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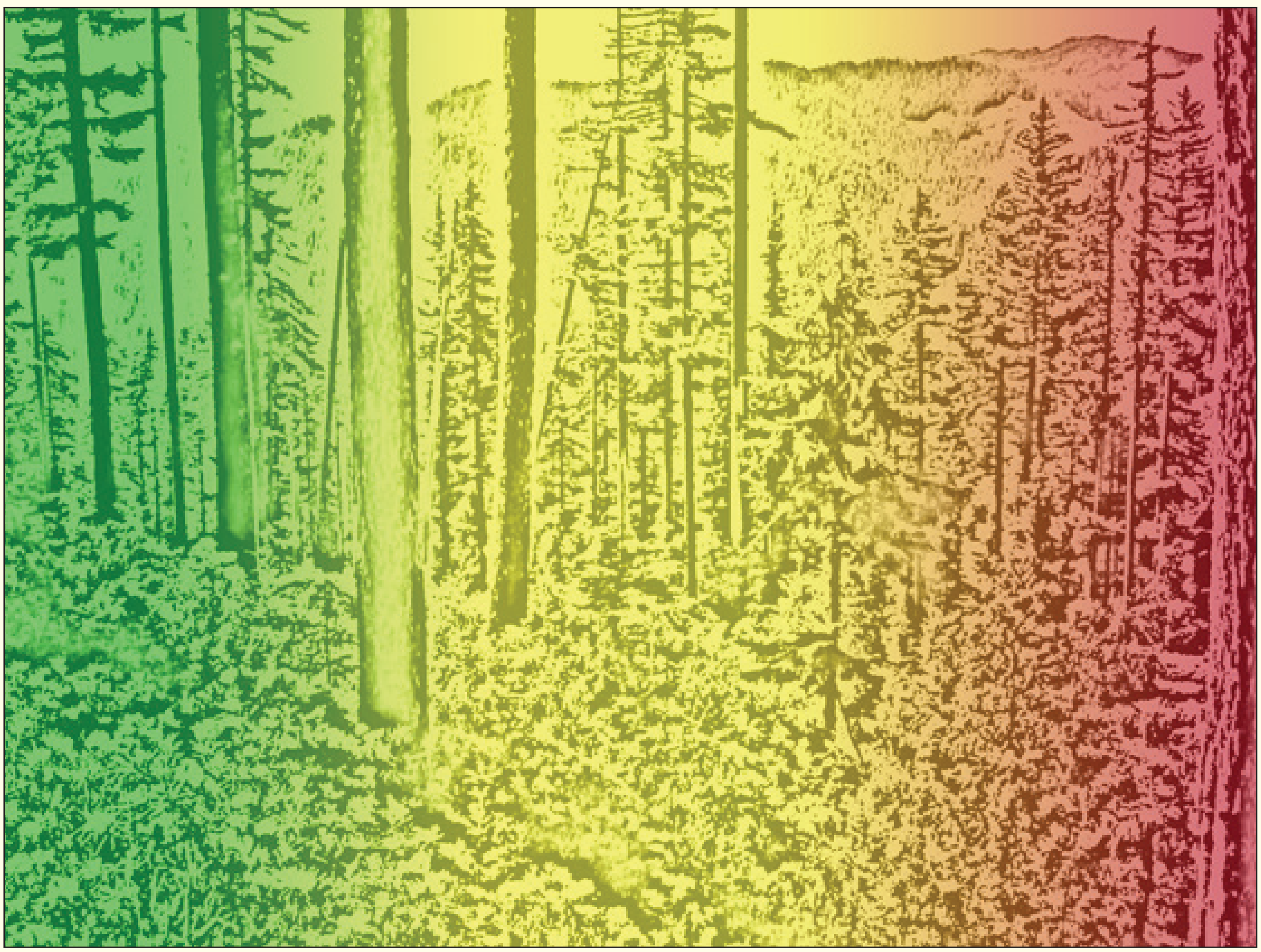
Effects of Climatic Variability and Change on Forest Ecosystems:

A Comprehensive Science Synthesis for the U.S. Forest Sector



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Effects of Climatic Variability and Change on Forest Ecosystems:

A Comprehensive Science Synthesis for the U.S. Forest Sector

James M. Vose, David L. Peterson, and Toral Patel-Weynand, Editors

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Abstract

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This report is a scientific assessment of the current condition and likely future condition of forest resources in the United States relative to climatic variability and change. It serves as the U.S. Forest Service forest sector technical report for the National Climate Assessment and includes descriptions of key regional issues and examples of a risk-based framework for assessing climate-change effects.

By the end of the 21st century, forest ecosystems in the United States will differ from those of today as a result of changing climate. Although increases in temperature, changes in precipitation, higher atmospheric concentrations of carbon dioxide (CO_2), and higher nitrogen (N) deposition may change ecosystem structure and function, the most rapidly visible and most significant short-term effects on forest ecosystems will be caused by altered disturbance regimes. For example, wildfires, insect infestations, pulses of erosion and flooding, and drought-induced tree mortality are all expected to increase during the 21st century. These direct and indirect climate-change effects are likely to cause losses of ecosystem services in some areas, but may also improve and expand ecosystem services in others. Some areas may be particularly vulnerable because current infrastructure and resource production are based on past climate and steady-state conditions. The ability of communities with resource-based economies to adapt to climate change is linked to their direct exposure to these changes, as well as to the social and institutional structures present in each environment. Human communities that have diverse economies and are resilient to change today will also be prepared for future climatic stresses.

Significant progress has been made in developing scientific principles and tools for adapting to climate change through science-management partnerships focused on education, assessment of vulnerability of natural resources, and development of adaptation strategies and tactics. In addition, climate change has motivated increased use of bioenergy and carbon (C) sequestration policy options as mitigation strategies, emphasizing the effects of climate change-human interactions on forests, as well as the role of forests in mitigating climate change. Forest growth and afforestation in the United States currently account for a net gain in C storage and offset approximately 13 percent of the Nation's fossil fuel CO_2 production. Climate change mitigation through forest C management focuses on (1) land use change to increase forest area (afforestation) and avoid deforestation, (2) C management in existing forests, and (3) use of wood as biomass energy, in place of fossil fuel or in wood products for C storage and in place of other building materials. Although climate change is an important issue for management and policy, the interaction of changes in biophysical environments (e.g., climate, disturbance, and invasive species) and human responses to those changes (management and policy) will ultimately determine outcomes for ecosystem services and people.

Although uncertainty exists about the magnitude and timing of climate-change effects on forest ecosystems, sufficient scientific information is available to begin taking action now. Building on practices compatible with adapting to climate change provides a good starting point for land managers who may want to begin the adaptation process. Establishing a foundation for managing forest ecosystems in the context of climate change as soon as possible will ensure that a broad range of options will be available for managing forest resources sustainably.

Keywords: Adaptation, carbon, climate change, climate-change effects, climate-smart management, ecological disturbance, forest ecosystems, mitigation, National Climate Assessment.

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English Equivalents

When you know:	Multiply by:	To find:
Millimeters (mm)	0.0394	Inches
Centimeters (cm)	.394	Inches
Meters (m)	3.28	Feet
Kilometers (km)	.621	Miles
Hectares (ha)	2.47	Acres
Square meters (m^2)	10.76	Square feet
Square kilometers (km^2)	.386	Square miles
Megagrams (Mg)	1	Tons
Teragrams (Tg)	1,102,311.31	Tons
Degrees Celsius (C)	1.8 °C + 32	Degrees Fahrenheit

Executive Summary

The forest sector technical report is a sector-wide scientific assessment of the current condition and likely future condition of forest resources in the United States relative to climatic variability and change. The assessment provides technical input to the National Climate Assessment (NCA) and serves as a framework for managing forest resources in the United States. The report provides technical input to the 2013 NCA developed by the U.S. Global Change Research Program (USGCRP).

The Global Change Research Act of 1990 requires the USGCRP to produce the NCA for the President and the Congress every four years, analyzing the effects of global change on multiple sectors and regions in the United States. The USGCRP is responsible for preparing the report based on technical information provided by public agencies and nongovernmental organizations. The NCA evaluates the effects of global change on agriculture, forests, energy production and use, land and water resources, transportation, human health and welfare, human social systems, and biological diversity, projecting major trends forward for up to 100 years.

In addition, the USGCRP is tasked with providing a coordinated strategy and implementation plan for assessing the effects of a changing climate on the Nation. This strategy is being developed to provide support to the NCA and establish a mechanism for an ongoing assessment capability beyond the 2013 report.

The forest sector technical report is the key technical input to the NCA forest sector chapter. To provide national stakeholder input to the forest sector technical report, a workshop was held in Atlanta, Georgia, on July 12–14, 2011, to solicit input from public, private, and tribal forest stakeholders, nongovernmental organizations, academics, professional organizations, private corporations, and federal agencies. These stakeholder suggestions helped to frame the subject matter content and management options in the report, ensuring relevance for decisionmakers and resource managers.

The forest sector technical report builds on the portion of the 2009 NCA that discussed forest ecosystems and incorporates new findings from scientific and management perspectives. The introduction provides an overview and discusses interrelated aspects of biophysical and socioeconomic phenomena in forested ecosystems that may be affected by climatic variability and change, followed by these chapters:

- Effects of Climatic Variability and Change
- Climate Change, Human Communities, and Forests in Rural, Urban, and Wildland-Urban Interface Environments
- Adaptation and Mitigation
- Improving Scientific Knowledge
- Future Assessment Activities
- Conclusions

It is difficult to conclude whether recently observed trends or changes in ecological phenomena are the result of human-caused climate change, climatic variability, or other factors. Regardless of the cause, forest ecosystems in the United States at the end of the 21st century will differ from those of today as a result of changing climate. Below we discuss the most important issues that have emerged from the report, including a brief summary of regional issues.

Effects of Climate Change on Ecosystem Structure, Function, and Services

A gradual increase in temperature will alter the growing environment of many tree species throughout the United States, reducing the growth of some species (especially in dry forests) and increasing the growth of others (especially in high-elevation forests). Mortality may increase in older forests stressed by low soil moisture, and regeneration may decrease for species affected by low soil moisture and competition with other species during the seedling stage. Most models preject that species habitat will move upward in elevation and northward in latitude and will be reduced in current habitats at lower elevations and lower latitudes. New climatic conditions may “move” faster in some locations than tree species can disperse, creating uncertainty about the future vegetation composition of these new habitats.

The high genetic diversity of most tree species confers tolerance of a broad range of environmental conditions, including temperature variation. Therefore, in many species, tree growth and regeneration may be affected more by extreme weather events and climatic conditions than by gradual changes in temperature or precipitation. Longer dry seasons and multiyear droughts will often become triggers for multiple stressors and disturbances (e.g., fire, insects, invasive species, and combinations thereof). These pulses of biophysical disturbance will change the structure and function of ecosystems across millions of hectares over a short period of time, focusing pressure on the regeneration stage of forest ecosystems. Increased atmospheric carbon dioxide (CO_2) and nitrogen deposition will potentially alter physiological function and productivity of forest ecosystems, with considerable variation in response among species and regions.

The effects of climate change on water resources will differ by forest ecosystem and local climatic conditions, as mediated by local management actions. Higher temperature during the past few decades has already decreased snow cover depth, duration, and extent, a trend that will probably continue with further warming. Decreased snow cover will exacerbate soil moisture deficit in some forests, which may decrease tree vigor and increase susceptibility of forests to insects and pathogens. As climatic extremes increase and forest ecosystems change, water produced from forest lands may become more variable and of lower quality.

Forest growth and afforestation in the United States currently account for a net gain in carbon (C) storage, offsetting approximately 13 percent of the Nation's fossil fuel CO_2 production. During the next few decades, Eastern forest ecosystems are expected to continue to sequester C through favorable response to elevated CO_2 and higher temperature, although retention of C will depend on maintaining or increasing total forest area. Western forest ecosystems may begin to emit C if wildfire area and insect disturbance increase as expected.

Future changes in forest ecosystems will occur on both public and private lands and will challenge our ability to provide ecosystem services desired by society, especially as human populations continue to grow and demands for ecosystem services increase. Climate change effects in forests are likely to cause losses of ecosystem services in some areas (e.g., timber production, water supply, recreational skiing), but they may improve and expand ecosystem services in others (e.g., increased growth of high-elevation trees, longer duration of trail access in high-snow regions). Some areas may be particularly vulnerable because current infrastructure and resource production are based on past climate and the assumption of steady-state natural resource conditions. Any change in forest ecosystems that affects water resources will typically result in a significant loss of ecosystem services.

Effects of Disturbance Regimes

The most rapidly visible and significant short-term effects on forest ecosystems will be caused by altered disturbance regimes, often occurring with increased frequency and severity. Interacting disturbances will have the biggest effects on ecosystem responses, simultaneously altering species composition, structure, and function. The type and magnitude of disturbances will differ regionally and will pose significant challenges for resource managers to mitigate and reduce damage to resource values:

- Wildfire will increase throughout the United States, causing at least a doubling of area burned by the mid-21st century.
- Insect infestations, such as the current advance of bark beetles in forests throughout the Western United States and Canada, will expand, often affecting more land area per year than wildfire.
- Invasive species will likely become more widespread, especially in areas subject to increased disturbance and in dry forest ecosystems.
- Increased flooding, erosion, and movement of sediment into streams will be caused by (1) higher precipitation intensity in some regions (e.g., Southern United States), (2) higher rain:snow ratios in mountainous regions (western mountains), and (3) higher area burned (western dry forests). These increases will be highly variable in space and time, affecting decisions about management of roads and other infrastructure, as well as access for users of forest land.
- Increased drought will exacerbate stress complexes that include insects, fire, and invasive species, leading to higher tree mortality, slow regeneration in some species, and altered species assemblages.

Managing Risk and Adapting to Climate Change

A risk-management framework for natural resources identifies risks and quantifies the magnitude and likelihood of environmental and other effects. Although risk management frameworks have been used (often informally) in natural resource management for many years, it is a new approach for projecting climate change effects, and some time may be needed for scientists and resource managers to feel comfortable with this approach. Risk assessment for climate change must be specific to a particular region and time period, and it needs to be modified by an estimate of the confidence in the projections being made.

Ecosystem services derived from forests are produced in (1) rural areas, where human population densities are low and forest cover dominates; (2) urban settings, where trees may exist in low densities but provide high value for direct ecosystem services; and (3) transition zones between rural and urban settings (wildland-urban interface [WUI]). Climate change will alter ecosystem services, perceptions of value, and decisions regarding land uses. Outcomes for people will be determined by the interaction between changes in biophysical environments (e.g., climate, disturbance, and invasive species) and human responses to those changes (management and policy). In recent years, C sequestration policy options and increased use of bioenergy emphasize both climate change-human interactions on forests and the role of forests in mitigating climate change.

Land use shifts in rural areas under climate change could involve conversion of forests to agricultural uses, depending on market conditions. Climate change is expected to alter productivity (local scale) and prices (market scale). The extent of WUI areas and urban areas are projected to increase, often at the expense of rural forests. Higher temperature coupled with population growth will increase the extent and value of urban trees for mitigating climate change effects, but these two factors may also increase the difficulty of keeping trees healthy in urban environments.

The ability of communities with resource-based economies to adapt to climate change is linked to their direct exposure to these changes, as well as to the social and institutional structures present in each environment. Human communities that have diverse economies and are resilient to change today will also be better prepared for future climatic stresses, especially if they implement adaptation strategies soon. Federal agencies have made significant progress in developing scientifically based principles and tools for adapting to climate change. These tools and techniques are readily available in recent materials that can be supplied to public, private, and tribal land owners and managers for their use in forest management.

Regional Effects of Climatic Variability and Change

The report incorporates a regional perspective and highlights key issues for the forest sector in the NCA regions.

Alaska

- Alaskan forests are regionally and globally significant, and changes in disturbance regimes will directly affect the global climate system through greenhouse gas emissions and altered surface energy budgets.
- Climate-related changes in Alaskan forests have societal consequences, because some forests are in proximity to (urban and rural) communities and provide a diversity of ecosystem services.
- In interior Alaska, the most important effects of climate change are permafrost thawing and changes in fire regimes.
- South-central Alaska will be sensitive to climate change because of its confluence of human population growth and changing disturbance regimes (insects, wildfire, invasive species).
- In southeast Alaska, climatic warming will affect forest ecosystems primarily through effects on precipitation (i.e., snow versus rain).

Hawaii and the Pacific Territories

- Pacific islands are vulnerable to climate change because of (1) the diversity of climate-related stressors; (2) low financial, technological, and human resource capacities to adapt to or mitigate projected effects; and (3) diverse and often more pressing concerns affecting island communities.

- Island societies and cultures based on traditional knowledge and institutions have provided resilience to these communities during past stressful periods. Contributing to resilience are locally based ownerships and management, subsistence economies, tight linkages between landowners and government, and opportunities for migration.
- The direct effects of changing climate on forests will be significant and strongly dependent on interactions with disturbances, especially novel fire regimes that are expanding into new areas because of flammable invasive species.
- For low-lying islands, enhanced storm activity and severity and sea level rise will cause the relocation of entire communities, with the first climate refugees already having to relocate from homelands in the region. For high islands, higher temperature, expanded cover of invasive species, and higher fire frequency and severity will affect ground-water recharge, downstream agriculture, urban development, and tourism.

Northwest

- Based on projections of distribution of tree species and forest biomes, widespread changes in the distribution and abundance of dominant forest species are expected, although the results of modeling studies differ. Forest cover will change faster via disturbance and subsequent regeneration responses, rather than through slow adjustment to gradual warming.
- 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.
- Area burned and biomass consumed by wildfire will greatly increase, leading to changes in ecosystem structure and function, resource values in the WUI, and expenditures for fire suppression and fuels management.
- A combination of higher temperature and dense, low-vigor stands have increased vulnerability to bark beetles and other insects, and mortality is currently high in some dry forests.

Southwest

- Disturbance processes facilitated by climatic extremes, primarily multiyear droughts, will dominate the effects of climatic variability and change on both short- and long-term forest dynamics.
- Although diebacks in species other than pinyon pine (*Pinus edulis* Engelm.) are not widespread, large fires and insect outbreaks appear to be increasing in frequency and spatial extent throughout the Southwest.
- Increased disturbance from fire and insects, combined with lower forest productivity at most lower elevation locations, will result in lower C storage in most forest ecosystems. The fire-insect stress complex may keep many low-elevation forests in younger age classes in perpetuity.
- Increased 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.

Great Plains

- Trees occur along streams, on planted woodlots, as isolated forests such as the Black Hills of South Dakota, and near the biogeographic contact with the Rocky Mountains and Eastern deciduous forests, providing significant value in riparian areas, at higher elevations, and within agroforestry systems.
- Tree species in mountainous regions are expected to gradually become redistributed to higher elevations, with disturbances mediating rapid change in some locations.
- Climate-driven changes in hydrology are expected to reduce the abundance of dominant, native, early-successional tree species and increase herbaceous, drought-tolerant, late-successional woody species (including nonnative species), leading to reduced habitat quality for riparian fauna.

- The potential for increased wildfire hazard, longer droughts, insect outbreaks, and fungal pathogens, individually and in combination, could significantly reduce forest cover and vigor. Reduced tree distribution will likely have a negative effect on agricultural systems, given the important role of shelterbelts and windbreaks in reducing soil erosion.

Midwest

- Northern and boreal tree species at the southern edge of their current range will decrease in abundance and extent as their current habitat becomes less suitable (and moves northward) and reestablishment in a warmer climate becomes more difficult. Some forested wetlands may also disappear as the climate warms. Some oak and hickory species tolerant of low soil moisture may become more abundant.
- Increased drought and fire occurrence are expected to have rapid and extensive effects on the structure and function of forest ecosystems. Oak decline and invasive species are expected to become more common, contributing to stress complexes that include nearly two centuries of land use activities.
- Increased disturbance will tend to fragment forest landscapes that are already highly fragmented in terms of species, structure, and ownerships. This will reduce habitat connectivity and corridors for species movement.
- The large amount of private land and fine-scale fragmentation of forest landscapes will make it challenging to implement climate change adaptation. Outreach to private land owners will be necessary to ensure that climate preparedness is effective.

Northeast

- Stress complexes are especially important in northeastern forests, where climate interacts with nitrogen (N) deposition, tropospheric ozone, land use, habitat fragmentation, invasive species, insects, pathogens, and fire.
- A warmer climate will cause a major reduction of spruce-fir forest, moderate reduction of maple-birch-beech forest, and expansion of oak-dominated forest. Projections of change in suitable habitat 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.
- Warmer temperature will increase rates of microbial decomposition, N mineralization, nitrification, and denitrification, resulting in higher short-term availability of calcium, magnesium, and N for forest growth, as well as elevated losses of these nutrients to surface waters.
- Migratory bird species that require forest habitat are arriving earlier and breeding later in response to recent warming, with consequences for the annual production of young and their survival. Many bird species have already expanded their ranges northward.

Southeast

- Red spruce (*Picea rubens* Sarg.) and eastern hemlock (*Tsuga canadensis* [L.] Carrière), already declining in some areas, are projected to be extirpated from the southeast by 2100 as a result of the combined stresses of warming, air pollution, and insects.
- The majority of the Nation's pulp and timber supply is produced in the southeast, but if temperature continues to increase and precipitation becomes more variable, conditions for pine growth may begin to deteriorate. Even if regional forest productivity remains high, the center of forest productivity could shift northward into North Carolina and Virginia, causing significant economic and social impacts.
- Increasing demand for water from a rapidly growing urban population, combined with increased drought frequency could result in water shortages in some areas of the Southeast.
- Warmer temperature may increase decomposition of soil organic matter and emissions of CO_2, reducing the potential for C sequestration.
- Increased fire hazard and insect outbreaks will provide significant challenges for sustainable management of forests for timber and other uses, but may also motivate restoration of fire-tolerant longleaf pine (*Pinus palustris* Mill.) forests.

An Imperative for Action

Climate change will generally reduce ecosystem services because most human enterprises are based on past climatic environments and the assumption of static natural resource conditions. Increased forest disturbance will, at least temporarily, reduce productivity, timber value, and C storage, and, in some cases, will increase surface runoff and erosion. Changes in forest ecosystems that affect hydrology and water supply will typically result in a significant loss of resource value. Scientific principles and tools for adapting to these climate change effects focus on education, vulnerability assessment of natural resources, and development of adaptation strategies and tactics. The hallmark of successful adaptation efforts in the United States has been science-management partnerships that work collaboratively within public agencies and externally with various stakeholders. Several recent case studies of adaptation for national forests and national parks are now available and can be emulated by other land management organizations.

Although uncertainty exists about the magnitude and timing of climate change effects on forest ecosystems, sufficient scientific information is available to begin taking action now. Managing simultaneously for C and for on-the-ground implementation of adaptation plans is challenging in both public and private sectors; however, implementation can be increased through effective exchange of information and success stories. Land managers are already using "climate-smart" practices, such as thinning and fuel treatments that reduce fire hazard, reduce intertree competition, and increase resilience in a warmer climate. Building on practices compatible with adapting to climate change provides a good starting point for land managers who may want to begin the adaptation process. Establishing a framework for managing forest ecosystems in the context of climate change as soon as possible will ensure that a broad range of options is available for managing forest resources sustainably.

We are optimistic that a proactive forest sector will make the necessary investments to work across institutional and ownership boundaries by developing, sharing, and implementing effective adaptation approaches. This will be accomplished by (1) embracing education on climate science for resource professionals and the general public; (2) ensuring accountability and infusing climate change into organizational efforts (e.g., management plans and projects); (3) implementing an all-lands approach through collaboration across administrative, political, and ownership boundaries; and (4) streamlining planning processes and establishing projects on the ground. The twofold challenge of adapting to climate change and managing C in the broader context of sustainable forest management will require creativity by future generations of forest resource managers. In the short term, management strategies that are relatively inexpensive, have few institutional barriers, and produce timely results can be rapidly implemented. For adaptation, examples include reducing nonclimatic stressors in forests (e.g., nonnative pathogens), implementing fuel reduction, and reducing stand densities. For C management, examples include reducing deforestation, increasing afforestation, reducing wildfire severity, increasing tree growth, and increasing use of wood-based bioenergy. Specific strategies and actions will differ by location, inherent forest productivity, and local management objectives.

Coordinating adaptation and C management will help optimize implementation across specific landscapes. For example, fuel reduction treatments can reduce wildfire severity in dry forests (adaptation) and provide material for local bioenergy use (C management). In the near term, we anticipate that federal agencies will continue to lead the development of science-management partnerships and collaborative approaches to climate-smart management, although (static) legal, regulatory, and institutional constraints will continue to deter timely responses to (dynamic) climate-caused changes in forest ecosystems. Successful adaptation strategies and C management will likely accelerate across large landscapes as community-based partnerships integrate climate change-related concerns into sustainable stewardship of natural resources.

Contents

Chapter 1

Introduction

David L. Peterson and James M. Vose[1]

Projected changes in climate (temperature and precipitation means and extreme events), increased atmospheric carbon dioxide (CO_2), and increased nitrogen deposition are likely to affect U.S. forests throughout this century. Effects will be both direct (e.g., effects of elevated CO_2 on forest growth and water use) and indirect (e.g., altered disturbance regimes), and will differ temporally and spatially across the United States. Some of these effects may already be occurring. For example, large insect outbreaks and large wildfires during the past decade (Bentz et al. 2009, Turetsky et al. 2010) are a wake-up call about the potential effects of a rapidly changing climate on forest ecosystems. Individually and in combination, these two major disturbance phenomena are reshaping some forest landscapes and may be causing long-term, possibly permanent changes in forest structure, function, and species composition (Hicke et al. 2012, McKenzie et al. 2004). Combined with other stressors, such as invasive species and air pollution (McKenzie et al. 2009), and a legacy of fire exclusion and other land management activities, maintaining resilience and restoring forest ecosystems in the face of climate change will be a major challenge for the 21st century and beyond (Peterson et al. 2011).

In this document, we provide a scientific assessment of the current condition and likely future condition of forest resources in the United States relative to climatic variability and change. This assessment, which is conducted periodically by the U.S. Global Change Research Program as a component of a broader assessment of the effects of climate change on natural resources (U.S. Global Change Research Program 2012), is scheduled to be completed in 2013. The most recent assessment of the forest sector (Ryan and Archer 2008) provides a foundation and point of departure for this document. We focus on the latest observations of effects of climatic variability and change in forest ecosystems, supported by scientific literature, with emphasis on issues and solutions relevant for sustainable management of forest resources.

It is difficult to conclude whether recently observed trends or changes in ecological phenomena are the result of human-caused climate change or climatic variability. Regardless of the cause, we emphasize the response of forest resources to climatic patterns observed over the past few decades because they are similar to climatic phenomena expected for the rest of the 21st century. Compared to most of the 20th century, these more recent patterns are associated with periods of warmer temperature throughout the United States, and to multiyear droughts (low soil moisture) in arid and semiarid regions of the Western United States and many areas of the Eastern United States (Karl et al. 2009). For example, Breshears et al. (2005) concluded that dieback of pinyon pine (*Pinus edulis* Engelm.) in the Southwestern

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United States was caused by “global-change type drought.” If extended drought will indeed be more common in the future, then it is reasonable to infer that this type of dieback will also be more common. “Global-change type” climatic phenomena provide a reasonable context for projecting the effects of climate change on forest ecosystems.

