Chapter 6
Regional Highlights of Climate Change

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6.1 Introduction

The U.S. Global Change Research Program provides a national framework for research and communication of scientific information and periodically develops syntheses through the National Climate Assessment (NCA) (http://globalchange.gov). Although the assessment is comprised of national syntheses for many

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Fig. 6.1 Regions of the United States as defined by the U.S. Global Change Research Program National Climate Assessment

different sectors (forests, agriculture, water resources, transportation, etc.), it also summarizes climate change effects in eight regions of the United States, each of which is represented by a separate regional assessment. Most of the information in this book is derived from a report for the forest sector of the NCA. Although the report and this book contain many specific examples, here we summarize and highlight some of the most important climate change issues facing each region of the United States, exploring a diversity of issues beyond the information contained in the preceding chapters.

The geographic domains of the regions as defined by the USGCRP are shown in Fig. 6.1. These regions may differ from other physical, biological, or political definitions of U.S. regions, but are used here for consistency with other components of the NCA. Each section provides different perspectives and a variable mix of biophysical and social issues. Despite the diversity of themes presented here, we note that climatic extremes, ecological disturbance, and their interactions are expected to have the biggest effects on ecosystems and social systems in most regions in the coming decades.

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6.2 Alaska

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Alaskan forests cover one-third of the state's 52 million ha of land (Parson et al. 2001), and are regionally and globally significant. Ninety percent of Alaskan forests are classified as boreal, representing 4 % of the world's boreal forests, and are located throughout interior and south-central Alaska. The remaining 10 % of Alaskan forests are classified as coastal-temperate, representing 19 % of the world's coastal-temperate forests (National Synthesis Assessment Team 2003), and are located in southeastern Alaska. Regional changes in disturbance regimes of Alaskan forests (Wolken et al. 2011) directly affect the global climate system through greenhouse gas (GHG) emissions (Tan et al. 2007) and altered surface energy budgets (Chapin et al. 2000; Randerson et al. 2006). Climate-related changes in Alaskan forests also have regional societal consequences, because some forests are in proximity to communities (both urban and rural) and provide a diversity of ecosystem services (Reid et al. 2005; Wolken et al. 2011).

In interior Alaska, the most important biophysical factors responding to changes in climate are permafrost thaw and changes in fire regime. The region is characterized by discontinuous permafrost, defined as ground (soil or rock) that remains at or below 0 °C for at least 2 years (Harris et al. 1988). Thawing permafrost may substantially alter surface hydrology, resulting in poorly drained wetlands and thaw lakes (Smith et al. 2005) or well-drained ecosystems on substrates with better drainage. Permafrost thaw may occur directly as a result of changes in regional and global climate, but it is particularly significant following disturbance to the organic soil layer by wildfire (Fig. 6.2). As permafrost thaws, large pools of stored carbon (C) in frozen ground are susceptible to increased decomposition, which will have not only regional effects on gross primary productivity (Vogel et al. 2009) and species composition (Schuur et al. 2007) but also feedbacks to the global C system (Schuur et al. 2008).

Recent changes in the fire regime in interior Alaska are linked to climate. The annual area burned in the Interior has doubled in the last decade compared to any decade since 1970, with three of the largest wildfire years on record also occurring during this time (Kasischke et al. 2010). Black spruce (*Picea mariana* [Mill.] Britton, Sterns & Poggenb.) forests, the dominant forest type in the Interior, historically burned in low-severity, stand-replacing fires every 70–130 years (Johnstone et al. 2010a). However, postfire succession of black spruce forests has recently shifted toward deciduous-dominated forests with the increase in wildfire

Fig. 6.2 In 2004, Alaska's largest wildfire season on record, the Boundary Fire, burned 217,000 ha of forest in interior Alaska (Photo by State of Alaska, Division of Forestry)

severity (Johnstone and Kasischke 2005; Kasischke and Johnstone 2005; Johnstone and Chapin 2006) (Fig. 6.2) and the reduction in fire-return interval (Johnstone et al. 2010a, b; Bernhardt et al. 2011) (see Chap. 4). With continued warming, changes in the fire regime will increase the risk to life and property for Interior Alaskan residents (Chapin et al. 2008).

South-central Alaska may be particularly sensitive to climate changes because of its confluence of human population growth and changing disturbance regimes (e.g., insects, wildfire, invasive species). Warmer temperatures have contributed to recent spruce beetle (*Dendroctonus rufipennis* Kirby) outbreaks in this region by reducing the beetle life cycle from 2 years to 1 year (Berg et al. 2006; Werner et al. 2006). Higher fuel loads resulting from beetle-caused tree mortality are expected to increase the frequency and severity of wildfires (Berg et al. 2006), which raises societal concerns of increased risks to life and property (Flint 2006). Most goods are shipped to Alaska via ports in south-central Alaska, so invasive plant species will probably become an increasingly important risk factor. Several invasive plant species in Alaska have already spread into burned areas (e.g., Siberian peashrub [*Caragana arborescens* Lam.], narrowleaf hawksbeard [*Crepis tectorum* L.], and white sweetclover [*Melilotus alba* Medik.]) (Lapina and Carlson 2004; Cortés-Burns et al. 2008).

Changes in surface hydrology in south-central Alaska have also been linked to warmer temperatures. In the Kenai lowlands in south-central Alaska, many water bodies have shrunk in response to warming since the 1950s and have subsequently been invaded by woody vegetation (Klein et al. 2005). Recently, the rate of woody invasion has accelerated as a result of a 56 % decline in water balance since 1968 (Berg et al. 2009). As a result of these combined effects of wetland drying and vegetation succession, wetlands are becoming weak C sources rather than strong C sinks, which has important consequences for the global climate system.

In southeastern Alaska, climatic warming has affected forest ecosystems primarily through effects on precipitation. Historically, this region has average winter temperatures close to 0 °C and long growing seasons, so even moderate warming could increase rain and reduce snow. Many glaciers extending from Glacier Bay and the Juneau ice field have receded since 1750, with observed reductions in snow (Motyka et al. 2002; Larsen et al. 2005). Continued warming and corresponding reductions in snow precipitation will influence the hydrologic cycle and thus alter fish and mammal habitat, organic matter decomposition, and the C cycle.

For the past 100 years, the culturally and economically important Alaska cedar (*Callitropsis nootkatensis* [D. Don] Oerst. ex. D.P. Little), also known as yellow-cedar, has been dying throughout southeastern Alaska (Hennon et al. 2006). The onset of this decline in 1880 (Hennon et al. 1990) is attributed to warmer winters and reduced snow, combined with early spring freezing events (Beier et al. 2008). The decline in Alaska cedar also has societal consequences because it is the highest valued commercial timber species exported from the region (Robertson and Brooks 2001). Native Alaskans also value this tree for ceremonial carvings; subsistence uses include fuel, clothing, baskets, bows, tea, and medicine (Schroeder and Kookesh 1990; Pojar and MacKinnon 1994). If cedar decline continues, it will alter the structure and function of forest ecosystems, as well as the lifeways of people in this region.

6.3 Hawaii and the U.S.-Affiliated Pacific Islands

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Hawaii and the U.S.-affiliated Pacific islands, including Guam, American Samoa, Commonwealth of Northern Marianas Islands, Federated States of Micronesia, Republic of Palau, and the Marshall Islands, contain a high diversity of flora, fauna, ecosystems, geographies, and cultures, with climates ranging from lowland tropical to alpine desert. Forest ecosystems range from equatorial mangrove swamps to subalpine dry forests on high islands, with most other forest life zones between. As a result, associated climate change effects and potential management strategies vary across the region (Mimura et al. 2007). The vulnerability of Pacific islands is caused by the (1) fast rate at which climate change is occurring; (2) diversity of climate-related threats and drivers of change (sea level rise, precipitation changes, invasive species); (3) low financial, technological, and human resource capacities to adapt to or mitigate projected effects; (4) pressing economic concerns affecting island communities; and (5) uncertainty about the relevance of large-scale projections for local scales. However, island societies may be somewhat resilient to climate change, because cultures are based on traditional knowledge, tools, and institutions that have allowed small island communities to persist during historical periods of biosocial change. Resilience is also provided by strong, locally based land and shore ownerships, subsistence economies, opportunities for human migration, and tight linkages among decision makers, state-level managers, and land owners (Barnett 2001; Mimura et al. 2007).

