuss et al. 2009 (P,T), Ryerson et al. 2003 (P), Volney and Fleming 2000 (T)
Defoliation by gracillariid leaf miners	*Archips pinus, Micrurapteryx salicifoliella, Phyllocnistis populiella*	*Populus tremuloides, Salix* spp.	Furniss et al. 2001, Wagner et al. 2008	
Defoliation by loopers	*Enypia griseata, Nepytia* spp.	*Abies* spp., *Pseudotsuga* spp., *Picea* spp., *Pinus* spp., *Thuja* spp.	Munroe 1963, Rindge 1967, Stevens et al. 1983	
Defoliation by tent caterpillars	*Malacosoma* spp.	*Prunus* spp., *Populus* spp., *Betula* spp., *Nyssa* spp., other broadleaved trees	Rejmánek et al. 1987	Frid and Myers 2002 (T), Lindroth et al. 1993 (CO_2), Volney and Fleming 2000 (T)
Infestations by Asian longhorned beetle	*Anoplophora glabripennis*[b]	*Acer* spp., *Ulmus* spp., *Populus* spp.	Cavey et al. 1998, Dodds and Orwig 2011	Keena 2006 (T), Keena and Moore 2010 (T), Peterson et al. 2004 (T)
Infestations by bronze birch borer	*Agrilus anxius*	*Betula* spp.	Nielsen et al. 2011	Jones et al. 1993 (P,T)
Infestations by emerald ash borer	*Agrilus planipennis*[b]	*Fraxinus* spp.	Cappaert et al. 2005	Crosthwaite et al. 2011 (T)
Infestations by goldspotted oak borer	*Agrilus auroguttatus*	*Quercus* spp.	Coleman et al. 2011	
Infestations by mountain pine beetle	*Dendroctonus ponderosae*	*Pinus* spp.	Safranyik and Carroll 2006	Bentz et al. 2010 (P,T), Powell et al. 2000 (T), Raffa et al. 2008 (P,T), Regnière and Bentz 2007 (T)
Infestations by pine engraver beetles	*Ips* spp.	*Pinus* spp.	Schenk and Benjamin 1969	Breshears et al. 2005 (P,T), Lombardero et al. 2000 (T), Raffa et al. 2008 (T)
Infestations by southern pine beetle	*Dendroctonus frontalis*	*Pinus* spp., chiefly southern pine	Reeve et al. 1995	Friedenberg et al. 2007 (T), Lombardero et al. 2000 (T), Tran et al. 2007 (T), Ungerer et al. 1999 (T), Waring et al. 2009 (T)
Infestations by spruce aphid	*Elatobium abietinum*[b]	*Picea* spp.	Lynch 2004	Powell 1974 (T), Powell and Parry 1976 (T)
Infestations by spruce beetle	*Dendroctonus rufipennis*	*Picea* spp.	Allen et al. 2006	Bentz et al. 2010 (T), Berg et al. 2006 (T), Hebertson and Jenkins 2008 (P,T)
Infestations by western pine beetle	*Dendroctonus brevicomis*	*Pinus* spp., chiefly *P. ponderosa*	Liebhold et al. 1986	Evangelista et al. 2011 (T)
Infestations by white pine weevil	*Pissodes strobi*	*Pinus* spp., *Picea* spp.	Lavallée et al. 1996	Sullivan 1961 (T)
Infestations by woolly adelgids	*Adelges piceae,*[b] *A. tsugae*[b]	*Abies fraseri, A. balsamea, Tsuga* spp.	McClure 1991	Butin et al. 2005 (T), Evans and Gregoire 2007 (T), McClure 1989 (T), Paradis et al. 2008 (T), Trotter and Shields 2009 (T)
Browsing by deer, elk, hares, and moose	*Odocoileus* spp., *Cervus canadensis*, *Alces alces*	Many broadleaved trees and some conifers	Gill 1992, Pease et al. 1979, Ross et al. 1970	Simard et al. 2010 (T)

[a] Letters following references denote studies considering the effects of precipitation (P), temperature (T), or carbon dioxide (CO_2).

[b] Nonindigenous to North America.

Source: Updated from Ayres and Lombardero (2000).

Table 2.2—Pathogens, parasites, and declines that are notable agents of biological disturbance in North American forests and therefore candidates for consequential changes to disturbance regimes as a result of climate change

			References	
Syndrome	**Pathogen/parasite/decline**	**Hosts**	**General references**	**Studies related to climate**[a]
Alder canker	*Valsa melanodiscus*	*Alnus* spp.	Worrall et al. 2009	Worrall et al. 2010 (T)
Annosum root rot	*Heterobasidion annosum*	Most conifers, some broadleaved trees	Stanosz et al. 1995	Boland et al. 2004, Witzell et al. 2011 (T)
Anthracnose leaf disease	*Discula destructiva, Glomerella cingulata, Colletotrichum gloeosporioides,* others	*Quercus* spp., *Fraxinus* spp., *Platanus* spp., *Cornus* spp.	Stanosz 1993	Chakraborty et al. 2000 (CO_2), Holzmueller et al. 2006 (P)
Armillaria root rot	*Armillaria* spp.	Broadleaved trees and conifers, e.g., *Acer* spp., *Picea* spp.	Entry et al. 1991, Smith et al. 1994	Dukes et al. 2009, Sturrock et al. 2011
Beech bark disease	*Nectria* spp. (and associated scale insects *Cryptococcus fagisuga*[b] *and Xylococculus betulae*)	*Fagus grandifolia*	Busby and Canham 2011, Garnas et al. 2011a	Dukes et al. 2009, Garnas et al. 2011b (P,T)
Butternut canker	*Ophiognomonia clavigignenti-juglandacearuma*[b] *(= Sirococcus clavigignenti-juglandacearumab)*	*Juglans cinerea*	Broders et al. 2011, Harrison et al. 1998	
Chestnut blight	*Cryphonectria parasitica*[b]	*Castanea dentata*	McKeen 1995	
Dothistroma needle blight	*Dothistroma septosporum* and *D. pini*	Many conifers, *Pinus* spp.	Welsh et al. 2009	Sturrock et al. 2011, Watt et al. 2009 (P,T), Woods et al. 2005 (P)
Dutch elm disease	*Ophiostoma novoulmi*[b] (and associated bark beetles *Hylurgopinus rufipes* and *Scolytus multistriatus*[b]*)*	*Ulmus* spp.	Holmes 1980	Boland et al. 2004
Dwarf mistletoe	*Arceuthobium* spp.	*Pinus* spp.	Synder et al. 1996	Brandt et al. 2004 (T), Stanton 2007 (P,T)
Fusiform rust	*Cronartium quercuum*	Pinus spp., chiefly southern pine	Doudrick et al. 1996, Nelson et al. 1996	Runion et al. 2010 (CO_2)
Laurel wilt	*Raffaelea lauricola* (and associated bark beetle *Xyleborus glabratus*)[b]	Lauraceae	Fraedrich et al. 2008, Harrington et al. 2011	Koch and Smith 2008 (T)
Oak wilt disease	*Ceratocystis fagacearum*	*Quercus* spp.	Juzwik et al. 2008	Boland et al. 2004, Tainter 1986 (T)
Phytophthora root disease	*Phytophthora cinnamomi*[b]	*Quercus* spp., *Castanea* spp., *Abies* spp.	Griffin et al. 2009, Hardham 2005	Zentmyer et al. 1979 (T), Bergot et al. 2004 (T)
Pitch canker	*Fusarium circinatum*[b]	*Pinus* spp.	Gordon et al. 1996	Ganley et al. 2009 (P,T), Inman et al. 2008 (T), Runion et al. 2010 (CO_2), Watt et al. 2011 (P,T)
Procera, black stain, and other *Leptographium* root diseases	*Leptographium* spp.	Many conifers, e.g., *Pinus* spp.	Harrington and Cobb 1983, Jacobi et al. 2008	
Scleroderris canker	*Gremmeniella abietina (= Sclerroderris lagerbergii* and *Ascocalyx abietina)*[b]	Conifers	Hamelin et al. 1993, Laflamme 2005	Boland et al. 2004, Donaubauer 1972, Venier et al. 1998 (P,T)
Sudden aspen decline	—	*Populus tremuloides*	Hogg and Schwarz 1999	Hogg et al. 2002, 2008 (P,T), Rehfeldt et al. 2009 (P,T), Worrall et al. 2008ab, 2010 (P,T)
Sudden oak death	*Phytophthora ramorum*[b]	*Quercus* spp., *Lithocarpus* spp.	Spaulding and Rieske 2011, Vaclavik et al. 2010	Venette and Cohen 2006 (P,T)
Swiss needle cast	*Phaeocryptopus gaeumannii*	*Pseudotsuga menziesii*	Hansen et al. 2000	Manter et al. 2005 (P,T), Stone et al. 2008 (P,T)
Thousand cankers disease	*Geosmithia morbida* (and associated bark beetle *Pityophthorous juglandis*)	*Juglans* spp.	Grant et al. 2011, Kolarik et al. 2011	
White pine blister rust	*Cronartium ribicola*[b]	Five-needle pines, e.g., *Pinus strobus, P. albicaulis*	Keane et al. 1990, Kinloch 2003	Sturrock et al. 2011
Alaska cedar decline	—	*Callitropsis nootkatensis*	Wooton and Klinkenberg 2011	Hennon et al. 2006 (P,T),Sturrock et al. 2011

— = none.

[a] Letters following references denote studies considering the effects of carbon dioxide (CO_2), precipitation (P), or temperature (T).

[b] Nonindigenous to North America.

Source: Updated from Ayres and Lombardero (2000).

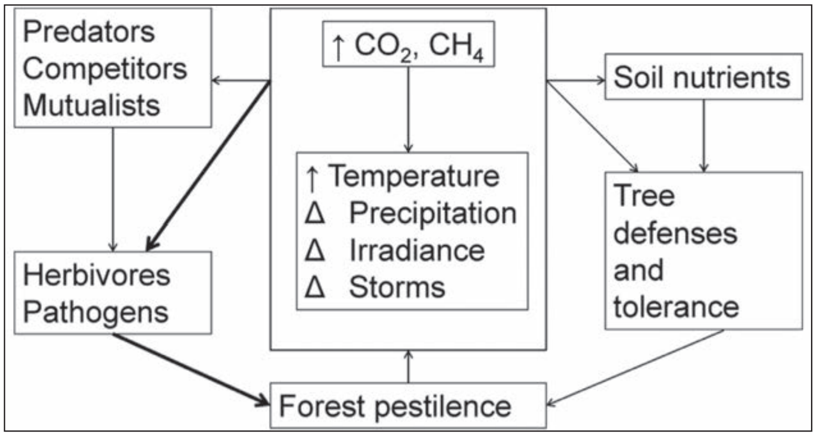


Figure 2.9—General pathways by which atmospheric changes associated with increasing greenhouse gases can influence forest disturbance from insects and pathogens. CO_2 = carbon dioxide, CH_4 = methane.

decades (Breshears et al. 2005, Raffa et al. 2008, Sherriff et al. 2011). Recent range expansions of bark beetles have been particularly notable (boxes 2.2, 2.3). Greater effects on forest ecosystems should be anticipated from these range expansions into areas with novel and naïve hosts (Cudmore et al. 2010). Mexican pine beetle (*D. mexicanus* Hopkins), previously known only in Mexico, has been recorded in the southwestern United States (Moser et al. 2005) and represents one of several species of Mexican bark beetles that may expand into U.S. forests with continued warming trends (Bentz et al. 2010, Salinas-Moreno et al. 2010). In general, climate change is anticipated to continue to reshape the patterns of bark beetle outbreaks in U.S. forests, with outbreak tendencies increasing for some species in some regions and decreasing in others (Bentz et al. 2010, Evangelista et al. 2011, Littell et al. 2010). For example, the unprecedented absence of southern pine beetle activity since the late 1990s in Louisiana and east Texas may be related to climatic warming (Friedenberg et al. 2008).

Defoliating insects—

Defoliating insects are another broad class of continentally important biotic disturbances in American forests (table 2.1). For example, western spruce budworm (*Choristoneura occidentalis* Freeman) is currently important in the West (USDA FS 2010), and eastern boreal forests have been affected by many cycles of spruce budworm (*C. fumiferana* Clemens) outbreaks (Candau and Fleming 2005). Other important defoliators include tussock moths, tent caterpillars, gypsy moths, and jack pine budworm (*Archips pinus* Freeman) (table 2.1). Like bark beetles, most of the important defoliating insects are indigenous to American forests (but not gypsy moths). Many have cyclical outbreak dynamics involving predators, parasitoids, and pathogens of the herbivore (Dwyer et al. 2004). Climatic effects on these predator-prey interactions remain largely unstudied (Klapwijk et al. 2012). In general, it is less clear than with the bark beetles how climatic patterns influence the frequency, extent, and geographic distribution of defoliators in American forests. There have been signals from some systems of climatic effects on winter populations (Kemp et al. 1985, Thomson and Benton 2007, Thomson et al. 1984, Williams and Liebhold 1995a; but see Reynolds et al. 2007), drought stress of host trees (Campbell et al. 2006, Williams and Liebhold 1995b), and phenological synchronization of larval emergence and bud break (Thomson et al. 1984). Considerable uncertainty remains about future responses of defoliators to climate change (Dukes et al. 2009, Rodenhouse et al. 2009).

Plant pathogens—

We identified 21 plant pathogens that are notable agents of disturbance in U.S. forests and therefore top candidates for consequential responses to climate change (table 2.2). Climatic effects on these agents are far less well studied than for forest insects, but it can be expected from first principles that the severity of at least some of these pathogens will be affected directly by climatic influences on sporulation and infection, indirectly by predisposing trees to infection, or both (Sturrock et al. 2011). For pathogens that involve associations with insects, climatic effects on the animal associates may also be important.

Examples of pathogens where there is some understanding of climatic effects include Swiss needle cast, caused by a foliar pathogen (*Phaeocryptopus gaeumannii* [T. Rohde] Petr.) in the Pacific Northwest and which is influenced by winter warming and spring precipitation. Climate projections suggest an increase in Swiss needle cast distribution and severity (Stone et al. 2008). The susceptibility of alder to a cankering pathogen is related to the phenology of the plant, the pathogen, and water availability (Grulke 2011, Rohrs-Richey et al. 2011). Quaking aspen (*Populus tremuloides* Michx.) and Alaska cedar (*Callitropsis nootkatensis* [D. Don] D.P. Little) are declining and experiencing elevated mortality in large areas in the United States. Sudden aspen decline appears to be related to drought stress (Worrall et al. 2010b), suggesting substantial future mortality with continued climate change in forests near the aridity limit for this species (Rehfeldt et al. 2009). Alaska cedar decline has been attributed to earlier snowmelt, which exposes roots to damage from lower temperatures (Hennon et al. 2010), and projected future warming is expected to cause additional mortality from freezing-induced root damage (Sturrock et al. 2011). Outbreaks of some virulent invasive pathogens can also be enhanced by climate (e.g., sudden oak death; Sturrock et al. 2011), whereas others are not very sensitive to climate (Garnas et al. 2011b).

The potential effects of climate change on root pathogens are difficult to project (Ayres and Lomardero 2000), but it will be important to understand this relationship because endemic root diseases are widespread in the United States and often have a major influence on forest dynamics and management. One would expect root diseases to be affected by both the distribution of host species and the effects of a changing climate on susceptibility of host species and prevalence of fungal pathogens. If a warmer climate increases physiological stress in a particular tree species, then it may be less resistant to some root diseases, potentially causing lower tree vigor, higher mortality in mature trees and seedlings, and lower C storage. Although some initial modeling of future changes in root pathogens has been attempted (*Armillaria* spp.; Klopfenstein et al. 2009), geographic specificity for host-pathogen relationships are highly uncertain based on current knowledge. Planting of nonhost species is a standard silvicultural approach to avoid root disease.

Nonnative and emerging insects and pathogens—On a global scale, biological invasions by nonindigenous species are at least as important as climate change for the sustainability of forest ecosystems and the goods and services that they provide (Seppälä et al. 2009). This pattern is evident in the United States, where invasive insects and pathogens are becoming an increasingly important component of forest disturbance (box 2.4) (Lovett et al. 2006). Warming, shifts in precipitation, and other alterations associated with climate change can affect forest vulnerability to these disturbance agents (Paradis et al. 2008, Sturrock et al. 2011). For example, the geographic range and incidence of dothistroma needle blight (*Dothistroma septosporum* [Dorog.] M. Morelet and *D. pini* Hulbary), which reduces growth of many conifers by causing premature needle defoliation, may shift with changing precipitation patterns (Woods et al. 2010).

The primary cause of biological invasions is from global commerce, not climate change. However, climate change is strongly connected to risks from continuing invasions. Increasing temperatures are generally expanding the geographic zones where potential invasive species could survive and reproduce if they arrive, for example, at ports of entry on the Eastern Seaboard and in the Great Lakes Waterway. The specter of global, climate-driven increases in invasion risks has prompted international organizations to discuss changes in trade restrictions to manage associated phytosanitation risks (Standards and Trade Development Facility 2009).

Outbreaks of lesser known forest insects have recently occurred in U.S. forests. Aspen leaf miner, (*Phyllocnistis populiella* Chambers) which reduces longevity of aspen leaves, has damaged 2.5 million ha of quaking aspen in Alaska since 1996 (Wagner et al. 2008). Large areas of willows were damaged during two eruptive outbreaks of the willow leafblotch miner (*Micurapteryx salicifoliella* Chambers)in the 1990s in two major river drainages in Alaska (Furniss et al. 2001); outbreaks of this leaf miner had not been previously reported. Substantial defoliation by Janet's looper (*Nepytia janetae* Rindge) of stressed trees in southwestern spruce-fir forests was preceded by uncharacteristically warm winters.[4] Defoliation by Janet's looper encouraged attack by opportunistic bark beetles. These examples are of previously rare native insects that displayed new eruptive behavior and caused notable forest disturbances. Our inability to anticipate disturbances by formerly innocuous native forest insects or pathogens is a major concern to forest health and monitoring.

Impacts and Interactions With Other Disturbances

Through their effects on tree growth and mortality, insects and pathogens have broad effects on ecosystem processes. Discussion of disturbance effects on biogeochemical cycling processes is presented in the "Effects of Climate Change on Forest Processes" section. In addition, insects and pathogens, by virtue of their host preferences, almost inevitably alter tree species composition within stands, can remove most host trees from many U.S. landscapes (tables 2.1, 2.2) (Lovett et al. 2006), and can modify forest types (e.g., from conifers to hardwoods) (Collins 2011, Orwig et al. 2002, Veblen et al. 1991). Because insects and pathogens often have size and age preferences for hosts, stands shift toward younger, smaller trees after biotic disturbances (Garnas et al. 2011a, Tchakerian and Couslon 2011, Ylioja et al. 2005). Wildlife habitat and biodiversity are altered by forest insects and pathogens, especially those that kill trees (Chan-McLeod 2006). Modified food supply, such as increases in insects and reductions in foliage, can affect multiple trophic levels (Chan-McLeod 2006, Drever et al. 2009). Both positive and negative effects occur depending on species, time since disturbance, surviving vegetation, ecosystem type, and spatial extent of outbreak (Chan-McLeod 2006).

