

Appendix 1: Regional Summaries

Alaska

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Introduction

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 percent of the world's boreal forests, and are located throughout interior and south-central Alaska (fig. A1-1). The remaining 10 percent of Alaskan forests are classified as coastal-temperate, representing 19 percent of the world's coastal-temperate forests (National Synthesis Assessment Team 2003), and are located in southeast Alaska (fig. A1-1). Regional changes in the disturbance regimes of Alaskan forests (Wolken et al. 2011) directly affect the global climate system through greenhouse gas 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).

Interior Alaska

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

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Figure A1-1—The boreal (interior and south-central) and coastal-temperate (southeast) forest regions of Alaska. The boreal forest ecoregions include the Alaska Range transition, coastal mountains transition, Pacific mountains transition, Kenai boreal, and intermontane boreal; the coastal-temperate forest includes the coastal rain forests ecoregion (Nowacki et al. 2001).

the organic soil layer by wildfire (fig. A1-2). 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). The observed warmer air and permafrost temperatures have important societal impacts, because transportation, water and sewer, and other public infrastructures may be damaged (Larsen et al. 2008, Nelson et al. 2002).

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 (fig. A1-2) also occurring during this time (Kasischke et al. 2010). Black spruce forests, the dominant forest type in the interior, historically burned in low-severity, stand-replacing



State of Alaska, Division of Forestry.

Figure A1-2—In 2004, Alaska's largest wildfire season on record, the Boundary Fire, burned 217 000 ha of forest in interior Alaska.

fires every 70 to 130 years (Johnstone et al. 2010a). However, postfire succession of black spruce (*Picea mariana* [Mill.] Britton, Sterns & Poggenb.) forests has recently shifted toward deciduous-dominated forests with the increase in wildfire severity (Johnstone and Chapin 2006, Johnstone and Kasischke 2005, Kasischke and Johnstone 2005) and the reduction in fire-return interval (Bernhardt et al. 2011; Johnstone et al. 2010a, 2010b). 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

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 aggressively into

burned areas (e.g., Siberian peashrub [*Caragana arborescens* Lam.], narrowleaf hawksbeard [*Crepis tectorum* L.], and white sweetclover [*Melilotus alba* Medik.]) (Cortés-Burns et al. 2008, Lapina and Carlson 2004), and these could proliferate further with the increase in wildfire potential. Changes in surface hydrology in south-central Alaska have also been linked to warmer temperatures. In the Kenai lowlands, a subregion of south-central Alaska (fig. A1-1), 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-percent 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.

Southeast Alaska

In southeast Alaska, climatic warming has affected forest ecosystems primarily through effects on precipitation (i.e., snow versus rain). 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 (Larsen et al. 2005, Motyka et al. 2002). 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 southeast 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 (Pojar and MacKinnon 1994, Schroeder and Kookesh 1990). If cedar decline continues, it will alter the structure and function of forest ecosystems, as well as the lifeways of people in this region.

Literature Cited

- Beier, C.M.; Sink, S.E.; Hennon, P.E. [et al.]. 2008.** Twentieth-century warming and the dendroclimatology of declining yellow-cedar forests in southeastern Alaska. *Canadian Journal of Forest Research*. 38: 1319–1334.
- Berg, E.E.; Henry, J.D.; Fastie, C.L. [et al.]. 2006.** Spruce beetle outbreaks on the Kenai Peninsula, Alaska, and Kluane National Park and Reserve, Yukon Territory: relationship to summer temperatures and regional differences in disturbance regimes. *Forest Ecology and Management*. 227: 219–232.
- Berg, E.E.; Hillman, K.M.; Dial, R.; DeRuwe, A. 2009.** Recent woody invasion of wetlands on the Kenai Peninsula Lowlands, south-central Alaska: a major regime shift after 18,000 years of wet Sphagnum-sedge peat recruitment. *Canadian Journal of Forest Research*. 39: 2033–2046.
- Bernhardt, E.L.; Hollingsworth, T.N.; Chapin, F.S. 2011.** Fire mediates climate-driven shifts in understory community composition of black spruce stands of interior Alaska. *Journal of Vegetation Science*. 22: 32–44.
- Chapin, F.S.; McGuire, A.D.; Randerson, J. [et al.]. 2000.** Arctic and boreal ecosystems of western North America as components of the climate system. *Global Change Biology*. 6: 211–223.
- Chapin, F.S., III; Trainor, S.F.; Huntington, O. [et al.]. 2008.** Increasing wildfire in Alaska's boreal forest: pathways to the potential solutions of a wicked problem. *BioScience*. 58: 531–540.
- Cortés-Burns, H.; Lapina, I.; Klein, S.C. [et al.]. 2008.** Invasive plant species monitoring and control-areas impacted by 2004 and 2005 fires in interior Alaska: a survey of Alaska BLM lands along the Dalton, Steese, and Taylor highways. BLM-BAER final report. Anchorage, AK: Bureau of Land Management, Alaska State Office. 162 p.
- Flint, C.G. 2006.** Community perspectives on spruce beetle impacts on the Kenai Peninsula, Alaska. *Forest Ecology and Management*. 227: 207–218.

- Harris, S.A.; French, H.M.; Heginbottom, J.A. [et al.]. 1988.** Glossary of permafrost and related ground-ice terms. Tech. Memo. No. 142. Ottawa, ON, Canada: National Research of Council Canada, Associate Committee on Geotechnical Research, Permafrost Subcommittee. 156 p.
- Hennon, P.E.; D'Amore, D.; Wittwer, D. [et al.]. 2006.** Climate warming, reduced snow, and freezing injury could explain the demise of yellow-cedar in southeast Alaska, USA. *World Resource Review*. 18: 427–450.
- Hennon, P.E.; Shaw, C.G.; Hansen, E.M. 1990.** Dating decline and mortality of *Chamaecyparis nootkatensis* in southeast Alaska. *Forest Science*. 36: 502–515.
- Johnstone, J.F.; Chapin, F.S. 2006.** Effects of soil burn severity on post-fire tree recruitment in boreal forest. *Ecosystems*. 9: 14–31.
- Johnstone, J.F.; Chapin, F.S.; Hollingsworth, T.N. [et al.]. 2010a.** Fire, climate change, and forest resilience in interior Alaska. *Canadian Journal of Forest Research*. 40: 1302–1312.
- Johnstone, J.F.; Hollingsworth, T.N.; Chapin, F.S.; Mack, M.C. 2010b.** Changes in fire regime break the legacy lock on successional trajectories in Alaskan boreal forest. *Global Change Biology*. 16: 1281–1295.
- Johnstone, J.F.; Kasischke, E.S. 2005.** Stand-level effects of soil burn severity on post-fire regeneration in a recently-burned black spruce forest. *Canadian Journal of Forest Research*. 35: 2151–2163.
- Kasischke, E.S.; Johnstone, J.F. 2005.** Variation in ground-layer surface fuel consumption and its effects on site characteristics in a *Picea mariana* forest complex in interior Alaska. *Canadian Journal of Forest Research*. 35: 2164–2177.
- Kasischke, E.S.; Verbyla, D.L.; Rupp, T.S. [et al.]. 2010.** Alaska's changing fire regime—implications for the vulnerability of its boreal forests. *Canadian Journal of Forest Research*. 40: 1313–1324.
- Klein, E.; Berg, E.E.; Dial, R. 2005.** Wetland drying and succession across the Kenai Peninsula lowlands, south-central Alaska. *Canadian Journal of Forest Research*. 35: 1931–1941.
- Lapina, I.; Carlson, M.L. 2004.** Non-native plant species of Susitna, Matanuska, and Copper River basins: summary of survey findings and recommendations for control actions. Final report. Anchorage, AK: U.S. Department of Agriculture, Forest Service, State and Private Forestry. 64 p.
- Larsen, C.F.; Motyka, R.J.; Freymueller, J.T. [et al.]. 2005.** Rapid viscoelastic uplift in southeast Alaska caused by post-Little Ice Age glacial retreat. *Earth and Planetary Science Letters*. 237: 548–560.
- Larsen, P.H.; Goldsmith, S.; Smith, O. [et al.]. 2008.** Estimating future costs of Alaska public infrastructure at risk from climate change. *Global Environmental Change*. 18: 422–457.
- Motyka, R.J.; O'Neel, S.; Conner, C.L.; Echelmeyer, K.A. 2002.** Twentieth century thinning of Mendenhall Glacier, Alaska, and its relationship to climate, lake calving, and glacier run-off. *Global and Planetary Change*. 35: 93–112.
- National Synthesis Assessment Team. 2003.** U.S. national assessment of the potential consequences of climate variability and change: educational resources: regional paper: Alaska. <http://www.usgcrp.gov/usgcrp/nacc/education/alaska/default.htm>. (3 September 2010).
- Nelson, F.E.; Anisimov, O.A.; Shiklomanov, N.I. 2002.** Climate change and hazard zonation in the circum-Arctic permafrost regions. *Natural Hazards*. 26: 203–225.
- Nowacki, G.J.; Spencer, P.; Fleming, M. [et al.]. 2001.** Unified ecoregions of Alaska [Ecoregion map]. Open-File Report 02-297. Anchorage, AK: U.S. Department of the Interior, Geological Survey.