In this document, we develop inferences from small-scale experiments (e.g., soil warming or CO_2 enrichment studies) and time series of natural resource data when available, while recognizing the challenges and uncertainties of scaling small-scale and site-specific studies in time and space (Peterson and Parker 1998). We also use the results of simulation modeling to project the effects of climate change on species distribution and abundance, ecosystem processes, ecological disturbance, and carbon dynamics. The results of both empirical (statistical) and process-based (mechanistic) models are presented, and we emphasize that these results are projections (proposed or calculated), rather than predictions (forecast or foretold about the future). Trends established by empirical data, combined with results from robust modeling, are a good combination on which to base inferences about climate change effects (e.g., Araújo et al. 2005). In this document, climate change effects are rarely projected beyond 2100, the limit for most current global climate models and emission scenarios (Solomon et al. 2007). We have high confidence in projections through the mid-21st century, beyond which agreement among global climate models diverges.

Forest ecosystems are inherently resilient to variability in climate at time scales ranging from daily to millennial. For example, forest species distribution and abundance have shifted over long time scales by responding individually to variability in temperature, precipitation (Brubaker 1986), and climatic influences on wildfire and other disturbance regimes (Prichard et al. 2009, Whitlock et al. 2008). Gradual changes in mean climate or atmospheric environment produce gradual changes in ecosystems. However, a rapid increase in temperature will increase the number of extreme climatic periods (e.g., extended droughts), leading to more frequent and intense ecological disturbances, which in turn lead to rapid change in the composition and dynamics of forests (McKenzie et al. 2009). Therefore, this assessment often focuses on extreme events and ecological disturbance, because these phenomena usually produce faster, larger, and more persistent changes than does a gradual increase in temperature.

Although the short-term effects of the El Niño-Southern Oscillation on natural resources have been well documented, the effects of dominant modes of climatic variability (Atlantic Monthly Oscillation, Pacific Decadal Oscillation, Pacific North American pattern) provide a better understanding about the potential effects of climate change, because periods of warmer (and cooler) and drier (and wetter) conditions are experienced over two to three decades at a time. For example, in some areas of the Western United States, the warm phase of the Pacific Decadal Oscillation is associated with more area burned by wildfire than in the cool phase (Hessl et al. 2004, Schoennagel et al. 2007). Studies of longer term modes of climatic variability thus provide a window into the nature of a permanently warmer climate, including quantitative relationships among temperature, precipitation, and area burned, on which projections of the effects of different climatic conditions can be based.

Forests that experience frequent disturbance often have characteristics that enhance their capacity to survive disturbance events (resistance) or facilitate recovery after disturbance (resilience) (Millar et al. 2007). Despite this inherent capacity, current thinking suggests that the rapid pace and magnitude of climate change will exceed the resistance and resilience capacity of many forests, and novel ecosystems without historical analogs will develop (Hobbs et al. 2009, Williams and Jackson 2007). A significant challenge for resource managers is to identify areas where forests are most vulnerable to change (i.e., have low resistance and resilience) and where the effects of change on critical ecosystem services will be greatest. Among the most obvious locations for vulnerable forest ecosystems (and species) are those near the limits of their biophysical requirements (Parmesan and Yohe 2003). However, the complexities of fragmented landscapes and multiple co-occurring stressors are likely to change response thresholds in many forest ecosystems (Fagre et al. 2009), with outcomes that may be unpredictable and unprecedented (Anderson et al. 2009, Scheffer et al. 2009). Under these conditions, traditional approaches to forest management that focus on historical conditions or protection of rare species or communities are likely to fail. Management approaches that instead anticipate and respond to change by guiding development and adaptation of forest ecosystem structures and functions will be needed to sustain desired ecosystem services and values across large landscapes and multiple decades (Millar et al. 2007, Seastedt et al. 2008). In this document, we discuss new management approaches along with specific tools and case studies. Uncertainty and risk are frequently discussed in this document, as mandated by general guidance for the National Climate Assessment. Important sources of uncertainty include short time series of climatological and forest effects data, limited spatial extent of many types of measurements, lack of understanding of complex ecological processes, and simulation models that cannot accurately represent a wide range of ecosystem dynamics. Risk is generally associated with the likelihood of exposure or effects at specific points in time, combined with the magnitude of the consequence of a particular biophysical change (Mastrandrea et al. 2010, Yohe and Oppenheimer 2011). Risk is inherently associated with human judgments and ranking (e.g., high, medium, low) and human values related to ecosystem services and perceptions (good vs. bad). When clearly articulated, in either qualitative or quantitative terms, uncertainty and risk are useful concepts for natural resource managers, decision-makers, and policymakers (Spittlehouse and Stewart 2003). Incorporating risk into discussions of climate change effects is relatively new for the forest resources community, but we are optimistic that doing so will improve our ability to apply climate change science to the management of forest ecosystems, including the development of adaptation options.

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Chapter 2

Effects of Climatic Variability and Change

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Introduction

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

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

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

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

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

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

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

Projected Changes in Future Climate

Scenarios for Projecting Future Climate

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

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

Temperature Projections

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

Box 2.1—Global climate models and emission scenarios

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

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

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

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

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

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

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

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

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

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

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

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

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

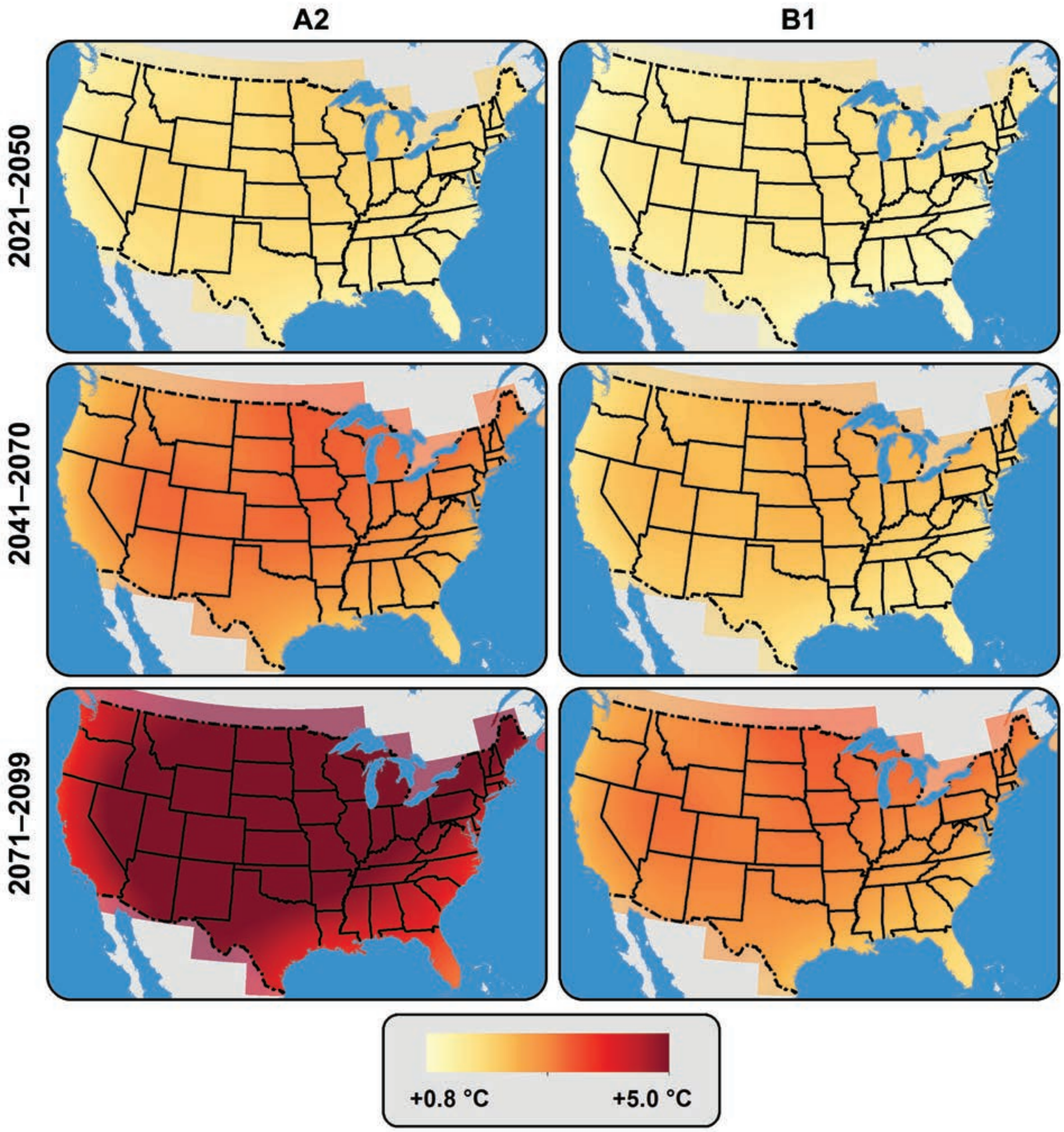

Figure 2.1—Multimodel mean annual differences in temperature between the three future periods compared to 1971 to 2000, from 15 global climate models using two greenhouse gas emission scenarios (A2 and B1). The A2 scenario is for higher greenhouse gas emissions than for B1 (see text). For most interior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

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

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

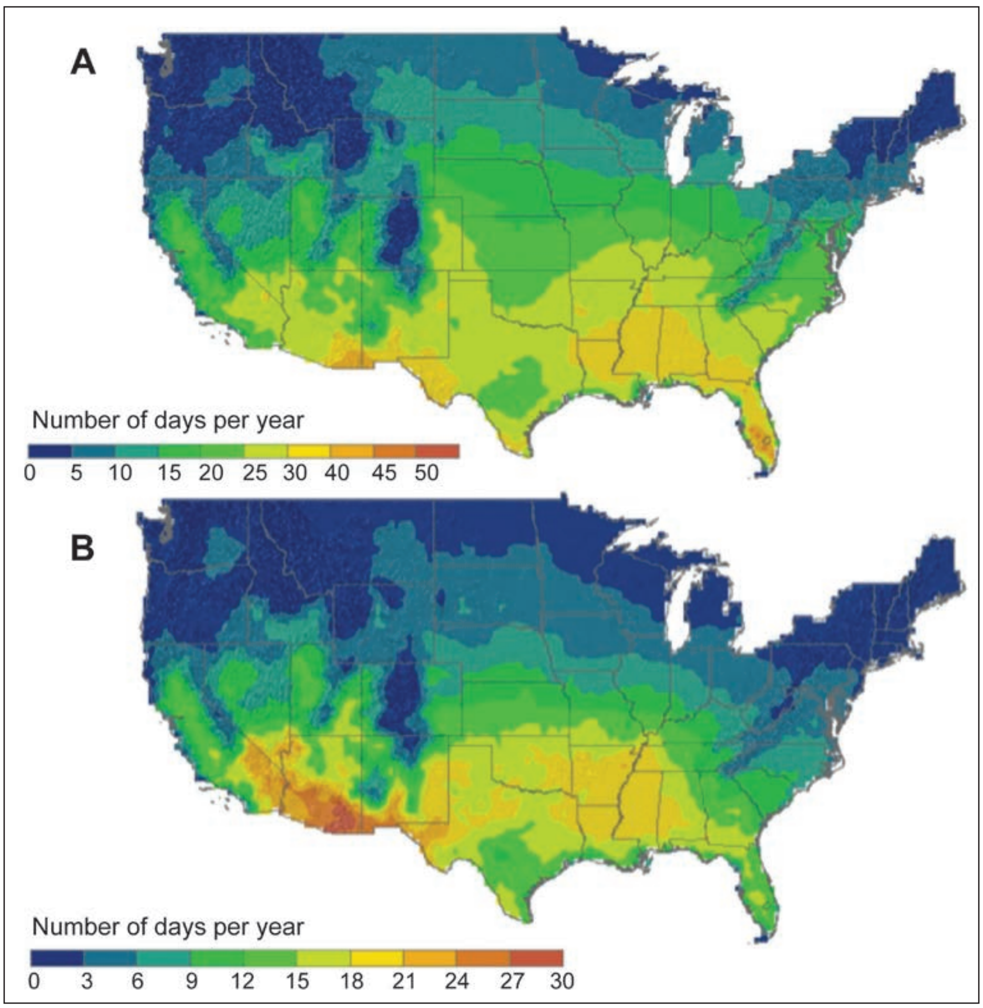

Figure 2.2—Spatial distribution of the mean change in the annual number of days with a maximum temperature above 35 °C (A), and in the annual number of consecutive days with a maximum temperature greater than 35 °C (B) between 1971 to 2000 and 2041 to 2070. Models project that much of the Southeastern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

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

Precipitation projections—

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

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

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

Drought projections—

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

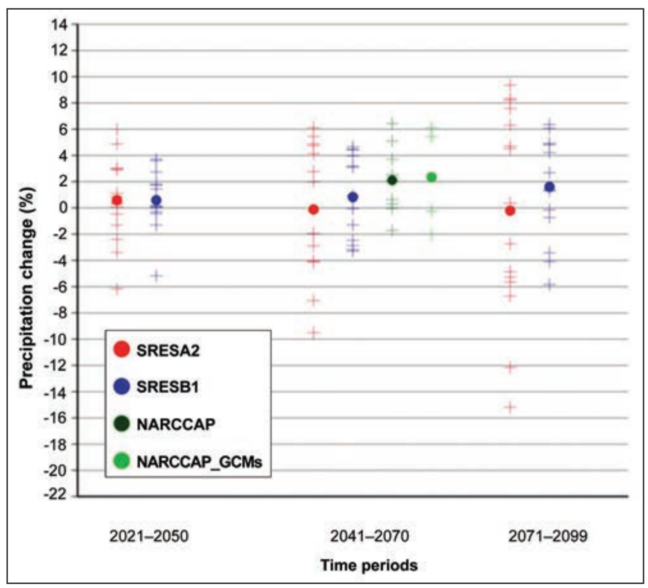

Figure 2.3—Mean annual percentage of precipitation change for three future time periods, relative to a 1971 to 2000 reference period. Little change in annual precipitation is projected for the continental United States as a whole, but individual model projections differ widely. Model projections for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

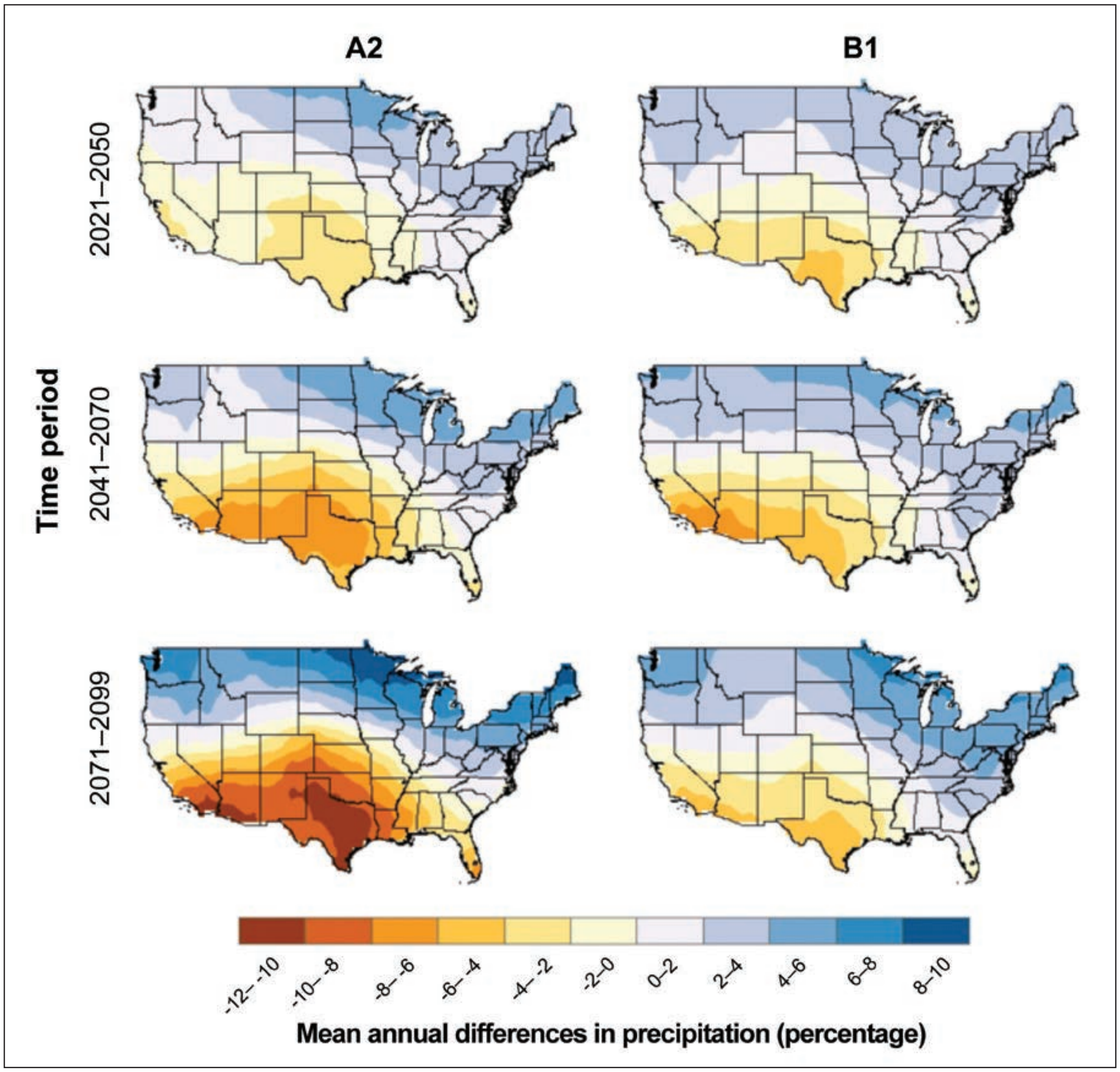

Figure 2.4—Mean percentage of annual differences in U.S. precipitation between three future periods relative to a 1971 to 2000 reference period. The Northeast, northern Midwest and Northwest are projected to have slightly more precipitation, and the Southwest is projected to have 2 to 12 percent less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

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

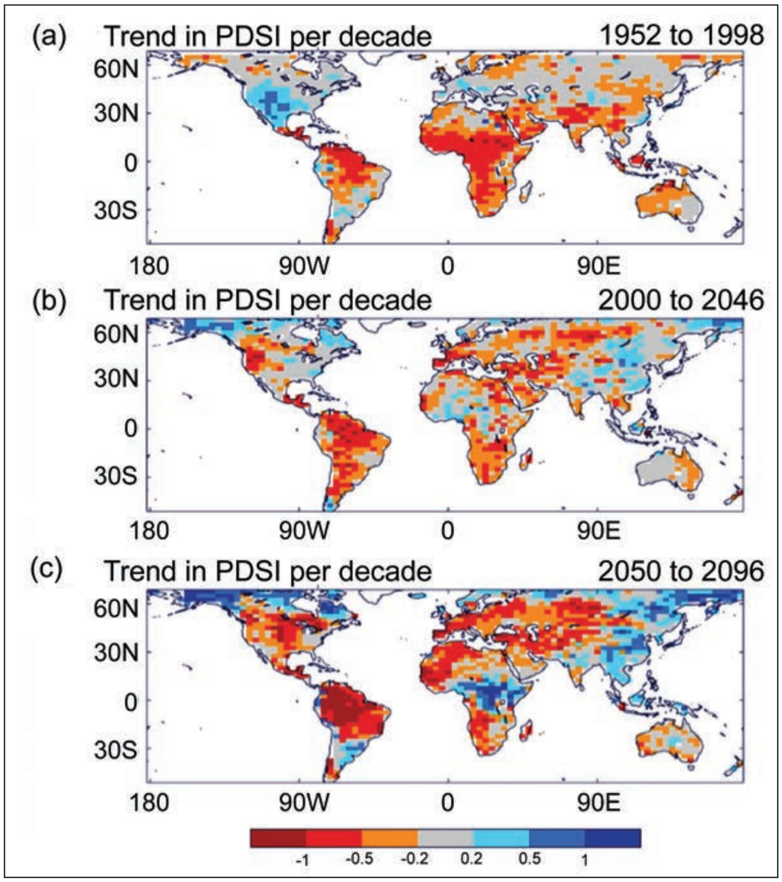

Figure 2.5—The trend in the Palmer Drought Severity Index (PDSI) per decade for (a) observed data and the mean of (b) the first half and (c) the second half of the 21st century. The PDSI is projected to decrease by 0.5 to 1 unit per decade for the period 2050–2096. For the PDSI, -1.9 to 1.9 is near normal,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

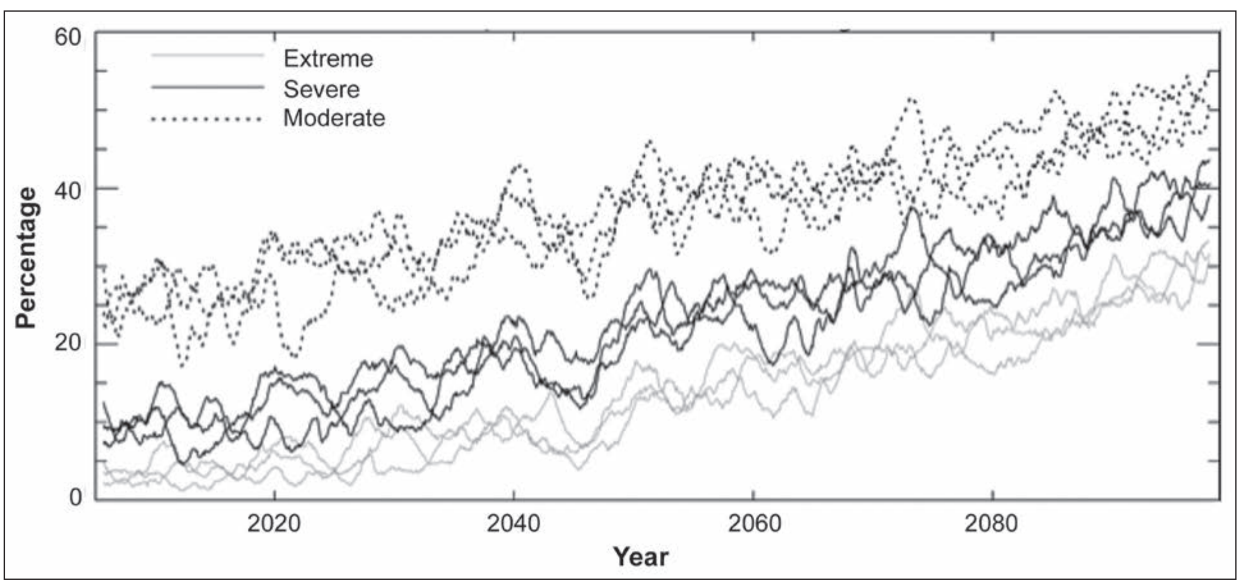

Figure 2.6—The projected average annual proportion of the global land surface in drought each month shows drought increasing over the current century. Drought is defined as extreme, severe, or moderate, which represents 1 percent, 5 percent, and 20 percent, respectively, of the land surface in drought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

Sea Level Rise

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

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

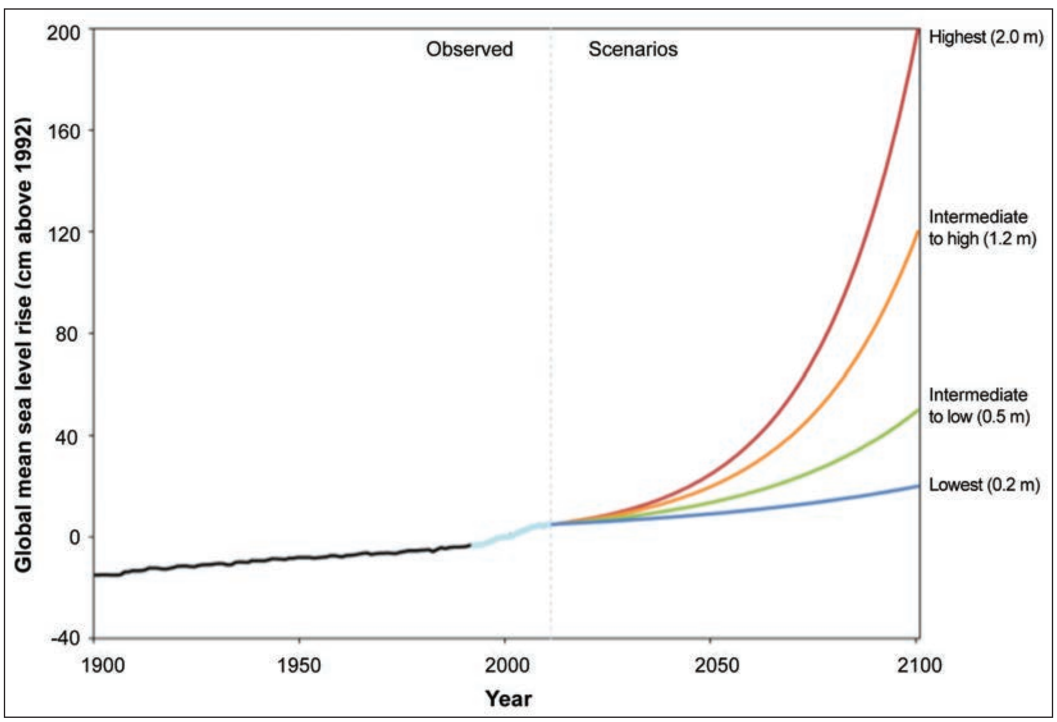

Figure 2.7—Four scenarios of projections of sea level rise from Parris et al. (2011) show sea level increasing from 0.2 to 2.0 m by 2100. The low scenario (dark blue line) is a quadratic extrapolation to the period 1990 to 2100 of historical trends in sea level rise over the entire period of observations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

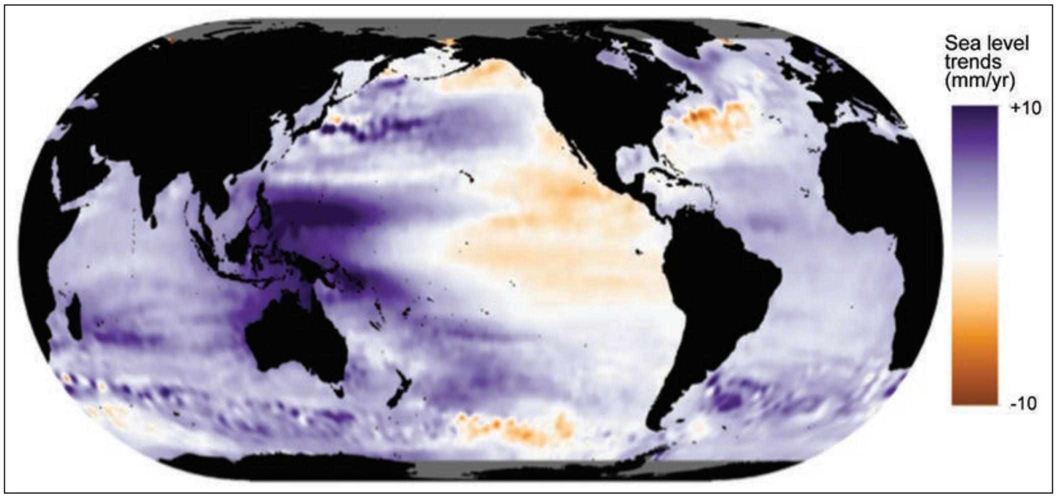

Figure 2.8—Geographic variability in the rate of global sea level change (1992–2010) based on three satellite records (TOPEX, Jason 1 and Jason 2) shows that little sea level rise occurred for the coastal United States during that period. Figure from NOAA Laboratory for Satellite Altimetry – Accessed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

Key Findings

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

Key Information Needs

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

Effects of Climate Change on Disturbance Regimes

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

Fire

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

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

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

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

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

Key Findings

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

Key Information Needs

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

Insects and Pathogens

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

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

General Concepts

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

Box 2.2—Mountain pine beetle and five-needle pines

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

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

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

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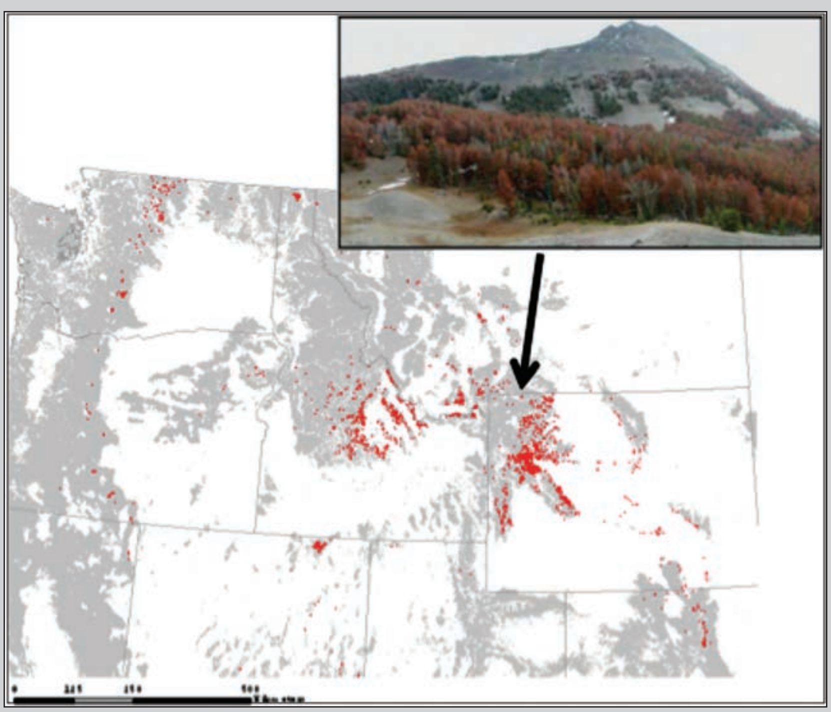
[1] *Greater Yellowstone Coalition v. State of Wyoming.* No. 09-361000, 10-35043, 10-35052, 10-35053, 10-35054. 16 U.S.C. 1533(a)(1)(D). (2011).