Trees damaged by insects and pathogens can have substantial socioeconomic effects. However, valuation of those effects remains a challenge because of nonmarket costs and accounting for long-term losses (Aukema et al. 2011; Holmes et al. 2010; Kovacs et al. 2011a, 2011b). The economic effect of forest disturbances is difficult to quantify because insect and pathogen outbreaks have immediate effects on timber and pulp supply to the market and, if the outbreak is extensive, influence the future economic potential of forests.

Valuation of forest resources is further complicated by difficulty in quantifying nonmarket values such as ecosystem services (Holmes et al. 2010). Regions with dead and dying trees have reduced aesthetic value (Sheppard and Picard 2006) and housing prices (Holmes et al. 2010, Price et al. 2010). Direct economic effects occur owing to tree removal and replacement, such as the $10 billion spent after emerald ash borer (*Agrilus planipennis* Fairmaire) infestations (Kovacs et al. 2010, 2011b). Indirect effects include reduced quality of life, enhanced perceived risk of wildfire and other infrastructure damage, and increased conflict regarding community responses (Flint 2006).

Fire and biotic disturbances interact in several ways. Fires lead to younger stands that may be less susceptible to attack, and killed trees provide a food resource for some insects and pathogens (Parker et al. 2006). Insect-killed trees influence fuels and therefore fire behavior, although the effect depends on a number of factors, including the number of attacked trees within a stand and time since outbreak (e.g., Ayres and Lombardero 2000, Jenkins et al. 2008, Simard et al. 2011), and fire-induced increases in tree defenses can mitigate bark beetle risks (Lombardero and Ayres 2011).

Extreme soil water deficits (drought) arise because of reduced precipitation and increased temperatures, and these strongly affect tree defenses against and tolerance of herbivores and pathogens (Lorio 1993). Although water limitations that reduce tree growth might also reduce tree defenses (Bentz et al. 2009, Sturrock et al. 2011), theory and data suggest that there may be either no effect (Gaylord et al. 2007, McNulty et al. 1997) or the opposite effect (Herms and Mattson 1992, Lombardero et al. 2000). Drought decreases inducible plant defenses even as it increases constitutive plant defenses (Lombardero et al. 2000). Thus, drought may increase tree susceptibility to pathogens, which generally evoke inducible defenses (Sturrock et al. 2011; Worrall et al. 2010a, 2010b). Drought facilitates population increases of western bark beetles. Some aggressive species such as

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[4] Ann Lynch. 2011. Personal communication. Research entomologist, Rocky Mountain Research Station, Southwest Forest Science Complex, 2500 S. Pine Knoll Road, Flagstaff, AZ 86001.

mountain pine beetle are able to maintain epidemics after return to normal conditions, whereas others such as pinyon ips decline with alleviation of drought stress (Raffa et al. 2008).

Future Vulnerabilities and Opportunities

Geographic changes in climate and disturbance place forests at risk, because mortality converts a large proportion of live biomass to dead, decomposing biomass, and because the new forest may have to establish under less climatically favorable conditions. Observations show, and theory predicts, that changing climate is altering biotic disturbance and will likely continue to do so. A changing climate may lead to more stressed trees that are susceptible to attack by insects and pathogens (Bentz et al. 2009, Sturrock et al. 2011). Climatic warming and elevated CO_2, through their positive effects on tree growth, may increase forest maturation rates in some regions of the United States (McMahon et al. 2010, Salzer et al. 2009, Wang et al. 2011), leading to a more rapid transition to stands susceptible to some disturbance agents. Decreased disturbance by individual species may occur in some regions when year-round temperatures lead to maladaptive conditions for some bark beetles (Bentz et al. 2010), such as the recent decreases in southern pine beetle damage (Friedenberg et al. 2008).

Changing climates also introduce practical problems for mitigation of disturbance, because geographic mismatches occur between risks and management expertise. For example, suppression of the pine beetle epidemic in New Jersey is hindered both by limited local experience with bark beetles and because administrative boundaries (physical and perceived) exist between different organizations.

Key Findings

- Tree mortality caused by forest insects and pathogens likely exceeds other causes of disturbance for U.S. forests.
- Climate change will likely increase epidemics of forest insects and pathogens and related tree mortality, with broad consequences for forest ecosystems and their services.

Key Information Needs

- Improved monitoring of biotic disturbance agents; more accurate quantification of the extent, severity, and types of effects to forests from biotic disturbance; evaluation of the efficacy of management responses to current epidemics.
- Increased understanding of how climate alters the abundance and effects of forest insects and pathogens, including interactions with other insects, pathogens, and disturbances, to project future biotic disturbance.
- Increased capacity to manage risks from potential new invasive species, including identifying the most likely pathways of entry.
- Better understanding of the socioeconomic costs associated with biotic disturbance to forests.

Invasive Plants

Invasive plants are recent introductions of nonnative, exotic, or nonindigenous species that are (or have the potential to become) successfully established or naturalized, and that spread into new localized natural habitats or ecoregions with the potential to cause economic or environmental harm (Lodge et al. 2006). This definition of “invasive” (1) does not consider native species that have recently expanded their range, such as juniper (*Juniperus* spp.) in the Western United States (Miller and Wigand 1994, Miller et al. 2005), (2) involves defined temporal and spatial scales, and (3) considers social values related to economic and environmental effects.

An estimated 5,000 nonnative plant species exist in U.S. natural ecosystems (Pimentel et al. 2005) (table 2.3). In general, the effects of invasive plants include a reduction in native biodiversity, changes in species composition, loss of habitat for dependent species (e.g., wildlife), changes in biogeochemical cycling, changes in ecosystem water use, and alteration of disturbance regimes. Billions of dollars are spent every year to mitigate invasive plants or control their effects (Pimentel et al. 2005). Negative environmental effects are scale-dependent (Powell et al. 2011), with some subtle beneficial properties (Sage et al. 2009), on ecosystem function (Myers et al. 2000, Zavaleta et al. 2001). For

Table 2.3—Summary of common invasive plant species and environmental impacts for forests and woodlands in the United States

Species	Common name	Growth form	Environmental impacts
Acer platanoides L.	Norway maple	Tree	Reduces abundance and diversity of native species; alters of community structure (e.g., shading of understory)
Ailanthus altissima Desf.	Tree of heaven	Tree	Alters ecosystem processes (e.g., increases soil nitrogen, alters successional trajectories); displaces native vegetation; allelopathic; roots can damage buildings and sewer lines; risk to human health (pollen allergies, sap-caused dermatitis)
Alliaria petiolata (m. Bieb.) Cavara and Grande	Garlic mustard	Biennial forb	Reduces abundance and diversity of native species; potentially allelopathic
Berberis thunbergii DC.	Japanese barberry	Shrub	Displaces native shrubs; changes soil properties (alters soil microbial composition, increases nitrate concentration); alters successional patterns; potentially increases fire risk (owing to increased biomass)
Bromus tectorum L.	Cheatgrass	Annual grass	May cause community type conversion; alters community structure, process, and function (e.g., decreases diversity, changes fire disturbance regime frequency, alters successional patterns and nutrient cycling)
Celastrus orbiculatus Thunb.	Oriental bittersweet	Vine	Alters soil chemistry (e.g., increased pH, increased calcium levels), plant succession and stand structure (e.g., shades out understory, increases continuity of overstory vegetation); decreases native plant diversity; reduces productivity in managed systems
Centaurea solstitialis L.	Yellow star-thistle	Annual forb	Displaces native plants, reduces native wildlife and forage; decreases native diversity; depletes soil moisture, altering water cycle; reduces productivity in agricultural systems (lowers yield and forage quality of rangelands)
Centaurea stoebe L.	Spotted knapweed	Biennial/perennial	Reduces plant richness, diversity, cryptogam cover, soil fertility; reduces forage production; poisonous to horses; increases bare ground and surface water runoff, and can lead to stream sedimentation; allelopathic
Cirsium arvense (L.) Scop.	Canada thistle	Perennial forb	Possible allelopathy; displaces native vegetation; alters community structure and composition; reduces diversity; reduces forage and livestock production
Cytisus scoparius (L.) Link	Scotch broom	Shrub	Interferes with conifer establishment; reduces growth and biomass of trees; alters community composition and structure (increases stand density, often creating monospecific stands); alters soil chemistry (increases nitrogen); toxic to livestock
Hedera helix L.	English ivy	Vine	Alters community structure; displaces native ground flora; weakens or kills host trees; potential to reduce water quality and increase soil erosion and soil nitrogen
Imperata cylindrical (L.) P. Beauv.	Cogongrass	Grass	Alters ecosystem structure (e.g., decreases growth and increases mortality of young trees) and function and decreases diversity; shortens fire return intervals and increases fire intensity, interferes with pine and oak regeneration
Ligustrum sinense Lour.	Chinese privet	Shrub	Interferes with native hardwood regeneration; alters species composition and community structure (forms dense monospecific stands)
Lonicera japonica Thunb.	Japanese honeysuckle	Vine	Alters forest structure and species composition; inhibits pine regeneration potentially weakens or kills host trees; suppresses native vegetation; provides food for wildlife; early- and late-season host for agricultural pests
Lygodium japonicum (Thunb. Ex Murr.) Sw.	Japanese climbing fern	Climbing fern	Reduces native understory vegetation; potentially weakens or kills host trees; interferes with overstory tree regeneration
Microstegium vimineum (Trin.) A. Camus	Japanese stiltgrass, Nepalese browntop	Annual grass	Reduces ecosystem function (alters soil characteristics and microfaunal composition, decreases diversity, alters stand structure); reduces timber production; possibly allelopathic
Pueraria montana var. *lobata*	Kudzu	Vine	Potentially eliminates forest cover; overtops, weakens, and kills host trees; reduces timber production; increases winter fire risk
Triadica sebifera (Willd.) Maesen and S.M. Alemeida ex Sanjappa and Predeep	Chinese tallow, tallowtree	Tree	Displaces native species and reduces diversity; increases soil nutrient availability; reduces fire frequency and intensity

example, some consider species in the genus *Tamarix* to be among the most aggressively invasive and detrimental exotic plants in the United States (Stein and Flack 1996), but others point out benefits, including sediment stabilization and the creation of vertebrate habitat in riparian areas that can no longer support native vegetation (Cohn 2005).

The spatial extent of many invasive plants at any point in time has been difficult to determine, limiting assessment of overall consequences of invasive plants. One assessment (Duncan et al. 2004) for the Western United States indicates that 16 invasive plants account for most current invasive plant problems. *Centaurea* species are particularly widespread and persistent in the West (table 2.3) (box 2.5). Cogongrass (*Imperata cylindrica* [L.] Raeusch.) (table 2.3), which has invaded extensive forested areas of the Southeast, is considered to be one of the most problematic invasive plants in the world (box 2.6). Mountain ecosystems tend to have fewer invasive plant species than other regions because of a short growing season, limited settlement history, relatively low frequency of seed sources, and prevalence of closed-canopy conifer forests that limit light in the understory and acidify the soil (Parks et al. 2005).

Box 2.5—*Centaurea* invasion in the Western United States

Eurasian forbs in the genus *Centaurea* are the most abundant invasive plants in the Western United States, covering over 7 million ha (3 million ha in California). Collectively known as knapweeds and star-thistles, 12 *Centaurea* species are listed as noxious in at least one U.S. state (5 species account for most of the damage). Although these species are usually associated with grasslands, they also affect forest ecosystems, particularly in open areas and after fire or other disturbances. In the northwestern United States, many forest ecosystems are susceptible to invasion by *Centaurea,* although some form of disturbance is typically required, particularly at higher elevations (Parks et al. 2005).

Yellow-star thistle has a strong growth and competitive response to elevated carbon dioxide (CO_2) (Dukes 2002). In one study, its aboveground biomass increased more than sixfold in response to elevated CO_2, which allowed it to compete aggressively with native species, although supplemental precipitation reduced its establishment in the field (Dukes et al. 2011). Predictive models project various changes in the range of *Centaurea* species in a warmer climate. Broennimann and Guisan (2008) projected a northern shift and reduced invasion extent for spotted knapweed by 2080 using the hot, dry HadCM-A1FI scenario, but Bradley et al. (2009) suggested that the distribution of yellow-star thistle was likely to increase in a warming West. Cumming (2007) found that small increases in temperature and precipitation would expand the suitable habitat for spotted knapweed in the short term, but large increases (4.5 °C, 10 cm) would decrease suitable habitat in Montana in the long term (several decades). Model output contains considerable uncertainty regarding potential changes in the geographic range of *Centaurea* species and thus represents potential vulnerabilities rather than predictions.

Tyrol knapweed (*Centaurea nigrescens* Willd.), shown here in a dry, mixed-conifer forest in eastern Washington, is listed as a noxious weed in four Western States. It is also found in the Midwestern and Northeastern United States, invading forests along roadsides and in disturbed or open areas. (Photos by Gabrielle Snider.)

Box 2.6—Invasive grasses, fire, and forests

Species such as cogongrass (*Imperata cylindrica* [L.] P. Beauv.) in the Southeastern United States and cheatgrass (*Bromus tectorum* L.) in the West are invaders that alter fire regimes and are some of the most important ecosystem-altering species on the planet. Cogongrass threatens native ecosystems and forest plantations in the southeast, generally invading areas after a disturbance (e.g., mining, timber harvest, highway construction, natural fire, or flood). It is a major problem for forest industry, invading and persisting in newly established pine plantations (Jose et al. 2002). In sandhill plant communities, cogongrass provides horizontal and vertical fuel continuity, shifting surface fire regimes to crown fire regimes and increasing fire-caused mortality in longleaf pine (*Pinus palustris* Mill.) (Lippincott 2000), potentially shifting a species-diverse pine savanna to a grassland dominated by cogongrass. Cogongrass does not tolerate low temperatures, but increased warming could increase the threat of cogongrass invasion into new areas. Empirical modeling has shown that climatic habitat for cogongrass could greatly increase, although it would still be restricted to the Gulf Coast (Bradley et al. 2010).

Cheatgrass is widely distributed in western North America and dominates many steppe communities (Mack 1981). After disturbance, this species can invade low-elevation forests (Keeley and McGInnis 2007, Keeley et al. 2003, Kerns et al. 2006), creating surface fuel continuity from arid lowlands into forested uplands. After establishment, cheatgrass contributes heavily to fine, continuous, and highly combustible fuel components that dry out early in the year, thus increasing the length of the fire season in some forests. Empirical modeling indicates that future changes in the climatic habitat of cheatgrass will depend on precipitation as well as temperature (Bradley et al. 2009). Climate models based on decreased precipitation, especially in summer, project expansion of cheatgrass area, and a reduction in the area of suitable climatic habitat, by up to 45 percent in Colorado, Montana, Utah, and Wyoming. Models based on increased precipitation, however, project reduction of cheatgrass area by as much as 70 percent. Elevated carbon dioxide increases cheatgrass productivity, a phenomenon that may already be contributing to the vigor and spread of this species (Ziska et al. 2005). Increased productivity causes higher fuel loads, potentially resulting in higher intensity fires. These consequences, in combination with more area burned by wildfire as caused by higher temperatures (Littell et al. 2009), can alter fire regimes in dry Western forests.

A severe infestation of cogongrass in a longleaf pine upland in central Florida. (Photo by James R. Meeker, U.S. Forest Service, available from Forestry Images, http://www.forestryimages.org/browse/detail.cfm?imgnum=3970058).

Interactions Between Climate Change and Plant Invasion

Plant invasions can be influenced by warmer temperatures, earlier springs and earlier snowmelt, reduced snowpack, changes in fire regimes, elevated N deposition, and elevated CO_2 concentrations. The responses of invasive plants to climate change should be considered separately from those of native species, because invasive plants (1) have characteristics that may differ from native species, (2) can be highly adaptive (Sexton et al. 2002), (3) have life-history characteristics that facilitate rapid population expansion, and (4) often require different management approaches than for native species (Hellmann et al. 2008). Successful invasion of a natural community depends on environment, disturbance, resource availability, biotic resistance, and propagule pressure (D'Antonio et al. 2001, Davis et al. 2000, Eschtruth and Battles 2009, Levine et al. 2004, Pauchard et al. 2009). Climate change may influence all of these drivers of invasion, with high variability across space and time.

Temperature, Precipitation, and Carbon Dioxide

Climate change will alter the abiotic conditions under which plant species can establish, survive, reproduce, and spread. Key environmental consequences of climate change are increased temperature, longer growing seasons, less snow, and more frequent drought. These effects are expected to increase plant stress and decrease survival in the drier, warmer, and lower elevation portions of species' ranges (Allen and Breshears 1998). Abiotic factors probably constrain the range of many invasive plants and limit their successful establishment (Alpert et al. 2000, Pauchard et al. 2009). With climate change, however, new habitat, once too cold or wet, may become available, enabling plants to survive outside their historical ranges and expand beyond their current ranges.

Many native plants are projected to move northward or upward in elevation with climate change. Examples of invasive plants projected to follow this pattern are rare, but information on species tolerances can provide insight on potential responses. For example, the northern limit of Japanese barberry (*Berberis thunbergii* DC.) (table 2.3), an invasive shrub in the Eastern United States, is determined by low temperature tolerance, the southern limit by cold stratification requirements for germination, and the western limit by drought tolerance (Silander and Klepeis 1999). The widespread invasive tree of heaven (*Ailanthus altissima* [P. Mill.] Swingle) is limited by cold and prolonged snow cover to lower mountain slopes, but it may be able to colonize during several successive years of mild climate, conditions that may become more common under climate change (Miller 1990). Temperature change is not the only driver for plant range expansion or contraction. Soil water availability and regional changes in climatic water balance may be important for plant invasions, particularly at lower elevations (Chambers et al. 2007, Crimmins et al. 2011). Besides changes in range, species growth, productivity, and reproduction may also change as climatic conditions change. For example, invasive plants may be better able to adjust to rapid changes in abiotic conditions by tracking seasonal temperature trends and shifting their phenologies (e.g., earlier spring warming) (Willis et al. 2010).