- Parson, E.A.; Carter, L.; Anderson, P. [et al.]. 2001.** Potential consequences of climate variability and change for Alaska. In: National Assessment Synthesis Team, eds. *Climate change impacts on the United States—foundation report*. Cambridge, United Kingdom: Cambridge University Press: 283–312.
- Pojar, J.; MacKinnon, A. 1994.** *Plants of the Pacific Northwest coast: Washington, Oregon, British Columbia, and Alaska*. Redmond, WA: Lone Pine Publishing. 528 p.
- Randerson, J.T.; Liu, H.; Flanner, M.G. [et al.]. 2006.** The impact of boreal forest fire on climate warming. *Science*. 314: 1130–1132.
- Reid, W.V.; Mooney, H.A.; Cropper, A. [et al.]. 2005.** *Ecosystems and human well-being: synthesis*. Washington, DC: Island Press. 155 p.
- Robertson, G.; Brooks, D. 2001.** Assessment of the competitive position of the forest products sector in southeast Alaska, 1985–94. Gen. Tech. Rep. PNW-GTR-504. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 29 p.
- Schroeder, R.F.; Kookesh, M. 1990.** Subsistence harvest and use of fish and wildlife resources and the effects of forest management in Hoonah, Alaska. Tech. Paper 142. Juneau, AK: Alaska Department of Fish and Game, Division of Subsistence. 326 p.
- Schuur, E.A.G.; Bockheim, J.; Canadell, J.G. [et al.]. 2008.** Vulnerability of permafrost carbon to climate change: implications for the global carbon cycle. *Bioscience*. 58: 701–714.
- Schuur, E.A.G.; Crummer, K.G.; Vogel, J.G.; Mack, M.C. 2007.** Plant species composition and productivity following permafrost thaw and thermokarst in Alaskan tundra. *Ecosystems*. 10: 280–292.
- Smith, L.C.; Sheng, Y.; MacDonald, G.M.; Hinzman, L.D. 2005.** Disappearing arctic lakes. *Science*. 308: 1429–1429.
- Tan, Z.; Tieszen, L.L.; Zhu, Z. [et al.]. 2007.** An estimate of carbon emissions from 2004 wildfires across Alaskan Yukon River Basin. *Carbon Balance and Management*. 2: 12. [doi:10.1186/1750-0680-2-12].
- Vogel, J.; Schuur, E.A.G.; Trucco, C.; Lee, H. 2009.** Response of CO₂ exchange in a tussock tundra ecosystem to permafrost thaw and thermokarst development. *Journal of Geophysical Research*. 114:G04018 [doi:10.1029/2008JG000901].
- Werner, R.A.; Holsten, E.H.; Matsuoka, S.M.; Burnside, R.E. 2006.** Spruce beetles and forest ecosystems in south-central Alaska: a review of 30 years of research. *Forest Ecology and Management*. 227: 195–206.
- Wolken, J.M.; Hollingsworth, T.N.; Rupp, T.S. [et al.]. 2011.** Evidence and implications of recent and projected climate change in Alaska’s forest ecosystems. *Ecosphere*. 2(11) art124 doi: <http://dx.doi.org/10.1890/ES11-00288.1>.

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 Mariana Islands, Federated States of Micronesia, Republic of Palau, and the Marshall Islands (fig. A1-3), 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 decisionmakers, state-level managers, and landowners (Barnett 2001, Mimura et al. 2007).

The distribution and persistence of different forest species are largely determined by temperature and precipitation and for coastal forests, sea level rise. Based on known historical climate-vegetation relationships, many forests are expected to experience significant changes in distribution and abundance by the end of the 21st 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). Streamflow decreased by 10 percent during the period 1973 to 2002 compared to 1913 to 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). This appears to be accurate for the central and western Pacific (Mimura et al. 2007), and at least for Hawaii, climatic forecasting suggests that this pattern will be more pronounced in drier areas of the state.

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. A1-4), 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). Combined with warmer and drier conditions, 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 streamflows 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).

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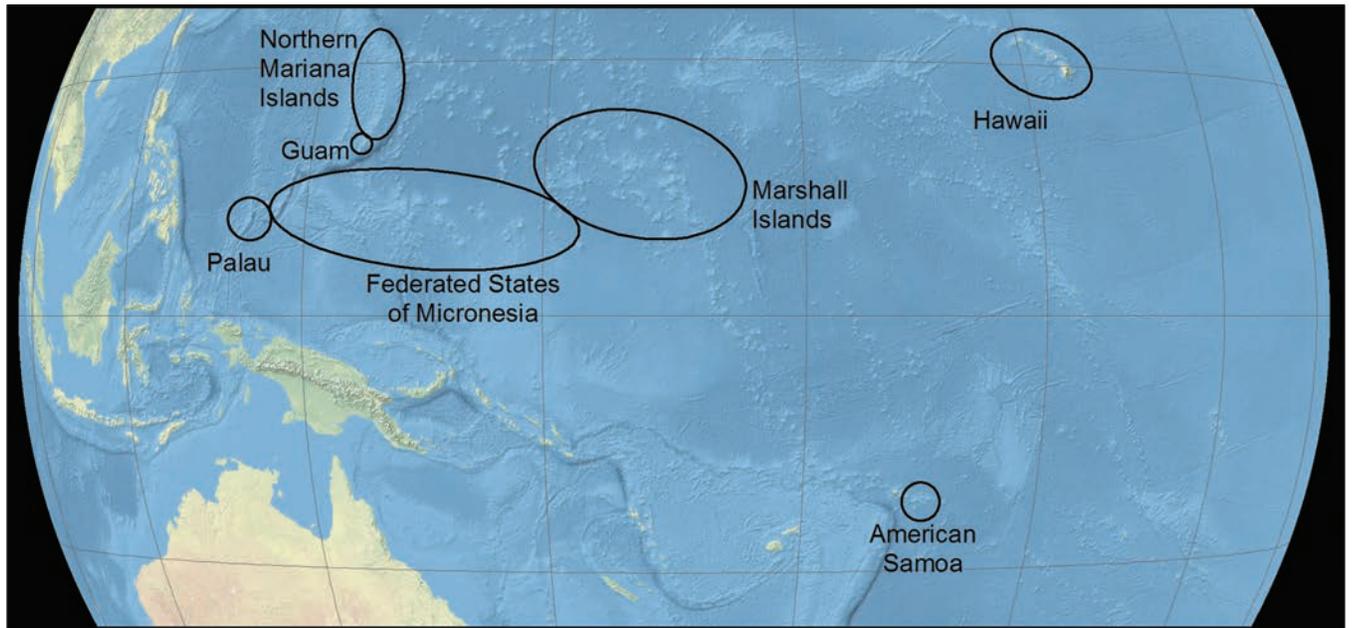


Figure A1-3—Hawaii and the U.S.-affiliated Pacific islands.



Figure A1-4—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.

Although Hawaii's Mauna Loa Observatory has been documenting the steady rise in atmospheric carbon dioxide (CO_2), the direct effects of elevated CO_2 in forests of the region have not been examined. However, most forests have at least some stimulatory effects from CO_2 (Norby et al. 2005), especially in younger, faster-growing species. Therefore, the effects of climate on fire regimes and streamflow 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 $2 \text{ mm} \cdot \text{yr}^{-1}$) (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 enhance 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.

Mimura et al. (2007) suggest high to moderately high confidence for anticipated diverse effects of climate change on island ecosystems (table A1-1). These effects 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 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). Climate-driven reductions in coastal forest area and functionality will increase population pressures on already limited natural resources, and the combination of inundation and enhanced storm damage will damage fragile economies (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 ground-water recharge as well as downstream agriculture, urban development, and tourism (Mimura et al. 2007).

Few options are available for managing climate-change effects on 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.

Table A1-1—Potential climate change related risks, and confidence in projections

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

Source: Mimura et al. 2007.

However, cost-effective prescriptions and examples of effective adaptation strategies are rare.

Several options for managing climate-change effects exist in Hawaii, because adequate financial resources and infrastructure are available. Hawaiian ecological relationships differ from those on other islands; for example, mangrove forests serve important shoreline conservation functions in the U.S.-affiliated Pacific islands, but mangrove species are not native to Hawaii and are considered problematic invasives. Land ownership in Hawaii is complex, requiring management for shoreline stabilization to rely on diverse native plant species and institutional partnerships. Because Hawaii has significant topographic relief, as well as moderately sophisticated management infrastructure, anticipatory planning and facilitation of inward species migration is already being practiced in some coastal wetlands.

For the majority of Hawaii's forest systems, sea level rise and storm surges are minor threats. Rather, key threats to native forest plant biodiversity include climate-driven acceleration of invasive species, resulting in displacement of native vegetation and in novel fire disturbance. This creates the potential for long-term conversion of native forests to grass and shrub lands dominated by invasive species. 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).

The region has lacked resources and expertise for conducting the research required to comprehensively manage climate change threats; research needs are particularly acute for the U.S.-affiliated Pacific islands. Throughout the region, research is needed to identify the 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 these thresholds (Barnett and Adger 2003). Stress complexes in forest systems affect thresholds; especially important are interactions among invasive species, altered fire regimes, insects, and pathogens. Silvicultural research is needed to understand how to treat extensive forest areas for invasive species that appear to use more water than native systems. Effective biocontrol agents are also needed to reduce the most damaging invasive species affecting regional forests. Expanded research in fire science (fire history, fire behavior, fuel characterization) would improve fuel maps and understanding about fire ecology and human dimensions of wildfire. Conservation genetics research would improve understanding of genotypic plasticity and diversity within species, restoration needs and adaptation potential, pathogen resistance in a changing climate, and locally relevant restoration practices that use genotypes and species suitable for future climate.

Literature Cited

- Asner, G.P.; Elmore, A.J.; Hughes, R.F. [et al.]. 2005.** Ecosystem structure along bioclimatic gradients in Hawai'i from imaging spectroscopy. *Remote Sensing of the Environment*. 96: 497–508.
- Asner, G.P.; Hughes, R.F.; Vitousek, P.M. [et al.]. 2008.** Invasive plants transform the three-dimensional structure of rain forests. *Proceedings of the National Academy of Sciences, USA*. 105: 4519–4523.
- Barnett, J. 2001.** Adapting to climate change in Pacific island countries: the problem of uncertainty. *World Development*. 29: 977–993.
- Barnett, J.; Adger, W. 2003.** Climate dangers and atoll countries. *Climatic Change*. 61: 321–337.
- Benning, T.L.; LaPointe, D.; Atkinson, C.T.; Vitousek, P.M. 2002.** Interactions of climate change with biological invasions and land use in the Hawaiian Islands: modeling the fate of endemic birds using a geographic information system. *Proceedings of the National Academy of Sciences, USA*. 99: 14246–14249.
- Cavaleri, M.A.; Sack, L. 2010.** Comparative water use of native and invasive plants at multiple scales: a global meta-analysis. *Ecology*. 91: 2705–2715.
- Chu, P.-S.; Chen, H. 2005.** Interannual and interdecadal rainfall variations in the Hawaiian Islands. *Journal of Climate*. 18: 4796–4813.
- Chu, P.-S.; Chen, Y.R.; Schroeder, T.A. 2010.** Changes in precipitation extremes in the Hawaiian Islands in a warming climate. *Journal of Climate*. 23: 4881–4900.
- D'Antonio, C.M.; Vitousek, P.M. 1992.** Biological invasions by exotic grasses, the grass/fire cycle, and global change. *Annual Review of Ecology and Systematics*. 23: 63–87.
- Giambelluca, T.W.; Diaz, H.F.; Luke, S.A. 2008a.** Secular temperature changes in Hawai'i. *Geophysical Research Letters*. 35, L12702, doi:10.1029/2008GL034377.
- Giambelluca, T.; Delay, J.; Asner, G. [et al.]. 2008b.** Stand structural controls on evapotranspiration in native and invaded tropical montane cloud forest in Hawai'i. [Abstract]. <http://adsabs.harvard.edu/abs/2008AGUFM.B43A0422G>. (11 December 2011).
- Mimura, N. 1999.** Vulnerability of island countries in the South Pacific to sea level rise and climate change. *Climate Research*. 12: 137–143.
- Mimura, N.; Nurse, L.; McLean, R.F. [et al.]. 2007.** Small islands. In: Parry, M.L.; Canziani, O.F.; Palutikof, J.P., eds. *Climate change 2007: impacts, adaptation and vulnerability. Contribution of Working Group II to the fourth assessment report of the Intergovernmental Panel on Climate Change*. Cambridge, United Kingdom: Cambridge University Press: 687–716.
- Norby, R.; DeLucia, E.; Gielen, B. [et al.]. 2005.** Forest response to elevated CO₂ is conserved across a broad range of productivity. *Proceedings of the National Academy of Sciences, USA*. 102: 18052–18056.
- Norton, C.W.; Chu, P.S.; Schroeder, T.A. 2011.** Projecting changes in future heavy rainfall events for Oahu, Hawaii: a statistical downscaling approach. [Abstract]. *Journal of Geophysical Research*. 116: D17110.
- Oki, D.S. 2004.** Trends in streamflow characteristics at long-term gaging stations, Hawaii. *Scientific Investigations Rep. 2004–5080*. Denver, CO: U.S. Geological Survey. 120 p.
- Safeeq, M.; Fares, A. 2011.** Hydrologic response of a Hawaiian watershed to future climate change scenarios. *Hydrological Processes*. doi: 10.1002/hyp.8328.