The distribution and persistence of most forest species are largely determined by temperature and precipitation, and coastal forests are also affected by sea level. Based on known historical climate-vegetation relationships, many forests are expected to experience significant changes in distribution and abundance by the end of the twenty-first century. Over the past 30 years, air temperature for mid-elevation ecosystems in Hawaii increased by 0.3 °C per decade, exceeding the global average rate (Giambelluca et al. 2008a). Stream flow decreased by 10 % during the period 1973–2002 compared to 1913–1972 (Oki 2004), which is similar to what is suggested by simulation modeling for a warmer climate (Safeeq and Fares 2011). Preliminary climatic downscaling for the Hawaiian Islands projects that continued warming and drying will be coupled with more intense rain events separated by more dry days (Chu and Chen 2005; Chu et al. 2010; Norton et al. 2011).

The direct effects of climate change on forests will be variable and strongly dependent on interactions with other disturbances, especially novel fire regimes that are expanding into new areas because of invasion by fire-prone exotic grass and shrub species (Fig. 6.3), such as fountain grass (*Cenchrus setaceus* [Forssk.] Morrone) and common gorse (*Ulex europaeus* L.) in Hawaii and guinea grass (*Urochloa maxima* [Jacq.] R.D. Webster) across the region (D'Antonio and Vitousek 1992). These invasions have the potential to alter or even eliminate native forests through conversion of forested systems to open, exotic-dominated grass and shrub lands. In wet forests, invasive plants can alter hydrologic processes by increasing water use by vegetation (Cavaleri and Sack 2010), and these effects may be more severe under warmer or drier conditions (Giambelluca et al. 2008b). Because invasive species have invaded most native-dominated ecosystems (Asner et al. 2005, 2008), anticipated direct (higher evapotranspiration) and indirect (increased competitive advantage of high water use plants) effects of climate change will modify stream flows and populations of stream organisms. Higher temperature will facilitate expansion of pathogens into cooler, high-elevation areas and potentially reduce native bird populations of Hawaii (Benning et al. 2002).

Most forests have at least some stimulatory effects from carbon dioxide CO_2 (Norby et al. 2005), especially in younger, faster-growing species. Therefore,

Fig. 6.3 In Hawaii's high-elevation forests (shown here) and in forests across the Pacific, projected warming and drying will increase invasive plants such as fire-prone grasses, resulting in novel fire regimes and conversion of native forests to exotic grasslands. For areas already affected in this way, climate change will increase the frequency and in some cases intensity of wildfire (Photo by Christian Giardina, U.S. Forest Service)

the effects of climate on fire regimes and stream flow described above may be accentuated by rising CO_2 through increased fuel accumulation and increased competitiveness of invasive species; higher water use across the landscape may be partially offset by higher water use efficiency in some species. For strand, mangrove, and other coastal forests, anticipated sea level rise for the region (about 2 mm year^{-1}) (Mimura et al. 2007) will have moderate (initial or enhanced inundation with expansion to higher elevation) to very large (extirpation of forest species in the absence of upland refugia) effects on the distribution and persistence of these systems. Enhanced storm activity and intensity in the region during some large-scale climatic events (e.g., El Niño Southern Oscillation) will increase the effects of storm surges on these coastal systems and increase salt water intrusions into the freshwater lens that human and natural systems require for existence (Mimura et al. 2007). A combination of sea level rise and increased frequency and severity of storm surges could result in extensive loss of forest habitat in low-lying islands.

Climate change effects on island ecosystems (Table 6.1) will extend across federal, state, tribal, and private lands, the most vulnerable being coastal systems and human communities. Sea level rise, apparent trajectories for storm intensity and frequency in the region, and warming and drying trends (for Hawaii) are based on robust measurements that suggest high confidence in projected ecological changes. Vulnerabilities and risks are most relevant in coastal zone forests, but all forests

Table 6.1 Potential climate change related risks, and confidence in projections (From Mimura et al. 2007)

Risk	Confidence level
Small islands have characteristics that make them especially vulnerable to the effects of climate change, sea level rise, and extreme events	Very high
Sea level rise is expected to exacerbate inundation, storm surges, erosion, and other coastal hazards, thus threatening infrastructure, settlements, and facilities that support the livelihood of island communities	Very high
Strong evidence exists that under most climate change scenarios, water resources in small islands will be seriously compromised	Very high
On some islands, especially those at higher latitudes, warming has already led to the replacement of some local plant species	High
It is very likely that subsistence and commercial agriculture on small islands will be adversely affected by climate change	High
Changes in tropical cyclone tracks are closely associated with the El Niño Southern Oscillation, so warming will increase the risk of more persistent and severe tropical cyclones	Moderate

of the region are at greater risk of degradation from secondary drivers of change, especially fire, invasive species, insects, and pathogens.

Island systems of the Pacific are home to some of the most intact traditional cultures on earth and communities that generally are strongly linked to forest resources. Sea level rise, increased storm frequency and intensity, and more severe droughts will reduce the habitability of atolls, representing a major potential impact in Pacific island countries (Barnett and Adger 2003). For low-lying Islands of the Pacific, enhanced storm activity and severity and sea level rise will cause the relocation of entire communities and even nations; the first climate refugees have already had to relocate from homelands in the region (Mimura et al. 2007). For high islands, warming and drying in combination with expanded cover of invasive species, and in some cases increased fire frequency and severity, will alter the hydrological function of forested watersheds, with cascading effects on groundwater recharge as well as downstream agriculture, urban development, and tourism (Mimura et al. 2007).

Few options are available for managing climate change effects in Pacific island ecosystems. For some very low-lying islands and island systems, such as the Marshall Islands where much of the land mass is below anticipated future sea levels, climate change will reduce fresh water supply and community viability. When fresh water becomes contaminated with salt water, the options for persisting in a location are logistically challenging and often unsustainable. For higher islands, adaptation practices include shoreline stabilization through tree planting, reduced tree harvest, facilitated upward or inward migration of forest species, and shoreline development planning (Mimura 1999). Because many Pacific island lands are owned and managed traditionally, adaptation and mitigation can be enhanced at the community level through education and outreach focused on coastal management and protection, mitigation of sea level rise, forest watershed protection, and restoration actions.

Because Hawaii has significant topographic relief, as well as moderately sophisticated management infrastructure, anticipatory planning and facilitation of inward species migration are already being practiced in some coastal wetlands. The spread of invasive species can be slowed by multifaceted management strategies (biocontrol, physical and chemical control) and restoration of areas with fire-prone invasives (green break planting, native species planting, physical and chemical control of weed species). To this end, management prescriptions for simultaneously addressing conservation objectives and climate change effects are being addressed by the Hawaii Department of Land and Natural Resources Watershed Initiative, U.S. Fish and Wildlife Service (USFWS) Pacific Island Climate Change Cooperative, Hawaii Restoration and Conservation Initiative, and Hawaii Conservation Alliance Effective Conservation Program, as well as individual climate change management plans (e.g., USFWS Hakalau Forest National Wildlife Refuge Comprehensive Conservation Plan). Because minimal scientific information is available for the U.S.-affiliated Pacific islands, research is needed to identify thresholds beyond which social-ecological systems in atolls will be permanently compromised, and the contributions of resource management, behavior, and biophysical factors to pushing systems across thresholds (Barnett and Adger 2003).

6.4 Northwest

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The state of knowledge about climatic effects on forests of the Northwest region was recently summarized in a peer-reviewed assessment of these effects in Washington (Littell et al. 2009, 2010) and a white paper on climatic effects on Oregon vegetation (Shafer et al. 2010). Recent modeling studies provide additional scenarios for effects of climate change on wildfire, insects, and dynamic vegetation in the Northwest. This summary describes evidence for such effects on climate-sensitive forest species and vegetation distribution, fire, insect outbreaks, and tree growth.

Based on projections of direct effects of climate change on the distribution of Northwest tree species and forest biomes, widespread changes in equilibrium vegetation are expected. Statistical models of tree species-climate relationships show that each tree species has a unique relationship with limiting climatic factors (McKenzie et al. 2003; Rehfeldt et al. 2006, 2008; McKenney et al. 2011). These relationships have been used to project future climate suitability for species in western North America (Rehfeldt et al. 2006, 2009; McKenney et al. 2007, 2011) and in Washington in particular (e.g., Littell et al. 2010 after Rehfeldt et al. 2006). Climate is projected to become unfavorable for Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco) over 32 % of its current range in Washington, and up to 85 % of the range of some pine species may be outside the current climatically suitable range (Rehfeldt et al. 2006; Littell et al. 2010). Based on preliminary projections from the global climate model (GCM) CCSM2 and the process model 3PG, Coops and Waring (2010) projected that the range of lodgepole pine (*Pinus contorta* var. *latifolia* Engelm. ex S. Watson) will decrease in the Northwest. Using similar methods, Coops and Waring (2011) projected a decline in current climatically suitable area for 15 tree species in the Northwest by the 2080s; 5 of these species would lose less than 20 % of this range, and the range of the other 10 species would decline up to 70 %.