Increased productivity in response to elevated CO_2 has been documented under controlled conditions for several invasive plant species, including cheatgrass (*Bromus tectorum* L.), Canada thistle (*Cirsium arvense* [L.] Scop.), spotted knapweed (*Centaurea melitensis* L.), yellow star-thistle (*C. solstitialis* L.), and kudzu (*Pueraria montana* [Lour.] Merr.) (Dukes et al. 2011, Ziska and Dukes 2011, Ziska and George 2004) (table 2.3) (boxes 2.5, 2.6). Response to CO_2 enrichment is less predictable when plants are grown in the field (Dukes and Mooney 1999, Ziska and Dukes 2011), where response may be limited by nutrients and water availability. Carbon dioxide enrichment can also increase efficiency of water use, which can partially ameliorate conditions associated with decreased water availability, particularly for C_3 plants (Eamus 1991).[5] This phenomenon may be partially responsible for global patterns of C_3 encroachment in grasslands dominated by C_4 plants or mixed species (Bond and Migdley 2000).[6]

Disturbance and Resource Availability

Disturbances such as fire, landslides, volcanic activity, logging, and road building open forest canopies, reduce competition, and expose mineral soil, increasing light and nutrient availability (D'Antonio et al. 1999, Elton 1958, Hobbs and Huenneke 1992). Invasive plants are generally well adapted to use increased resources. Fluctuating resource availability, coinciding with available propagules, facilitates regeneration and establishment of both native and exotic invasive species associated with forest development after disturbance (Davis et al. 2000, Halpern 1989, Parks et al. 2005). Opportunities for invasions may also be created by forest thinning, fuel treatments, and biofuel harvesting done to adapt or mitigate climate change (Bailey et al. 1998, Nelson et al. 2008, Silveri et al. 2001). However, the spatial extent of invasions may be limited (Nelson et al. 2008), especially for shade-intolerant species in closed-canopy western forests.

The reintroduction of fire is a high priority for restoration and management of fire-adapted forests such as ponderosa pine (*Pinus ponderosa* Douglas ex P. Lawson & C. Lawson), longleaf pine (*Pinus palustris* Mill.*)*, and loblolly pine (*Pinus taeda* L.). Invasive plants, especially annual grasses (box 2.6), can spread rapidly after fire, particularly in high-severity burns (D'Antonio 2000, Keeley and McGinnis 2007, Kerns et al. 2006). Forest sites treated with prescribed fire, which are often near the wildland-urban interface and roads, are also well positioned for invasive plant introduction and spread (Keeley et al. 2003).

Biotic factors—

The success of plant invasions is regulated by competition from resident plants, often measured as species richness and abundance (Levine 2000, Seabloom et al. 2003), and land managers can alter postdisturbance (logging, fire) invasive establishment by seeding to increase native plant competition. Although native plant competition can be overwhelmed by invasive plant seed abundance (D'Antonio et al. 2001, Lonsdale 1999), resistance related to soil properties is more likely to withstand seed abundance. Native plant competition with invasive plants can also be affected by the effects of predation, herbivory, and pathogens associated with native species. Native plant competition may change as temperature and ambient CO_2 increase; numerous studies have documented that weedy plants are more productive in an elevated CO_2 environment (Ziska and George 2004).

Propagule pressure—

Propagule pressure, which includes seed size, numbers, and temporal and spatial patterns, is perhaps the most important driver of successful invasions in forest ecosystems (Colautti et al. 2006, Eschtruth and Battles 2009, Simberloff 2009, Tilman 1997). For invasive plants, propagule pressure is largely controlled by factors other than climate. For example, the most critical factors projecting plant invasion in eastern hemlock (*Tsuga canadensis* [L.] Carrière) forests in the Eastern United States are overstory canopy disturbance and propagule pressure (Eschtruth and Battles 2009). However, little is known about how biotic and abiotic resistance factors interact with propagule supply to influence exotic plant invasion (D'Antonio et al. 2001, Lonsdale 1999).

Atmospheric CO_2 may influence seed production, through enhanced flowering under elevated CO_2, increasing the probability that a smaller seed can establish a viable population (Simberloff 2009). Of greater concern is how climate change may alter human activities that transfer seeds. For example, climate change could alter tourism and commerce, enhance survival of seeds during transport (Hellmann et al. 2008), and shift recreation to higher elevations. Changes in atmospheric circulation patterns could also alter wind-dispersed species, allowing new species to arrive in areas that previously had few seeds.

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[5] C_3 plants are those in which photosynthetic C fixation occurs in a metabolic process that converts CO_2 and ribulose biphospate into 3-phosphoglycerate. This phenomenon may be partially responsible for global patterns of C_3 encroachment in C_4 plants or mixed grassland species.

[6] Photosynthetic C fixation occurs in a metabolic process that uses the enzeyme PEP carboxylase to add CO_2 to phosphoeonlpyruvate, producing a 4-C compound prior to subsequent transport and use in the Calvin cycle.

Vulnerabilities—

Future climate change may increase the likelihood of invasion of forest lands for several reasons: the potential for increased ecological disturbance, the effect of warming on species distributions, enhanced competitiveness of invasive plants owing to elevated CO_2, and increased stress to native species and ecosystems (Breshears et al. 2005, Dukes and Mooney 1999, Pauchard et al. 2009, Ziska and Dukes 2011). The potential for warming itself to increase the risk of invasion in temperate mountainous regions is greater than in other regions because cold temperature has tended to limit the establishment of invasive plants in forests. Future actions to control invasive plants may also become less effective. Some herbicides are less effective on plants grown in elevated CO_2 (Ziska and Teasdale 2000), and some biocontrol methods may no longer be effective in a warmer climate (Hellmann et al. 2008).

Empirical models used to assess the potential change in suitable climatic conditions for invasive plants suggest that a warmer climate could result in both range expansion and contraction for common invasive plants (Bradley et al. 2009, Kerns et al. 2009, Pattison and Mack 2008, Sasek and Strain 1990). However, a weakness of empirically driven species distribution models is that they can be created without prior knowledge about species ecophysiology, autecology, synecology, and biotic interactions. Process-based models may ultimately prove more robust for prediction, although model parameters are quantified from experimental data or the research literature, which themselves have uncertainties. Regardless of whether the models are empirical or process-based, all model results have considerable uncertainty regarding their ability to project potential changes in the geographic range of invasive plants (Littell et al. 2011).

The idea that climate change may increase the success of biological invaders has been a key concept for more than a decade (Dukes and Mooney 1999), although empirical documentation of this phenomenon is largely absent (but see Willis et al. 2010). It is critical to understand the response of the most detrimental invasive plants to individual climatic factors, interactions between those factors, and interactions among diverse biological factors. For management responses to plant invasions to be cost-effective and successful, assertive action is needed in the early phase of invasion. A potentially useful approach is a climate change-based modification of the Early Detection and Rapid Response System (National Invasive Species Council 2001). For example, risk assessment could be done over broader geographic areas than has been performed in the past (Hellman et al. 2008). The successful control of invasive plants over broad areas of forest lands ultimately depends on knowledge about resistance of native species to invasion and our ability to limit propagule pressure.

Key Findings

- Climate change will likely increase the establishment of invasive plants in U.S. forests.
- Risk of exotic invasive plants entering forests is likely highest in mountainous ecosystems, where cooler temperatures and closed-canopy forests may have limited invasives under historical climate.

Key Information Needs

- Increased understanding of the responses of the most detrimental invasive plants to climate and biological factors.
- Better models for projecting potential distributions of invasive plants.

Erosion, Landslides, and Precipitation Variability

Based on analysis of recent climate records and the projections of climate change simulations, hydroclimate extremes will become more prominent with a warming climate (O'Gorman and Schneider 2009, Trenberth et al. 2009), with potential increases in flood frequency, droughts and low flow conditions, saturation events, landslide occurrence, and erosion. Ecosystems are expected to differ in their response to changes in precipitation intensity and interstorm length because of differences in geomorphic conditions, climate, species assemblages, and susceptibility to drought. For erosion, these differences may be predictable with a general mass balance framework, but other processes are poorly understood, such as the effect of drought on short- and long-term tree mortality, the resistance to insects and pathogens of vegetation, and subsequent feedbacks to erosion processes. The indirect effects of disturbances (e.g., fire, insect infestations, pathogens) to shifts in water balance will complicate the response of erosion and need to be incorporated into assessments of effects. Changing species composition will also have potential effects on forest ecosystem water balance, as discussed in the "Forest Hydrological Processes" section.

Erosion and Landslides

Changes in precipitation intensity, and in the magnitude and frequency of precipitation events that saturate the soil and cause runoff, will interact with mass wasting and erosion in both direct and indirect ways. Expected changes in annual precipitation differ across the United States and are uncertain over much of it, particularly at the local scale. Potential annual increases and decreases will directly contribute to the amount of water available to drive mass wasting at both seasonal and event scales. Increases in extremes of precipitation intensity (Easterling et al. 2000a, Karl and Knight 1998), rain-on-snow during mid-winter melt (Hamlet and Lettenmaier 2007, Lettenmaier and Gan 1990, Wenger et al. 2011), and transport of moisture in atmospheric rivers (Dettinger 2011, Ralph et al. 2006) are all likely mechanisms for increasing sufficient pore water pressure or hillslope, thus increasing the risk of landslides, erosion, and gully formation for individual storms. Seasonal to annual changes in precipitation will contribute to soil moisture and groundwater levels, which can amplify or mitigate individual events. Although we have a significant understanding of erosion and landslide processes, the ability to predict or manage high-risk areas is limited by uncertainty in predicting changes in precipitation amount, frequency, and location of extreme rainfall events.

Direct effects of some climatic changes on sediment yield and mass wasting may be overshadowed by longer term, indirect effects through vegetation response (Bull 1991, Collins and Bras 2008, Goode et al. 2011, Istanbulluoglu and Bras 2006, Kirkby and Cox 1995, Langbein and Schumm 1958, Tucker and Bras 1998). Although decreasing precipitation in some places might suggest reduced risks of erosion or landslides, this change may have indirect effects on mortality and thinning of vegetation and fire risk; these effects could have much greater consequences for erosion and landslides, through reductions in root reinforcement of soil and greater exposure of soil to precipitation effect and runoff. For example, paleoclimatic and paleoecological evidence links periods of drought and severe fire to severe erosion events (Briffa 2000, Marlon et al. 2006, Meyer and Pierce 2003, Meyer et al. 1992, Pierce et al. 2004, Swetnam and Betancourt 1998, Whitlock et al. 2003). At shorter time scales, years of widespread fire are linked to severely dry and warm years (Heyerdahl et al. 2008, Holden et al. 2011b, Littell et al. 2009, McKenzie et al. 2004, Morgan et al. 2008). As we shift toward a drier and warmer climate in the Western United States, more areas are likely to burn annually (e.g., Holden et al. 2011b, Littell et al. 2009, Running, 2006, Spracklen et al. 2009), with resulting postfire debris flows (Cannon et al. 2010, Luce 2005, Meyer and Pierce 2003, Moody and Martin 2009, Shakesby and Doerr 2006). Breshears et al. (2005) documented drought-induced canopy mortality of ponderosa pine, followed by erosional loss of topsoil and nutrients, with subsequent species replacement by pinyon pine and juniper. These types of state transitions may indicate the type of complex feedbacks that will lead to permanent canopy shifts, rather than to disturbance and recovery.

Adjustment of canopy density and root distributions to longer interstorm periods may increase the efficiency of use of rain or snowmelt (Brooks et al 2011, Hwang et al 2009). The response of both annual runoff and runoff from extreme events may be amplified or mitigated by forest canopy adjustment to temperature, moisture, N, and atmospheric CO_2. Increased precipitation intensity and amount, combined with lower root biomass from a drier climate, can yield more unstable slopes. An important interaction needing additional research is the effect of drought on adjustments of forest canopy leaf area and belowground allocation of C to hydrologic flowpaths and root reinforcement of soil. Shifts in species dominance can also result in major changes in root depth and cohesion (Hales et al. 2009). The spatial pattern of unstable slope conditions that can lead to landslides is influenced by interactions among the lateral redistribution of soil water in large events, the resulting pattern of high pore pressures with topographic slope, and root cohesive strength (Band et al. 2011).

Drought and Water Supply

Decreased precipitation and runoff is projected for substantial portions of the globe (Milly et al. 2005). Projections of the drought extent over the next 75 years show that the proportion of global land mass experiencing drought will double from 15 percent to 30 percent (Burke et al. 2006), and on most land masses, dry season precipitation is expected to decline by 15 percent (Solomon et al. 2009).

Many projected declines in precipitation are in semi-arid regions at mid-latitudes, where forests are at the limits of their ranges. Projections for the strongest declines in the United States are in the Southwest, strongly affecting water supply (Barnett and Pierce 2008, Rajagopalan et al. 2009). Further decreases in precipitation will probably increase both forest mortality (Allen et al. 2010, Holden et al. 2011a) and fire risk (Westerling et al. 2011); however, forest mortality may not substantially mitigate runoff reductions associated with decreased precipitation (Adams et al. 2011). Historical observations of interannual variability in precipitation in the Western United States have shown substantial increases in variability in the last half-century (Luce and Holden 2009, Pagano and Garen 2005), even in portions of the Western United States not projected to show precipitation declines. Short-term severe droughts have consequences for vegetation (Holden et al. 2011b, van Mantgem et al. 2009) and water supply. Although there has been interest in using forest harvest to augment water supplies, three factors limit the utility of the approach: (1) most increases in water yield after harvest occur in wet years (Brown et al. 2005, Ford et al. 2011b, Troendle and King 1987); (2) water yield increases in snow environments occur earlier in the year, exacerbating flow timing issues by climate change (Troendle and Leaf 1981, Troendle et al. 2010); and (3) in warmer and moister locations, increases in water yields can be replaced by decreases as young vegetation reestablishes within a few years (Brown et al. 2005, Ford et al. 2011b).

Key Findings

- Concentrating precipitation in more intense storms will likely increase erosion and landslide risk, but the ability to project effects at meaningful spatial and temporal scales is limited by uncertainties in projecting future precipitation regimes.
- Increases in drought frequency and severity will likely increase tree mortality and reduce streamflow.

Key Information Needs

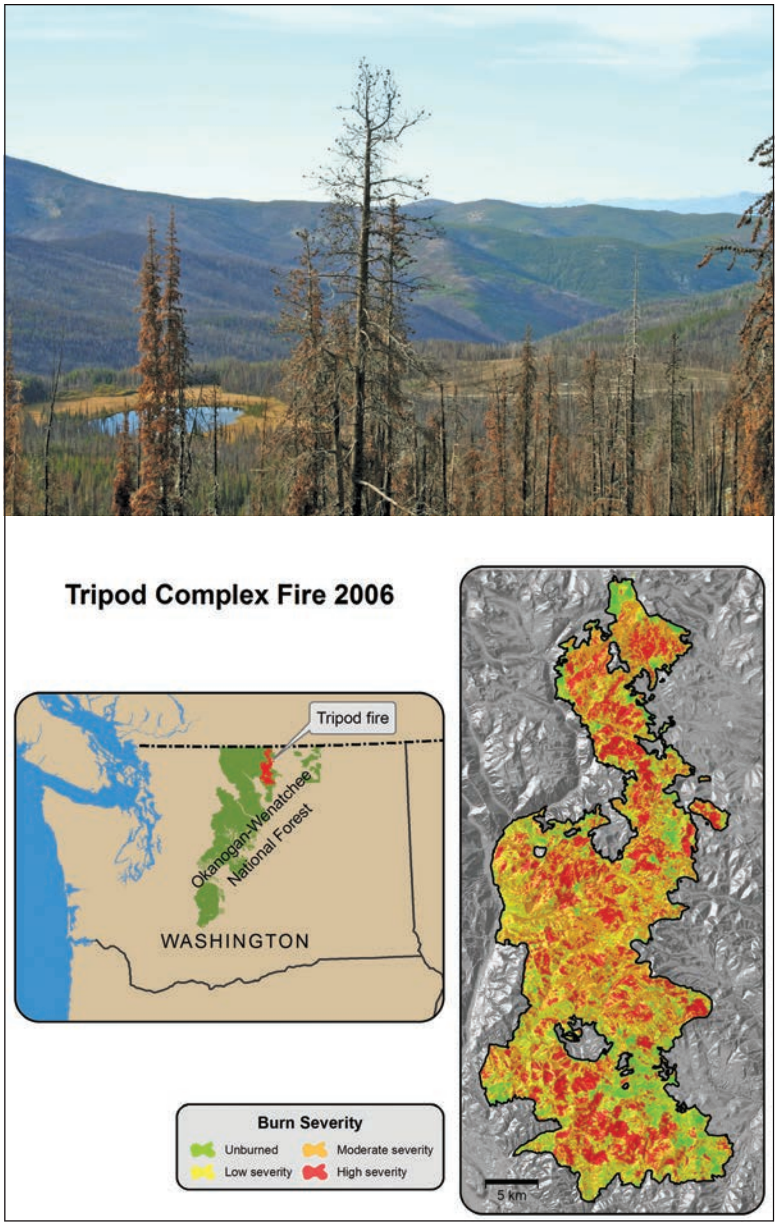
- Improved understanding of the effects of tree mortality and changing species composition on soil stability.
- Improved projections of changes in precipitation amount, and spatial and temporal distribution of extreme events.

Disturbance Interactions

A particular challenge is to understand interactions among disturbance regimes (Bigler et al. 2005, Busby et al. 2008). How will massive outbreaks of bark beetles, which kill drought-stressed trees by feeding on cambial tissues, increase the potential for large severe wildfires in a warming climate (fig. 2.10)? Interactions between processes can amplify or mute the overall effects of changes in complex forest ecosystems. The predominance of negative and positive feedbacks within and between processes will determine the stability or instability of the system.

Thresholds

Disturbance interactions may rapidly bring ecosystems to thresholds (Groffman et al. 2006). For example, Allen and Breshears (1998) and Breshears et al. (2005) documented rapid dieback of pinyon pine across the arid Southwest. Mature trees were pushed over a threshold by a combination of “global-change type drought” (Breshears et al. 2005) and an opportunistic bark beetle invasion. Regeneration of pinyon pine will determine whether this mortality represents a threshold for the ecosystem. Characteristic patterns of patchiness or continuity may indicate thresholds that have been approached or crossed (Scheffer et al. 2009) (table 2.4). For example, the invasion of sagebrush steppe by cheatgrass



Susan Prichard

Figure 2.10—Mountain pine beetle outbreak in the years before the Tripod Complex Fire (2006) in north-central Washington created a “perfect storm” in higher elevation lodgepole pine stands, which burned with exceptionally high intensity. This figure shows how the timing of other disturbances can exacerbate or mitigate wildfire severity.

Table 2.4—Characteristics of continuous versus abrupt thresholds

Cause/response	Predictable	Unexpected
Continuous	A tipping point is known from previous experience or modeling, and trends in the controlling factor(s) are measured. Example: gradual loss of habitat toward a point at which metapopulation models predict extirpation.	Controlling factor is changing gradually, but ecosystem effects or interactions of response variables are too complex to predict. Example: increases in an invasive nonnative species with unknown effects on biotic interactions of natives or grazing pressure.
Abrupt	Pulses in a controlling factor precipitate an inevitable response. Example: large disturbance or invasion (perhaps unprecedented) changes structure and composition of a landscape with a loss of 90 percent of potential nesting trees for northern spotted owls.	Pulses in a controlling factor (or very likely multiple controls) are expected to produce surprises. Example: changing fire frequency and mountain pine beetle outbreaks may have sudden consequences for vegetation, animals, or landscape pattern.