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 PNW and West-wide modeling studies provide additional scenarios for effects of climate change on wildfire, insects, and dynamic vegetation in Oregon and Washington. 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 (e.g., McKenzie et al. 2003) show that each tree species has a unique relationship with limiting climatic factors (McKenney et al. 2011; McKenzie et al. 2003; Rehfeldt et al. 2006, 2008). These relationships have been used to project future climate suitability for species in western North America (McKenney et al. 2007, 2011; Rehfeldt et al. 2006, 2009) 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 percent of its current range in Washington, and up to 85 percent of the range of some pine species may be outside the current climatically suitable range (Littell et al. 2010, Rehfeldt 2006). 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; five

of these species would lose less than 20 percent of this range, and the range of the other 10 species would decline up to 70 percent.

Various modeling studies project significant changes in species distribution in the Northwest, but with considerable variation within and between those studies. McKenney et al. (2011) summarized responses of tree species to climate change across western North America for three emissions 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 21st 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 carbon dioxide, 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, 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 (Littell et al. 2009, 2010; McKenzie et al. 2004). 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 to 2000, large increases are predicted by

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2100 in both area burned (76 to 310 percent, depending on climate and fire suppression scenario) and burn severities (29 to 41 percent).

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 21st 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 PNW, Bentz et al. (2010) simulated changes in adaptive seasonality for MPB and single-year offspring survival for spruce bark beetle (*Dendroctonus 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 21st century and in the north Cascades and central Idaho for the late 21st century. Single-year development of SBB offspring also increases at high elevations across the region in both the early and late 21st 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., Chen et al. 2009, Littell et al 2008, Peterson

and Peterson 2001). Douglas-fir is expected to grow more slowly in much of the drier part of its range (Chen et al. 2009) 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. In addition, projected changes in forest distribution, structure, and function need to be synthesized using recent GCM projections, including quantification of uncertainties about the effects of climate on forest processes.

Literature Cited

- Bentz, B.J.; Régnière, J.; Fettig, C.J. [et al.]. 2010.** Climate change and bark beetles of the Western United States and Canada: direct and indirect effects. *BioScience*. 60: 602–613.
- Berg, E.; Davidhenry, J.; Fastie, C. [et al.]. 2006.** Spruce beetle outbreaks on the Kenai Peninsula, Alaska, and Kluane National Park and Reserve, Yukon Territory: relationship to summer temperatures and regional differences in disturbance regimes. *Forest Ecology and Management*. 227: 219–232.
- Case, M.J.; Peterson, D.L. 2005.** Fine-scale variability in growth-climate relationships of Douglas-fir, North Cascade Range, Washington. *Canadian Journal of Forest Research*. 35: 2743–2755.
- Chen, P.-Y.; Welsh, C.; Hamann, A. 2010.** Geographic variation in growth response of Douglas-fir to interannual climate variability and projected climate change. *Global Change Biology*. 16: 3374–3385.
- Coops, N.C.; Waring, R.H. 2010.** A process-based approach to estimate lodgepole pine (*Pinus contorta* Dougl.) distribution in the Pacific Northwest under climate change. *Climatic Change*. 105: 313–328.
- Coops, N.C.; Waring, R.H. 2011.** Estimating the vulnerability of fifteen tree species under changing climate in Northwest North America. *Ecological Modelling*. 222: 2119–2129.
- Holman, M.L.; Peterson, D.L. 2006.** Spatial and temporal variability in forest growth in the Olympic Mountains, Washington: Sensitivity to climatic variability. *Canadian Journal of Forest Research*. 36: 92–104.
- Lenihan, J.; Bachelet, D.; Neilson, R.; Drapek, R. 2008.** Simulated response of conterminous United States ecosystems to climate change at different levels of fire suppression, CO₂ emission rate, and growth response to CO₂. *Global and Planetary Change*. 64: 16–25.
- Littell, J.S.; McKenzie, D.; Peterson, D.L.; Westerling, A.L. 2009.** Climate and wildfire area burned in western U.S. ecoprovinces, 1916–2003. *Ecological Applications*. 19: 1003–1021.
- Littell, J.S.; Oneil, E.E.; McKenzie, D. [et al.]. 2010.** Forest ecosystems, disturbance, and climatic change in Washington State, USA. *Climatic Change*. 102: 129–158.
- Littell, J.S.; Peterson, D.L.; Tjoelker, M. 2008.** Douglas-fir growth in mountain ecosystems: water limits tree growth from stand to region. *Ecological Monographs*. 78: 349–368.
- McKenney, D.W.; Pedlar, J.H.; Lawrence, K. [et al.]. 2007.** Potential impacts of climate change on the distribution of North American trees. *BioScience*. 57: 939–948.
- McKenney, D.W.; Pedlar, J.H.; Rood, R.B.; Price, D. 2011.** Revisiting projected shifts in the climate envelopes of North American trees using updated general circulation models. *Global Change Biology*. 17(8): 2720–2730.
- McKenzie, D.; Gedalof, Z.; Peterson, D.L.; Mote, P. 2004.** Climatic change, wildfire, and conservation. *Conservation Biology*. 18: 890–902.
- McKenzie, D.; Peterson, D.W.; Peterson, D.L.; Thornton, P.E. 2003.** Climatic and biophysical controls on conifer species distributions in mountain forests of Washington State, USA. *Journal of Biogeography*. 30: 1093–1108.
- Mote, P.W.; Salathé, E.P. 2010.** Future climate in the Pacific Northwest. *Climatic Change*. 102: 29–50.
- Peterson, D.W.; Peterson, D.L. 2001.** Mountain hemlock growth responds to climatic variability at annual and decadal time scales. *Ecology*. 82: 3330–3345.
- Powell, J.; Logan, J. 2005.** Insect seasonality: circle map analysis of temperature-driven life cycles. *Theoretical Population Biology*. 67: 161–179.

Rehfeldt, G.E.; Crookston, N.L.; Warwell, M.V.; Evans, J.S. 2006. Empirical analyses of plant-climate relationships for the western United States. *International Journal of Plant Sciences*. 167: 1123–1150.

Rehfeldt, G.E.; Ferguson, D.E.; Crookston, N.L. 2008. Quantifying the abundance of co-occurring conifers along Inland Northwest (USA) climate gradients. *Ecology*. 89: 2127–2139.

Rogers, B.M.; Neilson, R.P.; Drapek, R. [et al.]. 2011. Impacts of climate change on fire regimes and carbon stocks of the U.S. Pacific Northwest. *Journal of Geophysical Research*. 116: 1–13.

Shafer, S.L.; Harmon, M.E.; Neilson, R.P. [et al.]. 2010. The potential effects of climate change on Oregon's vegetation. In: Dello, K.D.; Mote, P.W., eds. *Oregon climate assessment report*. Corvallis, OR: Oregon State University, College of Oceanic and Atmospheric Sciences, Oregon Climate Change Research Institute: 175–210.

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 1 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.

Aridity dominates forest ecosystems in the Southwest, which encompass a wide range of topographic variability and Mediterranean, continental, and desert climates. Therefore, disturbance processes that are facilitated by climatic extremes, primarily multiyear droughts, dominate the potential effects of climatic variability and change on both short- and long-term forest dynamics (Allen and Breshears 1988). 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 to 18 percent 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 (Cayan et al. 2010, Weiss et al. 2009).

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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 percent; Colorado, 80 percent; New Mexico, 350 percent; and Utah, 300 percent. California and Nevada were projected to be relatively insensitive to temperature, but their data included extensive nonforest area. In a more recent analysis, Littell et al.² project the following increases for a 1 °C temperature increase: Arizona, 380 to 470 percent; California, 310 percent; Colorado, 280 to 660 percent; Nevada, 280 percent; New Mexico, 320 to 380 percent; and Utah, 280 to 470 percent. Applying the Parallel Climate Model to California, Lenihan et al. (2003) projected that area burned will increase at least 10 percent 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

² Littell, J.S. Relationships between area burned and climate in the Western United States: vegetation-specific historical and future fire. Manuscript in preparation. On file with: U.S. Geological Survey, Alaska Climate Science Center, 4210 University Drive, Anchorage, AK 99508.

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 one 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 (*Dendroctonus ponderosae* Hopkins) (USDA FS 2011). Facilitated by extended drought and warmer winters, mountain pine beetle outbreaks have focused primarily on older (stressed) lodgepole pine (*Pinus contorta* var. *latifolia* Engelm. ex S. Watson) forest. In Arizona and New Mexico, 7.6 to 11.3 percent 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 a business-as-usual (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 (*Pseudotsuga menziesii* [Mirb.] Franco), 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



U.S. Forest Service.

Figure A1-5—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.

(*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 (*Pinus contorta* var. *murrayana* [Balf.] Engelm.) 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 carbon 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 wildland-urban interface areas through assertive use of fuel treatments (Peterson et al. 2011), as shown recently in the Wallow Fire (Bostwick et al. 2011) (fig. A1-5). 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.