Various modeling studies project significant changes in species distribution in the Northwest, but with considerable variation within and between studies. McKenney et al. (2011) summarized responses of tree species to climate change across western North America for three emission scenarios. Projected changes in suitable climates for Northwest tree species ranged from near balanced (−5 to +10) to greatly altered species distribution at the subregional scale (−21 to −38 species), depending on the emissions scenario. Modeling results by Shafer et al. (2010) indicate either relatively little change over the twenty-first century under a moderate warming, wetter climate (CSIRO Mk3, B1), or, in western Oregon, a nearly complete conversion from maritime to evergreen needleleaf forest and subtropical mixed forest under a warmer, drier climate (HadCM3, A2). Lenihan et al. (2008) concluded that shrublands would be converted to woodlands, and woodlands to forest in response to elevated CO_2, a trend that would be facilitated by fire suppression.

Potential changes in fire regimes and area burned have major implications for ecosystem function, resource values in the wildland-urban interface (WUI), and expenditures and policy for fire suppression and fuels management. The projected effects of climate change on fire in the Northwest generally suggest increases in both fire area burned and biomass consumed in forests (McKenzie et al. 2004; Littell et al. 2009, 2010). Littell et al. (2010) used statistical climate-fire models to project future area burned for the combined area of Idaho, Montana, Oregon, and Washington. Median regional area burned per year is projected to increase from the current 0.2 million ha, to 0.3 million ha in the 2020s, 0.5 million ha in the 2040s, and 0.8 million ha in the 2080s. Furthermore, the area burned compared to the period 1980 through 2006 is expected to increase, on average, by a factor of 3.8 in forested ecosystems (western and eastern Cascades, Okanogan Highlands, Blue Mountains). Rogers et al. (2011) used the MC1 dynamic vegetation model to project fire area burned, given climate and dynamic vegetation under three GCMs. Compared to 1971–2000, large increases are predicted by 2100 in both area burned (76–310 %), and burn severities (29–41 %).

Tree vigor and insect populations are both affected by temperature: host trees can be more vulnerable because of water deficit, and bark beetle outbreaks are correlated with high temperature (Powell and Logan 2005) and low precipitation (Berg et al. 2006). Littell et al. (2010) projected relationships between climate (vapor pressure deficit) and mountain pine beetle (*Dendroctonus ponderosae* Hopkins) (MPB) attack in the late twenty-first century. They also projected potential changes in MPB adaptive seasonality, which suggested that the region of climatic suitability will move higher in elevation, eventually reducing the total area of suitability. Using future temperature scenarios for the Northwest, Bentz et al. (2010) simulated changes in adaptive seasonality for MPB and single-year offspring survival for the spruce beetle (*D. rufipennis* Kirby) (SBB). The probability of MPB adaptive seasonality increases in higher elevation areas, particularly in the southern and central Cascade Range for the early twenty-first century and in the north Cascades and central Idaho for the late twenty-first century. Single-year development of SBB offspring also increases at high elevations across the region in both the early and late twenty-first century.

Response of tree growth to climate change will depend on subregional-to-local characteristics that change the sensitivity of species along the climatic gradients of their ranges (e.g., Littell et al. 2008; Chen et al. 2010; Peterson and Peterson 2001). Douglas-fir is expected to grow more slowly in much of the drier part of its range (Chen et al. 2010) but may currently be growing faster in many locations in the Northwest (Littell et al. 2008). Although no regional synthesis of expected trends in tree growth exists, the projected trend toward warmer and possibly drier summers in the Northwest (Mote and Salathé 2010) is likely to increase growth where trees are energy limited (at higher elevations) and decrease growth where trees are water limited (at lowest elevations and in driest areas) (Case and Peterson 2005; Holman and Peterson 2006; Littell et al. 2008). Growth at middle elevations will depend on summer precipitation (Littell et al. 2008).

The effects of climate change on forest processes in the Northwest are expected to be diverse, because the mountainous landscape of the region is complex, and species distribution and growth can differ at small spatial scales. Forest cover will change faster via disturbance and subsequent regeneration over decades, rather than via gradual readjustment of vegetation to a new climate over a century or more. Additional data are needed on interactions between disturbances and on connections between climate-induced changes in forests and ecosystem services, including water supply and quality, air quality, and wildlife habitat.

6.5 Southwest

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Dying pinyon pines (*Pinus edulis* Engelm.) in New Mexico and adjacent states in the early 2000s became an iconic image of the effects of a warming climate in U.S. forests. Several consecutive years of drought reduced the vigor of pines, allowing pinyon ips (*Ips confusus* LeConte) to successfully attack and kill pines across more than one million ha (Breshears et al. 2005). The pinyon pine dieback was one of the most important manifestations of extreme climate in North America during the past decade, an indicator that a physiological threshold was exceeded because of the effects of low soil moisture (Floyd et al. 2009). Although this is not direct evidence of the effects of climate change, it demonstrates the effects of severe drought, a phenomenon expected more frequently in the future, on large-scale forest structure and function in arid environments.

Disturbance processes that are facilitated by climatic extremes, primarily multiyear droughts, dominate the potential effects of climatic variability and change on both short-term and long-term forest dynamics in the Southwest (Allen and Breshears 1998). Although diebacks in species other than pinyon pine have not been widespread, large fires and insect outbreaks appear to be increasing in both frequency and spatial extent throughout the Southwest. In Arizona and New Mexico, 14–18 % of the forested area was killed by wildfire and bark beetles between 1997 and 2008 (Williams et al. 2010). This forest mortality appears to be related to the current trend of increasing temperature and decreasing precipitation, at least in the southern portion of the region, since the mid 1970s (Weiss et al. 2009; Cayan et al. 2010).

In late spring 2011, following a winter with extremely low precipitation and a warm spring, the Wallow Fire burned 217,000 ha of forest and woodland in eastern Arizona and western New Mexico, receiving national attention for its size and intensity (Incident Information System 2011). The Wallow Fire was the largest recorded fire in the conterminous United States, and forced the evacuation of eight communities, cost $109 million to suppress (4,700 firefighters involved) and $48 million to implement rehabilitation measures, and resulted in high consumption of organic material and extensive overstory mortality across much of the burned landscape. A total of 880,000 ha burned in Arizona and New Mexico in 2011 (National Interagency Fire Center 2011). Large, intense fires illustrate how extreme drought can cause rapid, widespread change in forest ecosystems.

Recent large fires may portend future increases in wildfire. Using an empirical analysis of historical fire data on federal lands, McKenzie et al. (2004) projected the following increases in annual area burned for these Southwestern States, given a temperature increase of 1.5 °C: Arizona, 150 %; Colorado, 80 %; New Mexico, 350 %; and Utah, 300 %. California and Nevada were projected to be relatively insensitive to temperature, but their data included extensive non-forest area. In a more recent analysis, Littell et al. (n.d.) project the following increases for a 1 °C temperature increase: Arizona. 380–470 %; California, 310 %; Colorado, 280–660 %; Nevada, 280 %; New Mexico, 320–380 %; and Utah, 280–470 %. Applying the Parallel Climate Model to California, Lenihan et al. (2003) projected that area burned will increase at least 10 % per year (compared to historical level) by around 2100 (temperature increase of 2.0 °C).

The general increase in fire that is expected in the future, and that may already be occurring, will result in younger forests, more open structure, increased dominance of early successional plant species, and perhaps some invasive species. Because annual accretion of biomass is relatively low in this region, production of live and dead fuels in the understory in 1 year affects the likelihood of fire in the next year (Littell et al. 2009). The interaction of climate, fuel loading, and fuel moisture will contribute to both future area burned and fire severity.

The ongoing expansion of bark beetle outbreaks in western North America has been especially prominent in Colorado. Since 1996, multiple beetle species have caused high forest mortality on 2.7 million ha, of which 1.4 million ha were infested with mountain pine beetle (USDA FS 2011). Facilitated by extended drought and warmer winters, mountain pine beetle outbreaks have focused primarily on older (stressed) lodgepole pine forest. In Arizona and New Mexico, 7.6–11.3 % of forest and woodland area was affected by extensive tree mortality owing to bark beetles from 1997 through 2008 (Williams et al. 2010). As in other areas of the West, bark beetles appear to be attacking trees at higher elevations than in the past (Gibson et al. 2008).