(Fischer et al. 1996) and of the Sonoran Desert by buffelgrass (Esque et al. 2007) provide fuel continuity and the potential for much more extensive wildfires than noninvaded areas with patchy fuels.

A notable threshold response to multiple stressors is the reproductive cycle of mountain pine beetle (Logan and Powell 2001), whose outbreaks have killed mature trees across millions of hectares of pine in western North America. Within particular ranges of minimum winter temperatures and growing-season degree days, the reproductive cycle is synchronized to the seasonal cycle, permitting larvae to emerge at the right time to ensure maximum survival and therefore epidemic population size. This "adaptive seasonality," combined with drought-caused and age-related vulnerability of the host species, has brought an abrupt increase in mortality of lodgepole pine across the West (Hicke et al. 2006).

Conceptually, the thresholds are fairly well understood. Mathematical models abound, from early work on catastrophe theory and its associated hysteresis to identification of approaching thresholds via statistical properties. This modeling has by necessity taken place in simplified (often virtual) ecosystems, and a major challenge remains to apply such sophistication to real-world systems outside the specific examples chosen by modelers to test their hypotheses. A larger challenge will always be the unpredictability of the occurrence of contingent, interacting events that push systems across thresholds.

Stress Complexes: From Conceptual to Quantitative Models

In the context of the effects of climate change on ecosystems, sensitivity to disturbance interactions is extended to environmental drivers not usually identified as disturbances. For example, extreme temperatures, drought, and air pollution put forest ecosystems under stress, which may increase their vulnerability to "true" disturbances such as fire, insect outbreaks, and pathogens. Following McKenzie et al. (2009), we refer to interacting stresses as stress complexes and present three brief conceptual examples, from California, Alaska (both drawing on McKenzie et al. 2009), and the Southeast.

A striking feature of mixed-conifer forests in southern Sierra Nevada and southern California is ambient air pollution (fig. 2.11), particularly elevated ozone, which affects plant vigor by reducing net photosynthesis and therefore growth (Peterson et al. 1991) and is often concentrated at middle and upper elevations (Brace and Peterson 1998). Air pollution exacerbates drought stress from warmer temperatures, which amplifies biotic stresses such as insects and pathogens (Ferrell 1996). The stress complex for the Sierra Nevada is represented conceptually in fig. 2.12; interacting disturbances form the core of drivers of ecosystem change, modified by climate, management, and air pollution.

The state of Alaska has experienced massive fires in the last decade, including the five largest fires in the United States. Over 2.5 million ha burned in the interior in 2004. Concurrently (1990s), massive outbreaks of the spruce beetle occurred on and near the Kenai Peninsula in south-central Alaska (Berg et al. 2006) (fig. 2.13). Although periodic outbreaks have occurred throughout the historical record, both in south-central Alaska and the southwestern Yukon, these most recent outbreaks may be unprecedented in both extent and percentage of mortality (over 90 percent in many places) (Berg et al. 2006).

Both of these phenomena, wildfire and bark beetle outbreak, are associated with warmer temperatures in recent decades (Duffy et al. 2005, Werner et al. 2006). At the same time, major hydrological changes are underway from the cumulative effects of warming. Permafrost degradation is widespread in central Alaska, shifting ecosystems from birch

D. McKenzie

Figure 2.11—Air pollution in the Sierra Nevada foothills from the Central Valley in California.

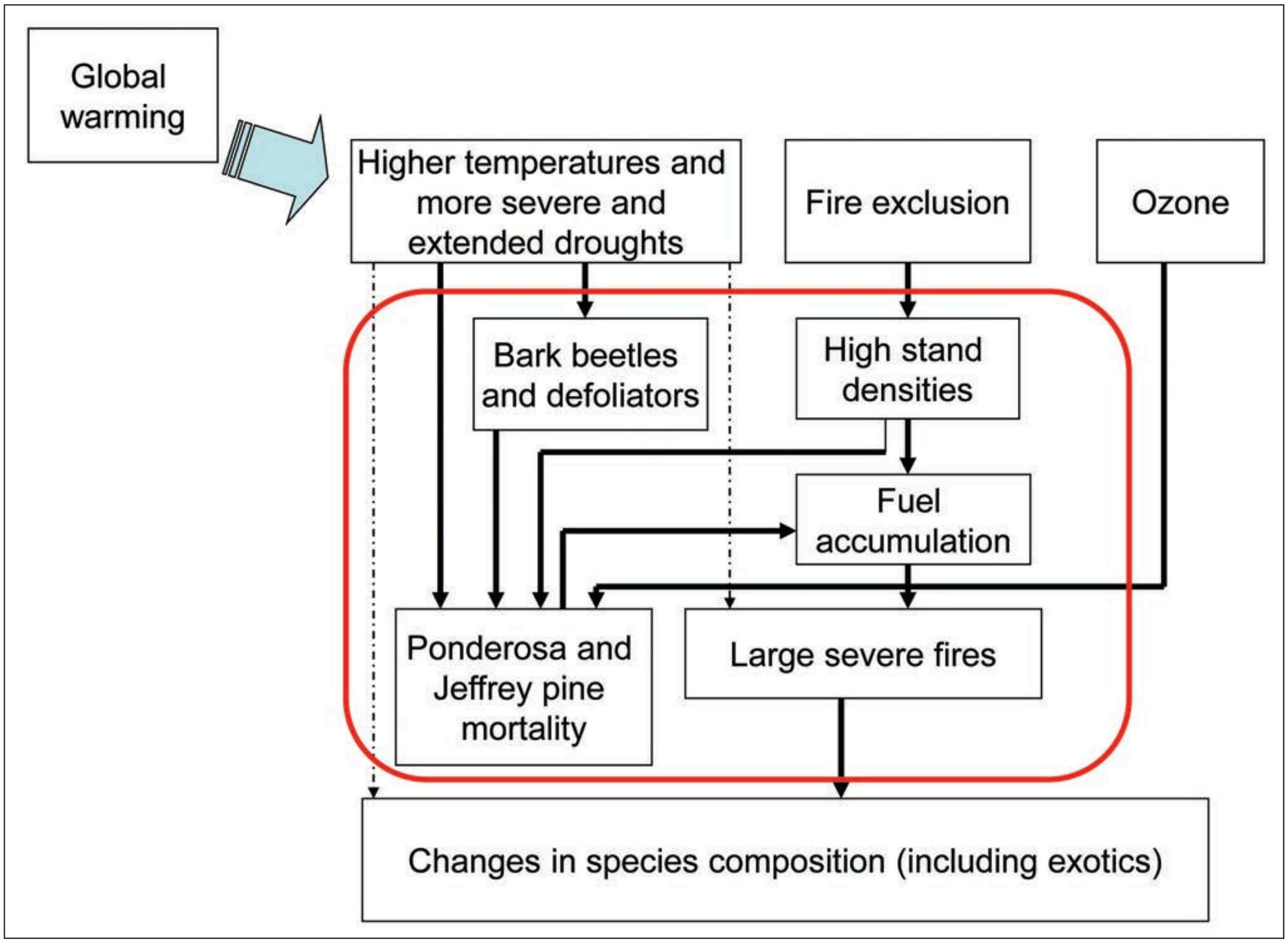


Figure 2.12—A conceptual model of stress complexes in Sierra Nevada and southern Californian mixed-conifer forests. The effects of insects and fire disturbance regimes (red box) and of fire exclusion are exacerbated by global warming. Stand-replacing fires and drought-induced mortality both contribute to species changes and exotic invasions. Modified from McKenzie et al. (2009).


W. M. Ciesla

Figure 2.13—Mortality of white spruce from bark-beetle attack on the Kenai Peninsula, Alaska.

forests to wetland types such as bogs and fens (Jorgensen et al. 2001). If broad-scale water balances become increasingly negative, peatlands may begin to support upland forest species (Klein et al. 2005). The stress complex for Alaska is represented conceptually in fig. 2.14; upland and lowland ecosystems may follow parallel but contrasting paths toward new structure and species composition.

Much of the forested landscape in the Southeast is adapted to frequent fire such that, unlike much of the West and Alaska, prescribed fire is a mainstay of ecosystem management. Fire-adapted inland forests overlap geographically with coastal areas affected by hurricanes and potentially by sea level rise (Ross et al. 2009), such that interactions between wildfires and hurricanes are synergistic (fig. 2.15). For example, dry-season (prescribed) fires may have actually been more severe than wet-season (lightning) fires in some areas, causing structural damage via cambium kill and subsequent increased vulnerability to hurricane damage (Platt et al. 2002). The stress complex for the Southeast is represented conceptually in fig. 2.16, where different disturbances "meet" in the outcomes for forest ecosystems.

Uncertainties

Our broad understanding of multiple stressors is mainly qualitative, despite case studies in various ecosystems that have measured the effects of interactions and even followed them over time (Hicke et al. 2012b). In our three examples, the directional effects of warming-induced stressors may be clear (e.g., in California, species composition shifts to those associated with frequent fire). However, the magnitudes of these effects are not, nor are the potentially irreversible crossings of ecological thresholds. Given the complexity

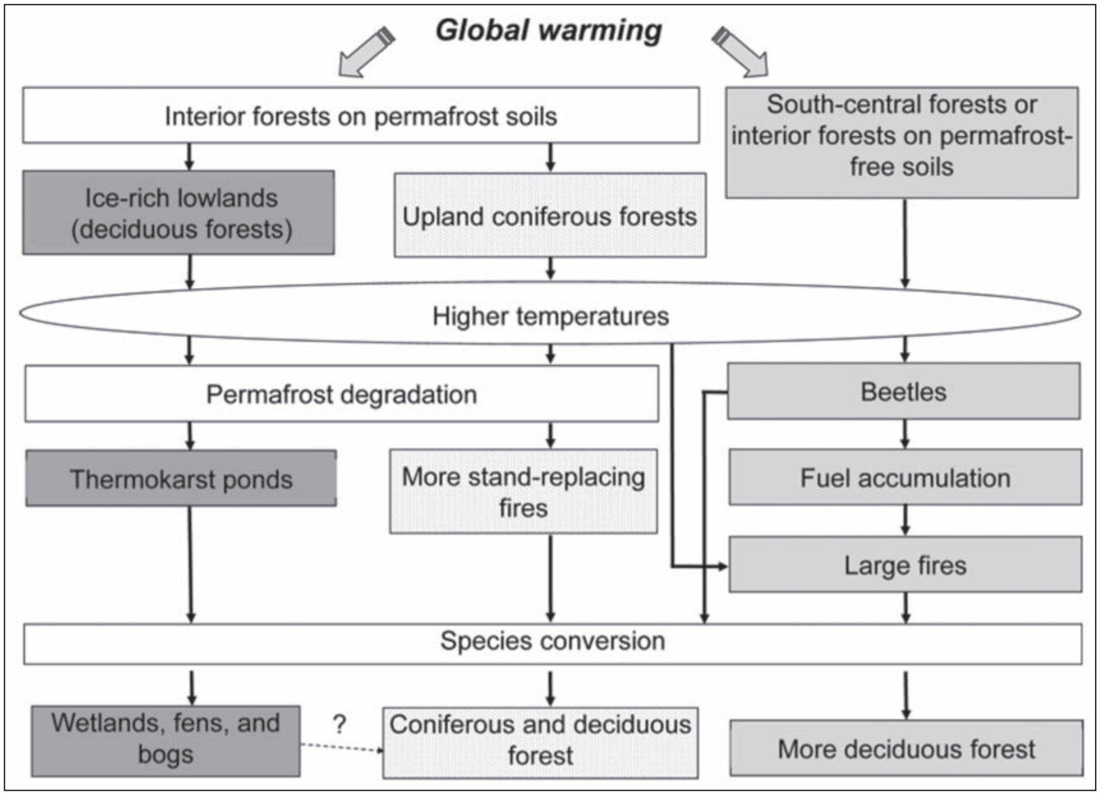


Figure 2.14—A conceptual model of stress complexes in the interior and coastal forests of Alaska. Rapid increases in the severity of disturbance regimes (insects and fire) are triggered by global warming. Stand-replacing fires, massive mortality from insects, and permafrost degradation contribute to species changes and conversion to deciduous life forms. Modified from McKenzie et al. (2009).

Courtesy of the Fire and Environmental Research Applications team, U.S. Forest Service, Digital Photo Series.

Figure 2.15—Interactions between wildfire and hurricanes are synergistic in the Southern United States. Figure depicts a longleaf pine/saw palmetto flatwoods on the Atlantic coastal plain, 2.5 years after a hurricane and with a previous history of prescribed fire.

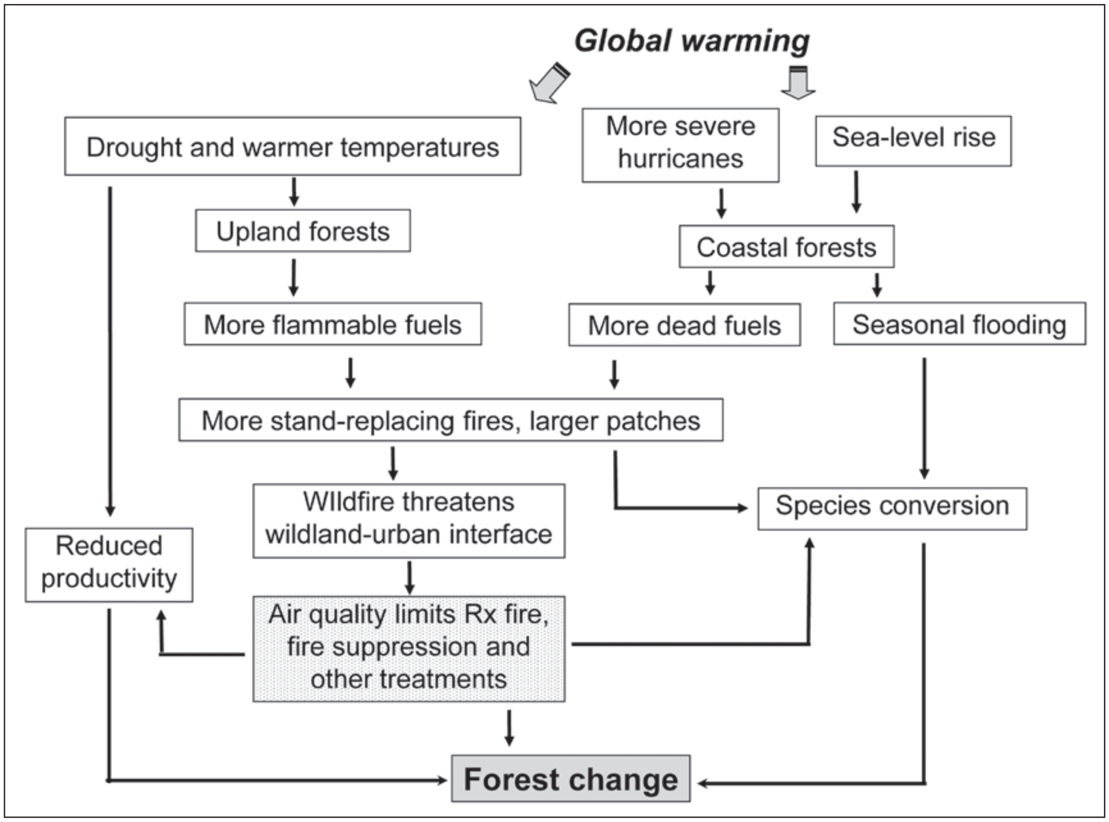


Figure 2.16—A conceptual model of stress complex in the interior and coastal forests of the Southeast. Increases in the severity of hurricanes are triggered by global warming, while sea level rises. Warmer and drier climate in uplands leads to longer periods with flammable fuels. Changes in fire and hydrologic regimes, and responses to them, lead to species change and altered C dynamics.

and diversity of potential interacting stressors in U.S. forests, a fruitful way to advance quantitative knowledge may be with explicit simulations with models of "intermediate complexity" to ascertain the sensitivity of ecosystems to the uncertainties associated with key parameters (e.g., the thickness of the arrows in figs. 2.12, 2.14, and 2.16).

Key Findings

- Interactions among ecological disturbance and stressors likely cause larger effects on ecosystems than any individual disturbance or stressor.
- Warmer temperature will generally exacerbate stress in drier forest ecosystems, partly through reduced vigor in vegetation but more importantly through increased disturbance.
- Climate-induced increases in wildfire occurrence and insect outbreaks across large landscapes will potentially cause rapid changes in the structure and function of forest ecosystems.

Key Information Needs

- Additional empirical data on stress interactions in a wide range of forest ecosystems.
- Transition from qualitative to quantitative analyses and models of how stressors and disturbances interact to affect forest ecosystems.

Effects of Climate Change on Forest Processes

Some of the changes to U.S. forests will be directly caused by the effects of an altered climate, such as increases in atmospheric CO_2 and N deposition on tree growth, increases in seed production, mortality, and regeneration, and shifting success among species as a result of altered outcomes of competition among species. Other changes will be indirectly caused by climate-induced changes in disturbances, such as droughts, fire, insect outbreaks, pathogens, and storms. Potential changes in the duration of snowpack (discussed below) will also affect disturbance and forest processes. In this section, we provide a synthesis of current knowledge of the potential direct and indirect effects of climate change on biogeochemical cycling (i.e., C, nutrients, and water) and forest tree distributions.

Carbon and Nutrient Cycling

The United States has 303 million ha of forest land, about 8 percent of the world's total. Forest C stocks and uptake or loss rates differ greatly with the wide range in environmental conditions, land use and land use history, and current human influences. Forests of the conterminous United States cover 281 million ha and contain 45,988 Tg of C. Estimates of the amount of the Nation's CO_2 emissions (1500 Tg of C in 2009) offset by forests and forest products vary with assumptions and accounting methods (e.g., from 10 to 20 percent) (McKinley et al. 2011), with 13 percent being the most recent and commonly used estimate for the United States (USEPA 2011). Ninety-four percent of forest C storage comes from growth on current forest lands, with the remaining 6 percent from a net positive conversion of other land uses to forests. Regional differences in forest C pools and storage rates are reported in McKinley et al. (2011), Woodbury et al. (2007), and U.S. Environmental Protection Agency (USEPA) (2011). Updates of the inventories used to estimate these pools and storage rates may be important to capture C losses in recent large fires, bark beetle outbreaks, and drought mortality. Also, certain components, such as dead wood and C in soil, are either sparsely measured or are only estimated (Woodbury et al. 2007).