Literature Cited

- Allen, C.D. 2007.** Interactions across spatial scales among forest dieback, fire, and erosion in northern New Mexico landscapes. *Ecosystems*. 10: 797–808.
- Allen, C.D.; Breshears, D.D. 1998.** Drought-induced shift of a forest-woodland ecotone: rapid landscape response to climate variation. *Proceedings of the National Academy of Sciences, USA*. 95: 14839–14842.
- Bostwick, P.; Menakis, J.; Sexton, T. 2011.** How fuel treatments saved homes from the 2011 Wallow fire. http://www.fs.fed.us/fire/management/fuel_treatments.pdf. (19 December 2011).
- Breshears, D.D.; Cobb, N.S.; Rich, P.M. [et al.]. 2005.** Regional vegetation die-off in response to global-change type drought. *Proceedings of the National Academy of Sciences, USA*. 102: 15144–15148.
- Cayan, D.R.; Das, T.; Pierce, T.P. 2010.** Future dryness in the southwest US and the hydrology of the early 21st century drought. *Proceedings of the National Academy of Sciences, USA*. 107: 21271–21276.
- Floyd, M.L.; Clifford, M.; Cobb, N.S. [et al.]. 2009.** Relationship of stand characteristics to drought-induced mortality in three southwestern piñon-juniper woodlands. *Ecological Applications*. 19: 1223–1230.
- Gibson, K.; Skov, K.; Kegley, S. [et al.]. 2008.** Mountain pine beetle impacts in high-elevation five-needle pines: current trends and challenges. R1-08-020. Missoula, MT: U.S. Department of Agriculture, Forest Service, Forest Health Protection. 32 p.
- Incident Information System. 2011.** Wallow. <http://www.incweb.org/incident/2262>. (19 December 2011).
- Lenihan, J.M.; Drapek, R.; Bachelet, D.; Neilson, R.P. 2003.** Climate change effects on vegetation distribution, carbon, and fire in California. *Ecological Applications*. 13: 1667–1681.

- Li, C.; Barclay, H.J.; Hawkes, B.C.; Taylor, S.W. 2005.** Lodgepole pine forest age class dynamics and susceptibility to mountain pine beetle attack. *Ecological Complexity*. 2: 232–239.
- Littell, J.S.; McKenzie, D.; Peterson, D.L.; Westerling, A.L. 2009.** Climate and wildfire area burned in western U.S. ecoprovinces, 1916–2003. *Ecological Applications*. 19: 1003–1021.
- McKenzie, D.; Gedalof, Z.; Peterson, D.L.; Mote, P. 2004.** Climatic change, wildfire, and conservation. *Conservation Biology*. 18: 890–902.
- National Interagency Fire Center. 2011.** Statistics: national year-to-date report on fires and acres burned by state. http://www.nifc.gov/fireInfo/fireInfo_stats_YTD2011.html. (19 December 2011).
- Peterson, D.L.; Halofsky, J.L.; Johnson, M.C. 2011.** Managing and adapting to changing fire regimes in a warmer climate. In: McKenzie, D.; Miller, C.; Falk, D., eds. *The landscape ecology of fires*. New York: Springer: 249–267.
- U.S. Department of Agriculture, Forest Service. 2011.** Rocky Mountain bark beetle. <http://www.fs.usda.gov/main/barkbeetle/home>. (19 December 2011).
- Weiss, J.L.; Castro, C.L.; Overpeck, J.T. 2009.** Distinguishing pronounced droughts in the southwestern United States: seasonality and effects of warmer temperatures. *Journal of Climate*. 22: 5918–5932.
- Williams, A.P.; Allen, C.D.; Millar, C.I. [et al.]. 2010.** Forest responses to increasing aridity and warmth in southwestern North America. *Proceedings of the National Academy of Sciences, USA*. 107: 21289–21294.

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. The Great Plains are divided here into three areas for discussion: northern Great Plains (North Dakota, South Dakota, Kansas, Nebraska), southern Great Plains (Oklahoma, Texas), and western Great Plains (Montana, Wyoming).

Forests in the northern Great Plains comprise less than 3 percent of the total land area within each state (Smith et al. 2009) (table A1-2). More than half of the forest land in South Dakota is in public land ownership in contrast to the other three states. Dominant forest types 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 (Haugen et al. 2009, Meneguzzo et al. 2008, Moser et al. 2008, South Dakota Resource Conservation and Forestry Division 2007). The decline of this species often leads to establishment of nonnative 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 (*Dendroctonus ponderosae* Hopkins) outbreaks. Eastern redcedar (*Juniperus virginiana* L.) is expanding in many states, the result of fire exclusion and prolonged drought conditions (Meneguzzo et al. 2008, South Dakota Resource Conservation and Forestry Division 2007). 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 (Haugen et al. 2009, Meneguzzo et al. 2008).

Forests in the southern Great Plains comprise less than 17 percent of the land area (table A1-2) (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 10 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 comprise less than 27 percent of the land area (Smith et al. 2009) (table A1-2), 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 (*Pseudotsuga menziesii* [Mirb.] Franco), lodgepole pine (*Pinus contorta* Douglas ex. Loudon var. *latifolia* Engelm. ex S. Watson), and ponderosa pine (Montana Department of Natural Resources and Conservation 2010).

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Table A1-2—Land area in the Great Plains 2007

	Total land area	Total forest land	Total timberland	Planted timberland	Natural origin	Reserved forest land	Other forest land	Other land
<i>Thousands of hectares</i>								
Northern Great Plains:								
Kansas	21 241	852	821	16	805	0	31	20 389
Nebraska	19 913	504	475	14	461	4	25	19 409
North Dakota	17 943	293	216	2	214	10	67	17 650
South Dakota	19 601	681	628	7	621	17	36	18 920
Total Northern Great Plains	78 697	2330	2140	39	2101	32	158	76 367
Southern Great Plains:								
Oklahoma	17 788	3102	2523	257	2265	18	561	14 686
Texas	67 863	6990	4799	1132	3668	46	2145	60 873
Total Southern Great Plains	85 651	10 092	7322	1389	5933	64	2706	75 559
Western Great Plains:								
Montana	37 760	10 123	8009	76	7933	1594	520	27 637
Wyoming	25 116	4632	2427	19	2407	1531	673	20 484
Total Western Great Plains	62 876	14 755	10 436	95	10 340	3125	1193	48 121
Total Great Plains	227 244	27 177	79 898	1523	18 374	3221	4057	200 047

Source: Smith et al. 2007.

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). 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 increased wildfire hazard, longer droughts, and increased risk of insect outbreaks, individually and in combination, could significantly modify Great Plains forest environments. Whereas most studies in this region have explored the potential influence of elevated carbon dioxide (CO₂) 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 CO₂ could be a potential factor in the expansion of forests from the Eastern United States into the Great Plains.

Literature Cited

- Bachelet, D.; Lenihan, J.; Drapek, R.; Neilson, R.P. 2008.** VEMAP vs VINCERA: A DGVM sensitivity to differences in climate scenarios. *Global and Planetary Change*. 64: 38–48.
- Barron, E. 2006.** State of the Texas forest 2005. College Station, TX: Texas Forest Service. 37 p.
- Bartlein, P.; Whitlock, C.; Shafer, S. 1997.** Future climate in the Yellowstone National Park region and its potential impact on vegetation. *Conservation Biology*. 11: 782–792.
- Guo, Q.; Brandle, J.; Schoeneberger, M.; Buettner, D. 2004.** Simulating the dynamics of linear forests in Great Plains agroecosystems under changing climates. *Canadian Journal of Forest Research*. 34: 2564–2572.
- Haugen, D.E.; Kangas, M.; Crocker, S.J. [et al.]. 2009.** North Dakota's forests 2005. Resour. Bull. NRS-31. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 82 p.
- Johnson, E.; Geissler, G.; Murray, D. 2010.** The Oklahoma forest resource assessment, 2010: a comprehensive analysis of forest-related conditions, trends, threats and opportunities. Oklahoma City, OK: Oklahoma Department of Agriculture, Food, and Forestry. 163 p.
- Koteen, L. 2002.** Climate change, whitebark pine, and grizzly bears in the Greater Yellowstone Ecosystem. In: Schneider, S.; Root, T., eds. *Wildlife response to climate change: North American case studies*. Washington, DC: Island Press: 343–414.
- Meneguzzo, D.M.; Butler, B.J.; Crocker, S.J. [et al.]. 2008.** Nebraska's forests, 2005. Resour. Bull. NRS-27. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 94 p.
- Montana Department of Natural Resources and Conservation. 2010.** Montana's state assessment of forest resources: base findings and GIS methodology. Missoula, MT: Montana Department of Natural Resources and Conservation. 29 p.
- Morgan, J.A.; Follett, R.F.; Allen, L.H. [et al.]. 2010.** Carbon sequestration in agricultural lands in the United States. *Journal of Soil and Water Conservation*. 65: 6A–13A.
- Moser, W.K.; Hansen, M.H.; Atchison, R.L. [et al.]. 2008.** Kansas forests, 2005. Resour. Bull. NRS-26. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 125 p.
- Perry, L.G.; Andersen, D.C.; Reynolds, L.V. [et al.]. 2012.** Vulnerability of riparian ecosystems to elevated CO₂ and climate change in arid and semiarid western North America. *Global Change Biology*. 18: 821–842.
- Smith, W.B., tech. coord.; Miles, P.D., data coord.; Perry, C.H., map coord.; Pugh, S.A., data CD coord. 2009.** Forest resources of the United States, 2007. Gen Tech. Rep. GTR-WO-78. Washington, DC: U.S. Department of Agriculture, Forest Service. 336 p.
- South Dakota Resource Conservation and Forestry Division. 2007.** South Dakota forest stewardship plan, 2007 revision. Pierre, SD: South Dakota Department of Agriculture, Resource Conservation and Forestry Division. 35 p.
- Westerling, A.L.; Turner, M.G.; Smithwick, E.A.H. [et al.]. 2011.** Continued warming could transform Greater Yellowstone fire regimes by mid-21st century. *Proceedings of the National Academy of Sciences USA*. 108: 13165–13170.
- Whitlock, C.; Shafer, S.; Marlon, J. [et al.]. 2003.** The role of climate and vegetation change in shaping past and future fire regimes in the northwestern U.S. and the implication for ecosystem management. *Forest Ecology and Management*. 178: 5–21.
- Wyckoff, P.H.; Bowers, R. 2010.** Response of the prairie-forest border to climate change: impacts of increasing drought may be mitigated by increasing CO₂. *Journal of Ecology*. 98: 197–208.
- Wyoming State Forestry Division. 2009.** Wyoming statewide assessment. Cheyenne, WY: Wyoming State Forestry. 74 p.
- Xu, W. 2002.** Economic impact of the Texas forest sector. Texas Agric. Exten. Pub. 161. College Station, TX: Texas AgriLife Extension Service. 19 p.