In a detailed analysis of tree growth data for the United States, Williams et al. (2010) found that growth in the Southwest was positively correlated with interannual variability in total precipitation and negatively correlated with daily maximum temperature during spring through summer, which suggests that increased future drought will have a profound effect on growth and productivity. Projecting an A2 emission scenario on these growth-climate relationships produced significant growth reductions for forests in Arizona, Colorado, and New Mexico after 2050, affecting primarily ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson), Douglas-fir, and pinyon pine. Projected growth decreases were larger than for any other region of the United States (Williams et al. 2010).

Simulation modeling of potential changes in vegetation in California suggests that significant changes can be expected by 2100 (Lenihan et al. 2003). Modeling results show that mixed-evergreen forest will replace evergreen conifer forest throughout much of the latter's historical range. This process may include gradual replacement of Douglas-fir–white fir (*Abies concolor* [Gordon & Glend.] Lindl. ex Hildebr.) forest by Douglas-fir–tanoak (*Lithocarpus densiflorus* [Hook. & Arn.] Rehd.) forest and the replacement of white fir-ponderosa pine forest by ponderosa pine-California black oak (*Quercus kelloggii* Newberry) forest in the Sierra Nevada. Tanoak-Pacific madrone (*Arbutus menziesii* Pursh)-canyon live oak (*Q. chrysolepis* Liebm.) woodland may replace blue oak (*Q. douglasii* Hook. & Arn.) woodlands, chaparral, and perennial grassland. In general, shrubland will replace oak woodland, and grassland will replace shrubland throughout the state. Evergreen conifer forest will advance into the high elevation subalpine forest in the Sierra Nevada, and species such as Shasta red fir (*Abies magnifica* A. Murray) and lodgepole pine may become more common in subalpine parklands and meadows. A high degree of regional variability in species changes can be expected, and large-scale transitions will need to be facilitated through fire disturbance that enables regeneration.

Increased disturbance from fire and insects, combined with lower forest productivity at most lower elevation locations because of a warmer climate, will probably result in lower C storage in most forest ecosystems. The fire-insect stress complex may keep many low-elevation forests in younger age classes in perpetuity. The normal cycle of fire followed by high precipitation (in winter in California, in early summer in much of the rest of the Southwest) may result in increased erosion and downstream sediment delivery (Allen 2007). In a warmer climate, it may be possible to reduce fire severity and protect WUI areas through assertive use

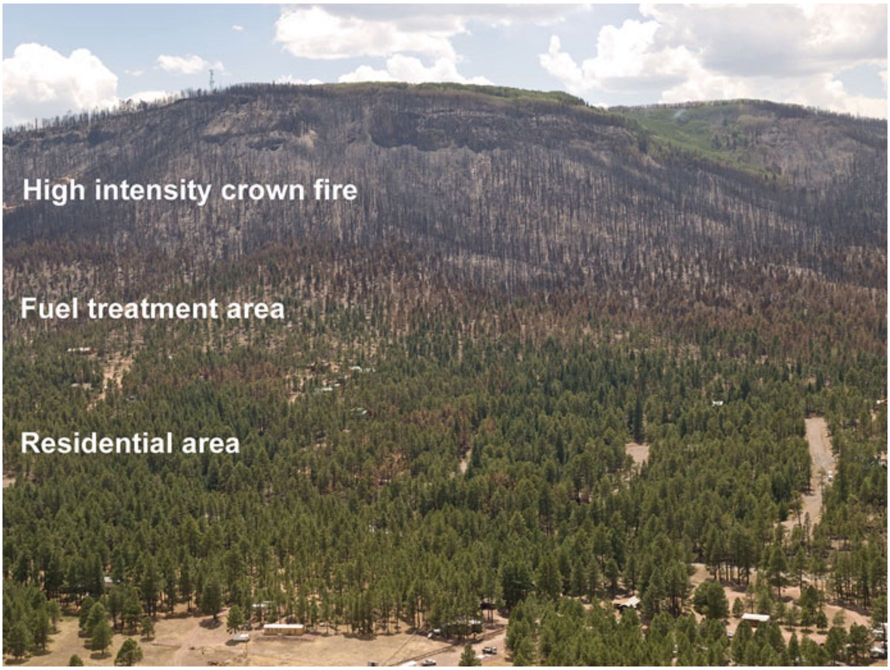

Fig. 6.4 The effectiveness of fuel treatments is seen in this portion of the 2011 Wallow Fire near Alpine, Arizona. High-intensity crown fire was common in this area, but forest that had been thinned and had surface fuels removed experienced lower fire intensity, and structures in the residential area were protected (Photo by U.S. Forest Service)

of fuel treatments (Peterson et al. 2011), as shown recently in the Wallow Fire (Bostwick et al. 2011) (Fig. 6.4). It may also be possible to reduce large-scale beetle epidemics by maintaining multiple forest age classes across the landscape (Li et al. 2005). Significant financial resources and collaboration across different agencies and landowners will be necessary to successfully implement these adaptive strategies.

6.6 Great Plains

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Natural vegetation of the Great Plains is primarily grassland and shrubland ecosystems with trees occurring in scattered areas along streams and rivers, on planted woodlots, as isolated forests such as the Black Hills of South Dakota, and near the biogeographic contact with Rocky Mountains and eastern deciduous forests.

Trees are used in windbreaks and shelterbelts for crops and within agroforestry systems, extending the tree-covered area considerably (e.g., over 160,000 ha in Nebraska) (Meneguzzo et al. 2008). Urban areas in the Great Plains benefit from trees providing wildlife habitat, water storage, recreation, and aesthetic value.

Forests in the northern Great Plains (North Dakota, South Dakota, Kansas, Nebraska) comprise less than 3 % of the total land area within each state (Smith et al. 2009). More than half of the forest land in South Dakota is in public land ownership in contrast to the other three states. Dominant forest species are ponderosa pine (*Pinus ponderosa* Lawson and C. Lawson var. *scopulorum* Engelm.), fir-spruce, and western hardwoods. Eastern cottonwood (*Populus deltoides* Bartr. ex Marsh.) forests are an important source of timber in North Dakota (Haugen et al. 2009) and Nebraska (Meneguzzo et al. 2008). Many cottonwood stands in this region are quite old, and regeneration has been minimal owing to infrequent disturbance (South Dakota Resource Conservation and Forestry Division 2007; Meneguzzo et al. 2008; Moser et al. 2008; Haugen et al. 2009). The decline of this species often leads to establishment of non-native species (Haugen et al. 2009) or expansion of natives such as green ash (*Fraxinus pennsylvanica* Marsh.), which is susceptible to the invasive emerald ash borer (*Agrilus planipennis* Fairmaire). In North Dakota, quaking aspen (*Populus tremuloides* Michx.) forests are generally in poor health and have minimal regeneration because of fire exclusion (Haugen et al. 2009). In South Dakota, forest land is dominated by ponderosa pine forest, which supports a local timber industry in the Black Hills area. Management concerns include densely stocked stands, high fuel loadings and fire hazard, and mountain pine beetle outbreaks. Eastern redcedar (*Juniperus virginiana* L.) is expanding in many states, the result of fire exclusion and prolonged drought conditions (South Dakota Resource Conservation and Forestry Division 2007; Meneguzzo et al. 2008). This presents opportunities for using redcedar for wood products, but also raises concerns about trees encroaching into grasslands and altering wildlife habitat (Moser et al. 2008). Land-use activities that support biofuel development, particularly on marginal agricultural land, may affect forests in this area (Meneguzzo et al. 2008; Haugen et al. 2009).

Forests in the southern Great Plains (Oklahoma, Texas) comprise less than 17 % of the land area (Smith et al. 2009), are often fragmented across large areas, and are mostly privately owned. In Texas, the forest products industry is one of the top ten manufacturing sectors in the state, with a fiscal impact of $33.6 billion on the state economy (Xu 2002). Loss of forest to urbanization, oil and gas development, and conversion to cropland and grassland has led to a permanent reduction in forest cover (Barron 2006; Johnson et al. 2010).