These forest C storage estimates are similar to those reported in a global study of forest sinks derived from the same sources (Pan et al. 2011). An analysis using eddy covariance flux measurements, satellite observations, and modeling estimated a C storage in the conterminous United States of 630 $Tg{\cdot}C{\cdot}yr^{-1}$ (Xiao et al. 2011), largely from forests and savannas. However, most agricultural lands either store little additional carbon or lose C (USEPA 2011). The large discrepancy between the biometric USEPA estimates and those of Xiao et al. (2011) is probably caused by two factors: (1) woodland encroachment (McKinley and Blair 2008, Pacala et al. 2001, Van Auken 2000) that is not measured by the USDA Forest Service Forest Inventory and Analysis used for the USEPA reporting, and (2) the poor performance of eddy covariance measurements to estimate ecosystem respiration, consistently leading to ~30 percent overestimates of ecosystem C storage (Barford et al. 2001, Bolstad et al. 2004, Kutsch et al. 2008, Lavigne et al. 1997, Wang et al. 2010). Other estimates for the conterminous United States are 1200 ± 400 Tg·C $\cdot yr^{-1}$ from inversion analysis (Butler et al. 2010) and 500 ± 400 Tg·C $\cdot yr^{-1}$ from three-dimensional atmospheric CO_2 sampling (Crevoisier et al. 2010).

Response of Forest Carbon Cycling Processes to Increased Temperature, Changes in Precipitation, Increased Carbon Dioxide, Nitrogen Deposition, and Tropospheric Ozone

Carbon storage in forest ecosystems results from the balance between growth of wood, foliage, and roots and their death or shedding and subsequent decomposition. Temperature, atmospheric CO_2 concentration, ecosystem water balance, and N cycling all interact to alter photosynthesis and growth. The critical issue is the balance among these factors affecting growth. For example, higher temperatures can benefit growth, but the most benefit would come with adequate nutrition and a positive water balance. Disturbance is the largest factor changing the balance between production and decomposition, but chronic changes in temperature, precipitation, CO_2, and N deposition over large areas can also alter the U.S. forest C balance.

Insights for the U.S. forest carbon balance from experiments and measurements—

Atmospheric concentrations of CO_2, currently about 390 parts per million (ppm), are expected to rise to 700 to 900 ppm by 2100, depending on future anthropogenic emissions and any changes in atmospheric uptake by terrestrial and aquatic ecosystems. Experimental results continue to confirm that the primary direct effect of elevated CO_2 on forest vegetation is an increase in photosynthesis (Norby et al. 2005), but individual studies show that photosynthetic enhancement, growth and C storage are moderated by the presence of drought or nutrient limitations (Finzi et al. 2006, Garten et al. 2011, Johnson 2006, Norby et al. 2010). A recent synthesis of free-air CO_2 enrichment studies (Norby and Zak 2011) showed that (1) elevated CO_2 does not increase the leaf area of forested sites, (2) net primary production is enhanced under elevated CO_2 only when water and nutrient supplies are abundant, (3) water use is reduced through stomatal closure (Leuzinger and Körner 2007, Warren et al. 2011) (see "Forest Hydrological Processes" below), and (4) CO_2-promoted increases in photosynthesis and net primary productivity do not always increase forest C storage. Despite the known limitations on tree response to elevated CO_2, a 19 percent increase in CO_2 over the past five decades may have increased aspen growth more than 50 percent (Cole et al. 2010).

Elevated atmospheric CO_2 will likely increase forest productivity, but because of the known limitations and uncertainties to the response, we do not know how much. Major uncertainties in projecting forest response to elevated CO_2 include projecting the responses of belowground processes such as soil C storage (Lukac et al. 2009), mature trees, and wetlands. Elevated CO_2 commonly enhances soil CO_2 efflux, suggesting that some of the additional photosynthesis is rapidly cycled back to the atmosphere (Bernhardt et al. 2006). An increase in labile C in soil pools may increase decomposition of more recalcitrant soil C and potentially reduce soil C storage (Hofmockel et al. 2011). For a mature forest, sustained increases in photosynthesis in response to elevated CO_2 (Bader et al. 2009) did not increase wood growth (Körner et al. 2005), soil respiration (Bader and Körner 2010), or root or soil C storage (Asshoff et al. 2006, Bader et al. 2009). For wetlands, elevated CO_2 can increase CO_2 and methane efflux (Ellis et al. 2009, Hutchin et al. 1995), but these fluxes strongly interact with precipitation, the water table, and potential species changes (Fenner et al. 2007).

Models project annual temperature to increase by 4 to 5 °C by 2070, with high-latitude boreal forests experiencing the largest increases in temperature. For temperate and boreal forests, modest increases in temperature tend to increase growth (Way et al. 2010). Warming will probably enhance upland forest growth for ecosystems with ample water, through changes in annual plant development and a longer growth season (Bronson et al. 2009, Gunderson et al. 2012, Hänninen et al. 2007). Growth in water-limited ecosystems will probably be reduced (Arend et al. 2011, Hu et al. 2010), and net C storage may be reduced (Cai et al. 2010). Observed changes in growth for these studies were not caused by increases in photosynthesis (Bronson and Gower 2010, Gunderson et al. 2010). Warming will also enhance microbial decomposition and nutrient mineralization in soils (Melillo et al. 2002), increasing plant nutrient availability (Melillo et al. 2011) (discussed below), but the long-term tradeoff between soil C loss and nutrient enhanced productivity is unknown. A longer growing season may increase the possibility of damage to trees from late frost events (Augspurger 2009, Gu et al. 2008).

Projected precipitation for 2070 to 2100 differs by region; the Southwest and areas of the Great Plains, Texas, Arkansas, and southern Missouri will receive lower summer precipitation, and precipitation in the East will increase in all seasons except the summer. Eastern forests, particularly on deep soils, are well buffered against substantial reductions in precipitation; forest growth, soil C storage, and nutrient availability show little effect of a chronic 12-year, 33-percent reduction in precipitation (Froberg et al. 2008, Hanson et al. 2007, Johnson et al. 2008). Western forests, particularly those that rely on snowmelt for their water, will probably show lower growth under drier conditions (Boisvenue and Running 2010, Hu et al. 2010). More frequent droughts in the Western United States will reduce tree growth and vigor and increase tree mortality (McDowell et al. 2008, McDowell 2011). A modeling study suggested that the amount of precipitation was more important for forest productivity than its frequency and intensity (Gerten et al. 2008).

Nitrogen deposition may increase in some regions and decrease or remain the same in other regions, depending on emissions associated with human population trends and the effectiveness of regulations to reduce N emissions. In areas where N deposition increases, it may enhance ecosystem C storage by increasing forest productivity (Churkina et al. 2009, de Vries 2009) and decreasing decomposition of soil organic matter (Janssens et al. 2010), but those gains may be offset by the concurrent release of nitrous oxide, a potent greenhouse gas (Zaehle et al. 2011). The potential for enhancing C gain would be low in regions where N deposition is already high (e.g., the Northeast) and high in regions where N deposition is low (e.g., the Southwest). In addition, tree species have a wide range of susceptibility to tropospheric ozone, which also varies regionally, and damage caused by ozone is not completely offset by elevated CO_2 (Karnosky et al. 2005).

Projections of the U.S. forest carbon balance from models—

Experimental manipulations of temperature and precipitation are rare for forest ecosystems, and ecological process models are needed to project how changes in multiple factors over large areas might affect forest C balance. Forests in different regions will probably respond differently to climate change because of differences in species composition, water and nutrient availability, soil depth and texture, and strength of other environmental factors such as ozone and N deposition. Model projections vary by region, just as projected changes in climate vary by region. Different models also produce different results.

Overall, in the Eastern United States, productivity or forest C storage is expected to increase with projected changes in climate, N, and CO_2. This is because the increased precipitation projected for many areas in the Eastern United States allows more photosynthesis under increased temperature and CO_2. For example, upland oak forests in Tennessee are projected to increase their current C storage rate by 20 percent for the climate and atmosphere predicted for 2100 (CO_2 concentration of 770 ppm, ozone concentration 20 ppb higher than today’s level, 4 °C temperature increase, and 20 percent more November–March precipitation) (Hanson et al. 2005). Globally, temperate forest and grassland net productivity is projected to increase 25 to 28 percent for CO_2 concentration of 550 ppm (Pinsonneault et al. 2011), an estimate that includes expected changes in climate. Based on a four-model simulation of the effects of increased temperature and CO_2 and altered precipitation, wet sites such as Eastern forests showed large absolute changes in net C storage rates and net ecosystem production (Luo et al. 2008). Modeled forest productivity also increased for New England forests (Campbell et al. 2009).

For the Western United States, many models project lower productivity or C storage for forests. Changes in climate and CO_2 are projected to turn Rocky Mountain forests into a C source by 2090 (Biome-BGC model) (Boisvenue and Running 2010), and decrease forest C storage for boreal aspen (Grant et al. 2006). However, other model studies project increases in C storage for Western U.S. forests (CENTURY Model) (Melillo et al. 2011, Pinsonneault et al. 2011, Smithwick et al. 2009). Carbon in northern bogs, peat lands, and permafrost regions may be lost with a warming climate (increasing methane production), depending on hydrology and other factors (Heijmans et al. 2008, Ise et al. 2008, Koven et al. 2011). Both global model simulations of climate change and ecosystem productivity (Friend 2010, Pinsonneault et al. 2011) projected higher C storage for both Eastern and Western U.S. forests, with the larger increase in the East. It is important to note that none of these local or global simulations include the effects of altered disturbance regimes in their projections.

Effects of changes in disturbance rates on the U.S. forest carbon balance—

For Western U.S. forests, climate-driven increased fire and bark beetle outbreaks are likely to substantially reduce forest C storage and storage rate (Metsaranta et al. 2010; Westerling et al. 2006, 2011), jeopardizing the current U.S. forest sink. Disturbances, mostly large-scale fires, have already turned Arizona and Idaho forests from a C sink into a C source (USEPA 2011). Tree mortality has increased globally, and large-tree mortality from drought and elevated temperature has promoted bark beetle outbreaks, with the consequence of a short-term C loss for Western U.S. forests (Allen et al. 2010). Tree mortality not caused by fire or insect outbreaks has also increased in the West (van Mantgem et al. 2009). We have no information on tree mortality trends in the Eastern United States, but tree mortality rates there are sensitive to air pollution exposure (Dietze and Moorcroft 2011). Tree regeneration after disturbance is critical for maintaining forest cover and associated C stocks (McKinley et al. 2011). Tree regeneration is uncertain for western montane forests, where fire intensity exceeds historical patterns (Bonnet et al. 2005, Haire and McGarigal 2010). Temperature and precipitation extremes are important for initiating disturbances, but the mean projections of the many GCMs used (and the individual models in general) do a poor job of predicting extreme events.

Effects on Eastern forests where precipitation is currently in excess—

In the next 30 years, projected changes in CO_2, temperature, and precipitation are not likely to change forest C storage and uptake from current levels or may even increase them, if tropospheric ozone levels are managed to remain at or below current levels (fig. 2.17). Changes in species composition through time will probably remain driven by competition between plants and interactions with pests and pathogens, except for sites with shallow or coarse textured soils that increase the effects of drought. Toward the end of this century, net C gain by Eastern U.S. forests will probably be reduced by a warming-induced increase in seasonal water deficits, but the effects will not be large. The beneficial

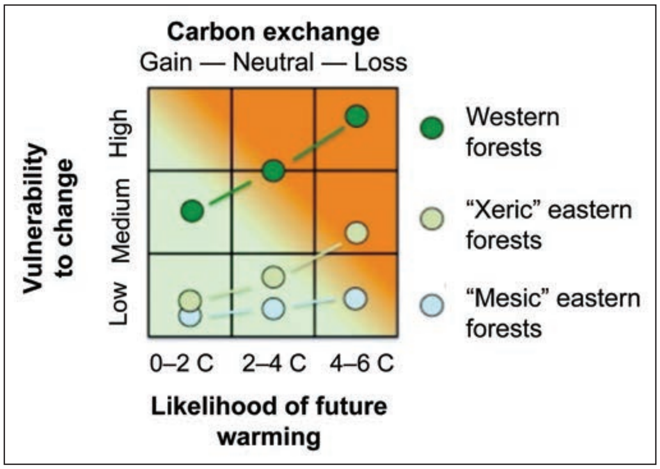


Figure 2.17—Risk analysis diagram for forest carbon cycle. Western forests are considered inherently limited by water demands that exceed precipitation supplies during substantial portions of the year. Xeric eastern forests include those growing on shallow or coarse-textured soils or those present at the western prairie forest transition zone that experience water deficits in some years. Mesic eastern forests experience severe water deficits only in occasional years and for relatively brief periods.

effects of elevated CO_2 and the extended growing season length will allow ample opportunity for C gain, even though the probability of water stress occurrence in the summer months will increase. On coarse-textured or shallow soils, the forest will show reduced annual C uptake (fig. 2.17).

Effects on forests in the Western, Southwestern, Lower Central, and Southeastern United States and Alaska—Changes in temperature and precipitation have already increased forest fire. Bark beetle outbreaks have increased forest C loss and are likely to continue to do so in the next 30 years, probably negating the increased productivity from warming temperatures and elevated CO_2 (fig. 2.17). After 2050, projected temperature and precipitation changes indicate that the survival of current tree species is uncertain, and that tree species change will be accompanied by disturbance and major C loss. An increased focus on forest regeneration after disturbance, perhaps with species adapted to the new climate, will probably be necessary to maintain forest cover and C stocks.

Effects on Nutrient Cycling

As noted, C cycling responses to elevated CO_2 and warming will be tightly linked to nutrient availability, especially N. Climate change is likely to have both direct and indirect effects on processes that regulate availability and fluxes. Biological processes that convert nutrients held in organic matter to available mineral forms are generally temperature-dependent. Experimental soil-warming studies confirm that N mineralization will increase in response to higher temperatures (Melillo et al. 2011), with an average increase in net N mineralization of about 50 percent (Rustad et al. 2001). These effects may be transient, however, because the supply of mineralizable substrates may not keep pace with opportunities for mineralization. Although experimental soil-warming studies have been critical to show potential effects of warming, results from these studies are limited by methodological constraints that make it difficult to scale results to the ecosystem level or incorporate whole-system interactions. Modeling approaches that scale to the ecosystem and incorporate interactions have generally confirmed patterns observed in soil-warming experiments (Campbell et al. 2009). Recent studies have used observed climate variability and corresponding measures of stream N in forested watersheds to infer changes in N-cycling processes. For example, in the Western United States, Baron et al. (2009) suggested that recent warming temperatures have melted glacial ice, subsequently flushing N from microbially active sediments. In the Eastern United States, Brookshire et al. (2011) found that seasonal variation in stream nitrate was tightly coupled to recent warming, and they used modeling to extrapolate effects of future warming on microbial activity and stream nitrate export. Brookshire et al. (2011) suggested that the consequences of elevated temperature will increase future N export threefold more than will projected changes in N deposition.

Effects of biotic disturbance on nutrient cycling in forests may also occur when species composition is changed because tree species affect belowground processes (Knoepp et al. 2011, Lovett et al. 2006). For example, forests afflicted with beech bark disease have increased litter decomposition, decreased soil C:N ratio, and increased extractable nitrate in the soil and nitrate in soil solution (Lovett et al. 2010). In eastern hemlock stands infested with hemlock woolly adelgid, litter N is increased, and N mineralization is accelerated even before tree mortality is observed (Orwig et al. 2008, Stadler et al. 2006). Defoliation also alters N pools and fluxes within forests (Lovett et al. 2002).

Key Findings

- Forest growth and afforestation offset 13 percent of fossil fuel CO_2 production in 2009 according to a recent analysis by the USEPA.
- In the Western United States, increased fire, bark beetle outbreaks, and droughts have likely reduced forest C storage, and these reductions will likely be larger in the future, slowing or halting the current C sink in the United States.
- In the Eastern United States, elevated CO_2 and temperature and sufficient water will increase forest growth and will likely increase C storage, except on sites with shallow soils or areas more subject to drought.
- Warmer temperatures will probably lead to increased nutrient cycling, promoting increased forest growth and elevated N levels in streams and rivers.

Key Information Needs

- More and longer term elevated CO_2 experiments in forests, especially in mature forests.
- More forest-scale warming experiments.
- More information on multifactor interactions and species changes, processes leading to tree mortality and species migration, and the cause and potential saturation of the current C sink in the United States.
- Analyses of long-term stream chemistry data to provide an integrated measure of nutrient cycling responses to climate variability, including more analyses across a wider range of ecosystems to understand variation in controls and response patterns.

Forest Hydrological Processes

Abundant and clean water are fundamental to the viability of aquatic ecosystems, human welfare, and economic growth and development throughout the world (Cech 2005, Jackson et al. 2001). The combination of increased demand for fresh water, changes in land use and cover, and climate change will place even greater demands on forest watersheds across the globe to meet the water resource needs of humans and aquatic ecosystems (Vörösmarty et al. 2010). Climate change will have both indirect and direct effects on forest water cycling. Indirect effects are associated with changes in atmospheric CO_2, increased temperature, altered soil water availability, climate-mediated changes in species composition, and changes in disturbance regimes or management and policy decisions that alter forest structure and composition. Indirect effects of climate change on forest water cycling work primarily through effects on forest evapotranspiration (ET), the combination of evaporation of water from plant and ground surfaces and transpiration. As discussed in the "Erosions and Landslides" section, direct effects are associated with more rainfall and more intense storms. These in turn increase base flows in streams (particularly intermittent streams), increase flood risk, accelerate erosion, and increase the potential for both landslides and increased interstorm periods and drought, along with climate-related changes in infiltration rate owing to extreme wildfire. Indirect and direct effects are interdependent.

Forest Evapotranspiration and Streamflow

Forest ET may be changing in response to changing climate (Gedney et al. 2006, Labat et al. 2004, Walter et al. 2004), but studies disagree about the direction of the change. Over relatively large areas and long temporal scales, streamflow is the balance between rainfall input and ET. Hence, the rainfall not used in ET is available for streamflow and groundwater recharge, and in many forest ecosystems, ET is a major regulator of streamflow and groundwater recharge. Walter et al. (2004) concluded that ET has been increasing across most of the United States at a rate of 10.4 mm per decade (inferred from U.S. Geological Survey records of precipitation and river discharge in six major basins in the United States). In contrast, river discharge across the globe has been increasing at a rate of 4 percent for each 1 °C increase in global temperature (Labat et al. 2004), suggesting a reduction in ET. Different response patterns are not surprising, because ET is affected by several co-occurring variables. For example, the increase in discharge in the study by Labat et al. (2004) has been attributed to the physiological effect of CO_2 decreasing ET, and not to the effect of changing land use (Gedney et al. 2006). Besides elevated atmospheric CO_2 concentration, some of the most important variables affecting forest ET are temperature, humidity, water availability, and species distributions. Potential effects of climate change on these variables and their interactions are discussed below, and they will probably result in differing patterns of change in ET at local and regional scales.