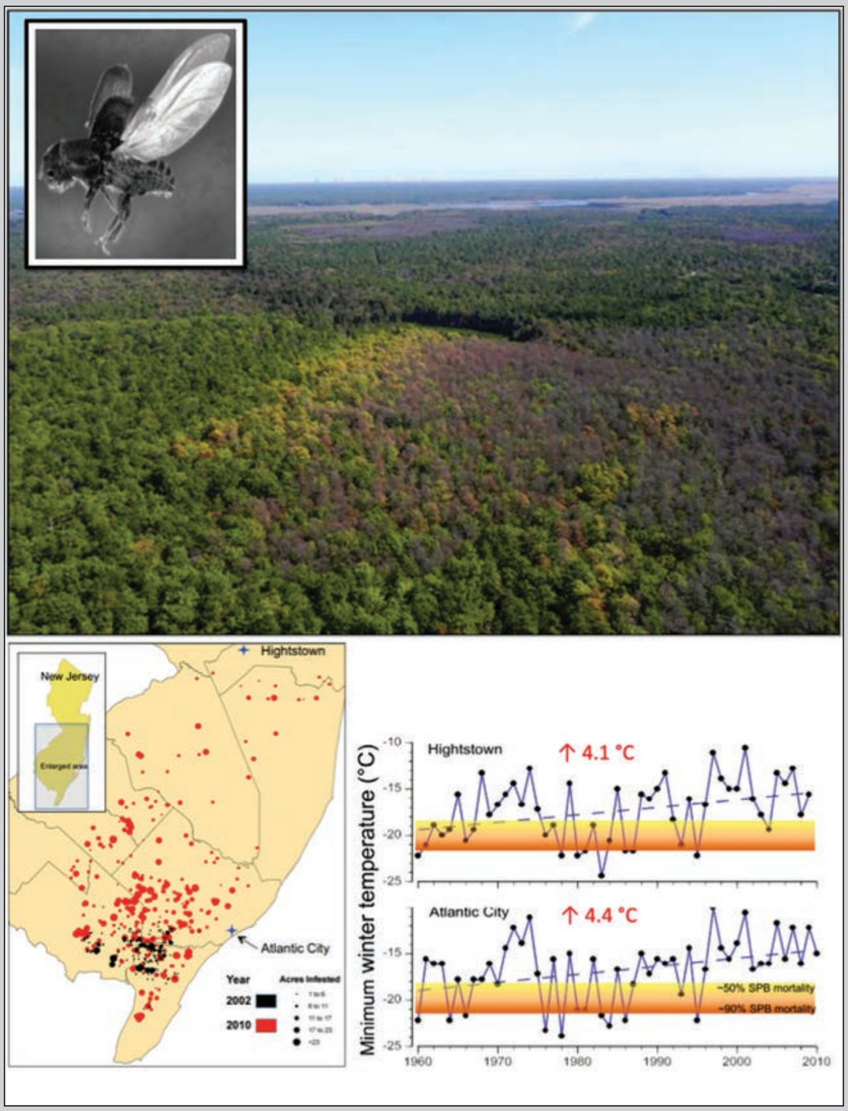


Area affected by mortality in stands of whitebark, limber, and bristlecone pine in 1997–2010 as detected by aerial surveys conducted by the USDA Forest Service. Affected area includes live and dead trees. Gray shading indicates locations of forest. Inset shows whitebark pine mortality in 2004 in Yellowstone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

Box 2.3—The southern pine beetle reaches New Jersey Pinelands

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

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



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

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

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

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

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

Climate and Biotic Disturbances

Bark beetles—

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

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

Box 2.4—Hemlock woolly adelgid

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

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

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

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

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

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

[a] Letters following references denote studies considering the effects of precipitation (P), temperature (T), or carbon dioxide (CO_2).
[b] Nonindigenous to North America.
Source: Updated from Ayres and Lombardero (2000).

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

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

— = none.

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

[b] Nonindigenous to North America.

Source: Updated from Ayres and Lombardero (2000).

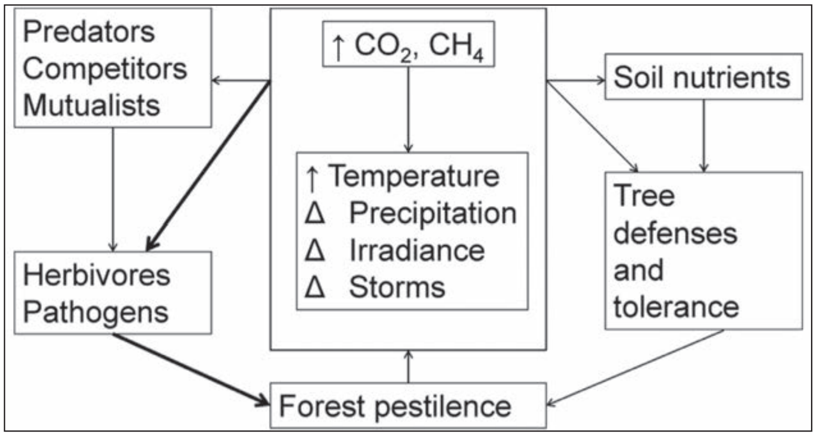


Figure 2.9—General pathways by which atmospheric changes associated with increasing greenhouse gases can influence forest disturbance from insects and pathogens. CO_2 = carbon dioxide, CH_4 = methane.

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

Defoliating insects—

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

Plant pathogens—

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

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

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

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

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

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

Impacts and Interactions With Other Disturbances

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

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

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

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

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

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

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

Future Vulnerabilities and Opportunities

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

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

Key Findings

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

Key Information Needs

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

Invasive Plants

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

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

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

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

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

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

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

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

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

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

Box 2.6—Invasive grasses, fire, and forests

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

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

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

Interactions Between Climate Change and Plant Invasion

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

Temperature, Precipitation, and Carbon Dioxide

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

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

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

Disturbance and Resource Availability

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

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

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

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

Biotic factors—

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

Propagule pressure—

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

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

Vulnerabilities—

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

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

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

Key Findings

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

Key Information Needs

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

Erosion, Landslides, and Precipitation Variability

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

Erosion and Landslides

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

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

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

Drought and Water Supply

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

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

Key Findings

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

Key Information Needs

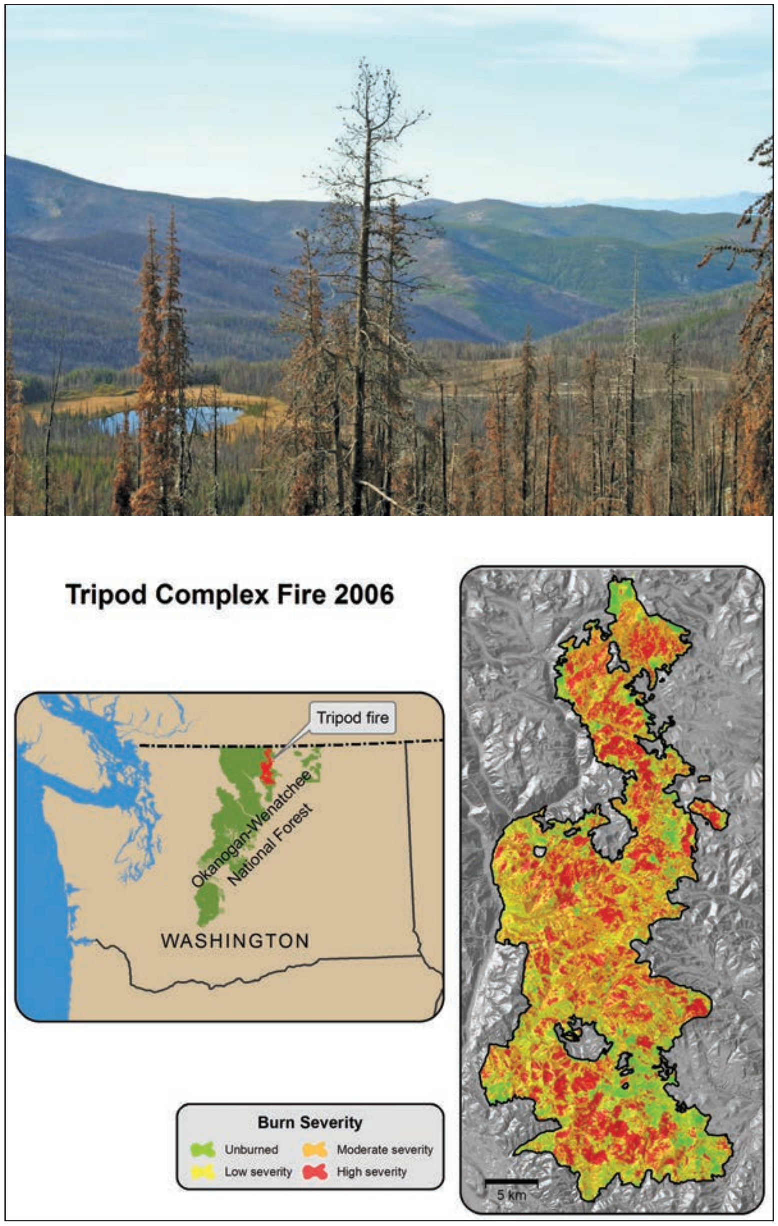
- Improved understanding of the effects of tree mortality and changing species composition on soil stability.
- Improved projections of changes in precipitation amount, and spatial and temporal distribution of extreme events.

Disturbance Interactions

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

Thresholds

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



Susan Prichard

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

Table 2.4—Characteristics of continuous versus abrupt thresholds

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

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

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

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

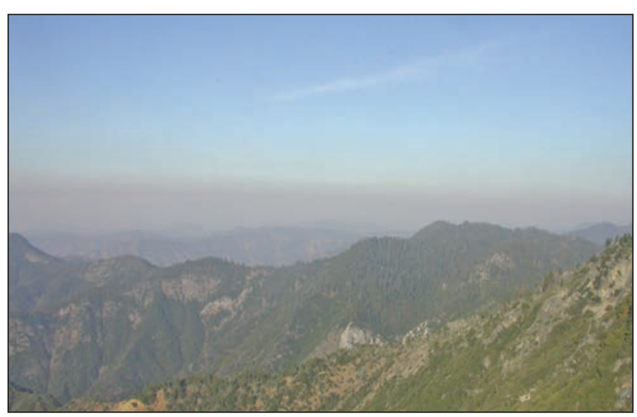
Stress Complexes: From Conceptual to Quantitative Models

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

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

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

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



D. McKenzie

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

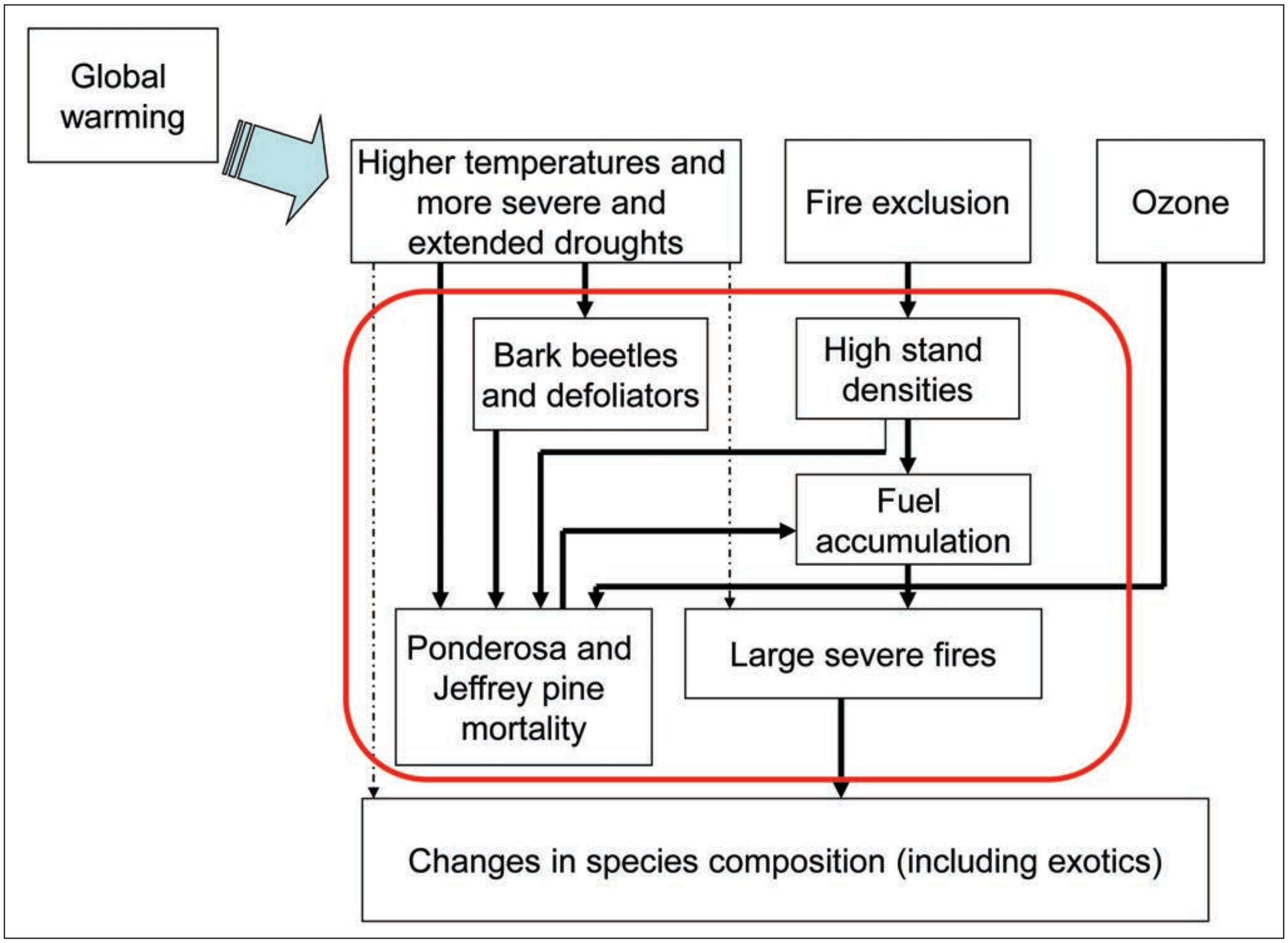


Figure 2.12—A conceptual model of stress complexes in Sierra Nevada and southern Californian mixed-conifer forests. The effects of insects and fire disturbance regimes (red box) and of fire exclusion are exacerbated by global warming. Stand-replacing fires and drought-induced mortality both contribute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

W. M. Ciesla

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

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

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

Uncertainties

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

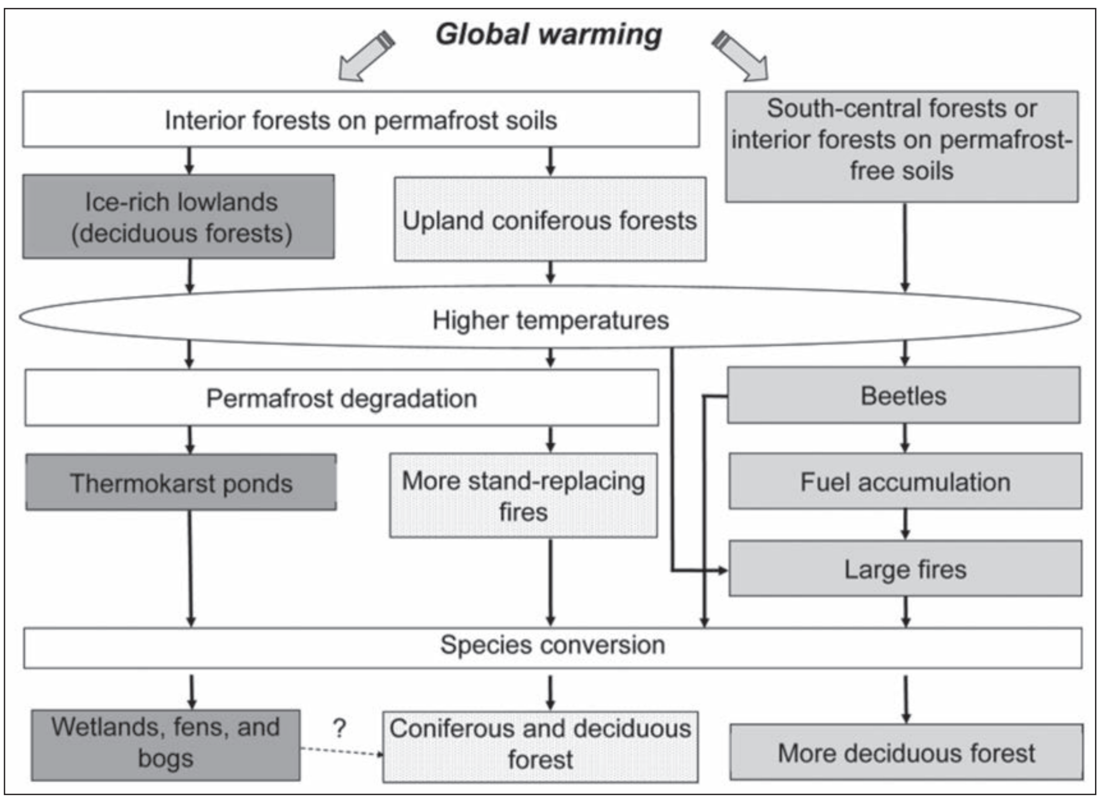


Figure 2.14—A conceptual model of stress complexes in the interior and coastal forests of Alaska. Rapid increases in the severity of disturbance regimes (insects and fire) are triggered by global warming. Stand-replacing fires, massive mortality from insects, and permafrost degradation contribute to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

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

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

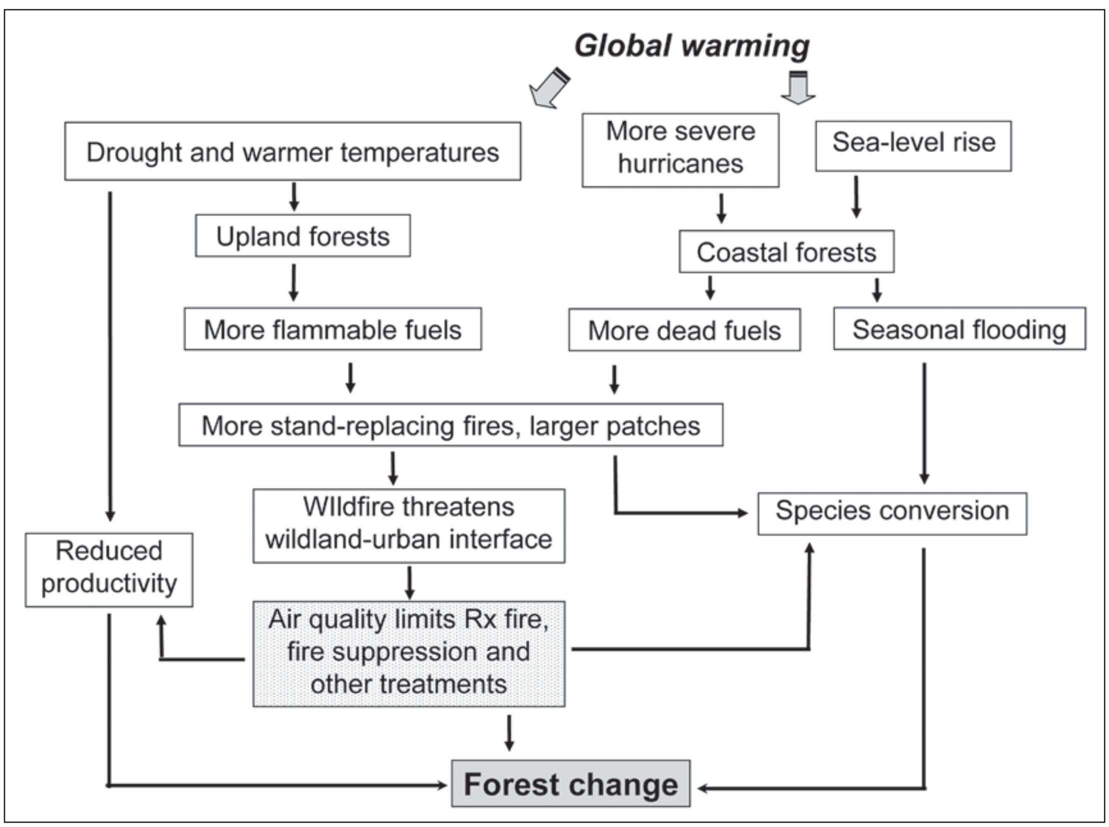


Figure 2.16—A conceptual model of stress complex in the interior and coastal forests of the Southeast. Increases in the severity of hurricanes are triggered by global warming, while sea level rises. Warmer and drier climate in uplands leads to longer periods with flammable fuels. Changes in fire and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

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

Key Findings

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

Key Information Needs

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

Effects of Climate Change on Forest Processes

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

Carbon and Nutrient Cycling

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

Effects on Eastern forests where precipitation is currently in excess—

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

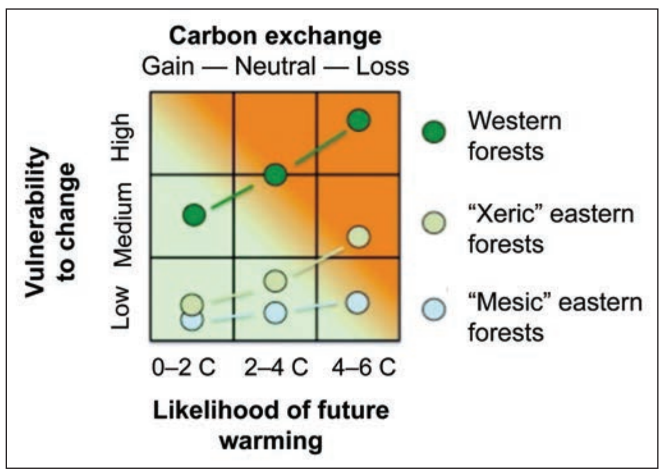


Figure 2.17—Risk analysis diagram for forest carbon cycle. Western forests are considered inherently limited by water demands that exceed precipitation supplies during substantial portions of the year. Xeric eastern forests include those growing on shallow or coarse-textured soils or those present at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

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

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

Effects on Nutrient Cycling

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

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

Key Findings

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

Key Information Needs

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

Forest Hydrological Processes

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

Forest Evapotranspiration and Streamflow

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

Elevated Atmospheric Carbon Dioxide

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

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

Warmer Temperatures and Drought

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

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

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

Changing Species Composition

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

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

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

Snowmelt

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

Soil Infiltration, Ground water Recharge, and Lateral Redistribution

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

Carbon and Water Tradeoffs

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

Key Findings

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

Key Information Needs

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

Forest Tree Species Distributions

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

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

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

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

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

Species Distribution Models

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

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

Process Models

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

Demography Studies

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

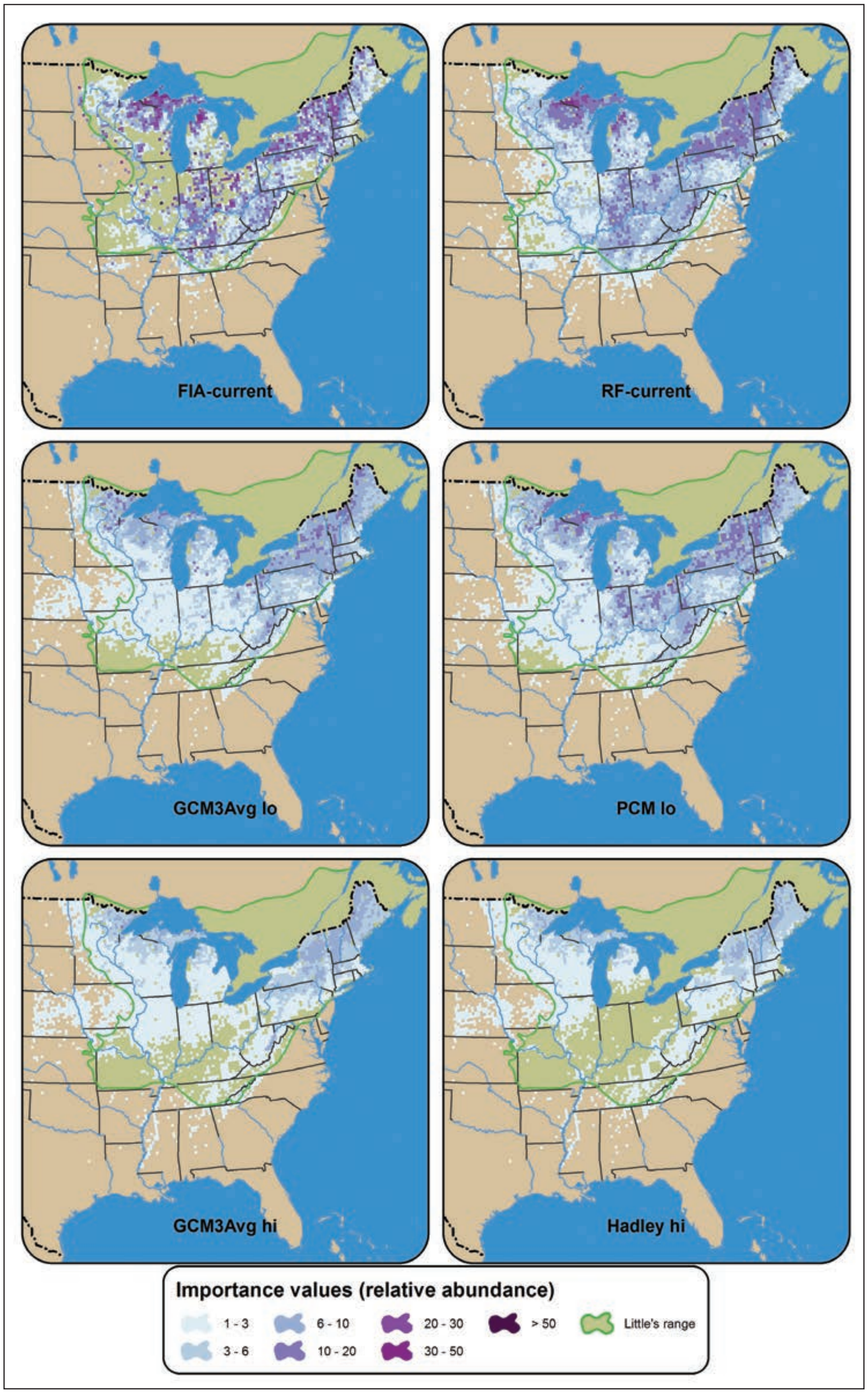


Figure 2.18—Maps of current and potential future suitable habitat for sugar maple show the expected northward migration of habitat with climate warming by 2100. The map includes the current inventory estimate of abundance from Forest Inventory and Analysis (FIA-current) sampling, the modeled current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