Forests in the western Great Plains (Montana, Wyoming) comprise less than 27 % of the land area (Smith et al. 2009), and most of this land is in public ownership. Montana has large contiguous areas of forest, particularly in the western part of the state where public land, forest industry, and private land intermingle. Both Montana and Wyoming have forested areas on mountains where the surrounding ecosystems are grassland and shrubland. The three major forest types in Montana are also the most commercially important species: Douglas-fir, lodgepole pine, and ponderosa pine (Montana Department of Natural Resources and Conservation 2010). Fire exclusion has caused higher fire hazard and more mountain pine beetle outbreaks. In recent years, the forest industry has been adversely affected by reduced timber supply and general economic trends. Wyoming forests are dominated by lodgepole pine, followed by spruce-fir and ponderosa pine, and land ownership is a mosaic of public, private, and industrial. Similar to Montana, the forest industry in Wyoming has faced several challenges but continues to be a significant component of the state economy (Wyoming State Forestry Division 2009). Both Montana and Wyoming have urban forests, riparian forests, and windbreaks and shelterbelts associated with agriculture. Tree species used in windbreaks and shelterbelts, including ponderosa pine and the nonnatives Scots pine (*Pinus sylvestris* L.) and Austrian pine (*P. nigra* Arnold) are being attacked by mountain pine beetles, and green ash is susceptible to the emerald ash borer. Similar to other parts of the Great Plains, some lower elevation riparian forests are in decline, because regeneration has been reduced by fire exclusion, water diversions, drought, agricultural activities, and urban development.

Little information is available on the potential effects of climate change on Great Plains forests. However, this area has been part of continental and national studies (Bachelet et al. 2008), and areas such as the Greater Yellowstone Ecosystem have a long history of research that has recently included climate change. Tree species in the Yellowstone area are expected to move to higher elevation in a warmer climate (Bartlein et al. 1997; Koteen 2002; Whitlock et al. 2003). However, projecting future vegetation distribution is complicated by the complex topography of Wyoming, which influences the microclimatic environment that controls vegetation distribution. Forests in this area and Montana are currently affected by insect outbreaks and wildfire, and changes in these disturbances under climate change could potentially disrupt ecosystems across large landscapes. A recent modeling study suggests that a warmer climate will increase the frequency and spatial extent of wildfire in the Yellowstone area (Westerling et al. 2011).

In a review of the literature on the effects of climate change in semiarid riparian ecosystems, Perry et al. (2012) noted that climate-driven changes in streamflow are expected to reduce the abundance of dominant, native, early-successional tree species and increase herbaceous, drought-tolerant, and late-successional woody species (including nonnative species), leading to reduced habitat quality for riparian fauna. Riparian systems will be especially important locations on which to focus monitoring for the early effects of climate change.

Reduced tree distribution in the Great Plains will likely have a negative effect on agricultural systems, given the important role of shelterbelts and windbreaks in reducing soil erosion. In these "linear forests," warmer temperatures are expected to reduce aboveground tree biomass and spatial variation in biomass at lower elevations, but may increase biomass on upland habitats (Guo et al. 2004). Whereas most studies in this region have explored the potential influence of elevated CO_2 on grassland, Wyckoff and Bowers (2010) analyzed the relationship between historical climate and tree growth and suggest that the interaction of climate change and elevated CO_2 could be a potential factor in the expansion of forests from the eastern United States into the Great Plains. Carbon sequestration through agroforestry has been suggested as a potential mitigation activity (Morgan et al. 2010).

Across the Great Plains, forests are currently exposed to many stressors. Common to all states in this region is a concern about land-use changes that would reduce the total area of forests, fragment intact forests, and alter forest dynamics. Current stressors such as insects, fungal pathogens, and altered hydrologic dynamics may be exacerbated by a warmer climate. The potential for stress complexes that include wildfire, longer droughts, and increased risk of insect outbreaks could significantly modify Great Plains forest environments (see Chap. 4).

6.7 Midwest

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Forests are a defining landscape feature for much of the Midwest, from boreal forests surrounding the northern Great Lakes to oak-hickory (*Quercus* spp., *Carya* spp.) forests blanketing the Ozark Highlands. Forests cover approximately 28 % of the area in the eight-state Midwest region and help sustain human communities ecologically, economically, and culturally. Most of the Midwest is contained within the Laurentian Mixed Forest, Eastern Broadleaf Forest (Continental and Oceanic), and Prairie Parklands ecoregions (Bailey 1995) (Fig. 6.5).

The broad diversity in species composition and structure across the Midwest will likely engender higher resilience to a changing climate than less diverse biogeographic regions, but each ecoregion might be best characterized by a few strong vulnerabilities. With this in mind, key vulnerabilities related to climate change are summarized below according to ecoregions. The term “vulnerability” refers to a decline in vigor and productivity, in addition to more severely altered community composition or ecosystem function, and a species or ecosystem may be considered vulnerable to climate change by virtue of significantly decreased well-being, even if it is not projected to disappear completely (Swanston et al. 2011).

The *Laurentian Mixed Forest* spans the northern areas of the Great Lakes states (Fig. 6.5), typified by a glaciated landscape with low relief covered with mesic broadleaf deciduous forests, sometimes mixed with conifers, and often grading to pure conifers on poor soils. Winters are cold and of long duration, often with heavy snowfall, and summers are warm and provide much of the annual precipitation. As a transitional zone between boreal forests in the north and broadleaf forests to the

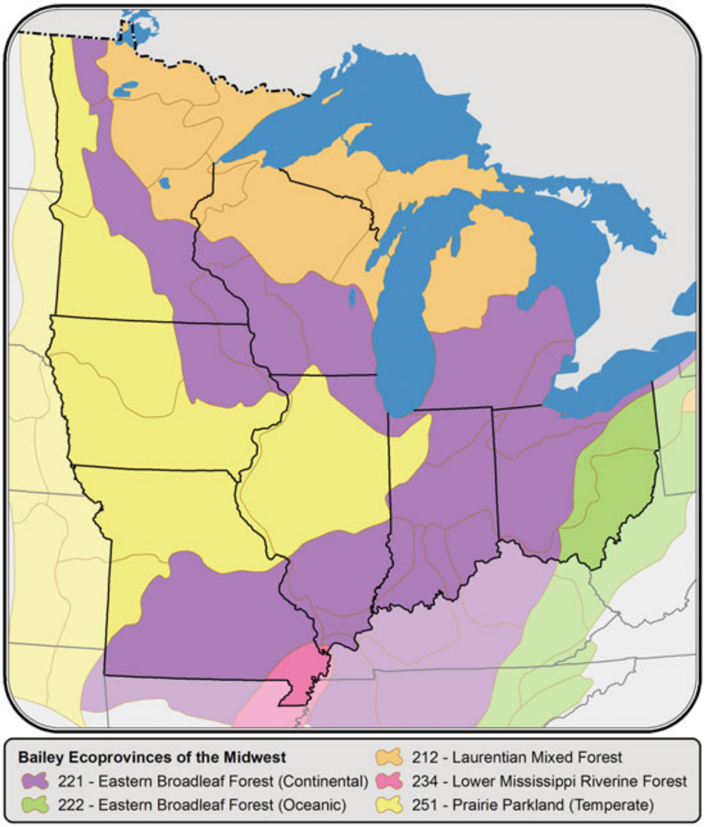

Fig. 6.5 Ecoregions in the Midwest, according to Bailey (1995)

south, the Laurentian forests are often dominated by boreal species at the southern edge of their suitable habitat range. Many of these species, such as black spruce, balsam fir (*Abies balsamea* [L.] Mill.), paper birch (*Betula papyrifera* Marsh.), and northern whitecedar (*Thuja occidentalis* L.), are projected to lose suitable habitat through much of their current range (Iverson et al. 2008a, b; Walker et al. 2002). Associated ecosystems may thus be more likely to experience stress and undergo more distinct community transitions (Swanston et al. 2011; Xu et al. 2012).

Forested wetlands, including peatlands, may be especially susceptible to a combination of range shifts and changes in hydrologic regimes (e.g., Swanston et al. 2011). These systems store a large amount of belowground C (Johnson and Kern 2003) that could be at risk if fire increases in drier conditions. Sub-boreal species such as sugar maple (*Acer saccharum* Marsh.) may be less affected than boreal species, but any effects may be more apparent aesthetically and economically owing to their prevalence on the landscape (Iverson et al. 2008b).