Elevated Atmospheric Carbon Dioxide

Over long time scales, higher CO_2 concentrations decrease stomatal density and aperture, both of which reduce transpiration (Beerling 1996, Ehleringer and Cerling 1995, Franks and Beerling 2009, Prentice and Harrison 2009). Indeed, both observational and experimental studies confirm long-term and large-scale changes in leaf stomatal conductance in response to elevated CO_2 (Lammertsma et al. 2011, Warren et al. 2011). As leaf stomatal conductance declines, ecosystem ET can also decline; however, any decline will depend on factors such as stand age, species composition, and leaf area. Empirical studies linking reduced stomatal conductance to reduced stand-level ET have not yet been possible, and most researchers have used modeling to make that linkage.

Warren et al. (2011) applied the Forest BGC model to data from several elevated CO_2 studies and projected that ET was reduced by 11 percent in older stands that did not experience an increase in leaf area. In younger stands, ET increased because of stimulation of leaf area. In a modeling study of deciduous forests in the northeastern United States, the estimated effect of elevated CO_2 on ET was modest, ranging from a 4-percent decrease to an 11-percent increase (Ollinger et al. 2008). In Mediterranean forest systems, changes in ET are also expected to be modest with increased temperature and CO_2, ranging from no change to a 10-percent decrease (Tague et al. 2009). Studies have not yet identified an increase in stand leaf area with elevated CO_2 (Norby and Zak 2011). Although the effects of elevated CO_2 on ET remain uncertain, studies agree that the direct effects will be modest (±10 percent) compared to the changes expected for other variables that affect ET, such as precipitation variability (Leuzinger and Korner 2010).

Warmer Temperatures and Drought

Although elevated CO_2 is likely to decrease ET, the increases in temperature and thus the increases in the vapor pressure deficit (VPD) between the inside of the leaf and the surrounding air may offset this effect, such that ET is affected little or not at all. As the air becomes drier, transpiration typically increases following an exponential saturation curve, with the rate of increase continually slowed by reduced stomatal opening. Most studies show that a physiological effect of reduced stomatal conductance in response to elevated CO_2 is observed only when the canopy air is very humid (low VPD). In a study of six deciduous tree species, a 22-percent reduction in transpiration occurred under elevated CO_2, but only at low VPD (Cech et al. 2003). These results support the idea that the physiological effects of elevated CO_2 on ecosystem water balance may depend on precipitation and atmospheric humidity.

Warming has changed the timing of foliage green-up and senescence, but the effects of these phenological changes on ET are complex and not well understood. Warming-induced lengthening of the growth season could increase ET and offset the reduction in stomatal conductance from elevated CO_2, but these effects are difficult to generalize across species and regions (Hänninen and Tanino 2011). Although the frost-free season across the United States has lengthened by about 2 weeks, resulting in a longer, warmer growing season, growth cessation in the autumn might come earlier with increasing temperatures for some boreal and temperate tree species (Kunkel et al. 2004). For other tree species, spring bud burst might be delayed by warmer temperatures (Zhang et al. 2007), perhaps because of not receiving the requisite chilling hours (Schwartz and Hanes 2010). In higher latitudes where chilling requirements are still being met, green-up is occurring sooner. Thus, springtime ET in the lower latitudes could be delayed while ET in the higher latitudes could be advanced.

The potential increase in ET owing to a lengthened growing season can be constrained by the water availability and drought that often arise late in the growing season (Zhao and Running 2010). Water limitations are a direct control on ET (lower water availability reduces transpiration), and many regions of the United States have experienced more frequent precipitation extremes, including droughts, over the last 50 years (Easterling et al. 2000b, Groisman et al. 2004, Huntington 2006, Solomon et al. 2007).

Changing Species Composition

Evapotranspiration is affected by the plant and tree species that comprise the canopy cover of a forest ecosystem. In general, pine forests are much more responsive to climatic variation than are deciduous forests (Ford et al. 2011a, Stoy et al. 2006). However, even within the same forest, growing season transpiration rates among canopy species (adjusted for differences in tree size) can vary by as much as fourfold, and co-occurring species can differ considerably in their responsiveness to climatic variation (Ford et al. 2011a). Characteristics of the xylem and sapwood, which vary by species, are among the most important determinants of stand transpiration in both observational (Vose and Ford 2011, Wullschleger et al. 2001) and theoretical studies (Enquist et al. 1998, Meinzer et al. 2005). Therefore, shifts in hydroclimate may be accommodated by changes in canopy leaf area, phenology, or species-based hydraulic efficiency.

Increased drought severity and frequency may contribute to rapid changes in forest species composition through two important processes. First, drought plays an important role in tree mortality (Allen et al. 2010); as soil water availability declines, forest trees either reduce stomatal conductance to reduce water loss (drought avoidance), or they experience progressive hydraulic failure (Anderegg et al. 2011). Second, some native insect outbreaks, and the mortality they cause, are also triggered by drought. Increasing temperatures are also expected to interact with drought. As temperatures increase, plant metabolism increases exponentially. If high temperature coincides with drought stress in forests, C starvation and mortality can occur more quickly than if these factors did not coincide (Adams et al. 2009). For example, Adams et al. (2009) projected a fivefold increase in the extent of pinyon pine mortality from an increase of 4.3 °C in temperature, based on historical drought frequency. If drought frequency increases as is expected under climate change scenarios (described in “Scenarios for Projecting Future Climate” section above), the projected mortality could be even greater.

Evapotranspiration will also change with changes in canopy density, canopy composition, water demand, and resulting energy partitioning in new communities, which will occur in response to species changes that accompany climate change, especially if large areas of forests experience mortality (Breshears et al. 2005). Insect and pathogen outbreaks and fire will be the likely primary forces behind large-scale and rapid changes in forest composition and structure; however, direct studies of these effects on hydrology are limited (Tchakerian and Couslon 2011). Potential biogeophysical effects from tree-killing biotic disturbances include (1) increased surface albedo, which will reduce the absorption of solar radiation; (2) decreased transpiration until the new forest is reestablished; and (3) decreased surface roughness, which affects atmospheric drag (Bonan 2008). Despite their importance as potential feedbacks to the atmospheric system (Adams et al. 2010, Rotenberg and Yakir 2010), little is known about how these processes have been altered by insect and pathogen outbreaks. After outbreaks that cause widespread tree mortality, streamflow increases, the annual hydrograph advances, and low flows increase (Potts 1984); at the same time, snow accumulation increases and snowmelt is more rapid after needle drop (Boon 2012, Pugh and Small 2011). According to one evaluation of radiative forcing effects from mountain pine beetle infestations, the cooling associated with increased albedo exceeded the warming associated with increased atmospheric CO_2, leading to a net cooling in the first 14 years after attack (O’Halloran et al. 2012). Increased surface albedo was especially pronounced in winter, when needle loss following tree mortality exposed more of the reflective snow surface. These studies show strong, but mostly indirect, evidence that large-scale forest mortality will alter water cycling processes; however, the magnitude and duration of responses will differ among species and across regions.

Snowmelt

Because of climate warming, snow cover in North America has shown a general reduction in duration, extent, and depth over the last few decades, with increased interannual variability (Barnett et al. 2008, Luce and Holden 2009, Mote et al. 2005, Pagano and Garen 2005, Regonda et al. 2005). A reduction in snowpack depth, persistence, and duration has significant effects on forest ecosystems, including water stress, disturbance, erosion, and biogeochemical cycling. In arid and semiarid systems, early and reduced snowmelt has led to increased water stress in the late growing season and increases in fire frequency and magnitude and the susceptibility of forest stands to infestation (Adams et al. 2011, Breshears et al. 2005, Holden et al. 2011b, Westerling et al. 2011). The rapid flush of water to the soil in spring snowmelt can release mobile solutes that have been slowly accumulating as a result of subnival biogeochemical cycling (e.g., Williams et al. 2009). These spring pulses can provide the major input of nutrients to aquatic ecosystems. Reductions in the spring flush, and increased rain in winter and early spring, can change the timing of N release from northern forests so that they resemble more southern ecosystems that lack the distinct seasonality of stream water N concentrations and export. Greater frequency and magnitude of rain-on-snow events may also increase soil erosion, sedimentation, and landslides.

Soil Infiltration, Ground water Recharge, and Lateral Redistribution

Forest ecosystems typically support high infiltration capacities because of large soil pores developed by root systems and soil fauna, so surface runoff is not common compared to other land cover. However, high-intensity precipitation or snowmelt events can rapidly move water in the soil to the unsaturated zone or ground water, or into the local stream, particularly in steep terrain (Brooks et al. 2011, Laio et al. 2001, Troch et al. 2009).[7] Increases in storm intensity projected for the future may increase peak streamflow and flooding through this process.

Carbon and Water Tradeoffs

Expanding C sequestration or wood-based bioenergy markets to offset fossil fuel emissions may affect water resources (Jackson et al. 2005), and these effects will depend on both the specific management activity and the scale of implementation. Planting fast-growing species for bioenergy production (or C sequestration) may reduce water resources (Jackson et al. 2005), but these reductions may be localized and minor if the planting area is small relative to the watershed size. In favoring certain species, the choice of species and the regional climate may influence overall effects. In wetter regions, where interception represents a higher proportion of ET, evergreen species may have a greater effect on site water balance. In drier regions, where transpiration represents the greatest proportion of ET, high water-use species such as *Populus* or *Eucalyptus* may have greater effects (Farley et al. 2005). Shortening rotation length might increase streamflow because the proportion of time that the stand is at canopy closure (when leaf area index is maximum and streamflow is lowest) will be reduced. In an analysis of forest plantations across the globe, Jackson et al. (2005) found the largest reductions in streamflow in 15- to 20-year-old plantations. If these plantations become widespread, irrigation of short-rotation forests would increase water use. A primary concern in the Western United States would be survival of plantations under drought.

Key Findings

- Effects of elevated CO_2 on transpiration will likely be modest (±10 percent), compared to the effects of precipitation variability on transpiration.
- Large-scale disturbances such as fire, bark beetle outbreaks, and defoliating insects will likely increase runoff.
- Increased temperatures have recently decreased snow cover depth, duration and extent and have advanced the timing of runoff. These effects will likely intensify as temperatures warm further. Fast-growth forests, if widely applied, may reduce streamflow.

Key Information Needs

- More information on interactions among hydrology, climate change, disturbance, and changing species composition and phenology.
- Better projections of future effects on hydrologic processes and water resources, which will require improved hydrologic models that can account for variation in species and stand structures, yet can be readily scaled to larger and more complex landscapes with mixed land uses.
- Planting fast-growing trees is an option to offset C emissions and increase sequestration, but studies are needed to rigorously evaluate the potential effects on water resources.

Forest Tree Species Distributions

The ranges of plant species have always shifted through time (Davis and Shaw 2001, Davis and Zabinski 1992, Webb

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[7] Hwang, T.; Band, L.E.; Vose, J.M. [N.d.]. Ecosystem processes at the watershed scale: hydrologic vegetation gradient as an indicator for lateral hydrologic connectivity of headwater catchments. On file with: Taehee Hwang, University of North Carolina, Institute for the Environment, Campus Box 1105, Chapel Hill, NC 27599-1105.

1992), but in recent decades, evidence is building that species are moving faster than in historical times (Chen et al. 2010, Dobrowski et al. 2011, Parmesan and Yohe 2003). For example, in a meta-analysis of 764 species range changes (mostly insects and no tree species), the average rate of northward migration was 16.9 km per decade (Chen et al. 2011). In contrast, an earlier meta-analysis, using 99 species of birds, butterflies, and alpine herbs, reported a northward migration of 6.1 km per decade (Parmesan and Yohe 2003). There is evidence of upward elevational migration of tree species (Beckage et al. 2008, Holzinger et al. 2008, Lenoir et al. 2008). However, to our knowledge, no study has documented northward latitudinal migration for trees in response to recent changes. Woodall et al. (2009) used forest inventory data to investigate surrogates for migration among 40 Eastern United States tree species. They used a comparison of the mean latitude of biomass of larger trees (>2.5 cm diameter at breast height [dbh]) relative to the mean latitude of density of seedlings (<2.5 cm dbh) across each species' range of latitude to detect possible future trends in distribution. For many of the species, this analysis indicated higher regeneration success at the northern edge of their ranges. Compared to mean latitude of tree biomass, mean latitude of seedlings was significantly farther north (>20 km) for the northern study species, southern species showed no shift, and general species showed southern expansion. Density of seedlings relative to tree biomass of northern tree species was nearly 10 times higher in northern latitudes than in southern latitudes. These results suggest that the process of northward tree migration in the Eastern United States is currently underway, with rates approaching 100 km per century for many species.

Pollen records suggest migration rates for some tree species of 2 to 2.5 km per decade during the period of roughly 5,000 to 6,000 years ago (Davis 1989), a time when species were not slowed by forest fragmentation (Iverson et al. 2004a, 2004b; Schwartz 1993). If the same rates applied today, these slow tree migration rates would make measuring species migration to current climate change logistically difficult. Instead, researchers must rely on modeling to project the potential effects of climate change on tree migration processes.

Two types of predictive models of vegetation change exist: (1) empirical, species distribution models that establish statistical relationships between species or life forms and (often numerous) predictor variables, and (2) process-based models, which simulate vegetation dynamics at the taxonomic resolution of species or life forms. There are well-recognized tradeoffs between using these different models to assess potential changes in species habitats resulting from forecasts of environmental change (Thuiller et al. 2008), and both approaches are widely used. When both approaches yield similar results for a particular area, confidence in model projections is improved. Other modeling approaches are used to inform these two approaches. Demography studies inform species distribution models (SDMs), and migration models are used with process-based models.

Species Distribution Models

Species distribution models are used to extrapolate species distributions in space and time, based on statistical models of habitat suitability (Franklin 2009). Species distribution models are built with observations of species occurrences along with environmental variables thought to influence habitat suitability and equilibrium species distribution. Predictive mapping of suitable habitat (but not whether a species will reach those habitats) in space and time are therefore possible by extension of these models. The SDMs have limitations, which include the assumptions that (1) the selected variables reflect the niche requirements of a species, (2) species are in equilibrium with their suitable habitat, (3) species will be able to disperse to their suitable locations, (4) projections can be made for novel climates and land covers, (5) the effects of adaptation and evolution are minimal, and (6) the effects of biotic interactions (including human interactions) are minimal (Ibanez et al. 2006, Pearson et al. 2006). However, SDMs can provide glimpses of probable futures useful for incorporating future conditions into conservation and management practices and decisions.

Species distribution models project a northward movement of tree species habitat in North America from 400 to 800 km by 2100 depending on the assumptions used in projecting future climate (Iverson et al. 2008, McKenney et al. 2011). Species distribution model projections also differ based on future scenarios and with time. For example, under a scenario of high greenhouse gas emissions (Hadley A1F1), about 66 species would gain and 54 species would lose at least 10 percent of their suitable habitat under climate change. A lower emission pathway would result in both fewer losers and gainers. Sugar maple (*Acer saccharum* Marsh.) would lose a large proportion of its habitat under the warmest scenario (fig. 2.18) (Iverson et al. 2008, Lovett and Mitchell 2004), but it would still maintain a presence of habitat in most areas. When multiple species were compiled together to create forest types, models project a severe loss of suitable habitat for spruce-fir (*Picea-Abies*), white-red-jack pine (*Pinus strobus* L., *P. resinosa* Aiton, *P. banksiana* Lamb.), and aspen-birch (*Populus-Betula*) suitable habitat, but a wide expansion of suitable habitat for oak-hickory (*Quercus-Carya*) (fig. 2.19) (Iverson and Prasad 2001, Iverson et al. 2008).

Process Models

To model species composition changes, a fully process-driven approach might be preferable to isolate mechanisms and to create “what-if scenarios.” However, such an approach is presently impossible because of (1) the necessity of detailed parameterization of species life histories and physiologies for more than 100 species, (2) the complexity of many interacting disturbance factors, and (3) the necessary high-resolution modeling over very large areas (Lawler et al. 2006). Dynamic global vegetation models (DGVM) operate at scales from regional (hundreds of kilometers) to global; these models can aggregate species into life forms or plant functional types (PFTs), using structural or functional attributes such as needleleaf vs. broadleaf and evergreen vs. deciduous (Bachelet et al. 2003, Bonan et al. 2003, Neilson et al. 2005). Most of these models project shifts to more drought-tolerant and disturbance-tolerant species or PFTs for future climates. This general shift in vegetation may be offset by physiological changes induced by CO_2 fertilization, as suggested by a DGVM (MC-1) that links water-use efficiency to CO_2-simulated expansion of forests into areas whose climate is currently too dry (Bachelet et al. 2003). This particular issue deserves further study to resolve the extent and duration of such mitigating effects of CO_2; these effects could change substantially depending on the outcome of climate-change projections. Ravenscroft et al. (2010) used the LANDIS model to simulate the potential effects of climate change to 2095 and found that mesic birch–aspen–spruce–fir and jack pine–black spruce (*Picea mariana* [Mill.] Britton, Sterns & Poggenb.) forest types would be substantially altered because of the loss of northerly species and the expansion of red (*Acer rubrum* L.) and sugar maple (fig. 2.20). Another promising modeling system that also includes climate variables is the Regional Hydro-Ecologic Simulation System (RHESSys) (Tague and Band 2004). Using this model in a Sierra Nevada mountain system, Christiansen et al. (2008) found significant elevational differences in vegetation water use and sensitivity to climate, both of which will probably be critical to controlling responses and vulnerability of similar ecosystems under climate change. Transpiration at the lowest elevations was consistent across years because of topographically controlled high moistures, the mid-elevation transpiration rates were controlled primarily by precipitation, and the high-elevation transpiration rates were controlled primarily by temperature (fig. 2.21).

Demography Studies

Demography studies track individuals over time, rather than use periodic plot-level inventories, to fully understand the role of climate relative to other risk factors like competition, variation in physiology and function, and vulnerability to insects and pathogens. Such demography data sets are rare, but one study has tracked more than 27,000 individuals of 40 species over 6 to 11 years to address these interactions over a portion of the southeastern United States (Clark et al. 2011). This study found that the primary climatic controls are spring temperature (regulating species fecundity) and growing season moisture, particularly for species of *Pinus*, *Ulmus*, *Magnolia*, and *Fagus*. Pitch pine (*Pinus rigida* Mill.) tracked both spring temperature and summer drought, yellow poplar (*Liriodendron tulipifera* L.) tracked neither, and sweetgum (*Liquidambar styraciflua* L.) tracked summer drought but not spring temperature (Clark et al. 2011). Overall, the effect of competition on growth and mortality exceeded the effects of climate variation for most species.