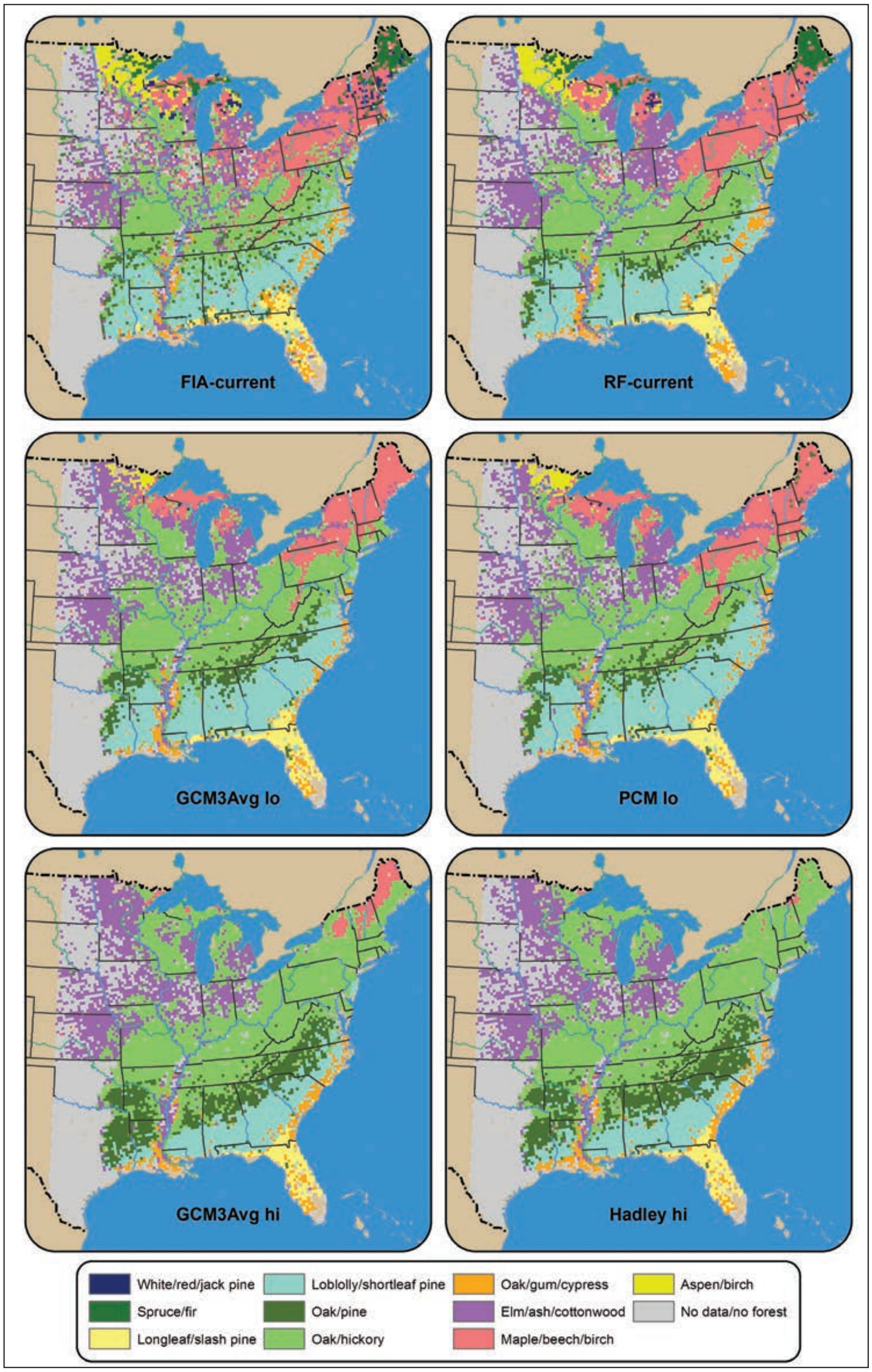


Figure 2.19—Maps of current and potential future suitable habitat for USDA Forest Service forest types (named according to dominant species) also show the major forests types moving northward with climate warming by 2100. The map includes the current inventory estimate of abundance from Forest Inventory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

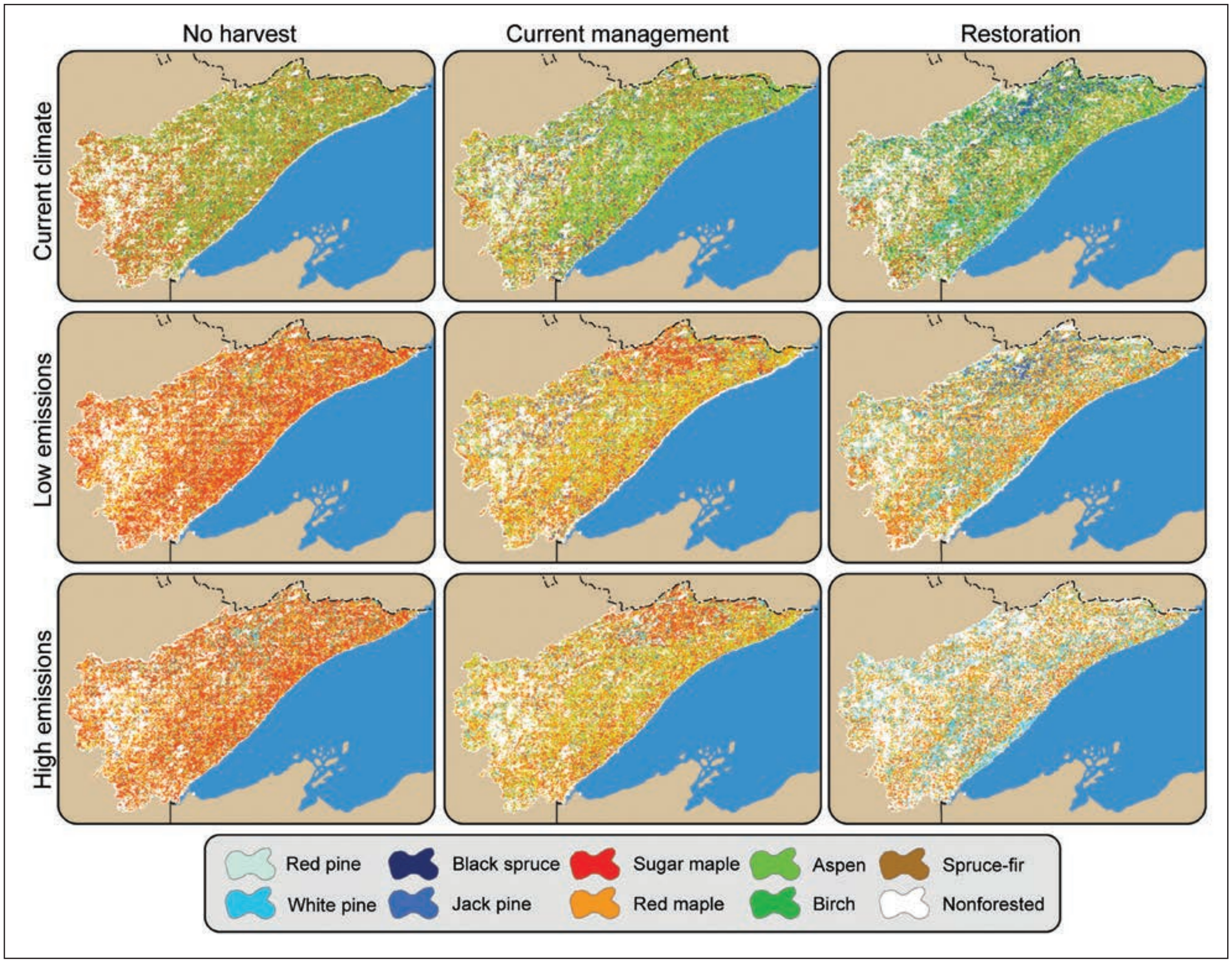


Figure 2.20—Maps of northern Minnesota forest types at year 2095 for nine scenarios (three emission levels × three management intensities). Sugar and red maple dominate the landscape by 2095 regardless of the emission scenario or management. Cells are classified to the forest type (named according to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

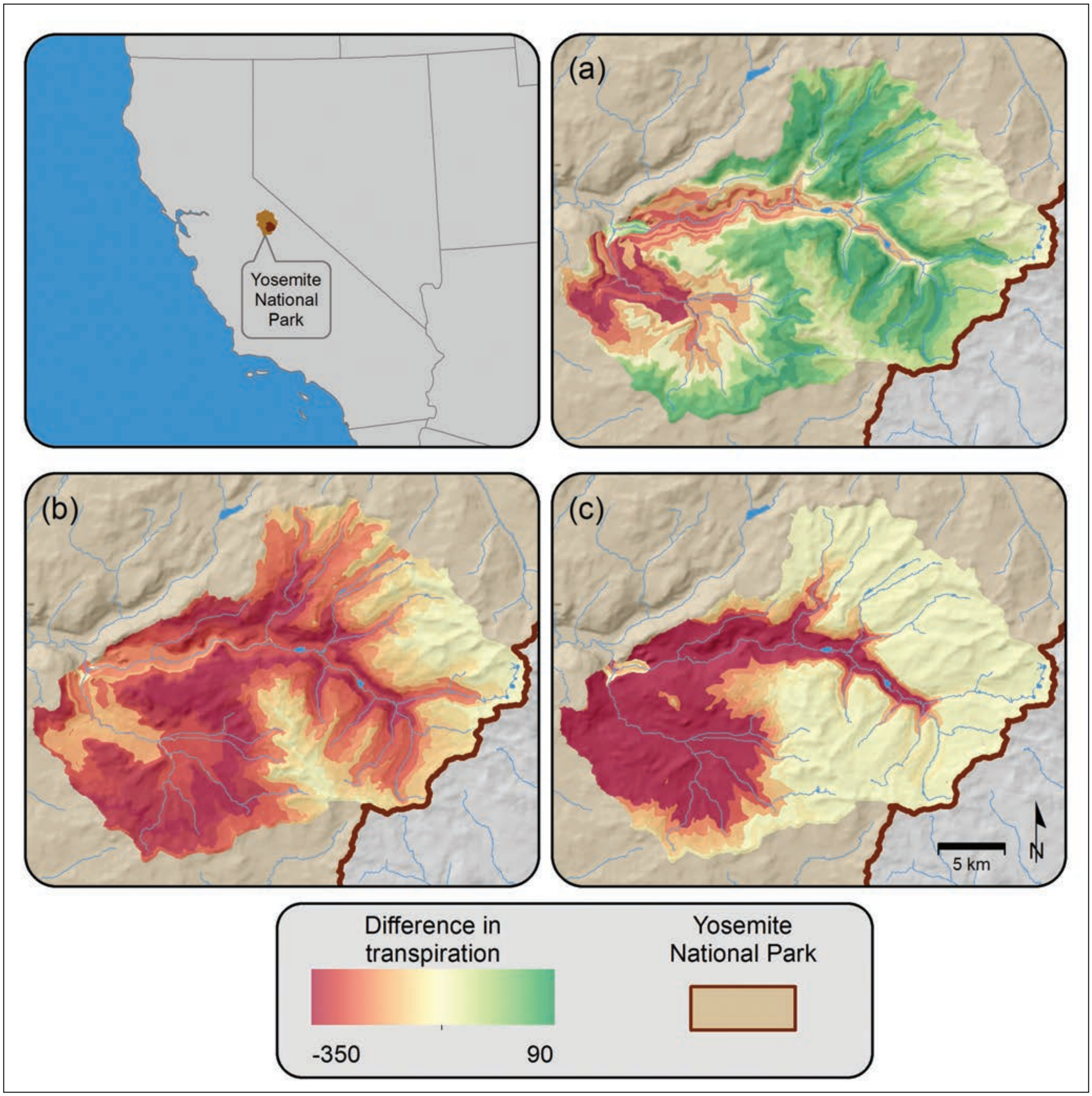


Figure 2.21—Maps of Upper Merced River watershed, Yosemite Valley, California, showing areas with greatest differences in transpiration between (a) warmest and coldest simulation years, (b) wet vs. average precipitation year, and (c) dry vs. average precipitation year. Greatest decreases in transpiration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

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

Dispersal and Migration Models

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

Assisted Migration

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

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

Key Findings

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

Key Information Needs

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

Effects of Altered Forest Processes and Functions on Ecosystem Services

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

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

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

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

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

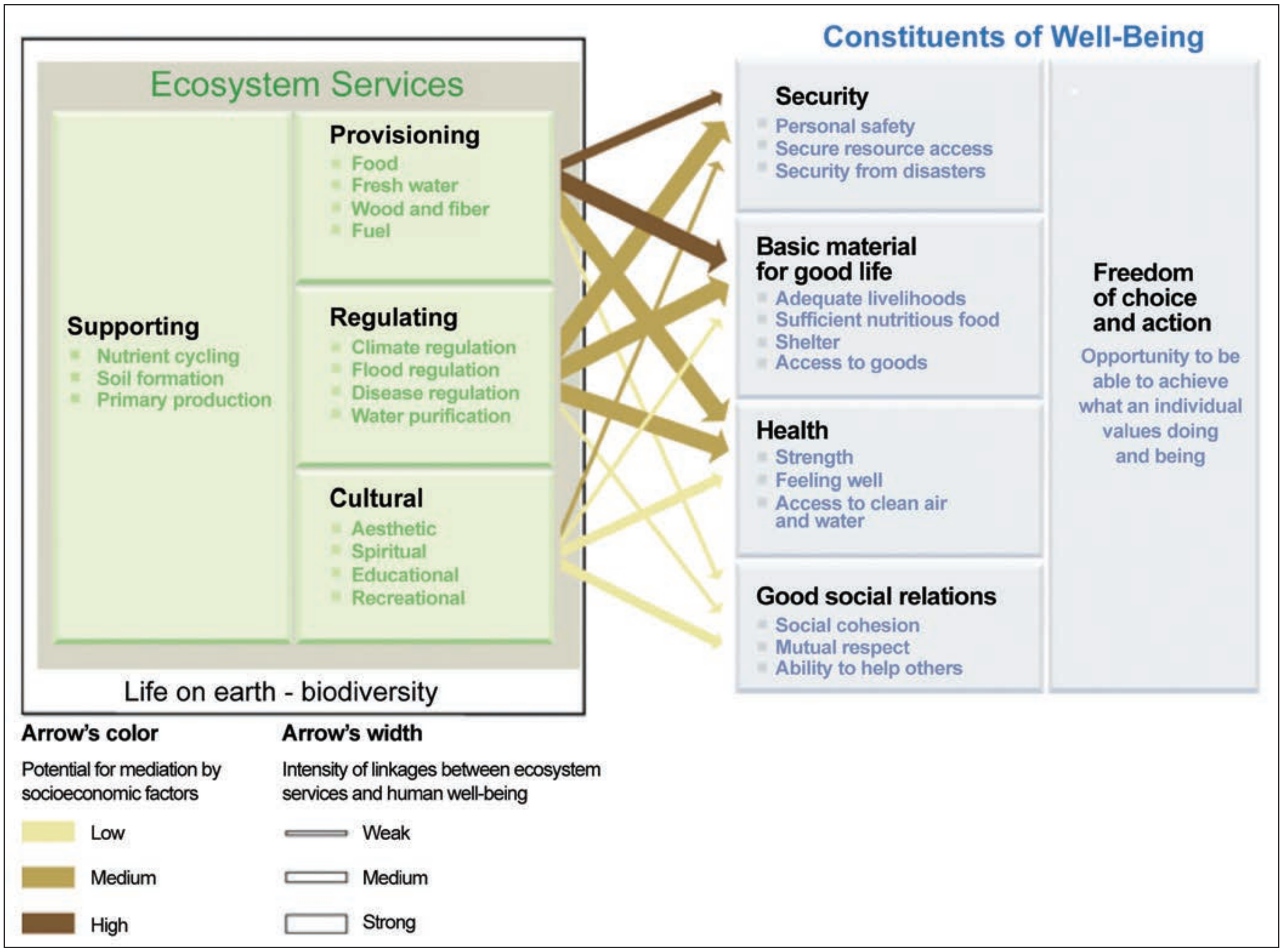


Figure 2.22—Linkages between ecosystem services and human well-being (World Resources Institute 2005).

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

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

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

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

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

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

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

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

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

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

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

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

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

Conclusions

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

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

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

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

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

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

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

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Chapter 3

Climate Change, Human Communities, and Forests in Rural, Urban, and Wildland-Urban Interface Environments

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Introduction

Human concerns about the effects of climate change on forests are related to the values that forests provide to human populations, that is, to the effects on ecosystem services derived from forests. Service values include the consumption of timber products, the regulation of climate and water quality, and aesthetic and spiritual values. Effects of climate change on ecological systems are expected to change service flows, people's perception of value, and their decisions regarding land and resource uses. Thus, social systems will adapt to climate changes, producing secondary and tertiary effects on the condition of forests throughout the world. This chapter explores how social systems might interact with changing climate conditions in determining the future of forested ecosystems in the United States.

Forests and derivative ecosystem services are produced and consumed in three types of environments. Most forested lands are in rural settings, where human population densities are low and forest cover dominates. In contrast, human populations dominate urban settings, where forests and trees may be scarce but their relative value, measured as direct ecosystem services, may be high. In urban areas within grassland biomes or in arid zones, tree cover may be highest where people live. Transition zones between rural and urban settings contain the wildland-urban interface (WUI), where forest settings comingle with human populations. These three settings pose different challenges for climate change-related resource management and policy, and each defines a unique set of opportunities to affect changes in forest conditions and service flows.

This chapter explores the interactions among forest condition, human value, policy, management, and other institutions, and the potential effects of these interactions on human well-being. We examine (1) the socioeconomic context (ownership structure, how value is derived, institutional context), (2) interactions between land use changes and climate change that affect forest ecosystems, and (3) social interactions with forests under climate change (climate factors, community structure, social vulnerability). In addition, forests have the potential to mitigate climate change through carbon (C) sequestration and through bioenergy production to substitute for fossil fuel energy. Hence, we also examine the potential influence of C mitigation on forest production, the forest economic sector, and forest land use.

Socioeconomic Context: Ownership, Values, and Institutions

In the United States, forest conditions and the flow of ecosystem services from forest land strongly reflect a long history of use and restoration as well as the influence of policy affecting both public and private forests (Williams 1989). Future forest management and policy, including responses to

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climate change, require an understanding of socioeconomic interactions with forests and how they might determine future conditions under different climate futures. Three key elements of the socioeconomic context of forests in the United States are (1) ownership patterns that define the institutional context of management, (2) forest contributions to human well-being through provision of various ecosystem services, and (3) the institutional settings that shape decisionmaking processes.

Forest Ownership Patterns in the United States

Forest owners, those who own and manage the land, comprise the individuals and groups most directly affected by, and most capable of mitigating, the potential impacts of global climate change on forests. Working within social and biophysical constraints, the owners ultimately decide the fate of the forest: whether it will remain forested, and whether and how it will be actively managed. Of the 304 million ha of forest land in the United States, 56 percent is privately owned by individuals, families, corporations, Native American tribes, and other private groups (fig. 3.1) (Butler 2008). The remaining forest land is publicly owned and controlled by federal, state, and local government agencies.

Ownership patterns differ significantly across the United States (fig. 3.2). In the East, where 51 percent of the Nation's forests are located, the extent of private ownership is much higher (81 percent) than in the West and in some states is as high as 94 percent. In contrast, the West is dominated by public, primarily federal, ownership (70 percent), with public forest ownership in some Western States as high as 98 percent (Butler 2008).

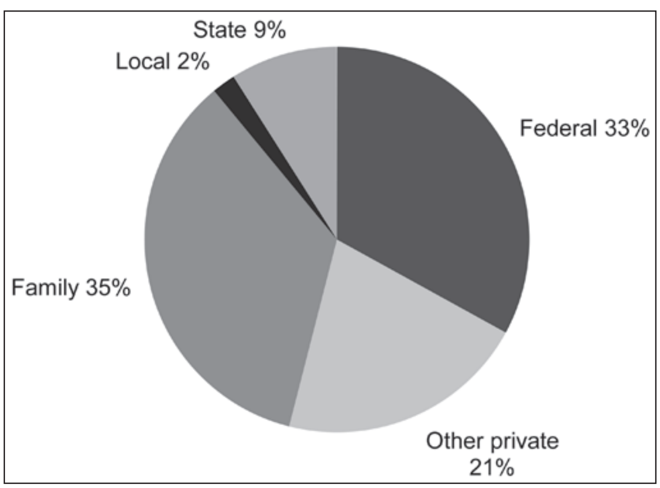

Figure 3.1—Forest ownership in the United States, 2006 (Butler 2008).

Public agencies have acquired land through various methods and manage them for diverse objectives. The federal government owns 33 percent of all forest land, with ownership dominated by the U.S. Forest Service (59 million ha) and the Bureau of Land Management (19 million ha). Other federal agencies with forest land holdings include the U.S. Fish and Wildlife Service, the National Park Service, and the Department of Defense. Public forests often have multiple uses, although one use may dominate at local scales (e.g., water protection, timber production, wildlife habitat, preservation of unique places, buffers for military exercises).

State agencies control 9 percent of all U.S. forest land, and county and municipal governments control 2 percent. Many state-owned forest lands are managed by forestry, wildlife, and park agencies. Other than military uses, most state and local uses mirror federal uses. Common objectives of many local land management agencies are water protection, recreation, and open space preservation.

Of the major forest ownership categories, families and individuals own a plurality (35 percent, 106 million ha) of the forest land in the United States. There are over 10 million of these ownerships, collectively called family forest ownerships.[3] The characteristics of their holdings differ, as do their reasons for owning them. Although most (61 percent) family forest ownerships are small (0.4 to 3.6 ha), 53 percent of the land in these ownerships is owned by those with 41 ha or more (fig. 3.3).

Most family forest ownerships own forest land for its amenity values, such as its beauty, legacy for future generations, and privacy. Financial motivations are not usually rated as important, although for a significant number of

[3] Defined by the U.S. Forest Service as families, individuals, trusts, estates, family partnerships, and other unincorporated groups of individuals who own forest land. The minimum forest holding size is 0.4 ha.

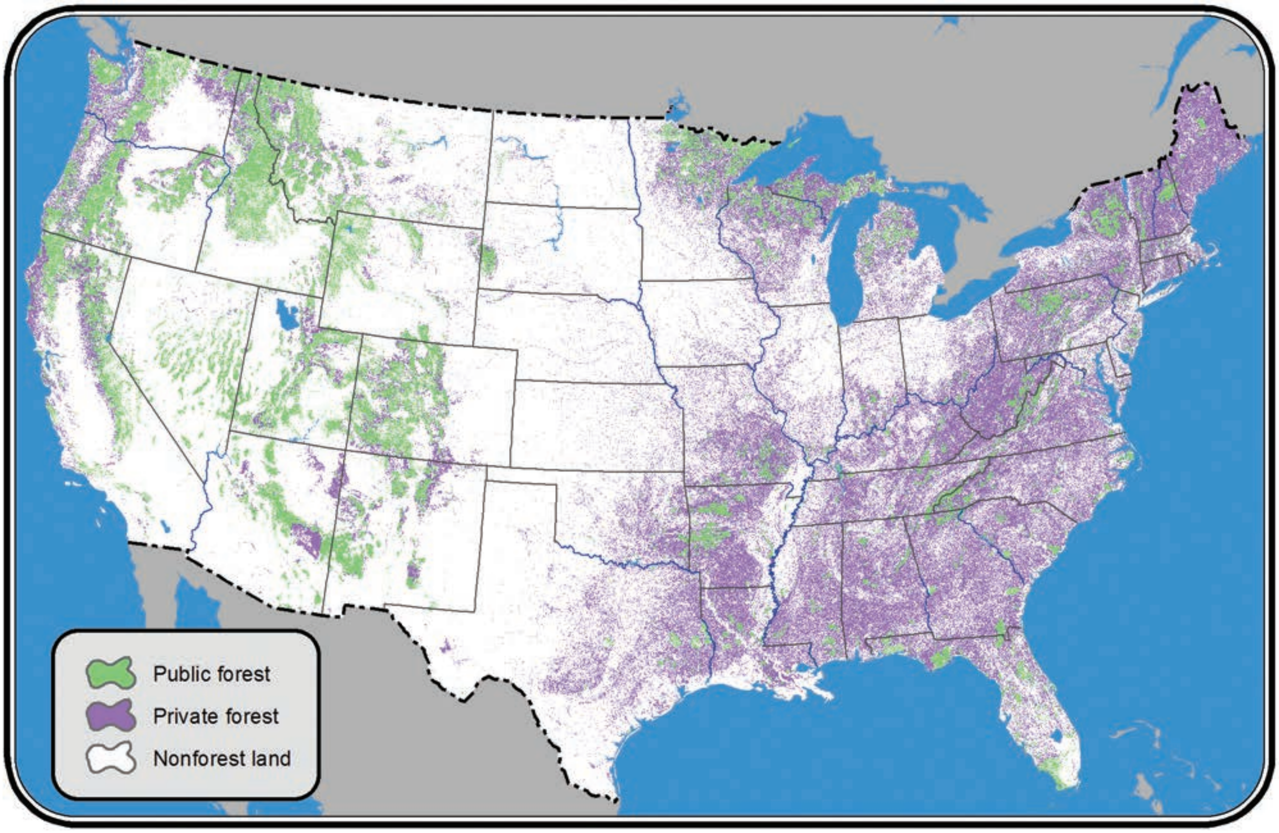

Figure 3.2—Distribution of public and private forest ownership in the United States.