Midwest

Christopher W. Swanston and Stephen D. Handler¹

Introduction

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. Savannas and open woodlands mark a major transition between forest and grassland biomes in the United States. Forests cover approximately 28 percent of the area in the eight-state Midwest region and help sustain human communities ecologically, economically, and culturally. Forest ecosystems are distributed according to patterns of climate, moisture, soils, and disturbance; ecoregions capture these broad patterns across the landscape. Most of the Midwest is contained within the Laurentian Mixed Forest, Eastern Broadleaf Forest (Continental and Oceanic), and Prairie Parklands ecoregions (Bailey 1995) (fig. A1-6).

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 (Swanston et al. 2011). In other words, a species or ecosystem may be considered vulnerable to climate change by virtue of significantly decreased well-being, even if is not projected to disappear completely from the landscape.

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Laurentian Mixed Forest

Key Vulnerabilities: Decline of Boreal Tree Species, Reduced Forested Wetlands

The Laurentian mixed forest spans the northern areas of the Great Lakes states (fig. A1-6), 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 long, often with heavy snowfall, and summers are warm and provide much of the annual precipitation. As a transitional zone between the boreal forests in the north and the broadleaf forests to the 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 (*Picea mariana* [Mill.] Britton, Sterns and Poggenb.), 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. 2008, 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, are a major feature of northern Lake States forests, and 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 carbon (Johnson and Kern 2003) that could be at risk if fire increases in drier conditions. Subboreal 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. 2008).

Eastern Broadleaf Forest

Key Vulnerabilities: Accelerated Oak Decline, Increased Wildfire, New Invasive Species

The Eastern Broadleaf Forest (fig. A1-6) mostly consists of the Continental ecoregion, with low rolling hills, some glaciation in the north, and the Ozark Highlands to the south.

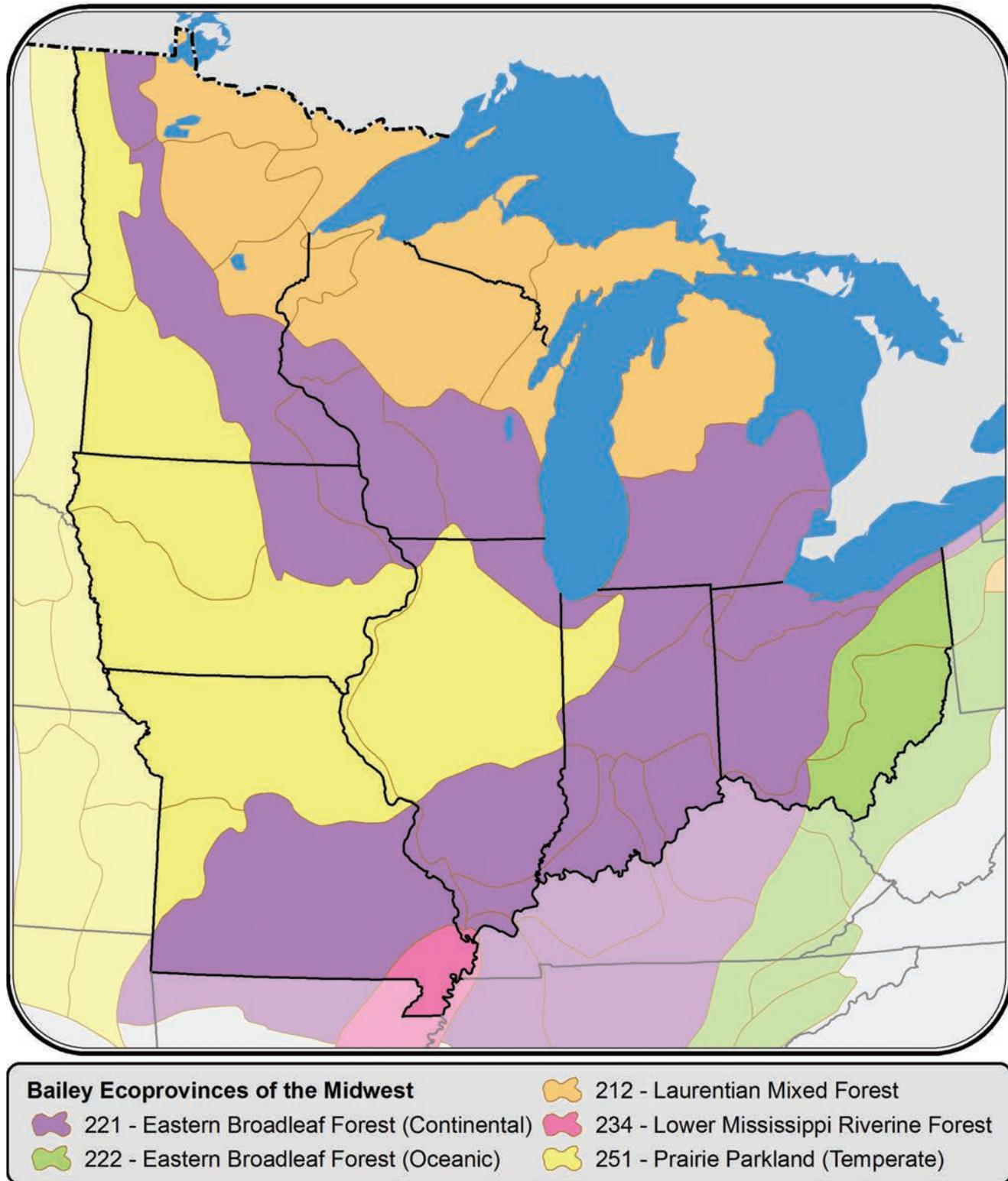


Figure A1-6—Ecoregions within the Midwest, according to Bailey et al. (1995).

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. 2008). 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* [Willd.] Ohwi, an aggressive vine) and Chinese and European privet (*Ligustrum sinense* Lour. and *L. vulgare* L., highly invasive shrubs), that may expand into the Midwest as winter minimum temperatures increase (Bradley et al. 2010).

Prairie Parklands

Key Vulnerabilities: Increased Wildfire, Fragmentation, Loss of Ecosystem Function

The Prairie Parklands (fig. A1-6) 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 (loss of continuity among forested areas) and parcelization (subdivision of

forest tracts into smaller ownerships) of forest ecosystems are more extreme in the Prairie Parklands than in other Midwest ecoregions. For example, over 90 percent 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 and Land Use

Key Issues: Parcelization, Poor Cross-Boundary Coordination, Ambivalent Stewardship

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. Forest ownership patterns differ greatly between the Western and Eastern United States, with 68 percent of forests in private ownership in the Midwest versus only 21 percent in the West (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, Mahmood 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.

Literature Cited

- Abrams, M.D. 1992.** Fire and the development of oak forests. *BioScience*. 42: 346–353.
- Bailey, R.G. 1995.** Description of the ecoregions of the United States. Washington, DC: U.S. Department of Agriculture, Forest Service. 108 p.
- Bengston, D.N.; Asah, S.T.; Butler, B.J. 2011.** The diverse values and motivations of family forest owners in the United States: an analysis of an openended question in the National Woodland Owner Survey. *Small-Scale Forestry*. 10: 339–355.
- Bradley, B.A.; Wilcove, D.S.; Oppenheimer, M. 2010.** Climate change increases risk of plant invasion in the eastern United States. *Biological Invasions*. 12: 1855–1872.
- Butler, B.J. 2008.** Family forest owners of the United States, 2006. Gen. Tech. Rep. NRS-27. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 72 p.
- DeCoster, L.A. 1998.** The boom in forest owners—a bust for forestry? *Journal of Forestry*. 96: 25–28.
- Flickinger, A. 2010.** Iowa’s forests today: an assessment of the issues and strategies for conserving and maintaining Iowa’s forests. In: Miller, E., ed. Des Moines, IA: Iowa Department of Natural Resources. <http://www.iowadnr.gov/Environment/Forestry/ForestryLinksPublications/IowaForestActionPlan.aspx>. (26 September 2012).
- Goldblum, D. 2010.** The geography of white oak’s (*Quercus alba* L.) response to climatic variables in North America and speculation on its sensitivity to climate change across its range. *Dendrochronologia*. 28: 73–83.
- Iverson, L.R.; Prasad, A.M.; Matthews, S. 2008.** Modeling potential climate change impacts on the trees of the northeastern United States. *Mitigation and Adaptation Strategies for Global Change*. 13: 487–516.
- Johnson, M.G.; Kern, J.S. 2003.** Quantifying the organic carbon held in forested soils of the United States and Puerto Rico. In: Kimble, J.M.; Heath, L.S.; Birdsey, R.A.; Lal, R., eds. The potential of U.S. forest soils to sequester carbon and mitigate the greenhouse effect. New York: CRC Press: 47–72.
- Kittredge, D.B. 2004.** Extension/outreach implications for Americas family forest owners. *Journal of Forestry*. 102: 15–18.
- Lenihan, J.M.; Bachelet, D.; Neilson, R.P.; Drapek, R. 2008.** Simulated response of conterminous United States ecosystems to climate change at different levels of fire suppression, CO₂ emission rate, and growth response to CO₂. *Global and Planetary Change*. 64: 16–25.
- Mehmood, S.R.; Zhang, D. 2001.** Forest parcelization in the United States: a study of contributing factors. *Journal of Forestry*. 99: 30–34.
- Mladenoff, D.J.; Pastor, J. 1993.** Sustainable forest ecosystems in the northern hardwood and conifer region: concepts and management. In: Aplet, G.H.; Olsen, J.T.; Johnson, N.; Sample, V.A., eds. Defining sustainable forestry. Washington, DC: Island Press: 145–180.
- Nelson, M.D.; Liknes, G.C.; Butler, B.J. 2010.** Map of forest ownership in the conterminous United States. [Scale 1:7,500,000]. Res. Map NRS-2. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station.
- Nowacki, G.J.; Abrams, M.D. 2008.** The demise of fire and “mesophication” of forests in the eastern United States. *BioScience*. 58: 123–138.
- Swanston, C.W.; Janowiak, M.; Iverson, L. [et al.]. 2011.** Ecosystem vulnerability assessment and synthesis: a report from the Climate Change Response Framework Project in northern Wisconsin, Version 1. Gen. Tech. Rep. NRS-82. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 142 p.
- Walker, K.V.; Davis, M.B.; Sugita, S. 2002.** Climate change and shifts in potential tree species range limits in the Great Lakes region. *Journal of Great Lakes Research*. 28: 555–567.
- Wang, C.; Lu, Z.; Haithcoat, T.L. 2007.** Using Landsat images to detect oak decline in the Mark Twain National Forest, Ozark Highlands. *Forest Ecology and Management*. 24: 70–78.
- Xu, C.; Gertner, G.; Scheller, R. 2012.** Importance of colonization and competition in forest landscape response to global climatic change. *Climatic Change*. 110: 53–83.