The *Eastern Broadleaf Forest* (Fig. 6.5) mostly consists of the Continental ecoregion, with low rolling hills, some glaciation in the north, and the Ozark

Highlands to the south. Precipitation generally comes during the growing season but decreases in the western ecoregion. Oak-hickory forest is dominant, grading to maple (*Acer* spp.), American beech (*Fagus grandifolia* Ehrh.), and American basswood (*Tilia americana* L.) in the north. Oak decline is increasing the mortality of oak species throughout the southern half of the Midwest and is correlated with drought periods (Wang et al. 2007). Species in the red oak (*Quercus rubra* L.*, Q. coccinea* Münchh.*, Q. velutina* Lam.) group are particularly susceptible to decline and make up a large proportion of upland forests in this ecoregion. White oak (*Q. alba* L.) may also be declining on the western margins of its range (Goldblum 2010), which may be further amplified by higher summer temperatures in the future (Iverson et al. 2008b). Oak decline could worsen if droughts become more frequent or severe, and elevated fine and coarse fuels could result from tree mortality, thereby increasing wildfire hazard.

Wildfire suppression has gradually favored more mesic species such as maple, leaving fire-adapted species like oaks and shortleaf pine (*Pinus echinata* Mill.) at a competitive disadvantage (Nowacki and Abrams 2008). With adequate moisture and continued fire suppression, these forests are likely to persist but may become increasingly susceptible to wildfire in a drier climate (Lenihan et al. 2008). A general decline in resilience, in combination with increased disturbances such as fire, could make these forests more susceptible to invasive species such as kudzu (*Pueraria lobata* [Lour.] Merr., an aggressive vine) and Chinese and European privet (*Ligustrum sinense* Lour. and *L. vulgare* L., highly invasive shrubs), that may to expand into the Midwest as winter minimum temperatures increase (Bradley et al. 2010).

The *Prairie Parklands* (Fig. 6.5) are predominantly covered by agriculture and prairie, with interspersed upland forests of oak and hickory. Forest stands are also found near streams and on north-facing slopes. Fragmentation and parcelization of forest ecosystems are more extreme in the Prairie Parklands than in other Midwest ecoregions. For example, over 90 % of forest land in Iowa is currently divided into private holdings averaging less than 7 ha (Flickinger 2010). Combined with extensive conversion of available land to agricultural monocultures, this ecoregion currently exists as a highly fragmented landscape for forest ecosystems, effectively impeding the natural migration of tree species. Model simulations indicate that factors such as increasing summer temperatures and dryness, coupled with inadequate fire suppression, could lead to loss of ecosystem function and transition to grasslands or woodland/savanna even under low emissions scenarios (Lenihan et al. 2008).

Human communities are an integral part of the landscape in the Midwest and have greatly shaped current forests and prairie-forest boundaries (Abrams 1992; Mladenoff and Pastor 1993). Contemporary land use and ownership patterns provide critical input to policy responses to ecological issues, including climate change. In the Midwest, 68 % of forests are in private ownership (Butler 2008; Nelson et al. 2010). Stewardship of private lands reflects diverse values and motivations (Bengston et al. 2011), providing a challenge to effective outreach (Kittredge 2004). Likewise, a coordinated response to forest ecosystem threats is further challenged by parcelization (DeCoster 1998; Mehmood and Zhang 2001). Fostering climate preparedness as a component of sustainable land stewardship will require significantly increased outreach and coordination to communicate relevant and credible information to private forest landowners. Conversely, inadequate attention to land stewardship will place this forest sector at greater risk of avoidable impacts of climate change.

6.8 Northeast

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Climate is a key regulator of terrestrial biogeochemical processes. A recent synthesis of climate-change effects on forests of the Northeast region concluded that changes in climate that are already underway will result in changes in forest species composition, length of growing season, and forest hydrology, which together exert significant controls on forest productivity and sustainability (Rustad et al. 2009). Since 1900, mean annual temperature in the region has risen by an average of 0.8 °C, precipitation has increased by 100 mm, the onset of spring (based on phenologic indicators) has advanced by approximately 4 days, streamflows have generally increased, and dates of river and lake ice melt have advanced by 1–2 weeks (Huntington et al. 2009). Projections for the twenty-first century suggest that temperature will increase by 2.9–5.3 °C, precipitation will increase by 7–14 % (with minimal change in summer precipitation), the onset of spring will advance by 10–14 days, riverflows will increase during winter and spring but decrease in summer because of increased frequency of short-term droughts, and winter ice and snow will diminish. Variability and intensity of weather are also expected to increase, with more precipitation during large events with longer intervening dry spells, and more frequent and severe extreme events.

Forests cover large areas of the land surface in the northeastern United States, from 59 % in Rhode Island to 89 % in Maine (National Land Cover Database 2001). These forests are currently dominated by (1) southern hardwoods (oak, hickory) and pines in the southernmost region; (2) northern hardwoods (American beech, paper birch, and yellow birch [*Betula alleghaniensis* Britt.], sugar maple, and red maple [*Acer rubrum* L.]) in the central part and at lower elevations throughout; and (3) boreal-conifer forests in the north and at higher elevations (red spruce [*Picea rubens* Sarg.], black spruce, and balsam fir. eastern hemlock (*Tsuga canadensis* [L.] Carrière), an important shade tolerant, late successional species, is found throughout the Northeast.

Paleoecological data reveal a strong climate signal in current species assemblages and show that tree species have shifted in response to a gradually changing climate over the past 12,000 years since deglaciation. Projecting how the distribution and abundance of species will shift in the future in response to climate change is com-

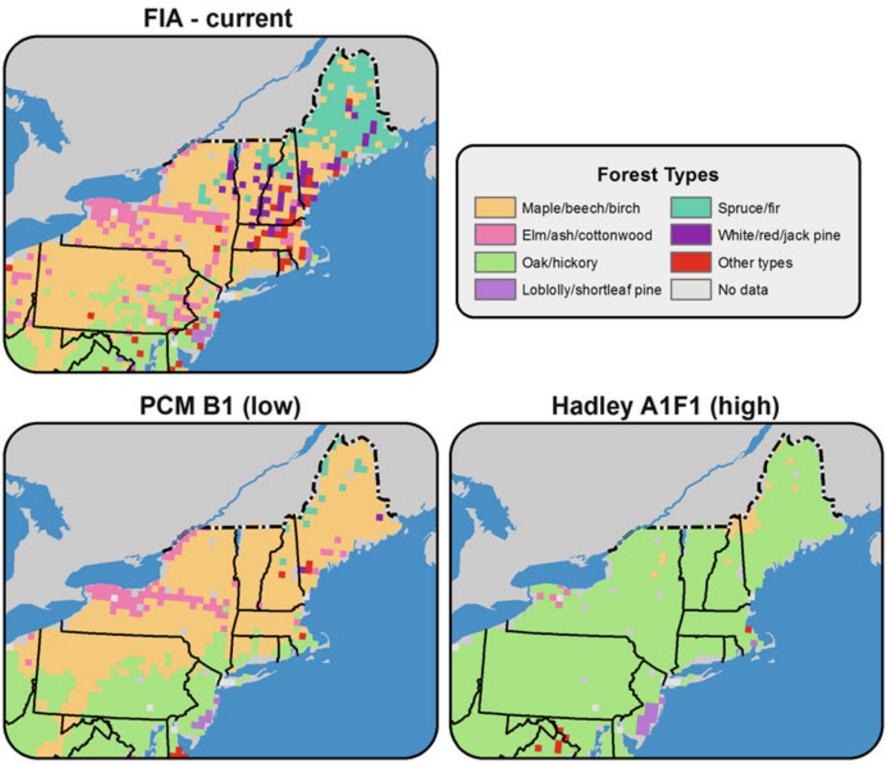

Fig. 6.6 Suitable habitat for forest vegetation in New England is expected to shift with changes in climate (year 2100) associated with different emissions scenarios (From Mohan et al. 2009, with permission)

plicated by the longevity of current individuals in the existing forest, robustness of the genetic pool to accommodate adaptation to new climatic conditions, limitations on regeneration and dispersal, and interactions with factors such as elevated nitrogen (N) deposition, elevated tropospheric ozone, land use change, habitat fragmentation, and changes in disturbance regimes caused by invasive species, pathogens, and fire.

In lieu of projecting future forest composition, some researchers have used "climatic envelopes," which combine information on current species distributions with climatic projections for the future, based on an ensemble of earth system models and emissions scenarios, to generate maps of "suitable habitat" for individual species and assemblages of species as forest types. For example, Iverson et al. (2008b) projected that a warming climate will result in a large contraction of suitable habitat for spruce-fir forest, moderate decline in suitable habitat for the maple-birch-beech forest, and expansion of suitable habitat for oak-dominated forest (Fig. 6.6). Projections of change in suitable habitat for individual tree species indicate that, of the 84 most common species, 23–33 will lose suitable habitat, 48–50 will gain habitat, and 1–10 will experience no change. The tree species predicted to have the most affected habitat include balsam fir, quaking aspen, paper birch (80–87 % decrease in suitable habitat), and black and white oak (greater than 100 % increase in suitable habitat) (Iverson et al. 2008b).