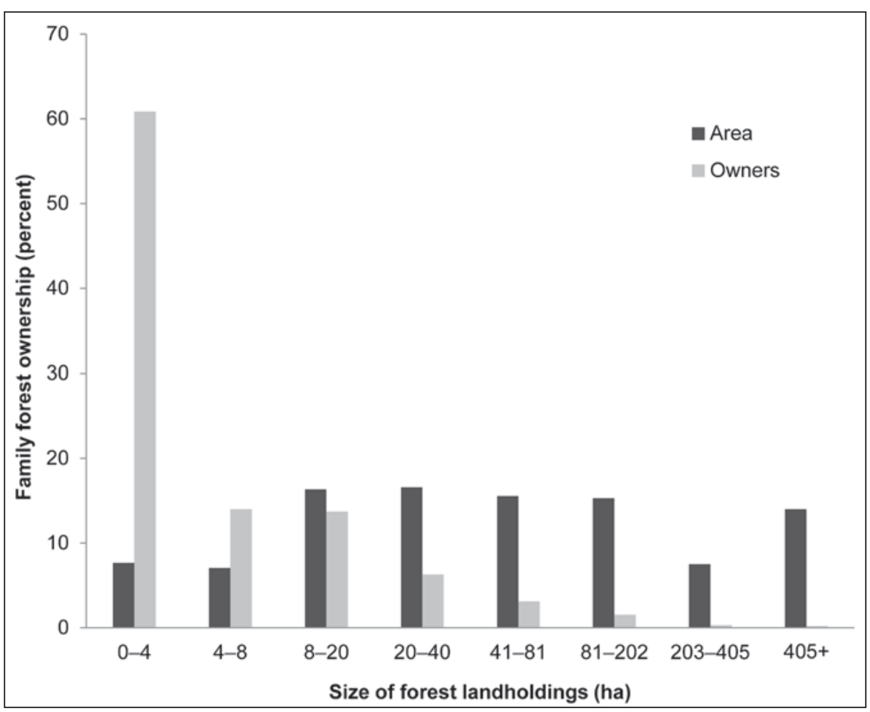

Figure 3.3—Family forest ownerships in the United States by size of forest holdings, 2006. (Butler 2008).

ownerships, especially with larger forest holdings, timber production and land investment are important.

Although timber production is not a primary ownership objective of most family forest owners, 27 percent of the family forest ownerships, owning 58 percent of the family forest land, have harvested trees. Few family forest owners have a written management plan (4 percent of family forest ownerships; 17 percent of family forest land), have participated in a cost-share program (6 percent; 21 percent), have their land green-certified (1 percent; 4 percent), or have a conservation easement on their land (2 percent; 4 percent) (Butler 2008). Nevertheless, evidence from landowner surveys indicates that most family forest owners have a strong land ethic and are conservation-minded (Butler et al. 2007).

Most other private forest land is controlled by corporations (56 million ha; 18 percent of all forest land). These include traditional forest industry and forest management companies, timber investment management organizations (TIMOs), and real estate investment trusts (REITs). Many other corporations also own forest land but do not have forest management as their primary ownership objective (e.g., utilities, mining companies, and those that happen to have forest acreage associated with a property, such as a manufacturing plant).

Native American tribes, nongovernmental organizations, clubs, and unincorporated partnerships control 8.5 million ha (3 percent) of the Nation's forest land. Some ownerships are explicitly for forest conservation (e.g., land trusts), others are largely for recreation (e.g., hunting clubs), and there are many other proposes.

From 1977 to 2007, U.S. forest land increased a net 8.9 million ha (4 percent) (Smith et al. 2009) (fig. 3.4). This increase occurred mostly in public, and in particular state, ownership. From 1997 to 2007, however, private forest land decreased a net 0.4 million ha. Over the next 50 years, U.S. forest land is projected to have a net loss of 9.3 million ha (Alig et al. 2003), mostly on private lands owing largely to urbanization.

Since the 1980s, the types of corporations that own forest land have undergone a major change. Traditionally, most corporate forest land was owned by vertically integrated

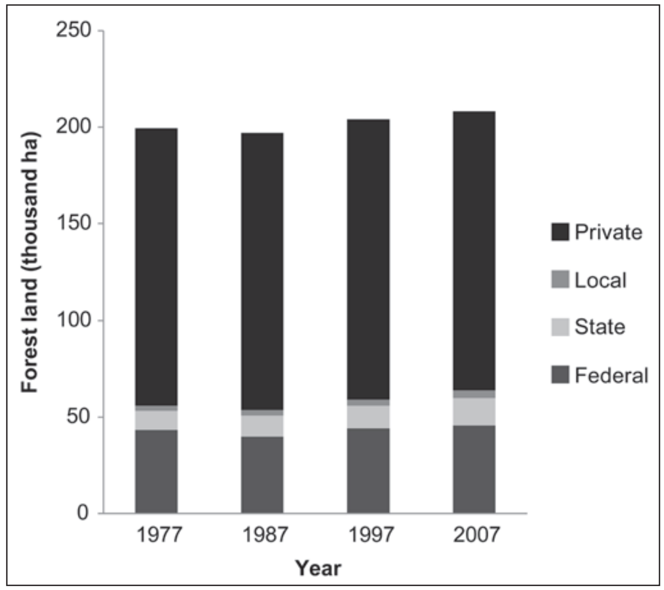

Figure 3.4—Trends in U.S. forest ownerships, 1977 through 2007 (Smith et al. 2009).

forest industry companies, which owned both forest land and the facilities to process the wood. Beginning in the 1980s and accelerating in the 1990s, most of these vertically integrated companies separated their forest holdings from their other assets, and many began to divest themselves of land. This decrease was paralleled by an increase in TIMOs and REITs. The vertically integrated companies were, at least in theory, more long-term-oriented and interested in supplying their mills from their lands. Conversely, TIMOs and REITs often have shorter investment time horizons and no need to supply mills, and hence they have different objectives.

Family forests have been undergoing parcelization, the dividing of larger parcels of land into smaller ones. If parcelization is accompanied by new houses, roads, or other changes, then forest fragmentation will increase, which in turn can harm ecosystem functions. Twenty percent of current family forest landowners are at least 75 years old, suggesting that a large amount of land will soon change hands. It is at this point of transfer that parcelization will probably occur, along with other changes in forest ownership objectives.

These forest ownership patterns have important implications for global climate change. It is especially notable that more than one-half of the forest land in the United States is currently owned by private landowners, thus these landowners could play a critical role in mitigating climate change effects. These ownership patterns, as well as the dynamics of change in forest land ownership, suggest the importance of engaging in a dialogue with landowners on the role of forest land management with respect to changes in both climate and land use. Such discussions might include the level of management necessary to sustain a suite of ecosystem services from forests (e.g., assisted migration of species, management of fire regimes) and to enhance the resilience of existing forest ecosystems. Policies that aim to mitigate the effects of climate change on forests must take into account the needs, desires, and resources of the owners.

Economic Contributions of Forests

Forest landowners have many reasons for owning forests, and forests deliver values in many forms to private landowners. Forests also provide a suite of ecosystem services that accrue to broader social well-being. For example, aesthetic values are usually not identified by the landowners as a monetary benefit of forest ownership. Likewise, forest owners may enhance wildlife habitat and use forest cover to protect watersheds without receiving financial returns.

In rural settings, forest cover can generally be equated with forest land use, because forests are a consequence of a decision either to dedicate land to growing trees, where other potential uses are not viable, or to allow land to return to a fallow condition. Rural forest ownership may provide direct returns, consumptive values, and monetary returns. Direct returns can accrue either through extractive activities (mainly commercial timber harvesting) or to the in situ value of forests (e.g., hunting leases, conservation easements). Consumptive values may accrue through direct use of forests for recreation, existence value, and aesthetics. Most monetary returns are generally confined to timber production, with some additional returns from recreation leases, conservation easements, and payments for other ecosystem services (generally through government programs such as state wetland mitigation programs guided by U.S. Environmental Protection Agency (USEPA) requirements under the Clean Water Act of 1977).

The United States produces more timber by volume than any other nation, and timber represents a significant source of value for forest landowners. Although the volume of roundwood used for industry and fuelwood nearly doubled between 1945 and the late 1980s, production since then has leveled off and declined (fig. 3.5). In 2006, the year before the latest recession, total timber production stood at about 90 percent of its peak value in 1988. The economic contribution of harvested timber has also declined. Between 1997 and 2006, the total value of shipments (the sum of net selling values of freight on board of all products shipped by the sector) fell by 7 percent, from a peak of $334 billion to $309 billion (Howard et al. 2010b).[4] Nearly the entire decline in the value of shipments over this period is explained by declines in the paper products sector. In 2006, production from the Eastern United States dominated this sector, representing 82 percent of the value of shipments in paper products industries and 74 percent of the value of shipments in wood products industries (79 percent of the total).

In contrast to declining production of wood products in the United States, consumption has been growing, implying increasing reliance on imported wood products. Consumption expanded from 0.37 billion m^3 in 1988 to about 0.57 billion m^3 in the 1990s and 2000s (Howard et al. 2010a). Although per capita consumption has been trending downward since the late 1980s, population growth has continued to push total consumption upward in recent years (fig. 3.6). Between 1957 and 2006, U.S. per capita consumption of wood products averaged 2 m^3 per person, peaking in the late 1980s (2.26 to 2.32 m^3 per person) and falling in the 2000s (1.95 m^3 per person). Nearly all the reduction over this period is explained by reductions in the consumption of fuelwood, leading to the conclusion that consumption levels of total wood and paper products in the United States have risen in direct proportion to population growth (Howard et al. 2010b).

[4] All dollar values in this section are measured as real 2005 dollars defined by the gross domesitic product price deflator.

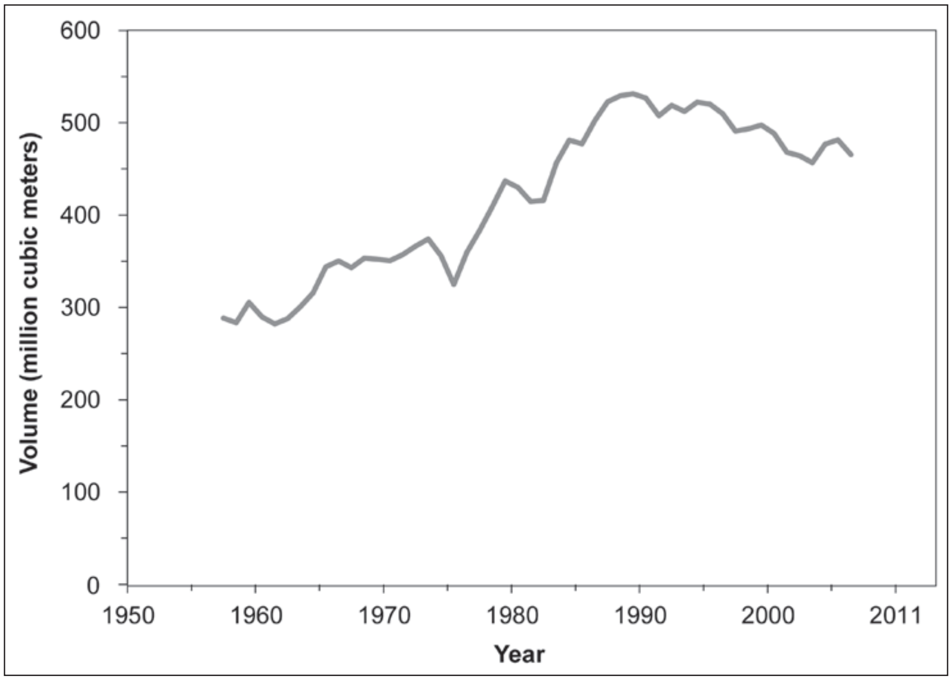

Figure 3.5—U.S. roundwood production, 1957 through 2006 (Howard et al. 2010a).

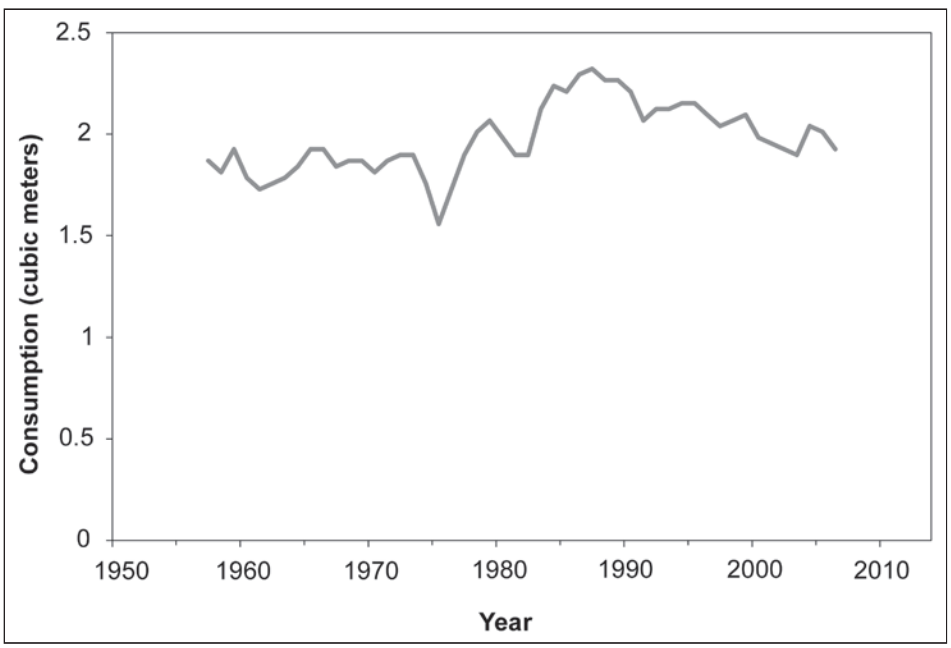

Figure 3.6—U.S. per capita apparent roundwood consumption, 1957 through 2006 (Howard et al. 2010b).

The value of U.S. timber production returned to forest landowners is difficult to assess because of data limitations. One estimate (USDA Forest Service 2011) puts this value at $22 billion in 1997, with 89 percent returned to private landowners. This is roughly 7 percent of the value of shipments for the wood products sectors (fig. 3.7). In 2006, the value of all wild-harvested nontimber resources was about $0.5 billion, and direct payments to landowners for forest-based ecosystem services was about $2 billion (USDA Forest Service 2011). Most payments for ecosystem services come from returns from conservation easements, hunting leases, and wetland mitigation banks. Total revenue to private forest landowners in 2006 was $20 million (about $119 ha^{-1}), representing an average capitalized value (at a discount rate of 5 percent) of about $2,347 ha^{-1} for all private forest land in the United States.

In rural settings, many ecosystem services from forest land provide benefits of forest ownership and use for which landowners are not compensated. For example, cultural values associated with forest areas—such as aesthetics, dispersed recreation, and spiritual needs—rarely lead to monetary compensation; nor do the benefits of protection of water quality and regulation of climate and flooding. Current policy initiatives (e.g., the 2008 Farm Bill [Food, Conservation, and Energy Act of 2008]) focus on providing payments, often through constructed markets, to compensate landowners for ecosystem services. An emerging area of engagement involves compensation from municipalities to landowners in municipal watersheds for activities that enhance or protect water quality (Brauman et al. 2007, Greenwalt and McGrath 2009). In urban settings, tree cover can affect environmental and aesthetic services for many people. Urban trees remove pollution, store C, and cool microclimates. Urban parks provide important recreation sites in the midst of human settlements. The forested area of the WUI is seen as an attractive environment in which to live, near rural or small-town settings. Here the extractive value of trees depends upon the size of the land and the landowner's preferences, but the trees in the WUI typically have little extractive value other than as fuel wood. The environmental and aesthetic services in the WUI differ from those of rural forests, because these environments are greatly influenced by the human activities in them.

Policy Context of Forest Management in Response to Climate Change

An institution is any rule or organization that governs the behavior of humans. In the context of forest management activities in response to climate change, relevant institutions include the structures of public and private land ownership,

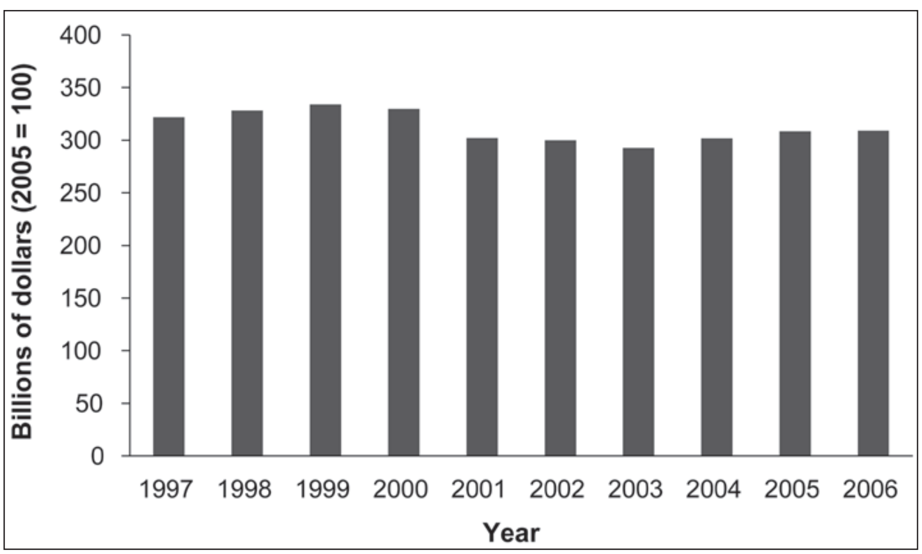

Figure 3.7—Value of shipments for forest products industries (NAICS codes 321, 322, and 337), 1997 through 2006 (Source: Howard et al. 2010a).

nongovernmental organizations addressing aspects of forest values, and policy instruments, such as forest management laws and taxes that influence land management decisions. Human behavior, expressed through land use decisions, is the dominant cause of landscape change; thus, institutions are crucial control mechanisms in determining future forest conditions and responses to climate change.

Forest management in the United States derives from the interaction of the two dominant institutional structures of private and public ownership. Private ownership affords extensive property rights held by autonomous landowners but is constrained by tax and regulatory policy. Under the right conditions, enlightened self-interest should guide landowners to allocate land to the highest-valued uses and, in the process, to effectively produce marketable goods and services. However, nonmarket ecosystem services, which deliver considerable value to society, are not likely to be fully valued in private transactions. On the other hand, the production of nonmarket goods is a primary rationale for public ownership of forest land (e.g., Krutilla and Haigh 1978). In theory, public land management aims to provide the "right mix" of all important goods and services.

Public ownership is not the only mechanism for providing nonmarket goods. Government can direct the actions of individual landowners toward producing other nonmarket benefits, by altering incentives (e.g., reforestation subsidies and severance taxes) and selectively restricting property rights (e.g., through forestry practice regulations). Nongovernmental organizations can directly affect changes in land use and resource allocation, through outright purchases of land or purchase of development or other rights using conservation easements. The use of all policy tools has its costs, including costs of both administration and of forgone market benefits. Balancing these regulatory costs against public benefits is a critical part of policy design.

Private and public forest ownerships offer up very different models of response to climate changes. For example, private forest owners might be expected to alter their management plans more rapidly in response to events such as observed or anticipated climate impacts, altered market signals (prices), or policy instruments that might affect the provision of nonmarket, ecosystem services. The extent of such a response is ultimately governed by the preference structure of private forest owners. Butler et al. (2007) found structural dissimilarities both between the objectives of corporate and family forest owners and among subgroups of the family forest owners. Still, the private forest sector has shown high responsiveness to market signals in harvesting timber and investing in future timber production, especially in the southeastern United States (Wear and Prestemon 2004). For example, the area of intensively managed pine plantations more than doubled in the South between 1990 and 2010 (Wear and Greis 2011) as production shifted from western to southern regions.

Overall, private forests in the United States have lower levels of forest inventory (reflecting a "younger" forest age class distribution), are generally more accessible, and are much more likely to be harvested or receive other forms of forest management. On the other hand, public management can attempt to maximize broad public welfare derived from forests and thus produce benefits not ordinarily provided by markets. By design, public forest management in the United States, especially at the federal level, can be slow to adjust, given the interplay between technical tradeoff analysis and an adversarial public process of resource planning (e.g., Wilkensen and Anderson 1987, Yaffee 1994). Overall, compared to private forests in the United States, public forests carry higher levels of forest inventory, are typically more remote, and are less likely to receive active management.

Future responses to climate signals, and especially to programs designed to mitigate greenhouse gas emissions, would probably be much larger on private lands, where market signals and direct policy instruments (incentives and disincentives) are readily translated into management actions. These responses could be in the form of increased harvesting (resulting from introduction of new product markets such as biofuels/bioenergy) and altered forest management (responding to demands for forest-based C storage), but they could also occur as forest area decreases or increases, depending on the comparative returns to land from forest and agricultural uses. As a result, we expect faster and larger policy impacts on greenhouse gas mitigation management activities to occur in the Eastern United States, where private ownership dominates and transportation and processing infrastructure for wood products are more extensive.

In the forest sector, policy responses to climate change have focused largely on mitigation actions that reduce either the use of fossil fuel, through bioenergy products, or the amount of carbon dioxide (CO_2) in the atmosphere, through C sequestration in forests. The use of woody biomass for the provision of energy could offset fossil fuels, substituting either ethanol for oil in transportation fuels or wood for coal in electricity production. In addition, C cap and trade initiatives focus on sequestering additional C through growth in forests and consumption of durable wood products. Potentially more significant, policies outside the forest sector could have secondary impacts on forest area and conditions, largely through land use changes. Federal agricultural policies, such as crop price support programs, likely affect the total area of cropland in production and therefore the area of forests. Local policies regarding land uses affect the rate at which forest land is converted to developed uses.

Interactions Among Forests, Land Use Change, and Climate Change

Land use changes are influenced by choices of landowners, market forces, and economic and environmental policies. These forces differ across the three types of forested environments (rural, urban, WUI). In rural environments, market forces influence shifts between agricultural uses and forest uses. Urban expansion converts forest land, with loss of some trees, and intensification of urban areas often leads to the loss of most trees. In the WUI, conversion of large forest tracts to residential areas is driven by home buyers who value the amenities of living in or near forests and are willing to pay more to do so. Such land use-induced changes in land cover can have local effects on climate, both temperature and precipitation (Fall et al. 2009). Hence, land use changes may interact with changes resulting from greenhouse gases and together strongly influence forest dynamics. This section focuses on understanding the nature of land use change and the role of landowner choices and institutional policies in retaining forest land cover under a changing climate.

Interactions Among Forests, Agricultural Land Use, and Climate Change in Rural Environments

United States forests produce more timber products than any other nation (United Nations Food and Agricultural Organization 2000), and U.S. agriculture produces 41 percent of the world's corn and 38 percent of the world's soybeans (Schlenker and Roberts 2009). History demonstrates a tradeoff between agricultural and forest uses in the United States, based on shifting advantages and returns between these two types of use. Both climate change and programs designed to subsidize land-based products may affect land use choices and the extent of forest area in the United States. Where people occupy rural areas, these areas will be affected not just by the dynamics influencing forest and agricultural use, but also by population growth.

Climate (e.g., precipitation amount and variability, air temperature, solar insolation, snow cover) is a key driver of agricultural productivity. Climate change could influence not only the returns to agriculture but also land use switching between crops, pasture, forest, and other uses. Modeling studies indicate that crop productivity is negatively related to temperature increases (for all seasons except fall) and positively related to nonfall precipitation (e.g., Mendelsohn et al. 1994). Climatic variability may also affect crop productivity (Mendelsohn et al. 2007). Estimates of potential climatic effects on productivity are influenced by how the model is specified. Schlenker and Roberts (2009) investigated nonlinear relationships between key climate variables and crop productivity that lead to critical thresholds in these relationships; for example, beyond a certain maximum temperature, small increases in temperature are related to large declines in crop productivity. Based on climate scenarios generated by the Intergovernmental Panel on Climate Change (IPCC), precipitous declines in productivity are projected for important crops in the United States, especially in the latter part of the 21st century (Schlenker and Roberts 2009).

Compared to assessments of climate effects on agriculture, estimates of its effects on forest productivity have been less definitive and emphatic. Unlike annual crops, forest ecosystems are long living and complex, and this may buffer some effects of climate variation. However, compared to agricultural crops, regeneration and mortality phases in the forest ecosystem are not well understood in relation to climate, and these are critical phases in forest establishment. Furthermore, disturbances such as wildfire, hurricanes, and intense rainfall and flood events can result in immediate changes to the forest ecosystem, including extensive mortality and erosion (for a full discussion, see chapter 2).

The history of land use in the United States indicates flexibility at the margins between agriculture and forests, but mainly in the East, where many states experienced agricultural abandonment and the recovery of forest cover through the 20th century (Ramankutty et al. 2010, Waisanen and Bliss 2002). At the same time, cropland expanded strongly in the Corn Belt and Northern Plains (Illinois, Iowa, Minnesota, North Dakota, and Montana) and in Florida. These trends are consistent with reduced transportation costs and increased integration of markets across and between regions, leading to consolidation of agricultural production in a few subregions (e.g., cereal crops in the Corn Belt, vegetable crops in Florida and California).

Changes in crop prices have affected changes in cropland allocation as well. In the 1970s, soybean markets drove conversion of forest land to cropland, especially in the Mississippi Alluvial Valley (Lubowski et al. 2006). Between 2000 and 2009, ethanol demands expanded U.S. corn production by 10 percent (2.91 million ha), and corn prices increased by about 75 percent (Congressional Budget Office 2009, Wallander et al. 2011). Expanded corn planting resulted largely from shifts out of soybeans, but with compensatory shifts toward soybeans and among other crop and hay land uses (Wallander et al. 2011). Although these crop substitutions moderated the push toward corn ethanol, they placed upward pressure on farm commodity and food prices in the United States and elsewhere (roughly 20 percent of increased prices were attributed to corn ethanol production) (Congressional Budget Office 2009).

Land use changes are not driven just by market factors such as price and transportation costs, but they may also be directly influenced by economic and environmental policies. United States agriculture is perhaps the most heavily subsidized sector of the U.S. economy, and changes in support prices and other programs could affect changes in land use allocation. Some policies directly encourage land use switching, as in the case of the Conservation Reserve Program (CRP). Established in the 1985 Farm Bill (Food Security Act of 1985), the CRP pays landowners to retire erodible cropland to natural cover; in February 2010, about 12.6 million ha were in the program (Hellerstein 2010). The most recent decline in cropland area in the United States coincides with the establishment of this program.

Future rural land uses are likely to adjust in response to a combination of three factors: population-driven urbanization, the comparative returns to agriculture and forestry, and policies that influence the expression of the first two factors. The recent Resources Planning Act (RPA) assessment (USDA FS 2012, Wear 2011) forecasts an increase in developed uses from about 30 million ha in 1997 to 54 to 65 million ha in 2060 (a gain of 24 to 35 million ha), based on alternative projections of population and income linked to IIPCC scenarios. The RPA assessment incorporates changes in relative returns to forests but holds agricultural returns constant over the forecast period.

Comparative returns to agriculture and forestry could be altered directly and indirectly by climate change. Direct effects derive from potential shifts in productivity, as examined above. At the margin, shifts in agricultural productivity would lead to land use switching between forests and crops. At a broader market scale, increased scarcity of crop output would drive up prices and overall demands for land in crops. Stronger shifts in comparative returns to forestry and agriculture would probably result from policy changes, especially those designed to encourage bioenergy production. The degree to which a bioenergy sector favors agricultural feedstocks, such as corn, or cellulosic feedstocks from forests will determine the comparative position of forest and crop returns to land use, and therefore land use allocations. The allocation among feedstock sources depends on energy policies at both federal and state levels, which could differentially affect rural land uses. For example, federal policy to date has subsidized corn ethanol production, but the 2008 Farm Bill and some state-level Renewable Portfolio Standard policies encourage use of wood in electricity generation. These policies would likely add to rather than supplant current emphasis on agricultural feedstocks. Policy initiatives to mitigate climate change through bioenergy and C sequestration may have more direct and immediate impacts on land use and the forest area than the impacts of climate change itself.