Northeast

Lindsey E. Rustad¹

Climate is a key regulator of terrestrial biogeochemical processes. For the Northeastern United States, the magnitude of climate change observed during the 20th century and expected for the 21st century has had, and will continue to have, profound effects on the structure, function, and biodiversity of the region's forests and the ecosystem services they provide. A recent synthesis of climate-change effects on forests of the 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).

According to an analysis of climatic data from the northeastern United States (Huntington et al. 2009), since 1900, mean annual temperature has risen by an average of 0.8 °C, precipitation has increased by approximately 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 to 2 weeks. Projections for the 21st century (based on climatic models and emission scenarios statistically downscaled for the region) suggest that temperature will increase by 2.9 to 5.3 °C, precipitation will increase by 7 to 14 percent (with little or no change in summer precipitation), the onset of spring will advance by 10 to 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, including hurricanes, winter rain, snow, ice storms, droughts, and heat waves.

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Forests cover large areas of the land surface in the Northeastern United States, from 59 percent in Rhode Island to upwards of 89 percent in Maine (National Land Cover Database 2001). These forests are currently dominated by (1) southern hardwoods (oak [*Quercus* spp.], hickory [*Carya* spp.]) and pines [*Pinus* spp.] in the southernmost region; (2) northern hardwoods (American beech [*Fagus grandifolia* Ehrh.], paper and yellow birch [*Betula papyrifera* Marsh., *B. alleghaniensis* Britt.], and sugar and red maple [*Acer saccharum* Marsh., *A. rubrum* L.]) in the central part and at lower elevations throughout; and (3) boreal-conifer forests in the north and at higher elevations (red and black spruce [*Picea rubens* Sarg., *P. mariana* {Mill.} Britton, Sterns & Poggenb.], and balsam fir [*Abies balsamea* {L.} Mill.]). Eastern hemlock (*Tsuga canadensis* [L.] Carrière), an important shade-tolerant, late-successional species, is found throughout the northeast. Paleocological data from the region 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 complicated 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. (2008) 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. A1-7). Projections of change in suitable habitat for individual tree species indicate that, of the 84 most common species, 23 to 33 will lose suitable habitat under low- and high-emission scenarios, 48 to 50 will gain habitat, and 1 to 10 will experience no change. Under a high-emission scenario, the tree species predicted to have the most affected habitat include balsam fir, quaking aspen (*Populus tremuloides* Michx.), paper birch (80 to 87 percent decrease in suitable habitat), and black and white oak (*Quercus velutina* Lam., *Q. alba* L.) (more than 100 percent increase in suitable habitat) (Iverson et al. 2008).

As climate and species composition change, so will forest productivity and carbon (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).

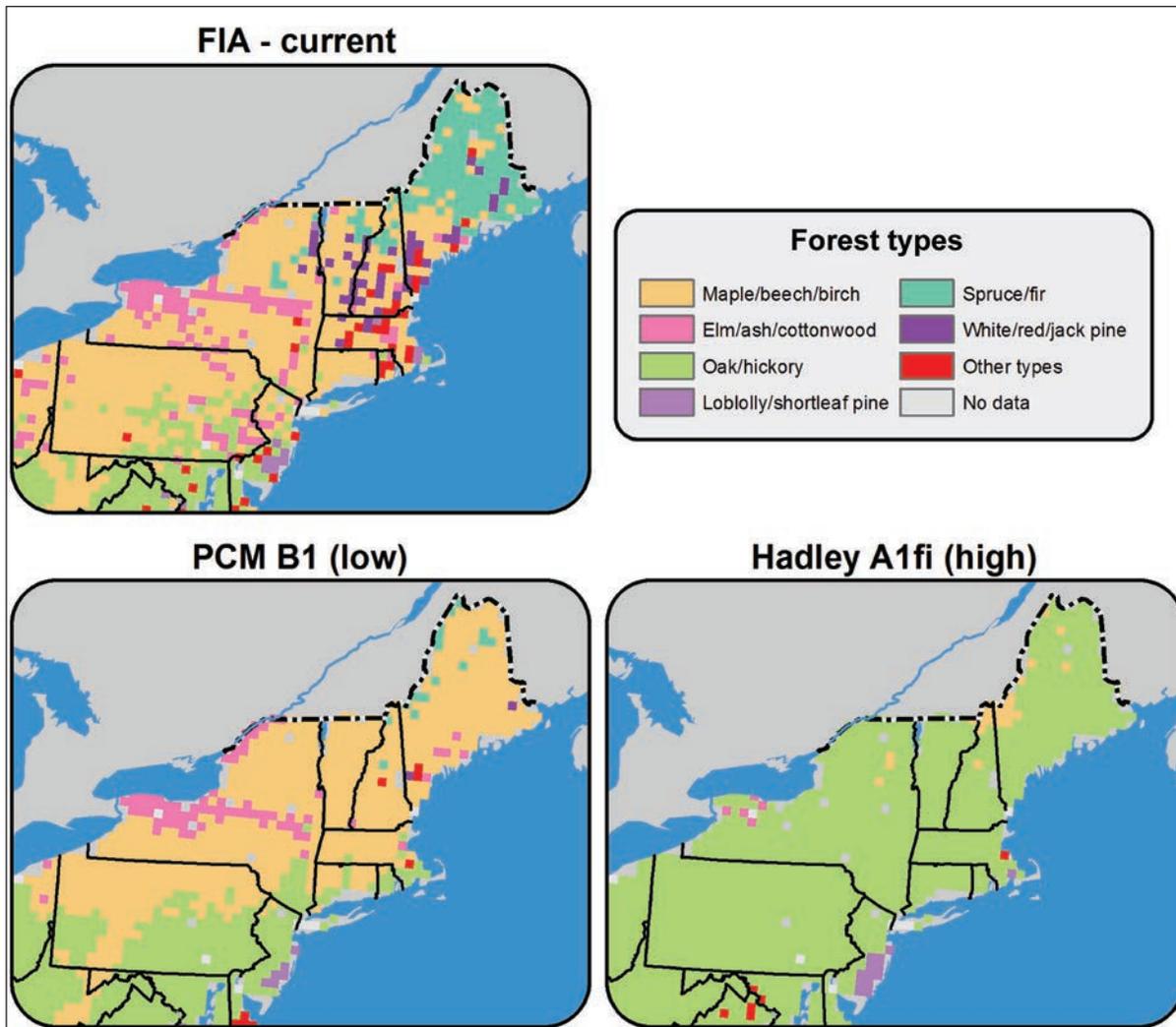


Figure A1-7—Suitable habitat for forest vegetation in New England is expected to shift with changes in climate associated with different emissions scenarios. From Mohan et al. (2009).

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 carbon dioxide (CO₂). For example, Ollinger et al. (2008) used the model PnET-CN to project that net primary productivity in deciduous forests would increase by 52 to 250 percent by 2100, depending on the global model and emissions 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 in forest ecosystems. 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). Even under a low emission scenario, 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 in the region. Warmer temperatures will also probably increase rates of root and microbial respiration, with an increased release of CO₂ from the soil to the atmosphere. Because soil respiration is the second-largest flux in the global C cycle (only primary productivity is larger), any increase in soil respiration may contribute to further warming. Soil-warming experiments in the Northeastern United States have confirmed this relationship (Rustad et al. 2000).

A major unknown in predicting warming-mediated biogeochemical responses is the potential interaction with projected future short- and long-term droughts, which tend

to reduce physiological activity or induce dormancy during periods of stress. Current projections suggest that future summers will be warmer, and total precipitation will remain constant but occur as larger events separated by longer dry periods (Huntington 2009). Coupled with potentially higher productivity, evaporation and transpiration will likely increase, resulting in lower soil moisture during the growing season. This has implications for direct effects on biological activity, as well as for wildfire frequency and severity, streamflow, and lake levels.

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, five are decreasing in abundance, including highly valued species such as ruffed grouse (*Bonasa umbellus* L.), and five show no change. Significant range expansions have also been observed, with 27 of 38 species studied expanding their ranges in a northward direction (fig. A1-8).

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 percent of the avifauna), an equal number are expected to increase as decrease. “Winners” (increased abundance) include pileated woodpecker (*Dryocopus pileatus* L.) (+15 to 50 percent), great horned owl (*Bubo virginianus* Gmelin) (+18 to more than 200 percent), and northern cardinal (*Cardinalis cardinalis* L.) (+20 to 33 percent). “Losers” (decreased abundance) include common loon (*Gavia immer* Brunnich) (-76 to -93 percent), winter wren (*Troglodytes hiemalis* Vieillot) (-42 to -73 percent), and rose-breasted grosbeak (*Pheucticus ludovicianus* L.) (-23 to -71 percent). Species such as Bicknell’s thrush