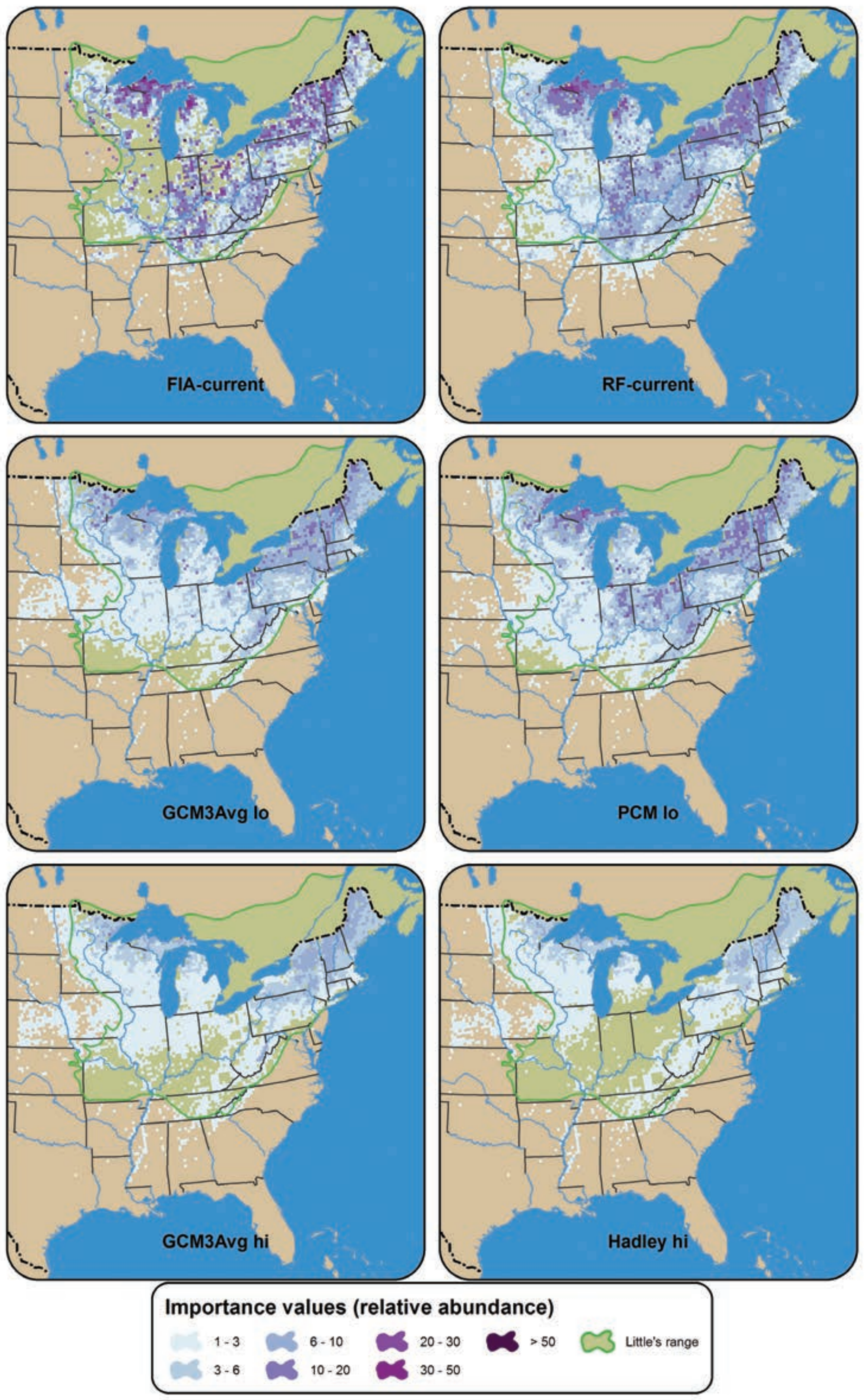

Figure 2.18—Maps of current and potential future suitable habitat for sugar maple show the expected northward migration of habitat with climate warming by 2100. The map includes the current inventory estimate of abundance from Forest Inventory and Analysis (FIA-current) sampling, the modeled current distribution (RF-current), and four model projections for future climate: (1) a low emissions scenario (B1) using the average of three global climate models (GCM3 Avg lo); (2) a low emissions scenario (B1) using the National Center for Atmospheric Research's Parallel Climate Model (PCM lo); (3) a high emissions scenario (A1F1) using the average of three global climate models (GCM3 Avg hi); (4) a high emissions scenario (A1F1) using the HadleyCM3 model (Hadley hi). Data from Prasad and Iverson (1999–ongoing).

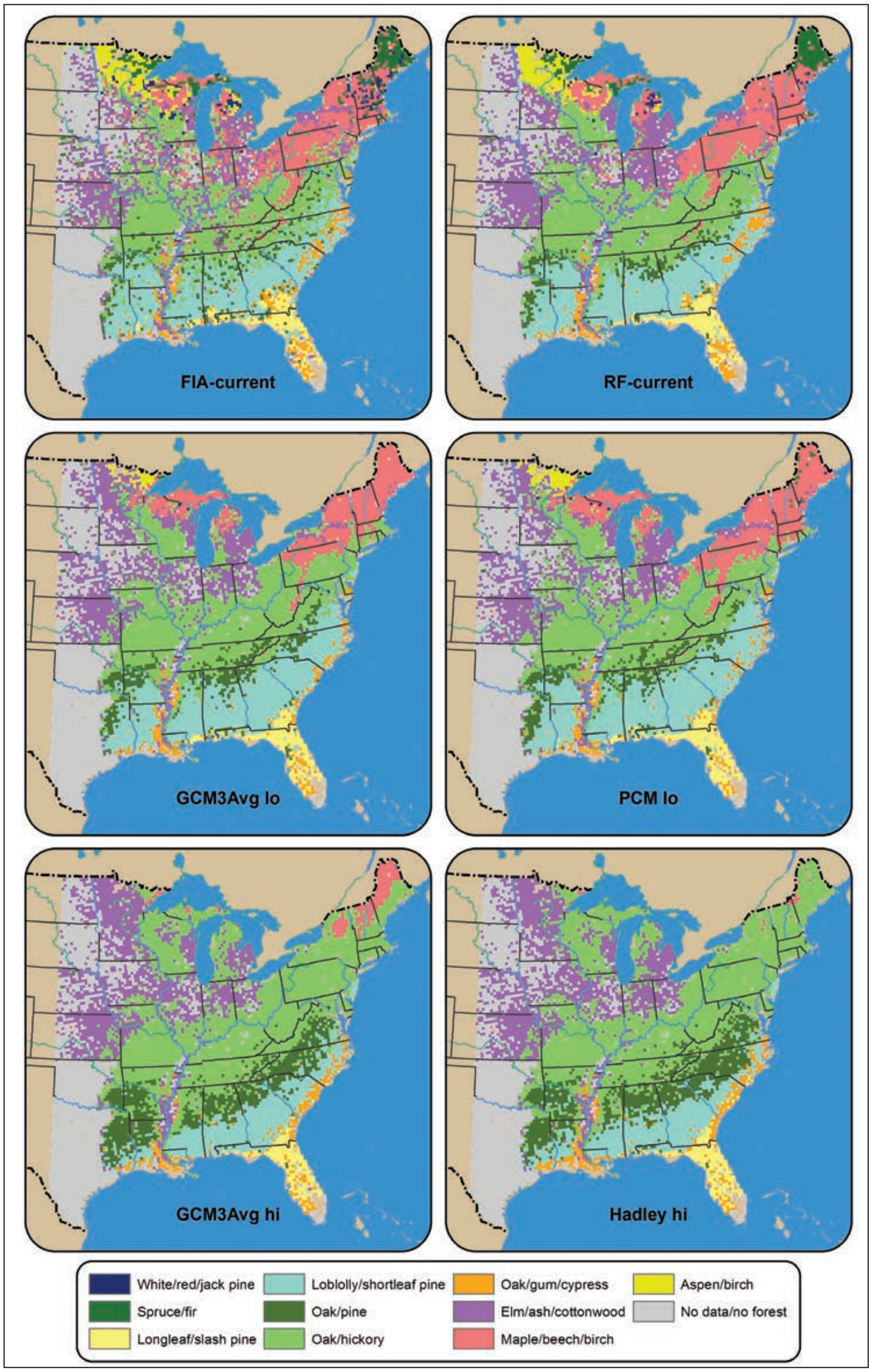

Figure 2.19—Maps of current and potential future suitable habitat for USDA Forest Service forest types (named according to dominant species) also show the major forests types moving northward with climate warming by 2100. The map includes the current inventory estimate of abundance from Forest Inventory and Analysis (FIA-current) sampling, the modeled current distribution (RF current) and four model projections for future climate: (1) a low emission scenario (B1) using the National Center for Atmospheric Research's Parallel Climate Model (PCM lo); (2) a low emission scenario (B1) usingthe average of three global climate models (GCM3 Avg lo); (3) a high emission scenario (A1F1) using the average of three global climate models (GCM3 Avg hi); (4) a high emission scenario (A1F1) using the HadleyCM3 model (Hadley hi). From Iverson et al. (2008).

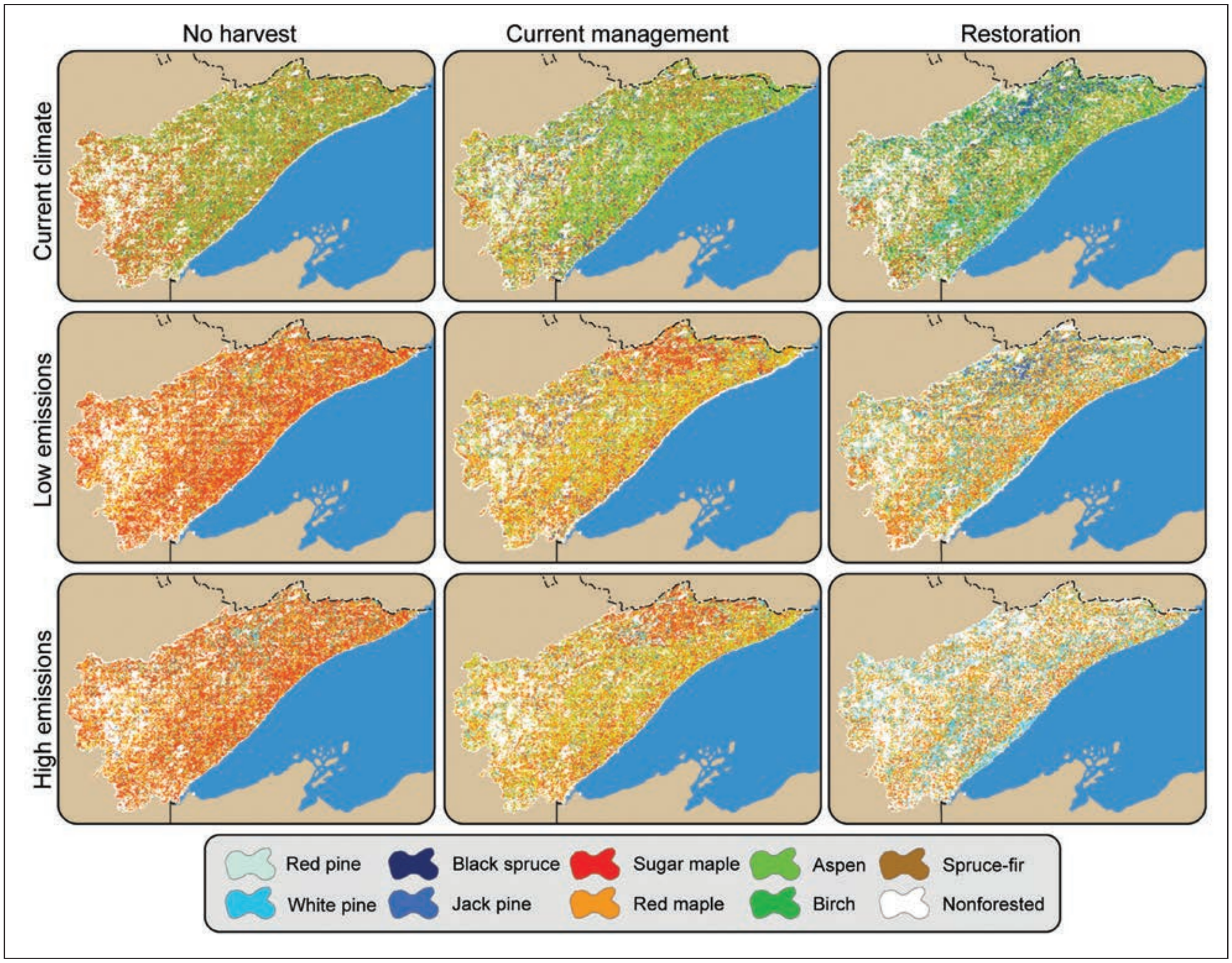

Figure 2.20—Maps of northern Minnesota forest types at year 2095 for nine scenarios (three emission levels × three management intensities). Sugar and red maple dominate the landscape by 2095 regardless of the emission scenario or management. Cells are classified to the forest type (named according to dominant species) with the highest total biomass, as generated via the LANDIS vegetation model. Climate simulations are done with the Hadley C3 global climate model with the A2 (high) and B1 (low) emission scenarios. From Ravenscroft et al. (2010).

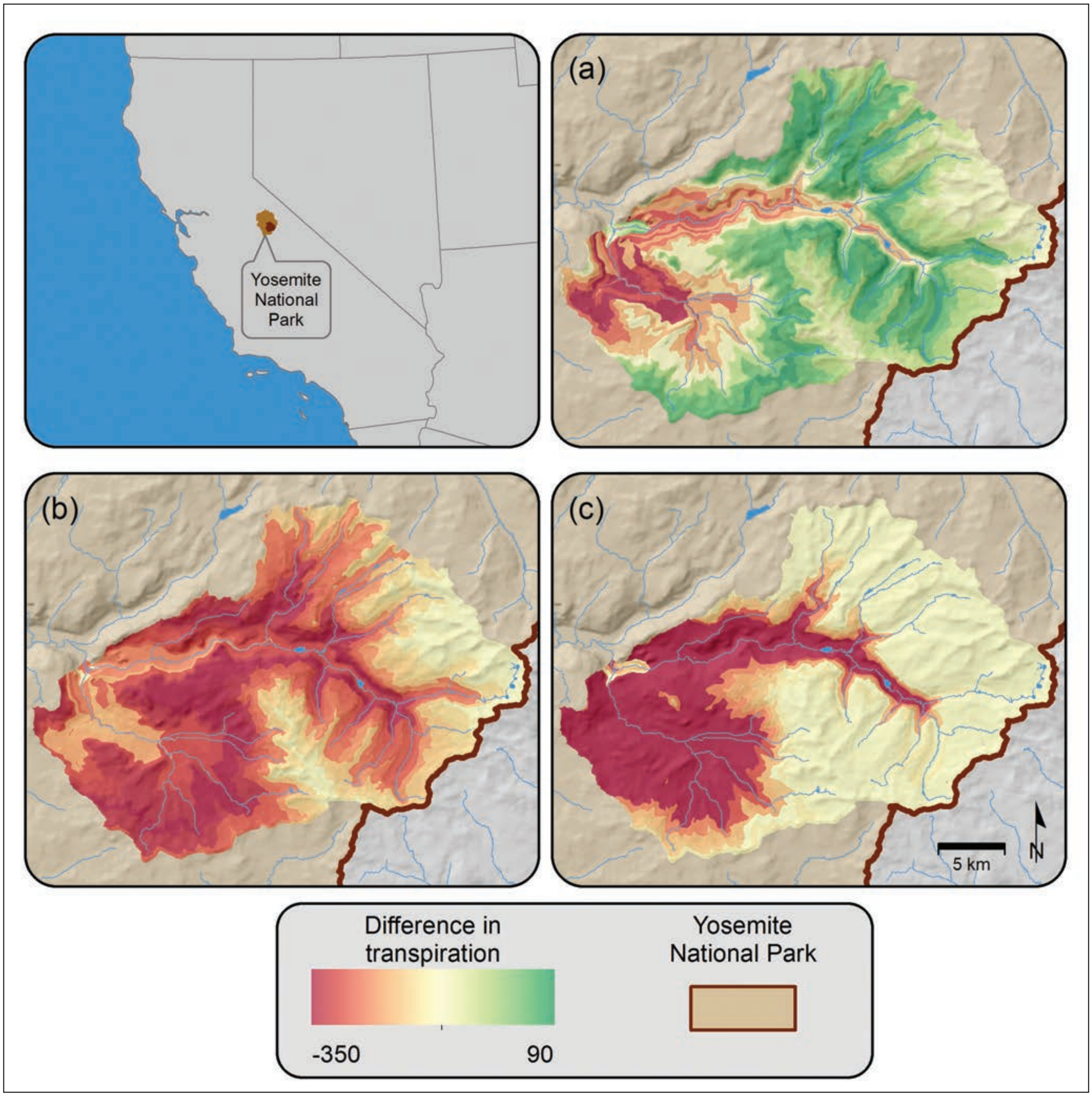

Figure 2.21—Maps of Upper Merced River watershed, Yosemite Valley, California, showing areas with greatest differences in transpiration between (a) warmest and coldest simulation years, (b) wet vs. average precipitation year, and (c) dry vs. average precipitation year. Greatest decreases in transpiration are shown in red; increases between years are shown in green. From Christensen et al. (2008).

Thus, the extra effort of demographic tracking can determine the vulnerability of individual trees to risk factors, including climate change over time, variation in abiotic variables over space, and competition (Clark et al. 2011).

Dispersal and Migration Models

To colonize suitable habitat resulting from a changing climate, each affected species will need to either migrate or be moved. Approaches used to model migration include reaction-diffusion models, phenomenological models, mechanistic models, and simulation models (Clark et al. 2003, Hardy 2005, Katul et al. 2005, Levin et al. 2003, Nathan et al. 2011). Recent advances in digital computation and more reliable data from seed dispersal studies have allowed improvement of these models so that they can begin to project the parameter values of seed dispersal curves as well as the seed distributions. For example, Nathan et al. (2011) modeled 12 North American wind-dispersed tree species for current and projected future spread according to 10 key dispersal, demographic, and environmental factors affecting population spread. They found a very low likelihood of the ability of any of the 12 species to spread 300 to 500 m yr $^{-1}$, the rate of change expected under climate change (Loarie et al. 2009). In contrast, the SHIFT model uses historical migration rates along with the strengths of the seed sources (its abundance within its current range) and potential future sinks (abundance of potential suitable habitat), rather than using poorly understood life history parameters (Iverson et al. 2004a, 2004b; Schwartz et al. 2001). When SHIFT model outputs of colonization potentials were combined with a species distribution model (DISTRIB) simulation of suitable habitat for five species—common persimmon (*Diospyros virginiana* L.), sweetgum, sourwood (*Oxydendrum arboreum* [L.] DC), loblolly pine, and southern red oak (*Quercus falcata* Michx.)—only 15 percent of the newly suitable habitat had any likelihood of being colonized by those species within 100 years (Iverson et al. 2004a, 2004b). These results both suggest that a serious lag will occur before species migration into the new habitat.