As climate and species composition change, so will forest productivity and C sequestration. More favorable climatic conditions for growth, particularly longer growing seasons, are correlated with higher productivity, whereas climatic extremes such as droughts, extreme cold or heat, and windstorms have been linked with tree diebacks and periods of lower productivity (Mohan et al. 2009). At Hubbard Brook Experimental Forest (New Hampshire), green canopy duration increased by 10 days over a 47-year period for a northern hardwood forest, suggesting a future longer period for growth and higher productivity (Richardson et al. 2006).

Model projections indicate that forest productivity for hardwood species is likely to be enhanced by future warmer temperatures, longer growing seasons, and increased concentrations of atmospheric CO_2. For example, Ollinger et al. (2008) used the model PnET-CN to project that net primary productivity in deciduous forests would increase by 52–250 % by 2100, depending on the global climate model and emission scenario used. The same model projected that current-day spruce forests are likely to show a climate-driven decrease in productivity along with a contraction of range. The effects of changing tree species assemblages and concurrent stress associated with forest fragmentation, atmospheric pollution, and invasive plant and animal species complicate these projections.

Changes in climate, hydrology, and forest tree species composition will have cascading effects on associated biogeochemical processes. Warmer temperatures and extended growing seasons will probably increase rates of microbial decomposition, N mineralization, nitrification, and denitrification, which will provide increased short-term availability of nutrients such as calcium, magnesium, and N for forest growth, as well as the potential for elevated losses of these same nutrients to surface waters (Campbell et al. 2009). Forests may respond to climate change with significant increases in nitrate leaching from soils to surface waters, with consequences for downstream water quality and eutrophication (Campbell et al. 2009). Potential accelerated loss of calcium and magnesium, especially from areas that have already experienced loss of these nutrients owing to decades of acidic deposition, may increase soil acidification. Warmer temperatures will also probably increase rates of root and microbial respiration, with an increased release of CO_2 from the soil to the atmosphere (Rustad et al. 2000).

Climate change will affect the distribution and abundance of many wildlife species in the region through changes in habitat, food availability, thermal tolerances, species interactions such as competition, and susceptibility to parasites and pathogens (Rodenhouse et al. 2009). Decades of survey data show that migratory birds are arriving earlier and breeding later in response to recent warming, with consequences for the annual production of young and survival (Rodenhouse et al. 2009). Among 25 species of resident birds studied, 15 are increasing in abundance, which is consistent with the observation that ranges of these species are limited by winter climate. Of the remaining species, 5 are decreasing in abundance and 5 show no change. Significant range expansions have also been observed, with 27 of 38

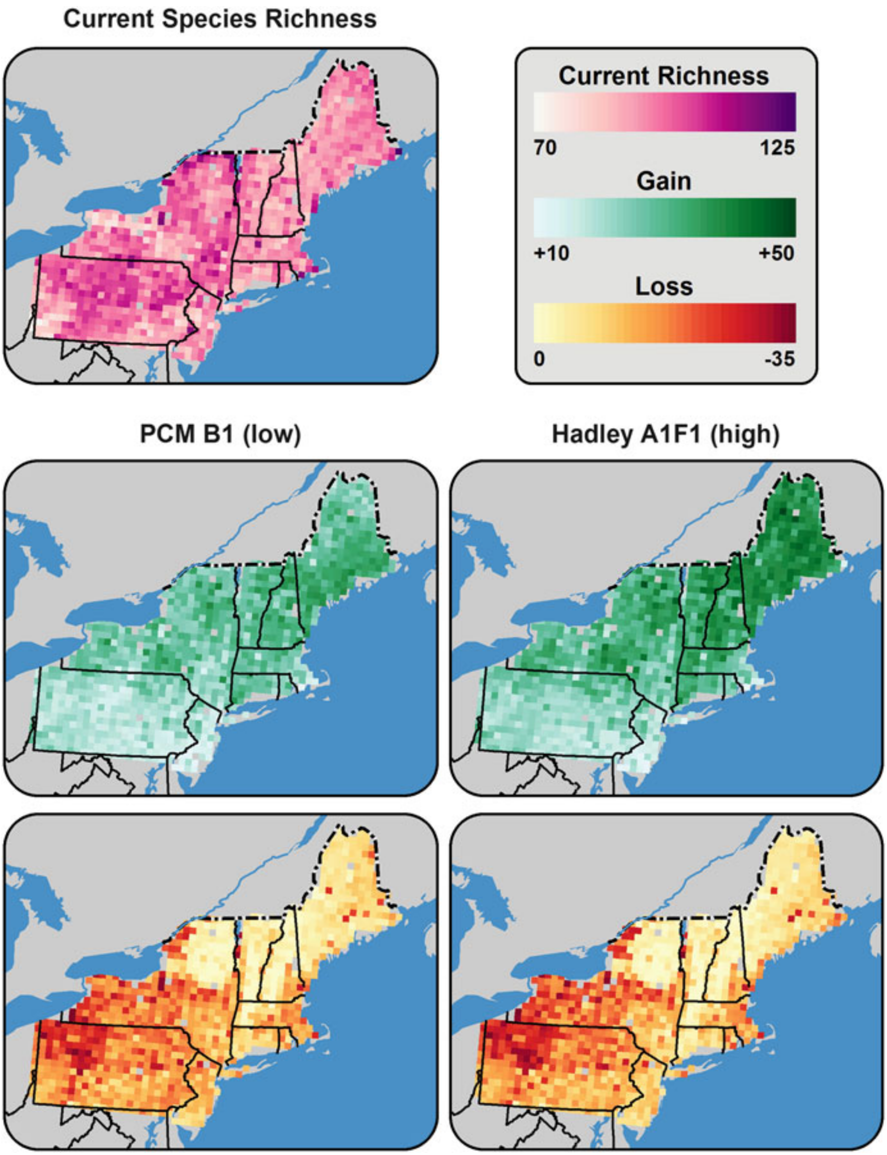

Fig. 6.7 Climate change (year 2100) is expected to affect bird species richness more intensely in some areas of the northeastern United States than in others (From Rodenhouse et al. 2008, with permission)

species studied expanding their ranges in a northward direction (Fig. 6.7). Using a climatic envelope approach, Rodenhouse et al. (2009) projected that twice as many resident bird species are expected to increase in abundance as decrease; for migrants (which comprise more than 85 % of the avifauna), an equal number are expected to increase as decrease.

Climate-related historical and future projected changes in native and introduced insects deserve special mention because these species contribute heavily to disturbance in Northeastern forests, and some species are particularly adept at adjusting to changing climatic conditions. Direct effects of climate change are likely to include summer warming-induced acceleration of reproductive and development rates, winter warming-induced increase in the ability to overwinter, and moisture-related changes in survival and fecundity. If minimum winter temperature increases as projected, this may allow the northward migration of many unwanted species, such as the hemlock wooly adelgid (*Adelges tsugae* Annand) (Skinner et al. 2003). Based on recent projections, climatic warming could allow the adelgid to spread throughout the range of eastern hemlock, potentially altering forest composition, nutrient cycling, and surface water quality (Dukes et al. 2009).

An increase in extreme weather events may have a larger effect on natural and managed systems than the more gradual change in mean climatic conditions. Legacies of past extreme windstorms and ice storms are apparent across the forested landscape of the region. It is imperative for the scientific and land management communities to better understand and anticipate the future occurrence and effects of these extreme events on forest composition, productivity, biogeochemistry, and fauna.

The twentieth century climate of the northeastern United States changed more rapidly than at any time since the last glaciation, and this rate of change is expected to continue throughout the twenty-first century. The direct and indirect effects of climate change on Northeastern forests, individually and in combination with other stressors such as acidic deposition, N and mercury deposition, tropospheric ozone, and various land uses, have the potential to cause significant changes in ecosystem structure and function (see Chap. 4). Additional research on indirect and interacting effects of these changes on forest ecosystems will be especially valuable for understanding potential effects of climate change, and for developing adaptation options that will enhance the sustainability of the diverse forests of this region.

6.9 Southeast

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Forests of the southeastern United States are a complex mixture of private and public land, interspersed with rapidly urbanizing areas and agriculture. A long history of active forest management, including intensive management such as forest plantations, fertilization, and prescribed fires, has created stand conditions and management regimes that differ from other areas of the United States. For example, relative to forests of the western United States, smaller tracts of accessible forest land may be more amenable to management actions that can be used to mitigate

C emissions or help forests adapt to climate change. On the other hand, the large private ownership of relatively small forest land holdings makes it challenging to implement uniform or coordinated large-scale management activities.