Interactions Between Trees and Climate in Urban Environments

Although it is common to distinguish between forest and developed land cover types, trees within developed areas may provide a disproportionately higher value of ecosystem services because of their proximity to human habitation. Trees in urban environments both influence and are influenced by climate change. As the area of urban use expands, the extent and importance of urban trees will increase. Climate change will likewise have important effects on these trees, and urban trees may be especially well positioned to provide critical services in moderating climate in urban environments.

In 2000, urban areas occupied 24 million ha (3.1 percent) of the conterminous United States and contained over 80 percent of the country's population (Nowak et al. 2005), and urban and community lands occupied 41 million ha (5.3 percent) (Nowak and Greenfield 2012). The definition of urban is based on population density using the U.S. Census Bureau's definition (2007): all territory, population, and housing units located within urbanized areas or urban clusters. The definition of community is based on jurisdictional or political boundaries delimited by U.S. Census Bureau definitions of incorporated or designated places (U.S. Census Bureau 2007). Community lands are places of established human settlement that may include all, some, or no urban land within their boundaries. Because urban land reveals the more heavily populated areas (population density-based definition) and community land has varying amounts of urban land that are recognized by their geopolitical boundaries (political definition), the category of "urban/community" was created to classify the union of these two geographically overlapping definitions where most people live.

Between 1990 and 2000, urban areas increased from 2.5 percent to 3.1 percent of U.S. land areas (an increase about the combined size of Vermont and New Hampshire), and they are projected to increase to around 8.1 percent in 2050 (an increase in area larger than Montana) (Nowak et al. 2005, Nowak and Walton 2005). Given a projected increase in urban land of 38.8 million ha between 2000 and 2050 and a concomitant conversion of about 11.8 million ha of forest to urban land (Nowak and Walton 2005), the current 20.8 billion Mg of C stored in U.S. urban forests (above and belowground biomass) nationally is projected to decrease to 20.1 billion Mg by 2050.[5] In addition, based on various climate change/development scenarios, percentage of tree cover nationally is projected to decrease by 1.1 to 1.6 percent between 2000 and 2060 (USDA FS 2012).

Urbanization can either increase or decrease tree cover depending where the urbanization occurs. The percentage of tree cover in urban/community areas tends to be significantly higher than in rural areas (i.e., lands outside of urban/community areas) in several predominantly grassland states, with the greatest difference in Kansas (17.3 percent) (Nowak and Greenfield 2012). In some cases, urban forest stewardship activities, both public and private, have helped to significantly increase and maintain forest area within cities. Urban/community land in most states in more forested regions had lower tree cover compared to rural lands, with the greatest difference in Kentucky (-37.9 percent). Within urban areas of the conterminous United States, percentage tree cover is declining at a rate of about 0.03 percent per year, which equates to 7900 ha or 4.0 million trees per year (Nowak and Greenfield 2012) out of an estimated 3.8 billion urban trees (Nowak et al. 2001). Analysis of 18 moderate- to large-sized U.S. cities reveals that percentage tree cover has declined, on average, by about 0.27 percent of city area per year (0.9 percent of tree cover) for these more densely populated areas (Nowak and Greenfield 2012). In urban/community areas of the conterminous United States, tree cover averages 35.1 percent (14.6 million ha), which is close to the national average (34.2 percent) (Nowak and Greenfield 2012). Cities developed in naturally forested regions typically have a

[5] Gavier-Pizarro, G.I.; Radeloff, V.C.; Stewart, S.I. [et al.]. [N.d.]. Seventy-year legacies of housing and road patterns are related to non-native invasive plant patterns in the forests of the Baraboo Hills, Wisconsin, USA. Ecosystems. Manuscript in preparation.

higher percentage of tree cover than cities developed in grassland or desert areas (Nowak and Greenfield 2012; Nowak et al. 1996, 2001).

The structural value of the urban trees (e.g., cost of replacement or compensation for loss of trees) in the United States is estimated at $2.4 trillion (Nowak et al. 2002b). Urban trees provide many additional benefits, such as air pollution removal and C storage and sequestration. Annual pollution removal (fine particulates, ozone, nitrogen dioxide, sulfur dioxide, carbon monoxide) by U.S. urban trees is estimated at 783 000 Mg ($3.8 billion value) (Nowak et al. 2006). Thus, as climates change, not only will these urban forests and their associated benefits be affected, but these forests will also help to mitigate the effects of climate change and reduce CO_2 emissions emanating from urban areas. For example, U.S. urban trees are estimated to store 771 million Mg of C ($14.3 billion value; based on a price of $20.3 per Mg of C), with a gross C sequestration rate of 25.1 million $Tg{\cdot}C \cdot yr^{-1}$ ($460 million yr^{-1}) (Nowak and Crane 2002).

Effects of Climate Change on Urban Trees

The greatest effects of climate change on urban trees and forests will likely be caused by warmer air temperature, altered precipitation, strengthening wind patterns, and extreme weather events, including droughts, storms, and heat waves. These changes, along with higher levels of CO_2, are likely to have significant implications for urban forests and their management.

In addition to regional and global climate changes, the urban environment creates local climatic changes that will affect urban forests. At the local scale, urban surfaces and activities (e.g., buildings, vegetation, emissions) influence local meteorological variables such as air temperature, precipitation, and windspeed. Urban areas often create what is known as the "urban heat island," where urban surface and air temperatures are higher than in surrounding rural areas. These urban heat islands can vary in intensity, size, and location and can lead to increases in temperatures of 1 to 6 °C (US EPA 2008). Urban areas also affect local precipitation percent (Shepherd 2005). For example, in some areas in the southeastern United States, monthly rainfall rates increase, on average, by about 28 percent (about 0.8 mm hr^{-1}) within 30 to 60 km downwind of city areas (maximum downwind increase of 51 percent), with a modest increase (5.6 percent) over the city area (Shepherd et al. 2002).

These environmental changes caused by the interaction of climate change and urbanization are likely to affect urban tree populations (Nowak 2010). Potential effects on urban tree populations include changes in (1) tree stress and decline in some species from changes in air temperature, precipitation, storm frequency and intensity, CO_2 levels and associated changes in air pollution, (2) changes in species composition owing to both changes in climate (e.g., Iverson and Prasad 2001) and human actions and invasive plant characteristics that are influenced by climate change, (3) insect and disease compositions and prevalence, and (4) management and maintenance activities focused on offsetting tree health and species compositional changes (Nowak 2000). Management activities to sustain healthy tree cover will alter C emissions (because of fossil fuel use), species composition, and urban forest attributes such as biodiversity, wildlife habitat, and human preferences and attitudes toward urban vegetation.

Effects of climate change may be accelerated or reduced in cities depending on whether managers alter plant populations toward better adapted species or attempt to minimize the effect of global climate change through enhanced maintenance activities (e.g., watering, fertilizing). The degree to which urban tree populations are affected by climate change will depend on actual changes in air temperatures, precipitation, and length of growing season, as well as on human activities in urban areas that affect outcomes such as pollution and CO_2 concentrations, disturbance patterns, and decisions related to vegetation maintenance, design, selection, plantings, and removals.

Effects of Urban Trees on Climate Change

Climate change can have both positive and negative effects on the urban forest. Management activities can produce healthy and sustainable urban forests to help offset impacts of climate change. Nowak (2000) proposed four main ways that urban forests affect global climate change:

Removing and storing carbon dioxide—
Trees, through their growth process, remove CO_2 from the atmosphere and sequester the carbon within their biomass (McKinley et al. 2011). The net C sequestered from afforestation or reforestation programs is mostly the C sequestered by the first generation of trees. Future generations of trees sequester back the C lost through decomposition of previous generations. Thus, the net C storage in a given area with a given tree composition will cycle through time as the population grows and declines. When forest growth (C accumulation) is larger than decomposition, net C storage increases. Some C from previous generations can also be locked up in soils. Management activities can enhance long-term C storage in several ways: with large, long-lived species that are adapted to local site conditions, minimized use of fossil fuels to manage vegetation, vegetation designs to reduce air temperature and energy use, and use of urban tree biomass in long-term products (or limits on wood decomposition after removal) and energy production (Nowak et al. 2002a).

Emitting atmospheric chemicals through vegetation maintenance practices—
Urban tree management often uses relatively large amounts of energy, primarily from fossil fuels, to maintain the vegetation structure. Thus, to determine the net effect of urban forests on global climate change, the emissions from maintenance/management activities need to be considered. For example, equipment used to plant, maintain, and remove vegetation in cities includes vehicles for transport or maintenance, chain saws, back hoes, leaf blowers, chippers, and shredders.

Altering urban microclimates—
Trees are part of the urban structure so they affect the local urban microclimate by cooling the air through transpiration, blocking winds, shading surfaces, and helping to mitigate heat island effects.

Altering energy use in buildings and consequently emissions from powerplants, by planting trees in energy-conserving locations around buildings—
Urban trees can reduce energy use in summer through shade and reduced air temperatures, and they can either increase or decrease winter energy use (Heisler 1986), depending on tree location around the building (e.g., providing shade, blocking of winter winds).

Interactions Between Climate Change and the Wildland-Urban Interface

The WUI is where homes and associated developments co-occur with wildland vegetation (Radeloff et al. 2005). This WUI zone is delineated under wildland fire policy in the United States, because the risk of wildland fire affecting homes and other structures is greatest here. However, the WUI has perhaps even greater significance beyond fire management and policy, in that it encompasses where people live in direct contact with forests and other wildlands, and where development of forested lands for residential and commercial uses has direct, ongoing effects on the forest. Key changes driven by climate change, population growth, and markets for land uses are especially concentrated in this zone.

Over time, these impacts are expected to increase, because growth in the WUI has outpaced growth outside the WUI, a trend expected to continue in coming decades, particularly in Western States. (Hammer et al. 2009). Theobald and Romme (2007) reached a similar conclusion; they estimated that Arizona, Colorado, Idaho, Montana, Nevada, and Utah will see the strongest WUI growth in the decades to come, and estimate at least 10 percent WUI growth in the United States by 2030. Analysis of historical growth patterns and projected growth rates in the United States generated estimates of 17 percent growth within 50 km of national parks, national forests, and wilderness areas by 2030 (Radeloff et al. 2010). The primary reason for continuing expansion of the WUI is that it reflects the affinity of many American home buyers for a house near or in the woods, and in a rural or small-town setting (McGranahan 1999). Forests are considered amenities, and home buyers prefer and pay more for home sites in or near the forest to maximize privacy, aesthetics, and recreational access.

Expected WUI growth differs across the United States. Population growth is a strong predictor of housing growth

(Liu et al. 2003), and areas where population is expected to grow are concentrated in the West and South (Hammer et al. 2009). Population growth is not the only factor driving housing growth; affluence, declining household size, and ownership of multiple homes also contribute to an expanding housing stock (Hammer et al. 2009). Although the housing market has changed dramatically since 2009, the downturn in home construction has been modest relative to the decades-long expansion of housing stock. Between 1940 and 2000, although nationwide population doubled, the number of housing units more than tripled (Hammer et al. 2009).

Not all rural areas are equally attractive (Johnson and Beale 2002, McGrannahan 1999); natural resources and other amenities add value, and protected area status is an added attraction to buyers because it guarantees that changes to the landscape (e.g., to species mix or forest age, conversion from forest cover to commercial, residential, or other use) will be modest. In studies that isolate just protected areas (i.e., the areas protected from development under various laws), the highest housing growth has occurred in proximity to protected areas (Radeloff et al. 2010, Wade and Theobald 2010). These lands have protected status in part to ensure that plant and animal species will be sustained, which makes their attractiveness for housing growth troubling from an ecological perspective (Gimmi et al. 2011).

Disturbances in the Wildland-Urban Interface

The WUI allows extensive contact between people and forests, through both simple proximity and intentional activity. Consequently, people in the WUI are more likely to be aware of fire and other forest disturbances that might be exacerbated by climate change, such as insect and disease outbreaks, severe weather damage, drought, and the spread of invasive plants. When forests are part of the residential setting, even less obvious changes (e.g., how early the maples leaf out, when migratory birds return) are more likely to be noticed, because the WUI resident sees the forest both daily and over long periods.

Among the best examples of how humans respond to disturbance and risk in the WUI is wildland fire, because it potentially threatens homes, typically a family's largest single investment of capital. Awareness and acceptance of the need to prepare for wildland fire has grown with the WUI. Since 2002, the Firewise program has enlisted communities across the country to develop and maintain their residential areas in ways that minimize fire risk (NFPA 2011). This program formalized ideas that had been developing over preceding decades in response to both growing losses in the WUI and empirical evidence regarding wildland fire safety (e.g., the safest configuration of vegetation surrounding the home, building materials, and home and yard maintenance) (Cohen 2000, NFPA 2011).

Ideally, entire communities would be "fire adapted," where fire should be able to pass through a community without causing extensive damage. The creation of fire-adapted human communities is now being based on an interagency cohesive wildland fire strategy that uses a risk-based approach and is grounded in scientific research (Calkin et al. 2011). The process of enlisting, encouraging, reminding, and assisting homeowners and communities in fire safety programs is far from simple. Basic research on psychology of risk, which measures and quantifies factors that affect how people judge risk, provides a basis for changing perceptions of risk and encouraging action to reduce risk (Slovik 1987), and some specific aspects of risk perception and response related to wildland fire have verified and extended this work (e.g., Cova et al. 2009). However, the same body of research also cautions that people are seldom willing to limit their frame of reference for a given risk as strictly as scientists might do and as policymakers might prefer. For example, even if asked by a scientist to focus on a specific source of risk without considering its context, most responses are shaped by additional factors (Slovik 1987). This phenomenon can be seen in forest management situations, for example when willingness to remove vegetation around the home is met with resistance, not because home owners do not understand risk, but rather, because they do not believe that removing vegetation would reduce their risk (McCaffrey 2009). What the manager considers relevant to the situation is not the same as what the homeowner considers relevant (McCaffrey and Winter 2011), and this lesson has implications for managing forests under climate change.

Climate change, like wildland fire, presents many challenges for the ability of people to understand, judge, and act on new information. Changes occur slowly and many threats are anticipated in the distant future, both of which attenuate the urgency of a response. In the broader cultural context, climate change beliefs are partisan and polarized and connect with deeply held convictions, such as whether humankind is the source of or a solution to problems, how to balance public good and private property rights, and whether science should be trusted or suspected. Long-term problems, large-scale solutions, and divisive underlying issues dramatically hurt chances for galvanizing public support for bold change. In contrast, specific, observable changes in forest resources, particularly in familiar and local forests, are best able to engage the attention and concern of people outside forest management and research communities. Local, place-specific solutions for problems are most likely to find support, especially if residents already know and trust local resource managers. Because people attach such great value to forests, the challenges of making climate change a salient issue and finding an engaged constituency are more modest than for climate change as a global concern.

Fire and other forest disturbances (e.g., insect outbreaks) are a source of concern to many homeowners, yet living in the WUI is itself the source of many direct and indirect forms of disturbance. For example, changes in the density and use of road systems have many negative effects (Hawbaker et al. 2006, Radeloff et al. 2010), and in the yards surrounding homes people modify vegetation for functional and aesthetic purposes (Cook et al. 2011). The plant species used in landscaping may include exotic invasive species, which can be evident decades after they are introduced by the homeowner (Rogers et al. 2009). Feeding wildlife and keeping pets can alter the trophic balance of forested ecosystems, particularly when domestic cats and dogs roam outdoors (Lepcyzk et al. 2003). Building and landscaping disturb soils and change light availability, which can facilitate expansion of highly invasive species, and yard maintenance intentionally changes the distribution of water and other nutrients, another source of indirect effects on forests (Cook et al. 2011). In short, the overall effect of residential land use on forests and their ecosystems is often negative.

Multiple Stressors on Wildlands in the Interface

Like climate change itself, development of and activity in the WUI results in various changes to the forest. Multiple stressors are more problematic than single or sequential stressors because they overtax the resilience of the forest. Although regulations such as zoning ordinances that limit housing density, and neighborhood covenants governing property management are intended (in part) to protect the environment, forests may still suffer because each individual stressor is dealt with as though it occurs alone. Taken together, however, the many small disturbances can overwhelm the ability of forests and wildlife to adapt by requiring too much change too quickly; this problem is not typically reflected in land use and other residential policies.

Awareness of the harm caused by multiple stressors is not apparent in the institutions that govern forests. A sobering possibility is that residential areas in or near forests could be well designed and governed under fully enforced and effective regulations, yet still sap the resilient capacity that the forest needs to adapt to climate change. An example would be a development plan that specifies what percentage of trees will be retained in a subdivision without accounting for their configuration, resulting in a fragmented forest, and disrupted wildlife habitat and corridors.

Human perception, unaided by science-based monitoring, will not easily detect the diffuse, slow-to-develop problems in the WUI stemming from multiple stressors, nor does this suite of stressors lend itself to simple explanation. For example, research suggests that housing development gives rise to an increase in native bird species richness, perhaps owing to the more varied habitat types (open areas, forest edges) created in a WUI development, but that higher levels of residential development decrease native bird richness (Lepczyk et al. 2003). This phenomenon, observed in relation to biodiversity of many species (McKinney 2002), illustrates how multiple stressors on the WUI may go largely unnoticed by the human communities responsible for them, yet have significant consequences. Once known and understood, however, resource management concerns that are conveyed effectively can change human behavior. Initial case-study research gives reason for optimism; in Fremont

County, Colorado, WUI residents actively learned from each other and were engaged in managing many complex WUI resource issues (Larsen et al. 2011). Given expectations for continued WUI growth, together with the impacts of climate change, such activities will be essential to maintaining enough forest health and resilience to adapt to whatever changes occur.

Social Interactions With Forests Under Climate Change

Social interactions with forests extend beyond the definition of forest area and service flows defined by land uses. As discussed, human activity can enhance or diminish the effects of a changing climate on forested ecosystems. People and the actions they take directly alter the capacity of forests to sequester carbon and to adapt to a changing climate. By reshaping the landscape, people alter the extent of forest and with it, forest health, sustainability, and capacity to meet the needs of other species. Society relies on forests for products and for wide-ranging ecosystem services, from life-sustaining ones (e.g., air and water filtering) to enhancements in quality of life (e.g., scenic vistas, recreation). Thus, people and societies mediate the relationship between forests and climate change both directly (by altering forests) and indirectly (by changing other physical and biological systems that in turn alter forests). The interaction between this social relationship with forests and climate change potentially will alter ecosystem services that people depend upon from forests and woodlands.

The relationship between forests and climate change in the United States cannot be understood without considering people and the communities in which they live. Some communities are embedded within social systems strongly linked to the condition and uses of natural resources. These natural resource-based communities, where the relationship is based on commodities such as timber or amenities such as recreation, may be disproportionately affected by interlinked climate and forest ecosystem changes. Another set of communities consists of tribal areas, which may become especially vulnerable to effects of climate change because of the relatively strong links between these communities, their economies, and their natural resource base. Unlike other sectors, the possibility of adaptation through migration is limited because of strong cultural ties to tribal lands. This section explores the extent and form of these two types of communities and their resilience to changes in interlinked climate and forest conditions. Assessing the resilience to climate change of both natural resource-based communities and tribal communities requires understanding of not only the economic and ecological vulnerabilities but also the social vulnerability of each. We propose a framework for exploring those vulnerabilities in light of climate change.

Natural Resource-Based Communities

Natural resource-based communities are closely linked with their geographic setting and environmental context. In these communities, people with collective, intersecting, and competing values interact because they are at the dynamic interface of societal and environmental processes (Flint and Luloff 2005). These communities also derive economic benefits from the surrounding natural resources and withstand their associated natural disturbances, such as wildfires and hurricanes. Natural resource-based communities are affected by both technological and macroeconomic changes. Using six categories, the U.S. Department of Agriculture, Economic Research Service (USDA ERS 2011) classified economic dependence by county. Farm dependency has declined considerably; in 2000, only 20 percent of nonmetropolitan counties were considered farming dependent (Dimitri et al. 2005); most are now centered in the Great Plains (fig. 3.8).[6] Other counties, particularly in the West, depend on federal

[6] Farm dependent, 1998 through 2000. Source: U.S. Department of Agriculture, Economic Research Service, County Typology Codes, using data from the U.S. Census Bureau and the Bureau of Economic Analysis. Type of data: Multiyear averages and point-in-time census data. Year(s): 1998 through 2000. Definition: Classification of counties by measures of farm earnings and employment, where 1 = farm-dependent county; 0 = all other counties; a county is defined as farm dependent if farm earnings accounted for an annual average of 15 percent or more of total county earnings during 1998 through 2000, or farm occupations accounted for 15 percent or more of all employed county residents reporting an occupation in 2000.

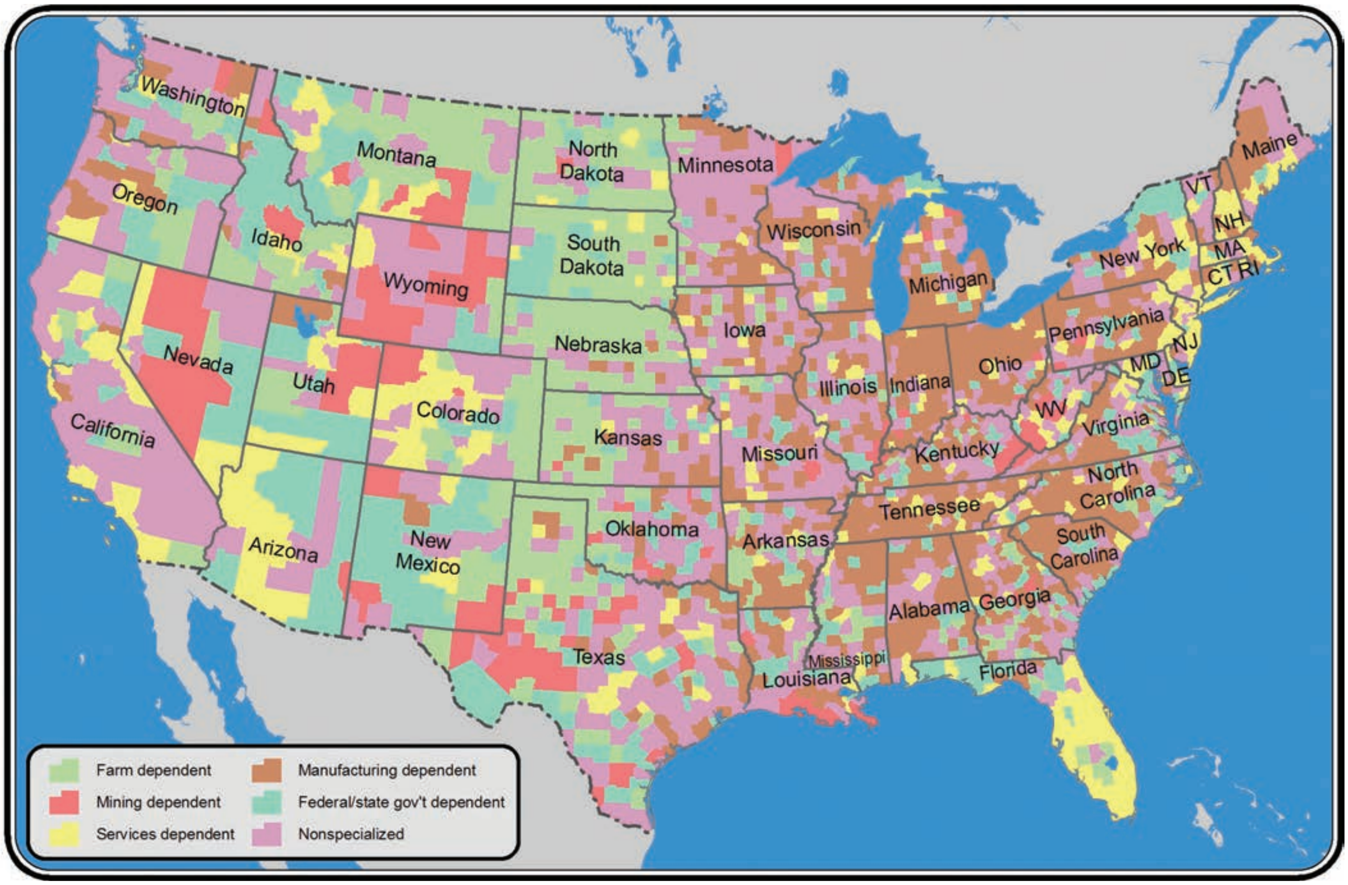

Figure 3.8—Economic dependence (USDA ERS 2011).

or state government and mining. Of the 368 recreation-dominated counties, 91 percent (334) were in rural areas (Lal et al. 2010).[7]

Structural changes in the timber industry have resulted in large-scale changes in land tenure, corporate consolidation in the timber industry, and separation of processing capacity ownership from timberland ownership (Bliss et al. 2010). Many forested areas previously owned by timber companies are now owned by timberland investors, who differ markedly from the industrial owners in, for example, their landholding objectives, time horizons, and management capacities. According to Bliss et al. (2010), these changes in the timber industry are dynamic, and any predictions about future ownership patterns and their implications for small-scale forestry and rural natural resource-based communities are highly speculative; however, they suggest three possible trajectories for future land uses: intensive timber production forestry, "highest and best use" parcelization and conversion, and conservation forestry.

Natural resource-based communities, such as those situated in or near forests or other expansive resources, often experience the consequences of natural disasters or environmental stresses sooner than do farther-removed

[7] Nonmetropolitan recreation-dependent, 1997–2000. Source: U.S. Department of Agriculture, Economic Research Service, County Typology Codes, using data from the U.S. Census Bureau and the Bureau of Economic Analysis. Type of data: Multiyear averages and point-in-time census data. Year(s): 1997–2000. Definition: Classification of nonmetropolitan counties by measures indicating high recreational activity, where 1 = recreation-dependent county; 0 = all other counties; measures of recreational activity were (1) wage and salary employment in entertainment and recreation, accommodations, eating and drinking places, and real estate as a percentage of all employment reported in the Census Bureau's County Business Patterns for 1999; (2) percentage of total personal income reported for these same categories by the Bureau of Economic Analysis; (3) percentage of housing units intended for seasonal or occasional use reported in the 2000 Census; and (4) per capita receipts from motels and hotels as reported in the 1997 Census of Business.

populations (Haque and Etkin 2007, Lynn and Gerlitz 2006, Lynn et al. 2011). These manifestations are most pronounced in developing areas of the world, including West Africa, South America, and portions of south Asia (Amisah et al. 2009, Laurance et al. 1998), where processes such as forest fragmentation, deforestation, and conversion of land from forests to agricultural uses have accelerated the rate of climate change and contributed to erratic rainfall patterns, increased temperature and wildfire frequency, and stressed water sources.