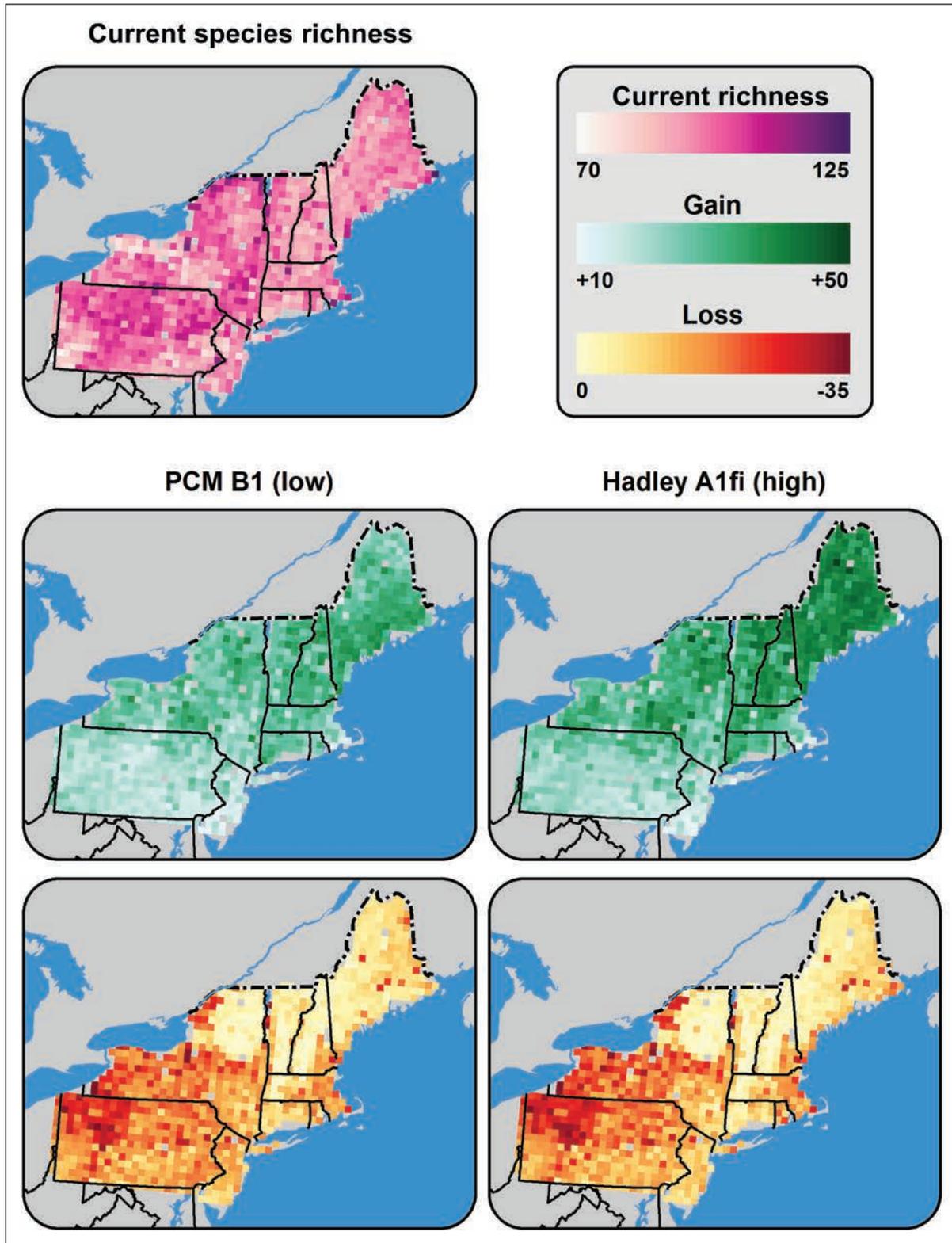


Figure A1-8—Climate change is expected to affect bird species richness more intensely in some areas of the Northeastern United States than in others. From Rodenhouse et al. (2008).

(*Catharus bicknelli* Ridgway), which inhabit high-elevation spruce-fir forests, are especially susceptible to climate change, because half the suitable habitat available for this species is projected to be lost with an increase of only 1 °C in mean annual temperature (Rodenhouse et al. 2008).

Climate-related historical and future projected changes in native and introduced insects and pathogens 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 (see table A1-3). Direct effects of climate change on these species 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. For example, hemlock woolly adelgid (*Adelges tsugae* Annand) (HWA) is distributed in areas where minimum winter temperatures stay above -28.8 °C (Skinner et al. 2003). Based on recent projections, climatic warming could allow HWA to

spread unimpeded throughout the range of eastern hemlock distribution. The potential effects of widespread hemlock mortality include changes in forest composition, structure, nutrient cycling, surface water quality, and populations of associated wildlife (Dukes et al. 2009). Indirect effects of climate change include changes in (1) nutrient supply, availability, and allocation in space and time; (2) distribution, life cycles, and phenology of insects, pathogens, and their hosts, predator associates, or competitors; (3) mismatches between location and optimal conditions, resulting in tree stress and increased pathogen susceptibility; and (4) creation of novel species assemblages or mismatches between locally or regionally coadapted genotypes that could enhance the potential for infestation and virulence.

Climate change is expected to be characterized by an increase in the prevalence and severity of extreme events, such as heat waves, cold waves, windstorms, floods, and droughts (Huntington et al. 2009, Solomon et al. 2007). A growing concern exists that these types of events can have a larger effect on natural and managed systems than the more gradual change in mean climatic conditions. Legacies of

Table A1-3—Common native and nonnative insects, pathogens, and invasive species of the northeastern United States (Dukes et al. 2009)^a

Insects	Pathogens	Invasives
Balsam woolly adelgid (<i>Adelges piceae</i> Ratzeburg)	Armillaria root rot (<i>Armillaria</i> spp.)	Tree-of-heaven (<i>Ailanthus altissima</i> [P. Mill.] Swingle)
Hemlock woolly adelgid (<i>Adelges tsugae</i> Annand)	White pine blister rust (<i>Cronartium ribicola</i> A. Dietr.)	Garlic-mustard (<i>Alliaria petiolata</i> [M. Bieb.] Cavara & Grande)
Emerald ash borer (<i>Agrilus planipennis</i> Fairmaire)	Chestnut blight (<i>Cryphonectria parasitica</i> [Murrill] Barr)	Japanese barberry (<i>Berberis thunbergii</i> DC.)
Asian longhorned beetle (<i>Anoplophora glabripennis</i> Motschulsky)	Beech bark disease (<i>Neonectria faginata</i> [M.L. Lohman, A.M.J. Watson & Ayers] Castl. & Rossman)	Japanese stiltgrass (<i>Microstegium vimineum</i> [Trin.] A. Camus)
Spruce budworm (<i>Choristoneura fumiferana</i> Clemens)	Dutch elm disease (<i>Ophiostoma ulmi</i> [Buisman] Nannf. and <i>O. novo-ulmi</i> Brasier)	Multiflora rose (<i>Rosa multiflora</i> Thunb.)
Gypsy moth (<i>Lymantria dispar</i> L.)	White trunk rot (<i>Phellinus</i> spp.).	Wine raspberry (<i>Rubus phoenicolasius</i> Maxim.)
Forest tent caterpillar (<i>Malacosoma disstria</i> Hübner)	Sudden oak death (<i>Phytophthora ramorum</i> S. Werres, A.W.A.M. de Cock)	

^a No relationship is implied from left to right in the table.

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 and productivity, biogeochemistry, wildlife, insects, pathogens, and invasive species.

The 20th century climate of the Northeastern United States has changed more rapidly than at any time since the last glaciation, and this rate of change is expected to continue throughout the 21st century. The direct and indirect

effects of climate change on Northeastern forests, individually and in combination with other stressors such as acid deposition, N and mercury deposition, tropospheric ozone, and various land uses, have the potential to cause significant changes in ecosystem structure and function. 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.

Literature Cited

- Campbell, J.L.; Rustad, L.E.; Boyer, E.B. [et al.]. 2009.** Consequences of climate change for biogeochemical cycling in forests of northeastern North America. *Canadian Journal of Forest Research*. 39: 264–284.
- Dukes, J.S.; Pontius, J.; Orwig, D. 2009.** Responses of insect pests, pathogens, and invasive plant species to climate change in the forests of northeastern North America: What can we predict? *Canadian Journal of Forest Research*. 39: 231–248.
- Huntington, T.G.; Richardson, A.D.; McGuire, K.J.; Hayhoe, K. 2009.** Climate and hydrological changes in the northeastern United States: recent trends and implications for forested and aquatic ecosystems. *Canadian Journal of Forest Research*. 39: 199–212.
- Iverson, L.; Prasad, A.; Matthews, S. 2008.** Potential changes in suitable habitat for 134 tree species in the northeastern United States. *Mitigation and Adaptation Strategies for Global Change*. 13: 487–516.
- Mohan, J.E.; Cox, R.M.; Iverson, L.R. 2009.** Composition and carbon dynamics of forests in northeastern North America in a future, warmer world. *Canadian Journal of Forest Research*. 39: 213–230.
- National Land Cover Database. 2001.** Multi-resolution land characteristics consortium: national land cover database. http://www.mrlc.gov/nlcd01_data.php. (23 December 2011).
- Ollinger, S.V.; Goodale, C.L.; Hayhoe, K.; Jenkins, J.P. 2008.** Potential effects of climate change and rising CO₂ on ecosystem processes in northeastern U.S. forests. *Mitigating Adaptation Strategies for Global Change*. 13: 467–485.
- Richardson, A.D.; Bailey, A.S.; Denny, E.G. [et al.]. 2006.** Phenology of a northern hardwood forest canopy. *Global Change Biology*. 12: 1174–1188.
- Rodenhouse, N.L.; Christenson, L.M.; Parry, D.; Green, L.E. 2009.** Climate change effects on native fauna of northeastern forests. *Canadian Journal of Forest Research*. 39: 249–263.
- Rodenhouse, N.L.; Matthews, S.N.; McFarland, K.P. [et al.]. 2008.** Potential effects of climate change on birds of the Northeast. *Mitigation and Adaptation Strategies for Global Change*. 13: 517–540.
- Rustad, L.E.; Campbell, J.L.; Cox, R.M. [et al.]. 2009.** NE forests 2100: a synthesis of climate change impacts on forests of the northeastern US and eastern Canada. 2009. *Canadian Journal of Forest Research*. 39: iii–iv.
- Rustad, L.E.; Melillo, J.M.; Mitchell, M.J. [et al.]. 2000.** Effects of soil warming on carbon and nitrogen cycling. In: Mickler, R.; Birdsey, R.; Hom, J., eds. *Responses of northern U.S. forests to environmental change*. New York: Springer-Verlag: 357–381.
- Skinner, M.; Parker, B.L.; Gouli, S.; Ashikaga, T. 2003.** Regional responses of hemlock woolly adelgid (*Homoptera: Adelgidae*) to low temperatures. *Environmental Entomology*. 32: 523–528.
- Solomon, S.; Qin, D.; Manning, M. [et al.], eds. 2007.** *Climate change 2007: the physical science basis: contribution of working group I to the fourth assessment report of the Intergovernmental Panel on Climate Change*. Cambridge, United Kingdom; New York: Cambridge University Press. 996 p.

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, often including intensive management such as forest plantations, fertilization, and prescribed fires, creates stand conditions and management regimes that differ from those in 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 carbon (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.

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 significant effects on southeastern forests. A single hurricane can reduce total forest C sequestration by 10 percent 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 (*Pinus 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).

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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 of the South (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 percent less carbon dioxide (CO₂) 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 et al. 1997). Most of the longleaf pine was cut during the 20th century and replanted with the faster growing loblolly pine, which is preferred by the timber industry but is less resistant to wildfire damage. 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). 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 trees that are more susceptible and to colonize trees to form new points of spread (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). If the trend of increasing frequency and severity of insect outbreaks continues, the productivity and large-scale structure of forest ecosystems will be altered significantly.