The majority of U.S. wood and fiber is produced in the Southeast, but climate change could significantly alter productive capacity in the region (Wertin et al. 2010). Loblolly pine is the most important commercial species, and although current air temperature is near optimal for growth across much of its range, as temperature continues to increase, conditions for pine growth may begin to deteriorate (McNulty et al. 1998b). Even if regional forest productivity remains high, the center of forest productivity could shift farther north into North Carolina and Virginia, causing economic and social effects in areas gaining and losing timber industry jobs (Sohngen et al. 2001).

Carbon sequestration is an increasingly valued component of forest productivity, and a large portion of the C stored in U.S. forests occurs in the Southeast (Pan et al. 2011). In addition to potentially reducing forest productivity (and therefore C uptake), climate change could increase decomposition of soil organic matter and CO_2 release (Boddy 1983). When added to the potential for increased wildfires, the potential for ecosystem C sequestration may decrease in the future, and the ecosystem value of sequestered forest C may shift from the southern to northern United States (Hurteau et al. 2008).

Wildfires, hurricanes, drought, insect outbreaks, and pathogen outbreaks have been a driving force for millennia in Southeastern forests. However, during the past two centuries, the type and magnitude of ecosystem stress and disturbance have changed and will likely continue to change as the climate warms (Dale et al. 2001). Wind and extreme precipitation events associated with hurricanes can have major effects on Southeastern forests. A single hurricane can reduce total forest C sequestration by 10 % in the year in which it occurs (McNulty 2002), although not all forest species are equally susceptible to wind damage. Longleaf pine (*Pinus palustris* Mill.) shows less damage than does loblolly pine (*P. taeda* L.) when exposed to an equal level of wind stress (Johnsen et al. 2009), suggesting that the former species would be more resistant to an increase in windstorms. Extreme precipitation events that accompany hurricanes can cause extended submersion of low-lying forests, which can kill tree roots by causing anaerobic soil conditions (Whitlow and Harris 1979).

Wildfires are a natural component of ecosystem maintenance and renewal in the Southeast, which has more area burn annually, with wildfire and prescribed fire, than any other region of the United States (except Alaska in some years) (Andreu and Hermansen-Baez 2008). However, decades of fire exclusion coupled with increasing air temperatures have increased the potential for crown fire in some Southeastern forests. Future fire potential is expected to increase from low to moderate in summer and autumn in eastern sections in the South, and from moderate to high in western portions (Liu et al. 2010). As fire seasons lengthen in the future, the window for prescribed burning may decrease because of increased fuel flammability, thus potentially affecting the management of fuels and C dynamics; fuel treatments with prescribed fire emit 20 % less CO_2 than wildfires, at least in the short term

(Wiedinmyer and Hurteau 2010). Historically, longleaf pine was a dominant species across the region. It is well adapted to drought, with thick bark and fast seedling growth, allowing it to thrive in habitats subjected to periodic wildfire (Brockway and Lewis 1997). Most of the longleaf pine was cut during the twentieth century, followed by replanting with the faster growing loblolly pine, which is preferred by the timber industry but is less resistant to wildfire. Land managers are reassessing the preferential use of loblolly pine, because longleaf pine would be more resistant to the increased fire, drought, and wind expected with climate change.

Insect and pathogen outbreaks are increasing in Southeastern forests (Pye et al. 2011), potentially threatening the long-term productivity and structure of forest ecosystems. Higher temperature has caused a longer growing season of at least 2 weeks compared to historical lengths, allowing additional time for insects and pathogens to find and colonize susceptible trees (Ayres and Lombardero 2000). In addition, timing of the predator–prey cycle may be changing. For example, when the growing season begins earlier, insects may be hatching and maturing before migratory insectivorous bird species return, allowing more insects to reach maturity, speed up the reproductive cycle, and locate susceptible host trees. Finally, higher temperature and subsequent soil drying increases stress in trees, reducing their physiological capacity to resist attack (McNulty et al. 1998a).

Some aspects of the high biodiversity in the Southeast may be susceptible to climate change (Thompson et al. 2009), particularly species that are near the environmental limit of their range. Red spruce and eastern hemlock are well adapted to the cool climates of the last glacial age. However, the extent and dominance of these two species have decreased greatly as a result of stress complexes that include warmer temperature, air pollution, and insects (McNulty and Boggs 2010; Elliott and Vose 2011). With further warming, red spruce and eastern hemlock are projected to be extirpated from the southern United States before 2100 (Prasad et al. 2007), and small populations of balsam fir will also be at risk. Altered tree species dominance will affect birds and other terrestrial vertebrate species that depend on forest habitat.

Cold water fish species, which are generally confined to northern and mountainous areas of the Southeast where cooler water (and air) temperatures allow dissolved oxygen contents to remain at sufficient levels, will likely face increased stress from higher temperature at the southern limit of their range. In addition, rainfall intensity has been increasing for over a century (Karl et al. 1995), which can in turn increase soil erosion and stream turbidity (Trimble 2008). A combination of higher air temperature and lower water quality may significantly reduce trout abundance across the Southeast during the coming decades (Flebbe et al. 2006).

Abundant, year-round rainfall has historically provided a sufficient supply of water for industrial, commercial, residential, agricultural, and hydro-electric use in the Southeast, but several factors may contribute to a shift in water abundance. The population of the Southeast is increasing and much ofthis increase is centered in

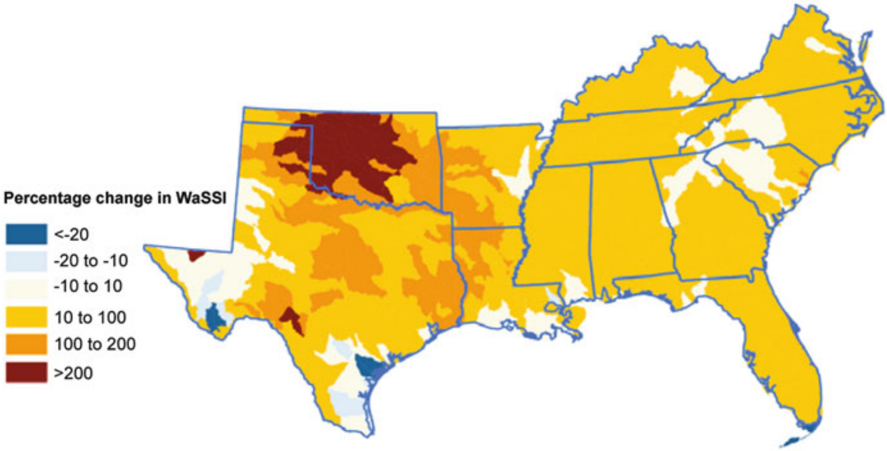

Fig. 6.8 Percentage change in water supply stress owing to climate change, as defined by the water supply stress index (*WaSSI*) for 2050 using the CSIROMK2 B2 climate scenario. WaSSI is calculated by dividing water demand by supply, where higher values indicate higher stress on watersheds and water systems. From Lockaby et al. (2011)

metropolitan areas, whereas much of the water originates in forested headwaters, often long distances from urban areas. On an annual basis, average water supply is approximately 20 times higher than demand, although short-term (1–3 years) drought can significantly increase pressure on available water (Lockaby et al. 2011) (Fig. 6.8). A combination of increased population, changing land-use patterns, and shifts in rainfall patterns could further amplify water shortages, and even if precipitation rates remain unchanged, higher tree water use in response to higher temperature, or shifting management regimes for new products such as biofuels, could contribute to water shortfalls (Sun et al. 2008; Lockaby et al. 2011). Seasonal timing of precipitation within the year could also affect water supply. If precipitation occurs in fewer, more intense events, then proportionally less water will be retained by forest ecosystems, and more will be lost as runoff, potentially causing flooding, soil erosion, and stream sedimentation (Trimble 2008).

The Southeast has diverse year-round recreational opportunities, some of which could be severely affected by climate change. Ski areas in the region are marginally profitable, and increased winter warming may increase the proportion of rain to snow and prevent snow making (Millsaps and Groothuis 2003). Reduced quality or quantity of the ski season could force most of the marginal ski areas to close. Similarly, cold water fisheries are a major recreational attraction, and revenues from lodging, food, and secondary activities are a major economic boost to local mountain economies. Therefore, extirpation of trout from these areas could significantly harm the recreation industry.

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