Although developing regions of the globe contain the best examples of forest-dependent communities vulnerable to climate change, these relationships also occur in developed regions of the Northern Hemisphere, highlighting the unevenness of climate vulnerability within developed nations. These vulnerabilities inherently relate to biophysical conditions of place, but they also manifest in terms of the socioeconomic and political milieu associated with many resource-dependent communities of the Northern Hemisphere. For example, individual and community vulnerability can be affected by characteristics such as income level, race, ethnicity, health, language, literacy, and land use patterns. Thus, the social vulnerability of natural resource-based communities to effects of climate change is important both to understand and to include in discussions of climate vulnerability, because the sociology of a given locale can compound or exacerbate biophysical vulnerabilities of place.

An analysis of forest-dependent communities in Canada suggests that specific social characteristics associated with forest-based communities increase climate change risks for such communities (Davidson et al. 2003). For example, human capital development is typically lower with respect to educational attainment in these areas, and there is a concentration in a specific skill set that makes it difficult for laborers to transfer skills to other occupations or contexts. The politicization of deforestation's role in climate change has also, in some cases, created a larger populace (often removed from place) that is unsympathetic to the labor dilemmas facing communities dependent on traditional forestry activities. Further, uncertainty about the exact nature of climate changes, coupled with the long-term planning horizon necessary for forest management, elevates risks associated with investments in forest-based industries, making such investments less appealing to potential investors. The result could lead to under-investments in communities primarily dependent upon a single sector economy. Moreover, climate change may not be perceived as such by local residents or key decisionmakers in forest-based communities, resulting in reluctance by communities to devise adaptive strategies to help mitigate current and future environmental stresses and hazards. Finally, methods used to assess climate risks may be inadequate in situations where climate change is occurring alongside other isolated environmental events.

Similar to biophysical vulnerabilities, social vulnerabilities differ spatially and are more prominent for certain sociodemographic groups such as racial and ethnic minorities, women, the elderly, the very young, and for people in specific geographical contexts such as forest-proximate communities. The fourth IPCC assessment (Pachauri and Resinger 2007, Solomon et al. 2007) addresses the spatiality of climate vulnerability: "There are sharp differences across regions and those in the weakest economic position are often the most vulnerable to climate change and are frequently the most susceptible to climate-related damages…. There is increasing evidence of greater vulnerability of specific groups such as the poor and elderly not only in developing but also in developed countries."

Tribal Forests

American Indians and Alaska Natives rely on reservation lands and access to traditional territories beyond the bounds of reservations for economic, cultural, and spiritual well being. Tribes have unique rights, including treaties with the federal government that reserved rights to water, hunting, fishing, gathering, and cultural practices (Lynn et al. 2011, Pevar 1992). We focus on the forests and woodlands on Indian reservations, how climate change will affect these lands, and the tribal communities that depend on these ecosystems.

Indian reservations contain 7.2 million ha of forest land, of which 3.1 million ha are classified as timberland and 2.3 million ha as commercial timberland (Gordon et al. (2003).

These forests are diverse, ranging from productive conifer forest in the Pacific Northwest to dry pine forest and juniper woodland in the Southwest, mixed hardwood-conifer forest in the Lake States, and spruce forest in the southern Appalachians (Gordon et al. 2003) (fig. 3.9). Of the 7.2 million ha of forest land, 4.1 million ha were classified as woodland (defined as less than 5 percent canopy cover of commercial timber species but at least 10 percent total canopy cover), of which 1.4 million ha are commercial woodland.

As part of a 10-year assessment of Indian forest management, surveys were conducted to identify the Indian vision for tribal forests. Gordon et al. (2003) described the Indian vision in terms of the major themes expressed by Native people: (1) natural, healthy, beautiful places; (2) integrated management; (3) self-governance and trust responsibility; (4) communication, tribal public involvement, and education. The same report also described resource management of tribal forests as moving close to attaining

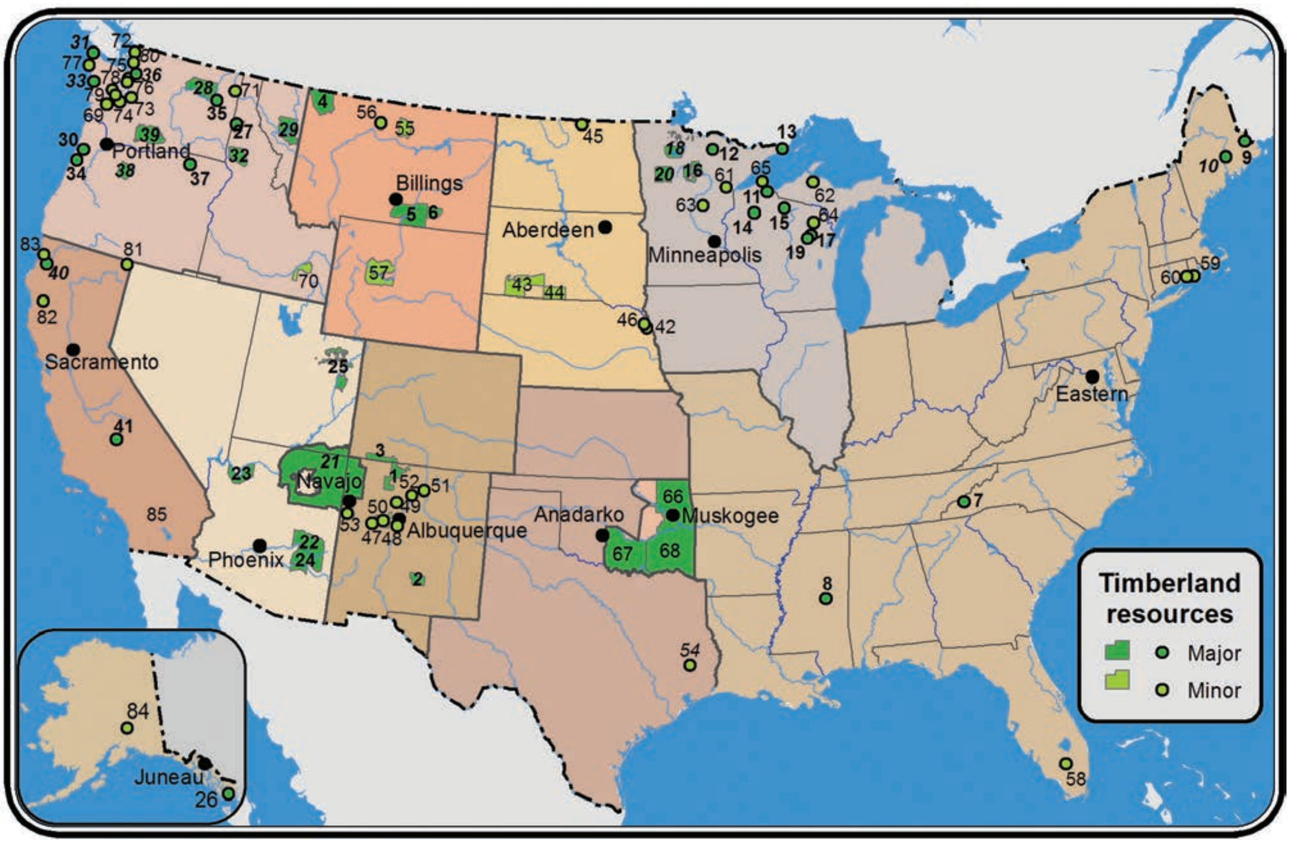

Figure 3.9—Reservations with significant timberland resources. Numbers 1 through 41 have over 4047 ha of commercial timberland, or over 2360 m^3 allowable cut, per reservation. Numbers 41 through 83 have less area in timberland, but what they have is economically viable. (1) Jicrailla, (2) Mescalero Apache, (3) Southern Ute, (4) Blackfeet, (5) Crow, (6) Northern Cheyenne, (7) Eastern Band of Cherokee, (8) Mississippi Choctaw, (9) Passamaquoddy, (10) Penobscot, (11) Bad River, (12) Bois Fort, (13) Grand Portage, (14) Lac Courte Oreilles, (15) Lac du Flambeau, (16) Leech Lake, (17) Menominee, (18) Red Lake, (19) Stockbridge/Munsee, (20) White Earth, (21) Navajo, (22) White Mtn. Apache, (23) Hualapai, (24) San Carlos, (25) Uintah and Ouray, (26) Annette Islands, (27) Cour d'Alene, (28) Colville, (29) Flathead, (30) Grand Ronde, (31) Makah, (32) Nez Perce, (33) Quinault, (34) Siletz, (35) Spokane, (36) Tulalip, (37) Umatilla, (38) Warm Springs, (39) Yakama, (40) Hoopa Valley, (41) Tule River, (42) Omaha, (43) Pine Ridge, (44) Rosebud, (45) Turtle Mountain, (46) Winnebago, (47) Acoma, (48) Isleta, (49) Jemez, (50) Laguna, (51) Picuris, (52) Santa Clara, (53) Zuni, (54) Alabama-Coushatta, (55) Fort Belknap, (56) Rocky Boy's, (57) Wind River, (58) Big Cypress, (59) Narragansett, (60) Pequot, (61) Fond du Lac, (62) L'Anse, (63) Mille Lacs, (64) Potawatomi, (65) Red Cliff, (66) Cherokee, (67) Chickasaw, (68) Choctaw, (69) Chehalis, (70) Fort Hall, (71) Kalispel, (72) Lummi, (73) Muckleshoot, (74) Nisqually, (75) Port Gamble, (76) Port Madison, (77) Quileute, (78) Skokomish, (79) Squaxin Island, (80) Swinomish, (81) Fort Bidwell, (82) Round Valley, (83) Yurok, (84) Alaska trust properties (Chugachmiut, Metlakatla, Tanana Chiefs Conference). Adapted, with permission, from the Intertribal Timber Council (IFMAT 2003).

this vision. Gordon et al. (2003) noted that the ecological condition and management of tribal forests has improved since the previous assessment (IFMAT 1993). Increasingly complex ecological approaches are being implemented, as well as increased fire management activities. In 2008, the U.S. Department of the Interior, Bureau of Indian Affairs (BIA) reported that an estimated 91 percent of the forest area had a forest management plan or an integrated resource management plan with forest management provisions.

In 2001, the total allowable annual cut on timberlands was reported as 1 840 000 m^3 and for woodlands was 230 000 m^3, with harvest volume at 1 430 000 m^3. Harvest value was estimated at $65.9 million, a 27 percent decline from the 1991 harvest value of $117.4 million (numbers adjusted for inflation, Gordon et al. 2003). The Northwest and the Lake States accounted for the greatest harvest volume and stumpage revenue in 2001. Tribal forestry faces challenges common to forestry, limited wood processing capacity, and, in the Western United States, poor markets for small wood products. In 2008, the BIA reported the effects of the continuing decline in the housing construction market on forestry-related products from tribal lands as well as the effects of rising fuel costs on transporting forestry-related products. The Mescalero Apaches had no market for small merchantable logs, severely hindering their forest management program. In the Northwest, traditionally stable tribal sawmills were having difficulties paying their bills. As with other forests in the United States, tribal forests face new challenges from invasive species, pest outbreaks, and large-scale fires initiating on tribal lands as well as spreading from adjacent forest lands. The BIA (2008) also reported the beginning of a projected decline in the number of professional foresters.

Tribal forests and woodlands provide jobs and revenue from timber production, nontimber forest products, grazing, and fishing and hunting. They also provide recreation opportunities, energy resources, and material for shelter, clothing, medicines, food, as well as places for religious ceremonies and solitude. In addition to the broader effects on forests discussed earlier, climate change effects on tribal forest and woodland ecosystems will have implications for treaty rights if culturally significant plant, animal, and fungi species ranges move outside reservation boundaries. Water resources and tribal water rights may be especially affected by climate change (Curry et al. 2011, Karl et al. 2009). Adaptation responses may be challenging, given fragmented tribal lands and the small size of some reservations. Lynn et al. (2011) also describe current adaptation approaches on tribal lands, including watershed management surrounding sacred waters, natural hazard management, and legislation to foster green jobs, such as farmers' markets to small-scale energy projects. Some tribes have begun to explore options to manage their forest lands for C sequestration. The fixed location of tribal lands defines important limits, however, to the adaptive capacity of tribal communities with regard to climate change.

Social Vulnerability and Climate Change

Generally, socially vulnerable populations are understood as marginal groups, in terms of material well being, which renders them relatively unable to anticipate, cope with, or recover from environmental stresses that occur within a geographically defined setting (Kelly and Adger 2000). A common conceptualization of vulnerability is informed by the widely held idea that interprets vulnerability not just in terms of susceptibility or sensitivity to loss arising from hazard exposure, but also as a function of three primary contributors: hazard exposure, sensitivity, and resilience or adaptive capacity (Brooks 2003, Polsky et al. 2007, Smit and Wandel 2006):

$$\text{social vulnerability} = f\,(\text{exposure, sensitivity, adaptive capacity}).$$

Exposure is understood as proximity to a physical hazard or stressor. Sensitivity is the susceptibility of humans in sociodemographic terms to physical hazard, which can also include sensitivities of the built environment, such as geography or land use change. Adaptive capacity is any mitigation and adaptation to hazard via sociodemographic factors or other means.

Birkmann (2006) identified at least 25 conceptualizations of vulnerability in terms of human populations. Definitions differ by disciplinary area and underlying assumptions concerning the nature of risk, disaster, and exposure. However, these variant understandings of vulnerability can be viewed analytically as either the "outcome" or "contextual" framing of vulnerability (Brooks 2003, Kelly and Adger 2000, O'Brien et al. 2007).

Outcome framing describes vulnerability as a resultant state that occurs after an exposure unit (e.g., individuals, communities) has experienced and adapted to an environmental stressor, such as more incremental changes in climate (Watson 2001). This understanding focuses attention on estimating or projecting a "future," an endpoint of vulnerability that comes about as a consequence of climate-changing emissions (e.g., greenhouse gases) and resultant climate scenarios. Biophysical impacts to humans or physical systems are then predicted from given scenarios, and finally adaptations to projected impacts are formulated. Implicit here is that vulnerabilities are not considered to be an inherent quality of place or community; rather, vulnerabilities arise after exposure to climate-altering processes or events. The definition of vulnerability above (Watson 2001) is an example of an outcome framing of vulnerability. The outcome perspective is also assumed in projects such as the U.S. Forest Service Forest Futures analysis, which examines the effect of future climate scenarios on forest resources. Most social vulnerability research related to forests has also used outcome framing.

Contextual vulnerability differs from outcome vulnerability in that it analyzes current vulnerabilities within the current social structure of a given place. An analysis of contextual vulnerability (e.g., economic reliance on river-based tourism) focuses on the relationships among political actors (elected officials), institutions (rules for concessionaires), socioeconomic well-being (workforce education level), and culture to identify how goods and information are distributed across society. From this evidence, the analysis predicts response to a future threat (e.g., whether guides will be able to maintain their concession for river rafting as in-stream flows decline). This approach assumes that human vulnerability to natural events depends entirely on the capabilities already existing in a social system. The efficacy with which communities cope with a range of current environmental and societal stressors determines how well they will respond to future stressors. The contextual vulnerability approach generates management implications; it suggests that currently vulnerable communities can be identified and management action taken to improve current adaptive capacity.

Contextual assessments are appealing because of the clarity of their implications, yet few such assessments have been undertaken. The most vulnerable human populations are often difficult to identify, and understanding the values and perceptions of risk that community members hold requires more than a review of existing social and economic conditions. For example, using data from the U.S. Census Bureau to identify forest-dependent communities based on low income or high unemployment would not suffice and could misclassify communities, though this approach holds obvious appeal to assessment teams with little time and few resources.

Providing better guidance for conducting applied vulnerability assessments was one goal of a workshop co-sponsored by the U.S. Forest Service and University of Montana, and attended by social scientists and resource managers from federal agencies and universities. The group developed an initial template for socioeconomic vulnerability assessments (SEVAs), which begins with a review of secondary data from Census Bureau and similar sources. Following this review, a SEVA will (1) briefly discuss the social history of the forest and its human geography, including both communities of place and communities of interest, (2) link current and expected biophysical changes to community-relevant outcomes, (3) determine stakeholders' perceptions of values at risk (e.g., resources, livelihoods, cultures or places threatened by climate change), and (4) prioritize threats to vulnerable communities and identify those that the landowner or land manager, singly or with their partners, can best address. This basic outline will need testing and refinement over time as land managers elaborate and improve on it, but it represents a first step toward bringing SEVAs within reach of any assessment team.

Conclusions

Although climate change has been identified as an important issue for management and policy, it is clear that the interaction among changes in biophysical environments (climate, disturbance, and invasive species) and human responses to those changes (management and policy) will determine outcomes of consequence to people. The ultimate effects on people are measured in terms of changes in ecosystem services provided by forested landscapes, including traditional timber products and new extractive uses, rural and urban recreation, cultural resources, the contributions of urban forests to human health, and the protection of water quality. Climate change has been linked to bioenergy and C sequestration policy options as mitigation strategies, emphasizing the effect of potential climate change-human interactions on forests as well as the role of forests in mitigating climate change. Any effect of climate change on forests will result in a ripple effect of policy and economic response affecting economic sectors and human communities in U.S. society.

The key mechanism of change in human-dominated landscapes is choice. Where private ownership dominates, choices regarding land use and resource production directly and indirectly affect changes in forest conditions and the flow of ecosystem services. The choices are directly influenced by shifts in land productivity, the prices of various products and ultimately the returns to different land uses. Land use shifts in rural areas under climate change could involve conversion between forests and agricultural uses, depending on market conditions. Climate changes are expected to alter productivity (local scale) and prices (market scale). Land use patterns dictate the availability of the full range of ecosystem services from forests and from trees within other land uses. Both WUI and urban areas are projected to increase, often at the expense of rural forests. Anticipated climate changes, coupled with population growth, strongly increase the extent and value of urban trees in providing ecosystem services and for mitigating climate change impacts at fine scales. However, climate change also increases the challenge of keeping trees healthy in urban environments.

Collective choice, in the form of various policies, also holds sway over land use and forest condition outcomes. Policies targeting climate mitigation, especially for bioenergy production and C sequestration, directly target forest extent and use. Implemented through markets, these policies would yield secondary and tertiary impacts to forest composition and structure through direct action and through resource input and product substitutions in related sectors. These and other policies (e.g., forest management regulations, land use restrictions, property taxes) also set the context for and potentially constrain the adaptive choices by private landowners.

Human communities living in environments along the gradient from urban to rural environments will experience changes to forests. Those communities dependent on forests for economic, cultural, or spiritual services are likely to see the effects of climate change first. The potential for human communities to adapt to potential climate changes is linked to their exposure to climate change, which differs along the rural-to-urban gradient, and also to the nature of the social and institutional structures in each environment. One can prepare for or mitigate future climate stresses in these environments by ensuring that the resilience of human communities in these environments are intact today, because the efficacy with which humans are presently able to deal with change will determine how well they will be able to respond to future stresses.

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Chapter 4
Adaptation and Mitigation

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Strategies for Adapting to Climate Change

Forest ecosystems respond to natural climatic variability and human-caused climate change in ways that are adverse as well as beneficial to the biophysical environment and to society. Adaptation refers to responses or adjustments made—whether passive, reactive, or anticipatory—to climatic variability and change (Carter et al. 1994). Many adjustments occur whether humans intervene or not; for example, plants and animals shift to favorable habitats resulting in range expansion or contraction, as well as changes in gene frequencies for traits that enable persistence in warm climates. Here we assess strategies and tactics resource managers can use in the process of reducing forest vulnerability and increasing adaptation to changing climates (Peterson et al. 2011). Plans and activities range from short-term, stop-gap measures, such as removing conifers that are progressively invading mountain meadows, to long-range, proactive commitments, such as fuels management to reduce the likelihood of severe wildfire or of beetle-mediated forest mortality.

Principles for Forest Climate Adaptation

The following principles apply broadly in developing new perspectives on forest climate adaptation:

Successful Climate Adaptation Planning and Implementation

In the context of this chapter, adaptation strategies, plans, and management actions are implicitly tied to broad goals of ecosystem sustainability. Restoration, maintenance, and promotion of natural ecological processes and ecosystem services define the mission of most public land-management agencies as well as many private (e.g., nongovernmental organizations) forest reserves. These goals often underlie economic and utilitarian goals of the forest industry and other special-use forest land owners as well. Climate adaptation efforts that benefit and promote goals of ecosystem sustainability are considered successful. Successful implementation of climate adaptation plans occurs when projects are developed and deployed for specific places with concrete treatments and prescriptions, explicit objectives, and for definitive time periods. Successful implementation also implies that monitoring and adaptive management schedules are integrated in out-year efforts, and are secured with funds and capacity needed for completion.

Education and Training

Given the limited inclusion until recently of climate-science and climate-effects courses in college curricula for forest

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managers, a knowledge vacuum exists among practitioners and decisionmakers about basic scientific principles. Training for practitioners in the fundamental concepts of climatology and physical and ecological sciences related to climate change is essential. Such knowledge will increase the institutional capacity to understand potential effects of climate change and associated irreducible uncertainty, and to construct appropriate strategies and actions. A multilevel approach facilitates climate change education and dialogue for practitioners. A regional education program in the Northern United States incorporated several elements (Peterson et al. 2011), including basic education, intensive training, and discipline-specific and targeted workshops (fig. 4.1). Short (1- to 2-day) basic educational seminars convey fundamental principles of climate change and the effects of climate change on ecosystems and generate discussion of how different resources under management consideration can adapt to projected changes. Intensive training includes week-long courses providing indepth information and detailed explanations of fundamental climate processes and interactions, as well as greater detail on mechanisms of forest response to climate stressors. Participants have the opportunity to evaluate issues or resources by using available (e.g., online) tools. Discipline-specific trainings allow focused presentation and discussion of climate change implications for specific resource issues (e.g., silviculture, wildlife).

Science-Management Partnerships

Partnerships between scientists and managers are needed to improve understanding of climate science and increase experience in developing adaptation strategies. These collaborations can develop in different forms. For example, science information might reside with staff within an agency, but in different program areas than those traditionally involved with forest management. University extension specialists have a long history of spanning boundaries between science and applications (e.g., providing genetic expertise in developing seed-transfer rules), and can be brought into partnerships. Research scientists with universities and agencies increasingly participate in resource management collaborations. A key element in all collaborations is that they maintain interactive dialogue, with managers and scientists reciprocally learning from and informing each other about relevance.

Risk and Uncertainty

Given the environmental complexities of forest ecosystems, and their diverse and often conflicting institutional and societal roles, decisionmakers have long confronted challenges of risk and uncertainty. Climate change adds further dimensions of uncertainty, increasing the complexity of risk analyses. Although trends in climate and ecosystem response

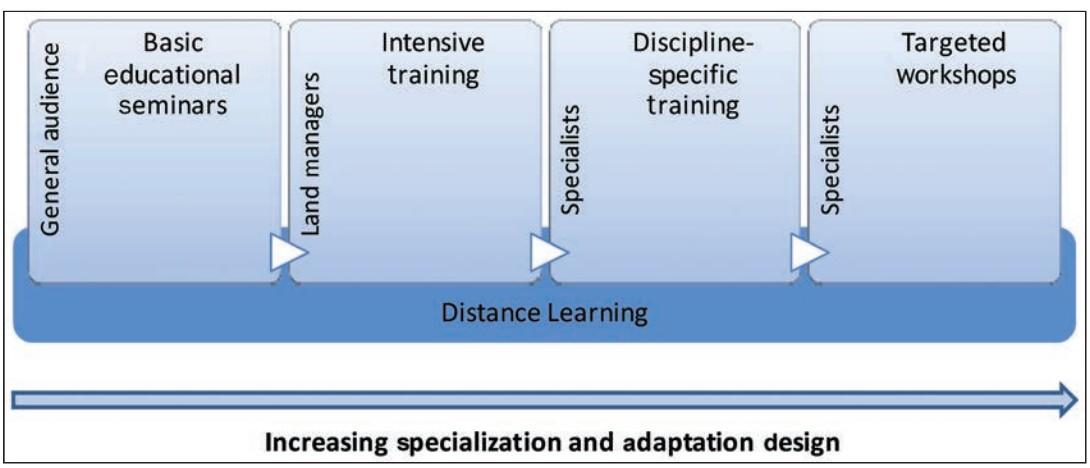

Figure 4.1—Conceptual diagram of educational and training efforts leading to increased complexity of adaptation planning and activities. These elements are integrated but need not be taken consecutively. Distance learning can be incorporated into all activities. (From Peterson et al. 2011.)

usually can be bounded with probabilistic envelopes, these are often wide and should be considered as only a guide for evaluating local decisions; unexpected conditions and surprises are especially important at local scales. In developing forest adaptation strategies, effort should be made to (1) be aware of risks, (2) assess vulnerabilities, (3) develop adaptation responses that are realistic yet minimize uncertainties, and (4) incorporate new knowledge and learning gained over time to modify decisions as appropriate (adaptive management) (Moser and Luers 2008). Adaptation responses to risk include (1) no action—continue conventional practices, (2) contingency planning—develop a response strategy (e.g., to anticipated major disturbance), and (3) anticipatory and proactive strategies—curtail or diminish potential impacts (e.g., of a major disturbance) while optimizing attainment of goals (Joyce et al. 2008).

Toolkit Approach

Novelty and surprise in climate-change effects, combined with a diversity of management objectives and of spatial and temporal management scales, mean that no single approach will fit all situations. A toolkit approach to adaptation strategies recognizes that, from a wide array of available methods in the literature or in practice, the best strategy will require selecting appropriate methods for the specific situation. Tools include resource management practices, educational and reference modules, decision-support aids, and qualitative and quantitative models that address adaptation of natural and cultural resources to climate change (Peterson et al. 2011). Tools include existing management practices, perhaps used in new ways, as well as novel approaches developed specifically to meet climate challenges.

No-Regrets Decisionmaking

"No-regrets" decisionmaking refers to actions that result in a variety of benefits under multiple scenarios and have little or no risk of socially undesired outcomes. This would include (1) implementing fuel treatments in dry forests to reduce fire hazard and facilitate ecological restoration, while creating resilience to increased fire occurrence in a warmer climate, and (2) installing new, larger culverts in locations where peak flows during flooding are expected to be higher in a warmer climate, thus protecting roads and reducing maintenance costs. These types of actions benefit resources and values regardless of climate-change effects and can be implemented in the near term (Swanston and Janowiak 2012).

Flexibility and Adaptive Learning

Uncertainty about future climates and ecosystem responses, and limited experience to date in developing forest adaptation strategies, imply that flexibility, experimentation, and adaptive learning should be incorporated into all efforts to develop adaptation strategies. Ideally a formal adaptive-management program will be developed in conjunction with projects implemented, but other approaches to monitoring that enable change of management practices are also appropriate.

Mixed-Models Approach

Climate- and ecosystem-response models are proliferating rapidly. Regional and locally downscaled climate-change scenarios logically seem useful for conducting vulnerability analyses and developing adaptation responses at scales relevant to forest sector needs. However, given the many processes about which we know very little, output from projections should be used cautiously and in conjunction with other filters. Models are often useful for examining forest-response correlates with recent historical events and for attributing influence or causality (e.g., dissecting climatic factors that might have influenced large wildfires or insect outbreaks). Models are often less useful for forecasting at small spatial scales or over long time periods, and in regard to complex biological processes. Output from models is useful as background information for envisioning a range of potential futures rather than to project a single outcome. The use of different types of models to address the same area and issue is recommended, such as models built with different assumptions, process interactions, and input data. Both quantitative (algorithm-based) and qualitative models (e.g., flow charts, indices, and verbal tools) should also be considered, and differences and similarities in projected futures can be evaluated. In recent years, it has been suggested that, if a model (or several models) hindcasts observed historical conditions well, it will also accurately predict future conditions. This is not necessarily true, because a given result can be reached via multiple pathways; in other words, a model can produce a correct historical reconstruction for the wrong reasons (Crook and Forster 2011), which means that forecasts could also be wrong. The experience and judgment of resource professionals are also important for evaluating potential future climate conditions and ecosystem responses. A recently developed summary of frequently asked questions (Daniels et al. 2012) can guide the effective use of models.