Some aspects of the exceptionally 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 (*Picea rubens* Sarg.) and eastern hemlock (*Tsuga canadensis* [L.] Carrière) are well adapted to the cool climates of the last glacial age. However, during a period of postglacial warming, the extent and dominance of these two species have decreased greatly owing to stress complexes that include warmer temperature, air pollution, and insects (Elliott and Vose 2011, McNulty and Boggs 2010). With further warming, red spruce and eastern hemlock are projected to be extirpated from the Southern United States before the end of the 21st century (Prasad et al. 2007), and small remnant populations of balsam fir (*Abies balsamea* [L.] Mill.) will also be at risk. Birds

and other terrestrial vertebrate species that depend on forests dominated by these trees for habitat and food must adapt, migrate, or face a similar fate.

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).

The majority of the Nation’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 in the southeast, and although current air temperature is near optimal for growth across much of its range, as temperature

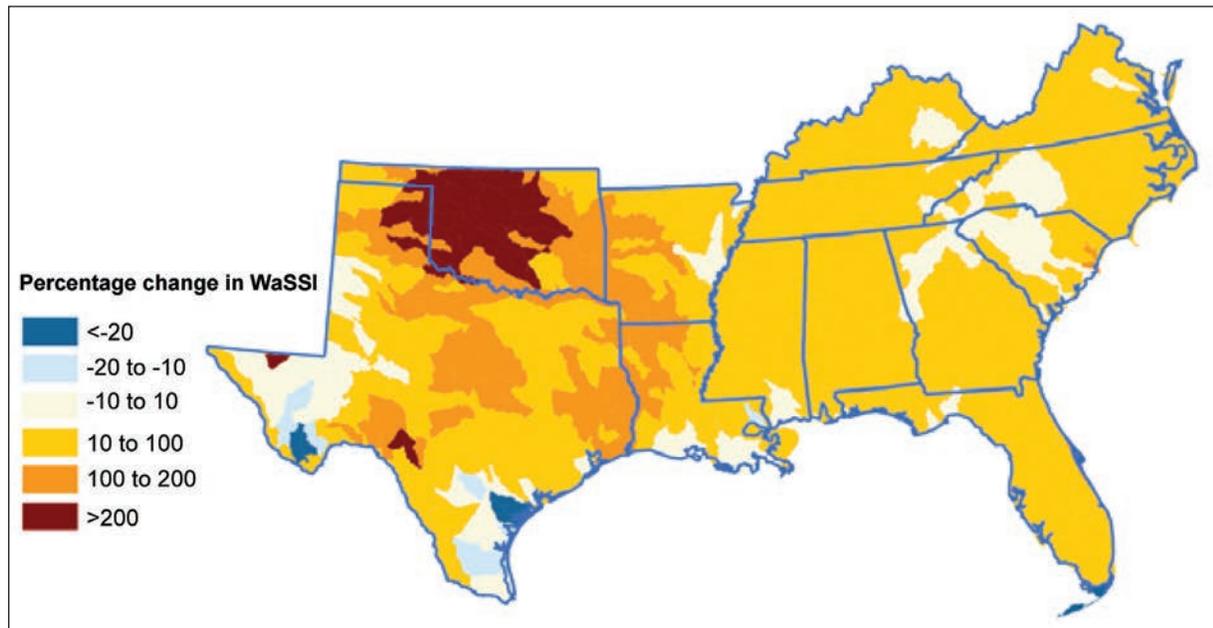


Figure A1-9—Percentage change in water supply stress owing to climate change, as defined by the water supply stress index (WaSSI) for 2050 using the CSIROMK2 B2 climate scenario. WaSSI is calculated by dividing water demand by supply, where higher values indicate higher stress on watersheds and water systems. From Lockaby et al. 2011.

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 significant economic and social effects in those areas gaining and losing timber industry jobs (Sohnngen et al. 2001).

Carbon sequestration is an increasingly valued component of forest productivity. Globally, forests sequester up to 16 percent of all the CO₂ generated from the burning of fossil fuels, and in the United States, much of this storage occurs in Southeastern forests (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 CO₂ release in the Southeast (Boddy 1993). 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).

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 of this increase is centered on metropolitan areas, whereas much of the water originates in forested headwaters, often long distances from urban areas. On an average

annual basis, water supply is approximately 20 times higher than demand, although short-term (1 to 3 years) drought can significantly increase pressure on available water (Lockaby et al. 2011) (fig. A1-9). 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 with higher air temperature, or shifting management regimes for new products such as biofuels, could contribute to water shortfalls (Lockaby et al. 2011, Sun et al. 2008). The 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. Many Southeastern ski areas 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 permanently 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.

Literature Cited

- Andreu, A.; Hermansen-Baez, L.A. 2008.** Fire in the South 2: The Southern wildfire risk assessment. A report by the Southern Group of State Foresters. 32 p. <http://www.southernwildfirerisk.com/reports/FireInTheSouth2.pdf>. (10 September 2012).
- Ayres, M.P.; Lombardero, M.J. 2000.** Assessing the consequences of global change for forest disturbance from herbivores and pathogens. *The Science of the Total Environment*. 262: 263–286.
- Boddy, L. 1983.** Carbon dioxide release from decomposing wood: effect of water content and temperature. *Soil Biology and Biochemistry*. 15: 501–510.
- Brockway, D.G.; Lewis, C.E. 1997.** Long-term effects of dormant-season prescribed fire on plant community diversity, structure and productivity in a longleaf pine wiregrass ecosystem. *Forest Ecology and Management*. 97: 167–183.
- Dale, V.H.; Joyce, L.A.; McNulty, S.G. [et al.]. 2001.** Climate change and forest disturbances. *BioScience*. 59: 723–734.
- Elliott, K.J.; Vose, J.M. 2011.** The contribution of the Coweeta Hydrologic Laboratory to developing an understanding of long-term (1934–2008) changes in managed and unmanaged forests. *Forest Ecology and Management*. 261: 900–910.
- Flebbe, P.A.; Roghair, L.D.; Bruggink, J.L. 2006.** Spatial modeling to project southern Appalachian trout distribution in a warmer climate. *Transactions of the American Fisheries Society*. 135: 1371–1382.
- Hurteau, M.D.; Koch, G.W.; Hungate, B.A. 2008.** Carbon protection and fire risk reduction: toward a full accounting of forest carbon offsets. *Frontiers in Ecology and the Environment*. 6: 493–498.
- Johnsen, K.H.; Butnor, J.R.; Kush, J.S. [et al.]. 2009.** Hurricane Katrina winds damaged longleaf pine less than loblolly pine. *Southern Journal of Applied Forestry*. 3: 178–181.
- Karl, T.R.; Knight, R.W.; Plummer, N. 1995.** Trends in high-frequency climate variability in the twentieth century. *Nature*. 377: 217–220.
- Liu, Y.; Stanturf, J.; Goodrick, S. 2010.** Trends in global wildfire potential in a changing climate. *Forest Ecology and Management*. 259: 685–697.
- Lockaby, B.G.; Nagy, C. [et al.]. 2011.** Water and forests. In: *Southern forest futures technical report*. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. Chapter 13. <http://www.srs.fs.usda.gov/futures/reports/draft/pdf/Chapter%2013.pdf>. (26 September 2012).
- McNulty, S.G. 2002.** Hurricane impacts on U.S. forest carbon sequestration. *Environmental Pollution*. 116: s17–s24.
- McNulty, S.G.; Boggs, J.L. 2010.** A conceptual framework: redefining forest soil’s critical acid loads under a changing climate. *Environmental Pollution*. 158: 2053–2058.
- McNulty, S.G.; Lorio, P.L.; Ayres, M.P.; Reeve, J.D. 1998a.** Predictions of southern pine beetle populations using a forest ecosystem model. In: Mickler, R.A.; Fox, S.A., eds. *The productivity and sustainability of Southern forest ecosystems in a changing environment*. New York: Springer-Verlag: 617–634.
- McNulty, S.G.; Vose, J.M.; Swank, W.T. 1998b.** Predictions and projections of pine productivity and hydrology in response to climate change across the southern United States. In: Mickler, R.A.; Fox, S.A., eds. *The productivity and sustainability of Southern forest ecosystems in a changing environment*. New York: Springer-Verlag: 391–406.
- Millsaps, W.; Groothuis, P.A. 2003.** The economic impact of North Carolina ski areas on the economy of North Carolina: 2002–2003 season. North Carolina Ski Areas Association. <http://www.goskinc.com>. (15 September 2012).
- Pan, Y.; Birdsey, R.A.; Fang, J. [et al.]. 2011.** A large and persistent carbon sink in the world’s forests, 1990–2007. *Science*. 333: 988–993.

- Prasad, A.M.; Iverson, L.R.; Matthews, S.; Peters, M.**
2007-ongoing. A climate change atlas for 134 forest tree species of the eastern United States. Delaware, OH: U.S. Department of Agriculture, Forest Service, Northern Research Station. [Database] <http://www.nrs.fs.fed.us/atlas/tree>. (15 September 2012).
- Pye, J.M.; Holmes, T.P.; Prestemon, J.P.; Wear, D.N.**
2011. Economic impacts of the southern pine beetle. In: Coulson, R.N.; Klepzig, K.D., eds. Southern pine beetle II. Gen. Tech. Rep. SRS-140. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station: 213–222.
- Sohnen, B.; Mendelsohn, R.; Sedjo, R.** **2001.** A global model of climate change impacts on timber markets. *Journal of Agricultural and Resource Economics*. 26: 326–343.
- Sun, G.; McNulty, S.G.; Moore Myers, J.A.; Cohen, E.C.**
2008. Impacts of stresses on water demand and supply across the southeastern United States. *Journal of the American Water Resources Association*. 44: 1441–1457.
- Thompson, I.; Mackey, B.; McNulty, S.; Mosseler, A.**
2009. Forest resilience, biodiversity, and climate change. Technical Series no. 43. Montreal, Canada: Secretariat of the Convention on Biological Diversity. 67 p.
- Trimble, S.W.** **2008.** Man-induced soil erosion on the Southern Piedmont: 1700–1970, 2nd ed. Ankeny, IA: Soil and Water Conservation Society. 80 p.
- Wertin, T.M.; McGuire, M.A.; Teskey, R.O.** **2010.** The influence of elevated temperature, elevated atmospheric CO₂ concentration and water stress on net photosynthesis of loblolly pine (*Pinus taeda* L.) at northern, central and southern sites in its native range. *Global Change Biology*. 16: 2089–2103.
- Whitlow, T.H.; Harris, R.W.** **1979.** Flood tolerance in plants: a state-of-the-art review. Washington, DC: U.S. Department of Commerce, National Technical Information Service: 1–161.
- Wiedinmyer, C.; Hurteau, M.D.** **2010.** Prescribed fire as a means of reducing forest carbon emissions in the western United States. *Environmental Science and Technology*. 44: 1926–1932.