Assisted Migration

Many species will be unable to migrate to suitable habitat within 100 years (Iverson et al. 2004a, 2004b) and may face serious consequences if they cannot adapt to new climatic conditions. Assisted migration may help mitigate climate change by intentionally moving species to climatically suitable locations outside their natural range (Hoegh-Guldberg et al. 2008, McLachlan et al. 2007). Assisted migration has been controversial, with some advocating for it (Minteer and Collins 2010, Vitt et al. 2010) and some against (Ricciardi and Simberloff 2009). Proponents state that these drastic measures are needed to save certain species that cannot adapt or disperse fast enough in an era of unprecedented global change. The main concern of opponents is that the placement of species outside their range may disturb native species and ecosystems when these ‘‘climate refugees’’ establish themselves in new environments. The uncertainty of climate in the future and the complexity and contingency associated with ecosystem response also argue against assisted migration.

One way to resolve the debate is to subdivide assisted migration into “rescue-assisted migration” and “forestry-assisted migration.” As the names imply, rescue assisted migration moves species to rescue them from extinction in the face of climate change, and this type is the source of most of the controversy. Forestry-assisted migration is aimed more at maintaining high levels of productivity and diversity in widespread tree species that are commercially, socially, culturally, or ecologically valuable (Gray et al. 2011, Kreyling et al. 2011). With forestry-assisted migration, maintaining forest productivity and ecosystem services are the most obvious desired outcomes. Given the broad distribution of most tree species, and the relatively short distances proposed for tree seed migration, forestry-assisted migration typically involves transfers within or just beyond current range limits to locations where a population’s bioclimatic envelope is expected to reside within the lifetime of the planted population (Gray et al. 2011). The introduction of genotypes to climatically appropriate locations may also contribute to overall forest health by establishing vigorous plantations across the landscape that are less susceptible to forest pests and pathogens (Wu et al. 2005). If realized, such an outcome would help ensure the continued flow of the many ecosystem services provided by forests, such as wildlife habitat, erosion prevention, and C uptake (Kreyling et al. 2011). If practiced in a manner where genotypes are transferred within or just beyond current range limits, forestry-assisted migration may be a viable tool for adaptation to climate change, especially if limited to current intensively managed plantations. Turning extensive areas of forest in the United States from lightly managed forests into managed plantations would likely be unpopular.

Key Findings

- Models project that species habitat for most species will move up in elevation and northward in latitude and be reduced or disappear from current habitats in lower elevations and lower latitudes.
- Habitats will probably move faster than tree species can disperse, creating uncertainty about the future vegetation composition of these new habitats.

Key Information Needs

- Studies on the mitigating effects of elevated CO_2 on drought stress and subsequent effects on projections of future habitat suitability.

Effects of Altered Forest Processes and Functions on Ecosystem Services

Ecosystem services link the effects of altered forest processes, conditions, and disturbance regimes to human well-being (World Resources Institute 2005). A broad range of utility and values derive from four types of ecosystem services: (1) provisioning or products from ecosystems, (2) regulation of ecosystem processes, (3) cultural or nonmaterial benefits, and (4) supporting services required for the production of all other ecosystem services (Joyce et al. 2008) (fig. 2.22). Anticipated climate changes portend changes in all types of ecosystem services derived from forests. Because the assessment endpoint for ecosystem services is human well-being, we are ultimately concerned about the potential effects of climate change on the ecosystem services that forests provide. This subsection explores these changes and provides a linkage between climate change effects on biophysical processes and human well-being.

Ecosystem services differ across temporal and spatial scales but are most often assessed and recognized for watersheds and regions (or subregions). However, they can also be meaningful at the forest stand or national scales. Disturbances (natural and human) and stressors can control delivery of ecosystem services across variable timeframes. Ecosystem services occur in forests not as a single service but rather as a suite or bundle of services. The bundle of services changes with time and in response to disturbance regimes and stressors.

The vulnerability of ecosystem services to climate change will vary widely, depending not only on the service of concern (e.g., wood products or flood regulation) and location (defined by region, such as the Southwest versus the northeastern), but also on the location in reference to human condition, such as rural versus urban settings. The value of the affected service multiplied by the likelihood of effect defines the risk to ecosystem services and provides a framework for understanding potential consequences and prioritizing actions.

Climate-related mechanisms of change in the Nation’s forests could alter ecosystem services in ways that are not yet fully understood, and estimating these effects introduces another layer of uncertainty. That is, climate regulates forest processes that control future forest conditions that determine future ecosystem services. Still, the potential effects of climate change on forest ecosystems could have profound and mostly disruptive consequences for ecosystem services with important implications for human well-being. Ecosystem services also depend on the interactions with land use, human demographics and economies, which may simultaneously adjust to climate stimuli (see chapter 3).

Forests in the United States consist of both managed (active) and unmanaged (passive) ecosystems (Ryan et al. 2008) held in public and private ownerships. Some public forests or wildlands are withdrawn from active management (national parks, state parks, wilderness areas and wild and

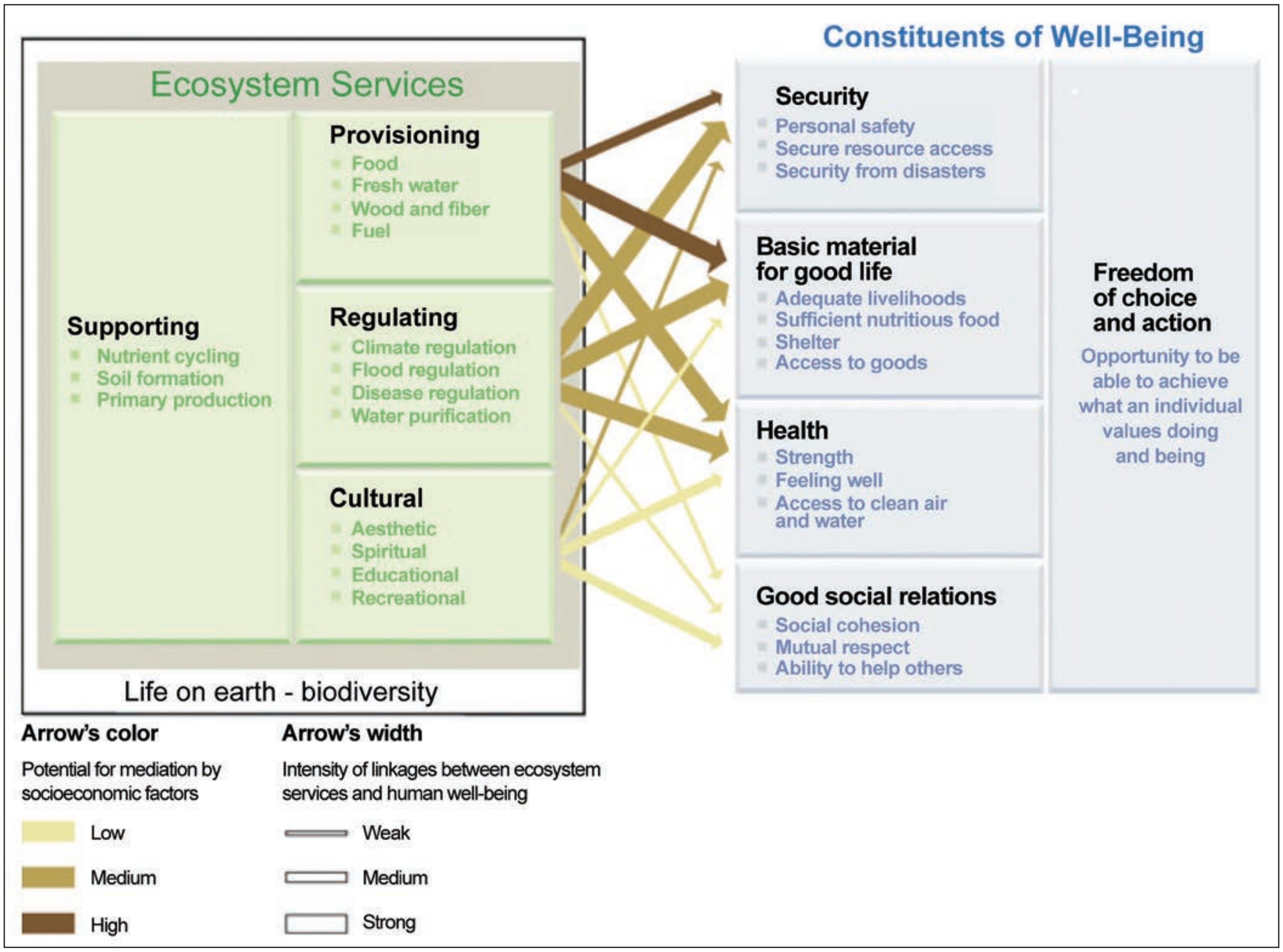

Figure 2.22—Linkages between ecosystem services and human well-being (World Resources Institute 2005).

scenic rivers), but most of them are managed for multiple-use goals (e.g., most national forests and Bureau of Land Management lands). Public land management in the United States is largely focused on nonmarket ecosystem services, including recreation, aesthetic values, and water purification. Most forest management for timber production occurs on private forest lands in the United States (Smith et al. 2009), using both capital-intensive (short- rotation, pine plantation silviculture) and land-extensive approaches (occasional harvesting followed by natural regeneration). Private lands also provide the full spectrum of ecosystem services, either by design through conservation easements, or as a byproduct of other management objectives (see Butler et al. 2007). In many cases, private forest lands provide ecosystem services that accrue to broader social (human) well-being without equitable financial compensation. Public ownership dominates Western forests, and private ownership dominates Eastern forests.

The bundle of ecosystem services provided under current climate conditions differs across the assortment of public and private and managed and unmanaged forests. As a consequence of the regional distribution of anticipated future climate change, the provision of ecosystem services from these lands could also change and be modified by adaptation and mitigation strategies. Social perception of the risks to ecosystem services will be determined by the rate of change in these services (flows) as well as by an understanding of the adaptation and mitigation strategies applied in response to climate change. Social systems will adapt to climate change and affect the condition of forests in the United States and throughout the world.

Several mechanisms of change in forest ecosystems described in the preceding subsections hold implications for ecosystem services. First, climate change could alter the amount and distribution of forest biomass in forests, either through shifts in productivity associated with atmospheric C concentrations or through altered forest disturbance regimes. Changes in forest biomass directly influence the supply of wood products from lumber to fuel for electricity production (provisioning services), and they alter the amount of C stored in forest pools (a regulating service). Future productivity and disturbance effects would probably be focused in the Rocky Mountain and Intermountain West and Alaska, where only a small portion of U.S. timber production occurs, suggesting declines in timber supply from these regions. Declines would be small in the context of national markets, but they could represent substantial shares of local rural economic activity in these regions. The scale of timber effects would likely be local.

Changes in tree cover will affect microclimate conditions (e.g., the cooling of urban heat islands), whereas shifts in C stocks through accumulation of biomass could affect changes in global climate trajectories. Projections of accelerated emissions related to elevated insect epidemics and fire activity could represent a substantial effect on the Nation's forest C reservoirs. The scale of C storage effects could shift U.S. forests from net C sinks to net C emitters (Wear et al. 2012).

Because productivity effects related to atmospheric changes are ambiguous and highly uncertain, their influence on timber markets or C stocks are generally unknown. However, if forest productivity were to increase in the Eastern United States and decrease in the Western United States, this could accelerate the shift in timber production from West to East, and especially to the Southeast.

Disturbance regimes affect other ecosystem services as well. Estimates of the economic consequences of insect and pathogen outbreaks focus either on timber market effects (e.g., southern pine beetle, [Pye et al. 2011]) or on the influence of tree mortality on property values (e.g., hemlock woolly adelgid, [Holmes et al. 2010]). These measures of market effects for price-based services address one element of a complex of values affected by forest disturbances. In the case of forest insects, management decisions already account for a certain level of expected tree mortality, so the more relevant question is whether effects significantly exceed the "background" losses associated with endemic insects and pathogens.[8] Property values define the effect of disturbance and related mortality on the ecosystem services delivered to private property owners, but they cannot capture the "public good" aspects of changes to forest aesthetics for people who view forests. To illustrate, widespread tree mortality related to pine beetle epidemics on national forests can reduce the aesthetic values for millions of people. These quality of life effects represent real value losses, but they are difficult to estimate and may even be transitory as regrowth occurs and society adjusts expectations regarding what constitutes a natural or aesthetically appealing condition.

Climate change could alter the complex of interactions between forest conditions and waterflow and quality. Forest cover and condition constitute only one element of a complex system, so effects may be difficult to isolate, but forest condition appears to be strongly related to both flood protection (a regulating service) and water quantity and quality (a provisioning service). More variable precipitation patterns (stronger drought and extreme rainfall events) both increase the service value of forests in protecting against flooding and landslides, but they also change forest conditions in ways that reduce soil-protecting qualities. This negative feedback suggests potential for accelerated losses of flood protection services of forests in many places. Reduced supplies of these services would coincide with strong growth in the demand for water services caused by population growth and associated water needs for personal and commercial uses.

The longer term and less certain effects of climate change on forest conditions discussed above suggest a growing area of forests in a state of disequilibrium with

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[8] Studies of economic effects of endemic pests such as Pye et al. (2011) may best be viewed as providing estimates of background losses (i.e., the "business as usual" case).

disjoint species-climate associations. The notion of "novel" conditions suggests "unknowable" implications, especially regarding the supply and demands for ecosystem services and the reactions of private landowners and government to increasing scarcity of important services. However, economic factors will likely drive responses. It is also likely that the risks of climate change to forests may open public dialogue regarding the costs and benefits of providing ecosystem services as well as how changes in forest policy may better align the producers and consumers of these services on both private and public forest lands (e.g., providing compensation for private landowners' provision of scarce ecosystem services, as anticipated by the 2008 Farm Bill [Food, Conservation, and Energy Act of 2008]).

Adaption and mitigation strategies for forests will alter the provisioning of ecosystem services and involve explicit tradeoffs between services. For example, thinning and fuel treatment to reduce the vulnerability of forests to disturbance regimes and stressors defines a specific tradeoff between short-term changes in C stocks to enhance the long-term sequestration of C.

Adaptation strategies in forests can build resistance to climate-related stressors, increase ecosystem resilience by minimizing the severity of climate change effects, or facilitate large-scale ecological transitions in response to changing environmental conditions. Resistance, resilience, and transitions are tiered to increasing levels of environmental change and time scale. Each one of these adaptation strategies will result in a changing bundle of ecosystem services.

Because effects on ecosystem services connect climate change to human well-being, they provide a key metric of costs to society. The scarcity of ecosystem services may provide an impetus for adaptive actions, either through individual or company decisions where private goods are affected, or through government intervention where ecosystem services represent public goods. Future provision of forest-related ecosystem services will develop not only from climate-induced changes in forested ecosystems but also from the response of these social choice systems to perceived scarcity of ecosystem services. Furthermore, policy choices designed to mitigate climate change imply important effects on forest conditions and the provision of ecosystem services. For example, increased demands for wood-based bioenergy would alter forest biomass, C, and species composition and would change the distribution of rural land uses. The interaction between climate change and social systems in determining the future of forest ecosystem services is the topic of chapter 3.

Conclusions

As documented in the U.S. Climate Change Science Program Synthesis and Assessment Product 4.3 (Backlund et al. 2008), climate change is occurring and we are observing many effects on forests. Some of the most notable observed effects occur in the Western United States and include an increase in the size and intensity of forest fires, bark beetle outbreaks killing trees over enormous areas, accelerated tree mortality from drought, and earlier snowmelt and runoff. Predictions of the effects of climate change on forests have changed little since the 2008 report, but additional research has provided wider documentation of the effects of climate change and better tools for projecting future effects.

If climate change proceeds as simulated by models, the United States will be warmer, changes in precipitation will differ by region, and precipitation will become more variable. Global climate model simulations predict that the average annual temperature in the United States will increase 2.5 to 5.3 °C by 2100 above average temperatures in 1971 through 2000, with the largest temperature increase in the northern and interior regions. By 2100, precipitation will likely decrease 6 to 12 percent in the Southwest and increase 6 to 10 percent in Northern states, and droughts will become more common. Sea level will rise between 0.2 and 2.0 m by 2100, depending on the emissions scenario and model. Days with temperature greater than 35 °C will also increase.

The projected changes will likely lead to even more disturbance from insect outbreaks, forest fire, and drought, and the tree mortality from these disturbances may switch the United States from a current C sink (offsetting 13 percent of U.S. fossil fuel greenhouse gas emissions) to a source. Such a switch would provide a positive feedback by accelerating atmospheric CO_2 concentrations and climate warming. Carbon losses from tree mortality caused by disturbance may be partially offset by increased growth in Eastern U.S. forests, where water is sufficient and elevated atmospheric CO_2 and N deposition promote tree growth. Fire suppression efforts will likely be more costly as climate warms and fires become larger and more frequent, and as more houses are built in the wildland-urban interface. Interactions among disturbances are currently difficult to project, but will likely increase overall disturbance.

Habitat for species will likely move northward and upward in elevation, and the movement of suitable habitat may be faster than species can disperse to the new habitats. Climate change will likely accelerate the establishment of invasive species in forests, with perhaps the highest risk in mountainous ecosystems.

Direct and indirect effects of climate change will affect the hydrological cycle. The effects of elevated CO_2 on transpiration will likely be less than ± 10 percent, a relatively small change compared to the effects of precipitation variability on transpiration. More frequent droughts will reduce streamflow, and concentrating precipitation in intense storms will likely increase the risk of erosion and landslides. Tree mortality from disturbances will likely increase runoff, and elevated temperatures will probably decrease snow cover depth, duration, and extent and advance the timing of runoff.

Predictions for many other ecosystem effects remain much less certain than those presented above. Some effects, like the response of mature trees to elevated CO_2 or the combined effects of temperature and drought, currently lack the science to make good projections. Other projections, such as the movement of individual species, the location and timing of insect and pathogen outbreaks, and the success of specific invasive species, result from complex interacting factors and contingencies—perhaps too complex to ever be predictable. Local predictions for future climate remain uncertain, which makes local projections of effects on ecosystems also uncertain.

Despite some uncertainty about how future ecosystems will evolve, what their specific species complement will be, and the timing and exact frequency of future disturbance, climate change will bring more disturbance to forests and with it, significant management challenges. Approaches to dealing with many of these management challenges are presented in chapter 4 of this assessment.

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