Integration With Other Priorities and Demands of Forest Management

Mitigation, involving actions to reduce human influence on the climate system, is another fundamental approach for addressing climate challenges (Metz et al. 2001), and integrating mitigation activities with adaptation strategies is important. The best approach is usually to address mitigation and adaptation goals concurrently, although in some situations, strategies may conflict, and compromise choices may be required. Climate change remains only one of many challenges confronting forest management, and other priorities must be evaluated at different temporal scales. For example, managing under the Endangered Species Act of 1973 (ESA) can invoke actions that, by regulatory imperative, are required in the short term but make little sense, given long-term projections of the effects of climate change. For forest lands where ecological sustainability is the central goal, ecosystem-based management as practiced in land management since the late 1980s (e.g., Kohm and Franklin 1997, Lackey 1995) provides a foundation for addressing most aspects of climate-change effects. Ecosystem-based management acknowledges that natural systems change continuously and that such dynamics bring high levels of uncertainty. Ecosystem-based management concepts are therefore appropriate foundational principles in developing forest adaptation strategies.

Placing Adaptation in Context

Forest ecosystems in the United States occur in diverse environmental, institutional, and regulatory contexts. Socially beneficial outcomes for climate adaptation depend on matching the best strategy with the context.

Biogeography and Bioclimate

Composition, structure, and processes of forests are influenced by their location, which determines the continental-to-local climatic regimes of forest ecosystems, physical context (geomorphology, soils, tectonics, topography), biogeographic constraints and opportunities (corridors or barriers to movement), ecological legacy (historical and prehistorical ecosystems), and a myriad of societal influences, such as land ownership, regulatory context, and land use histories. Adaptation strategies will differ in detail, if not always overall approach, for forest ecosystems in different parts of the United States.

Scale

Climate change affects forest ecosystems at many temporal and spatial scales, for example, from its influence on timing of bud burst to the evolution of ecotypes, and from trophic interactions on a rotting log to shifts in biome distribution across continents. The longevity of forest trees, combined with their significant influence on the physical landscape (e.g., soil development, watershed quality) and role as habitat, adds complexity to scale issues. Analysis at the correct spatial scales is especially important for assessing trends of climate change and ecological response, given that averages and trends on broad scales (e.g., continental) can mask variability at fine scales (e.g., watershed).

An adaptation framework based on appropriate temporal and spatial scales (e.g., Peterson and Parker 1998) ensures that plans and activities address climate effects and responses effectively. Because scales are nested, the best strategies focus on the scale of the relevant project and include evaluation of conditions and effects at scales broader than the project level, as well as analysis of effects at finer scales (tables 4.1 and 4.2). Broad-scale analysis establishes context,

Table 4.1—Factors that affect the relevance of information for assessing vulnerability to climate change of large, intermediate, and small spatial scales

	Relevance by spatial scale		
Factor	**Large**[a]	**Intermediate**[b]	**Small**[c]
Availability of information on climate and climate change effects	High for future climate and general effects on vegetation and water	Moderate for river systems, vegetation, and animals	High for resource data, low for climate change
Accuracy of predictions of climate change effects	High	Moderate to high	High for temperature and water, low to moderate for other resources
Usefulness for specific projects	Generally not relevant	Relevant for forest density management, fuel treatment, wildlife, and fisheries	Can be useful if confident that information can be downscaled accurately
Usefulness for planning	High if collaboration across management units is effective	High for a wide range of applications	Low to moderate

[a] More than 10 000 km^2 (e.g., basin, multiple national forests).
[b] 100 to 10 000 km^2 (e.g., subbasin, national forest, ranger district).
[c] Less than 100 km^2 (e.g., watershed).
Source: Modified from Peterson et al. 2011.

Table 4.2—Factors that affect the relevance of information for assessing vulnerability to climate change of large, intermediate, and small time scales

	Relevance by time scale		
Factor	**Large**[a]	**Intermediate**[b]	**Small**[c]
Availability of information on climate and climate change effects	High for climate, moderate for effects	High for climate and effects	Not relevant for climate change and effects predictions
Accuracy of predictions of climate change effects	High for climate and water, low to moderate for other resources	High for climate and water, moderate for other resources	Low
Usefulness for specific projects	High for temperature and water, low to moderate for other resources	High for water, moderate for other resources	Low owing to inaccuracy of information at this scale
Usefulness for planning	High	High for water, moderate for other resources	Low

[a] More than 50 years.
[b] 5 to 50 years.
[c] Less than 5 years.
Source: Modified from Peterson et al. 2011.

including recognition of processes and effects that manifest only at large scales (e.g., species decline, cumulative watershed effects), potential undesired consequences that could be alleviated by early action, and the need for large management units and collaboration across ownerships.

Institutional and Regulatory Contexts

Forests are managed for many goals. Most publically administered forest lands are managed for long-term ecological and physical sustainability. Within that broad goal, emphasis differs by designation for protection level (parks, wilderness, and reserves) and ecosystem services (national and state forests, Bureau of Land Management [BLM] forest and woodlands, and tribal forest lands). The focus on maintenance of ecological and environmental conservation on public lands is subject to strict legal and regulatory direction, such as the National Environmental Policy Act [NEPA] of 1969, Clean Air Act of 1970, Clean Water Act of 1977, Endangered Species Act [ESA] of 1973, and their state counterparts. The goals, tactics, and time horizons of climate adaptation strategies for lands under these jurisdictions and legal mandates differ considerably from those of private forest lands. Adaptation on industrial forest land focuses on strategies most effective to sustain productive output over the period of economic analysis (Sedjo 2010), whereas adaptation on nonindustrial private forest lands differs by the diverse goals and capacities of landowners. Other institutional considerations that influence adaptation relate to educational and technological capacities, staff resources, and funding. Choices also depend on the quality of collaboration, because support, trust, and interaction among stakeholders influence the type of risk accepted and commitment to novel or experimental approaches.

Adaptation Strategies and Implementation

Overview of Forest Adaptation Strategies

The literature on conceptual approaches to forest adaptation strategies (Baron et al. 2008, Joyce et al. 2008, Peterson et al. 2011, Swanston and Janowiak 2012) (table 4.3) includes broad conceptual frameworks, approaches to specific types of analyses (e.g., vulnerability assessments, scenario planning, adaptive management), and tools and guidance for site-specific or issue-specific problems. An umbrella approach for addressing adaptation at the highest conceptual level in

Table 4.3—Climate adaptation guides relevant to the forest sector

Category	Emphasis	Reference
Adaptation framework	General options for wildlands	Millar et al. 2007
	Options for protected lands	Baron et al. 2008, 2009
	Adaptation guidebooks	Peterson et al. 2012, Snover et al. 2007, Swanston and Janowiak 2012
Vulnerability analysis	Climate change scenarios	Cayan et al. 2008
	Scenario exercises	Weeks et al. 2011
	Forest ecosystems	Aubry et al. 2011, Littell et al. 2010
	Watershed analysis	Furniss et al. 2010
Genetic management	Seed transfer guidelines	McKenney et al. 2009
	Risk assessment	Potter and Crane 2010
Assisted migration	Framework for translocation	McLachlan et al. 2007, Riccardi and Simberloff 2008
Decisionmaking	Silvicultural practices	Janowiak et al. 2011b
	Climate adaptation workbook	Janowiak et al. 2011a
Priority setting	Climate project screening tool	Morelli et al. 2011b

forest ecosystems focuses on resistance, resilience, response, and realignment strategies (Millar et al. 2007) (box 4.1). These general principles help in early phases of planning to dissect the range and scales of appropriate options at the broadest levels (Spittlehouse 2005), and they apply to many management and land-ownership contexts, but they do not provide guidance for developing site-specific plans. Similarly, broad discussions focus on other fundamental principles relative to forest adaptation planning, such as reinterpreting the role of historical variability, ecological change over time, and use of historic targets in management and restoration (Harris et al. 2006, Jackson 2012, Milly et al. 2008).

Special concerns for adaptation in parks and protected areas were developed by Baron et al. (2008, 2009) and Stephenson and Millar (2012), who emphasize the need to acknowledge that future ecosystems will differ from the past, and that fundamental changes in species and their environments will be inevitable. Given anticipated nonanalog climates and ecosystem responses, science-based adaptation will be essential. Baron et al. (2009) emphasized the need to identify resources and processes at risk, define thresholds and reference conditions, establish monitoring and assessment programs (adaptive management), and conduct scenario planning. They emphasize that preparing for and adapting to climate change is as much a cultural and intellectual challenge as an ecological issue. Diverse regulatory and value contexts dictate what will be desired for future ecosystem conditions, which drive decisions about goals, strategies, and actions.

The reality of change and novelty in future forest ecosystems under changing climates underscores the importance of vulnerability assessments in developing adaptation strategies (Aubry et al. 2011, Johnstone and Williamson 2007, Lindner et al. 2010, Littell and Peterson 2005, Littell et al. 2010, Nitschke and Innes 2008, Spittlehouse 2005). Vulnerability assessments can differ in terms of subject matter, geographic focus, level of detail, and quantitative rigor, but usually require a science-management partnership to ensure that current science is used to evaluate climate-change effects. Regional-scale assessments can be cautiously downscaled to smaller management units, recognizing there will be tradeoffs in accuracy. Some of the most detailed approaches to vulnerability assessment in response to climate challenges have focused on watersheds (e.g., Furniss et al. 2010), as described in the examples below. Scenario planning as a tool for vulnerability assessment has been well developed for forested ecosystems in U.S. national parks (Weeks et al. 2011). Tools developed for setting priorities in forest planning and for assessing risks are especially applicable for near-term decisionmaking (Janowiak et al. 2011a, Morelli et al. 2011).

Several efforts have taken comprehensive approaches to incorporate both conceptual strategies and specific tools into integrated guidebooks for developing adaptation strategies in the forest sector (Peterson et al. 2011, Swanston and Janowiak 2012). These guidebooks encourage education and training in the basic climate sciences and describe how to proceed from assessment to on-the-ground practices.

Box 4.1

A general framework for adaptation options suitable for conditions of forested ecosystems. Options range from short-term, conservative strategic approaches to strategies for long-term, proactive plans. (From Millar et al. 2007.)

Promote resistance

Actions that enhance the ability of species, ecosystems, or environments to resist forces of climate change and that maintain values and ecosystem services in their present or desired states and conditions.

Increase resilience

Actions that enhance the capacity of ecosystems to withstand or absorb increasing impact without irreversible changes in important processes and functionality.

Enable ecosystems to respond

Actions that assist climatically driven transitions to future states by mitigating and minimizing undesired and disruptive outcomes.

Realign highly altered ecosystems

Actions that use restoration techniques to enable ecosystem processes and functions (including conditions that may or may not have existed in the past) to persist through altered climates and in alignment with changing conditions.

Strategic steps for forest climate adaptation—
The following steps represent broad consensus that has emerged on developing forest climate adaptation strategies (review in Swanston and Janowiak 2012).

Step 1: Define location (spatial extent), management goals and objectives, and timeframes—Determining spatial and temporal scales and site-specific locations is essential for developing appropriate strategies. Management goals and objectives (box 4.2) for climate adaptation should be explicit and integrated with mitigation and other nonclimate-related management goals. This does not necessarily mean that goals are stated in narrowly specific quantitative terms; indeed, many forest adaptation goals and objectives can be defined broadly (e.g., sustaining ecosystem services).

Step 2: Analyze vulnerabilities—Vulnerability to climate change can be defined as “the degree to which geophysical, biological, and socio-economic systems are susceptible to, and unable to cope with, adverse impacts of climate change” (Solomon et al. 2007). Vulnerability is a function of the degree to which a system is exposed to a change in climatic conditions, its sensitivity to that change, and its adaptive capacity (Gallopín 2006, IPCC 2001, Solomon et al. 2007). Climate-vulnerability assessments are a central step in developing adaptation strategies and can take different forms (Glick et al. 2011, USGCRP 2011). Whichever approach is used, the intent is to determine how climatic variability and change might affect resources of concern, and to aid in developing appropriate priorities, strategies, and timeframes for action.

Step 3: Determine priorities—Priority actions for climate adaptation often differ from those for traditional forest management contexts. Furthermore, given rapidly changing conditions and emerging understanding of trends, priorities need to be reassessed regularly. When conditions are urgent and resources limited (e.g., a species in rapid decline), triage methods can be useful (Joyce et al. 2008); in longer term planning, no-regrets assessments (National Research Council 2002, Overpeck and Udall 2010) minimize risk.

Step 4: Develop options, strategies, and tactics—Swanston and Janowiak (2012) present a framework approach for developing adaptation plans. This process begins at a broad conceptual level and steps down to regional and local, site-specific project planning, as reflected by the increasing specificity of the following terms (fig. 4.2). **Adaptation options** are fundamental concepts and the broadest and most widely applicable level in a continuum of

Box 4.2

Management Goals

Management goals are broad, general statements that express a desired state or process to be achieved. They are often not attainable in the short term and provide the context for more specific objectives. Examples of management goals include:

- Maintain and improve forest health and vigor
- Maintain wildlife habitat for a variety of species

Management Objectives

Management objectives are concise, time-specific statements of measurable planned results that correspond to preestablished goals in achieving a desired outcome. These objectives include information on resources to be used for planning that defines precise steps to achieve identified goals. Examples of management objectives include:

- Regenerate a portion of the oldest aspen forest type through clearcut harvest in the next year to improve forest vigor in young aspen (*Populus* spp.) stands.
- Identify and implement silvicultural treatments within 5 years to increase the oak (*Quercus* spp.) component of selected stands and enhance wildlife habitat.

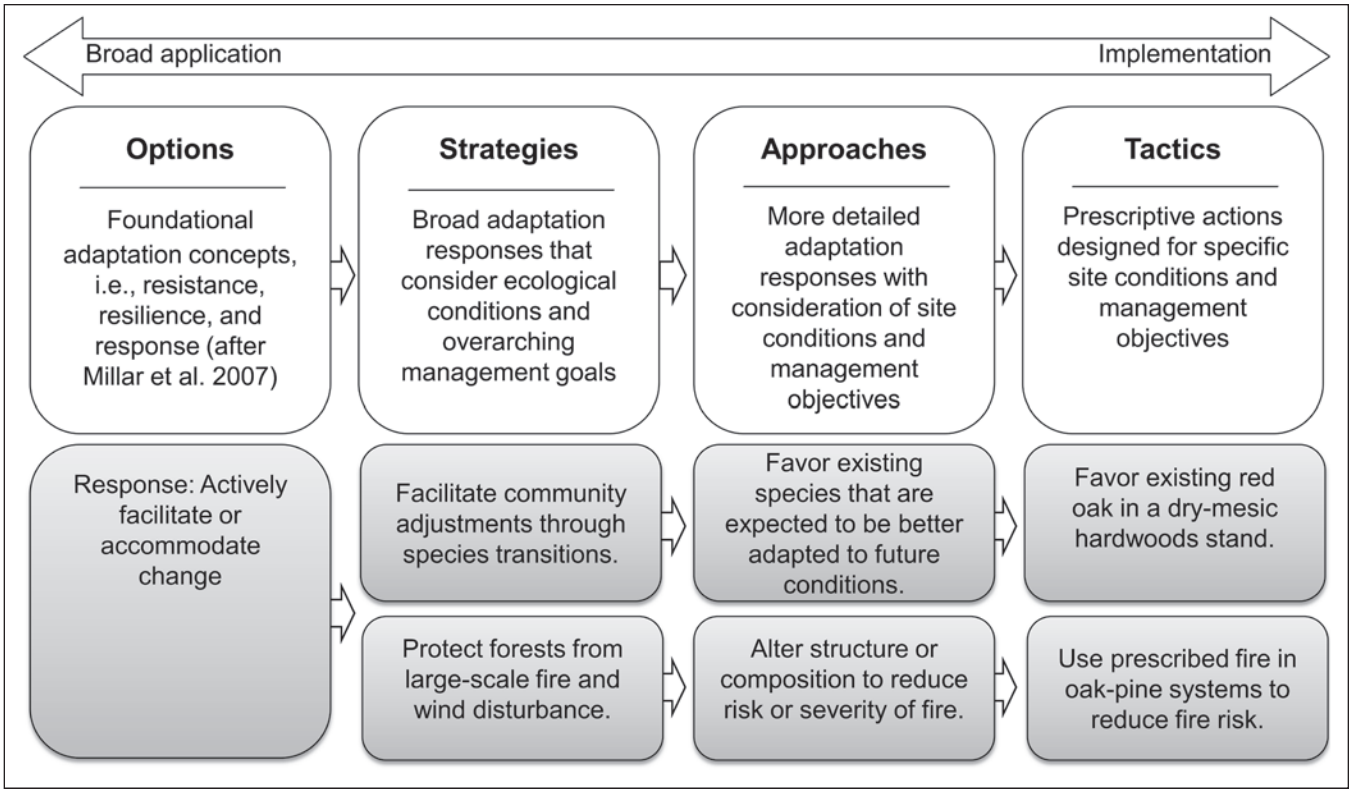

Figure 4.2—A continuum of adaptation options to address needs at appropriate scales, and examples of each (shaded boxes). (From Janowiak et al. 2011.)

management responses to climate change. Options include resistance, resilience, response, and realignment, which reflect conservative, short-term categories to proactive, long-range ones (Millar et al. 2007) (see box 4.1); they can be general or specific and focused on the local situation. **Adaptation strategies** illustrate ways that options can be used. Similar to options, strategies are broad and can be applied in many ways across different forest landscapes (table 4.4). **Approaches** provide greater detail on how forest managers can respond, and differences in application among specific forest types and management goals become evident. **Tactics** are the most specific adaptation response, providing prescriptive direction in how actions are applied on the ground. The culmination of this process is development of a plan, such as a NEPA document or other project plan, prescription, or treatment description.

Step 5: Implement plans and projects**—**Implementation of projects should include replication, randomization, and other experimental design elements, as possible, which sets up the value of the final step.

Step 6: Monitor, review, adjust**—**Formal adaptive management is often advocated as a key element in forest climate-adaptation planning (Baron et al. 2008, 2009; Joyce et al. 2008). Adaptive management involves a comprehensive set of steps developed in an experimental framework. Monitoring is tied to predefined thresholds and other target goals. These are developed to test hypotheses about project effectiveness and appropriateness, and, if thresholds are exceeded, trigger review and adjustment of plans (Joyce et al. 2008, 2009; Margoluis and Salafsky 1998; Walters 1986). In practice, many constraints exist to implementing the full adaptive management cycle in forest ecosystems (Joyce et al. 2009). However, informal monitoring keyed to assessing treatment effectiveness and enabling adjustment of practices is essential because of dynamic conditions driven by climate change.

Table 4.4—Climate change adaptation strategies under broad adaptation options

Strategy	Resistance	Resilience	Response
Sustain fundamental ecological conditions	X	X	X
Reduce the impact of existing ecological stressors	X	X	X
Protect forests from large-scale fire and wind disturbance	X		
Maintain or create refugia	X		
Maintain or enhance species and structural diversity	X	X	
Increase ecosystem redundancy across the landscape		X	X
Promote landscape connectivity		X	X
Enhance genetic diversity		X	X
Facilitate community adjustments through species transitions			X
Plan for and respond to disturbance			X

Source: Butler et al. 2012.

Tools and Resources for Adaptation and Implementation

Until recently, few guides to implementing climate adaptation plans were available, but many active projects now exist, including in the forest sector. The examples below are not exhaustive, but represent the type of tools available and the meta-level databases and Web resources that assist in finding relevant tools for specific locations and needs.

Web Sites

Climate Change Resource Center (CCRC)—
Described more below, the CCRC (http://www.fs.fed.us/ccrc) is a U.S. Forest Service-sponsored portal dedicated to compiling comprehensive, credible information and resources relevant to forest resource managers (USDA FS 2011a).

Climate Adaptation Knowledge Exchange (CAKE)—
This is a joint project of Island Press and EcoAdapt (CAKE 2011) (http://www.cakex.org). Its main feature is a retrievable knowledge base that can assist in managing natural systems in the face of rapid climate change by compiling relevant information. The CAKE maintains an interactive online platform, creating a directory of practitioners to share knowledge and strategies, and identifying data tools and information available from other sites. Case studies, toolboxes, and reference materials are relevant to forest sector issues.

NaturePeopleFuture.org—
This is The Nature Conservancy (TNC) knowledge base for climate adaptation (TNC 2011a) (http://conserveonline.org/workspaces/climateadaptation). The Web site is used to collect input on climate-adaptation projects, summarize relevant products and ideas, and communicate about TNC efforts to draw together scientific research and innovative conservation projects. Geographically diverse forest ecosystem situations are presented, and adaptation tools and the methods discussed are relevant to forest sector issues.

Tribes and Climate Change—
Developed by the Institute for Tribal Environmental Professionals and Northern Arizona University, Tribes and Climate Change (http://www4.nau.edu/tribalclimatechange) summarizes information and resources to help Native people better understand climate change and its effects on their communities (NAU 2011). The site provides basic climate science information, including climate change scenarios and vulnerability assessment background, profiles of tribes throughout the United States that are addressing climate change effects, audio files of elders discussing adaptation from traditional perspectives, and resources and contacts to develop adaptation strategies. A section is devoted to forest ecosystems.

Tools

Climate Wizard—
Sponsored and developed by TNC, Climate Wizard is a Web-based tool that uses select climate projections relevant to the time and space resolution of inquiries, enabling users to visualize modeled changes at several time and spatial scales (TNC 2011b). Used with scenario exercises, the Wizard can assist development of forest adaptation strategies.

Vegetation Dynamics Development Tool (VDDT)—
Developed by the TNC Southwest Forest Climate Assessment Project, VDDT is a user-friendly computer tool for forest resource managers (ESSA 2011). VDDT provides a state-and-transition landscape modeling framework for examining the role of various disturbance agents and management actions in vegetation change. It allows users to create and test descriptions of vegetation dynamics, simulating them at the landscape level. VDDT provides a common platform for specialists from different disciplines to collectively define the roles of various processes and agents of change on landscape-level vegetation dynamics, and allows for rapid gaming and testing of ecosystem sensitivity to alternative assumptions.

Template for Assessing Climate Change Impacts and Management Options (TACCIMO)—
This Web-based tool connects forest planning to climate-change science providing access to relevant climate-change projections and links to peer-reviewed scientific statements describing effects and management adaptation options (North Carolina State University 2011). The tool is intended for all forest planners with a need for public and private land management information. Input is given by the user on management conditions and capabilities to address climate change, which is linked with available physical and biological information on climate impacts and management options. TACCIMO produces a customized report that synthesizes user input needs with available science and related planning options.

Climate Project Screening Tool (CPST)—
This verbal interview tool helps resource managers explore options for ameliorating the effects of climate in resource projects (Morelli et al. 2011b). The CPST also acts as a priority-setting tool, allowing managers to assess relative vulnerabilities and anticipate effects of different actions. It also helps managers identify and assess projects that are soon to be implemented but have not benefited from serious consideration of climate influence. Through a set of guided questions and development of answers based on available climate and ecosystem information, the CPST reduces uncertainty by identifying possible effects of both climate change and adaptation actions on resources.

Climate Change Adaptation Workbook—
The Climate Change Adaptation Workbook is designed to help forest managers more effectively bring climate change considerations to the spatial and temporal scales where management decisions are made (Janowiak et al. 2011a). The workbook is an analytical process built on a conceptual model for adaptation derived from adaptive management principles. It draws on regionally specific information, filtering climate and vegetation projections through professional judgment and experience. Using a five-step process, the workbook can help incorporate climate change in resource management at different spatial scales (e.g., stand, large ownership) and levels of decisionmaking (e.g., planning, problemsolving, implementation). By defining current management goals and objectives in the first step, the process integrates climate change adaptation into existing management efforts. It is not intended to provide specific guidance or replace other forms of management planning; rather, it relies on the expertise of natural resource professionals and complements existing management planning and decision-making.

System for Assessing Vulnerability of Species (SAVS)—
This verbal index tool identifies relative vulnerability or resilience of vertebrate species to climate change (Bagne et al. 2011). Designed for resource managers, SAVS uses a questionnaire with 22 predictive criteria to create vulnerability scores. The user scores species attributes relating to potential vulnerability or resilience associated with projections for their region. Six scores are produced: (1) an overall score denoting level of vulnerability or resilience, (2) four categorical scores (habitat, physiology, phenology, and

biotic interactions) indicating source of vulnerability, and (3) an uncertainty score, which reflects user confidence in the predicted response. The SAVS provides a framework for integrating new information into climate change assessments and developing adaptation plans.

Institutional Responses

President's Directive

Executive Order 13514 (2009), "Federal Leadership in Environmental, Energy, and Economic Performance," directs each federal agency to evaluate climate change risks and vulnerabilities to manage the short- and long-term effects of climate change on the agency's mission and operations. An interagency climate change adaptation task force includes 20 federal agencies and develops recommendations for agency actions in support of a national climate change adaptation strategy. The task force recommended that federal agencies establish climate change adaptation policies, increase agency understanding of how climate is changing, apply understanding of climate change to agency mission and operations, develop an adaptation plan and implement at least three adaptation actions in 2012, and evaluate and share "lessons learned" with other agencies.

Some of the more successful adaptation efforts to date have involved collaboration among different institutions. Collaboration can take many forms, such as between federal agencies, between federal and state agencies, between various agencies and Native American tribes, and between various land management agencies and a wide range of stakeholders. There is no standard model, and effective collaborations will differ by landscape and local institutional relationships.

U.S. Forest Service

The U.S. Forest Service climate response is led by the climate change advisor's office, which develops guidance and evaluates progress toward climate adaptation. Agency goals and actions are described in a strategic framework document (USDA FS 2008). Forest Service research and development also has a climate change strategic plan (Solomon et al. 2009). These documents state the conceptual visions for science-based adaptation on the 175 national forests and national grasslands. Of seven key goals in the overall framework, five pertain to climate adaptation:

Science—
Advance understanding of the environmental, economic, and social implications of climate change and related adaptation activities on forests and grasslands.

Adaptation—
Enhance the capacity of forests and grasslands to adapt to the environmental stresses of climate change and maintain ecosystem services.

Policy—
Integrate climate change, as appropriate, into policies, program guidance, and communications and put in place effective mechanisms to coordinate across and within deputy areas.

Education—
Advance awareness and understanding regarding principles and methods for sustaining forests and grasslands, and sustainable resource consumption, in a changing climate.

Alliances—
Establish, enhance, and retain strong alliances and partnerships with federal agencies, state and local governments, tribes, private landowners, nongovernmental organizations, and international partners to provide sustainable forests and grasslands for present and future generations.

Tactical approaches and implementation are outlined in the National Roadmap for Responding to Climate Change (USDA FS 2011b). The roadmap identifies 10 steps along four major dimensions, namely, agency and organizational capacity, partnerships and conservation education, adaptation, and mitigation (fig. 4.3). The process includes (1) science-based assessments of risk and vulnerability; (2) evaluation of knowledge gaps and management outcomes; (3) engagement of staff, collaborators, and partners through education, science-based partnerships, and alliances; and (4) management of resources via adaptation and mitigation. To assist in these tasks, the CCRC (USDA FS 2011a) serves as a reference Web site with information and tools to address

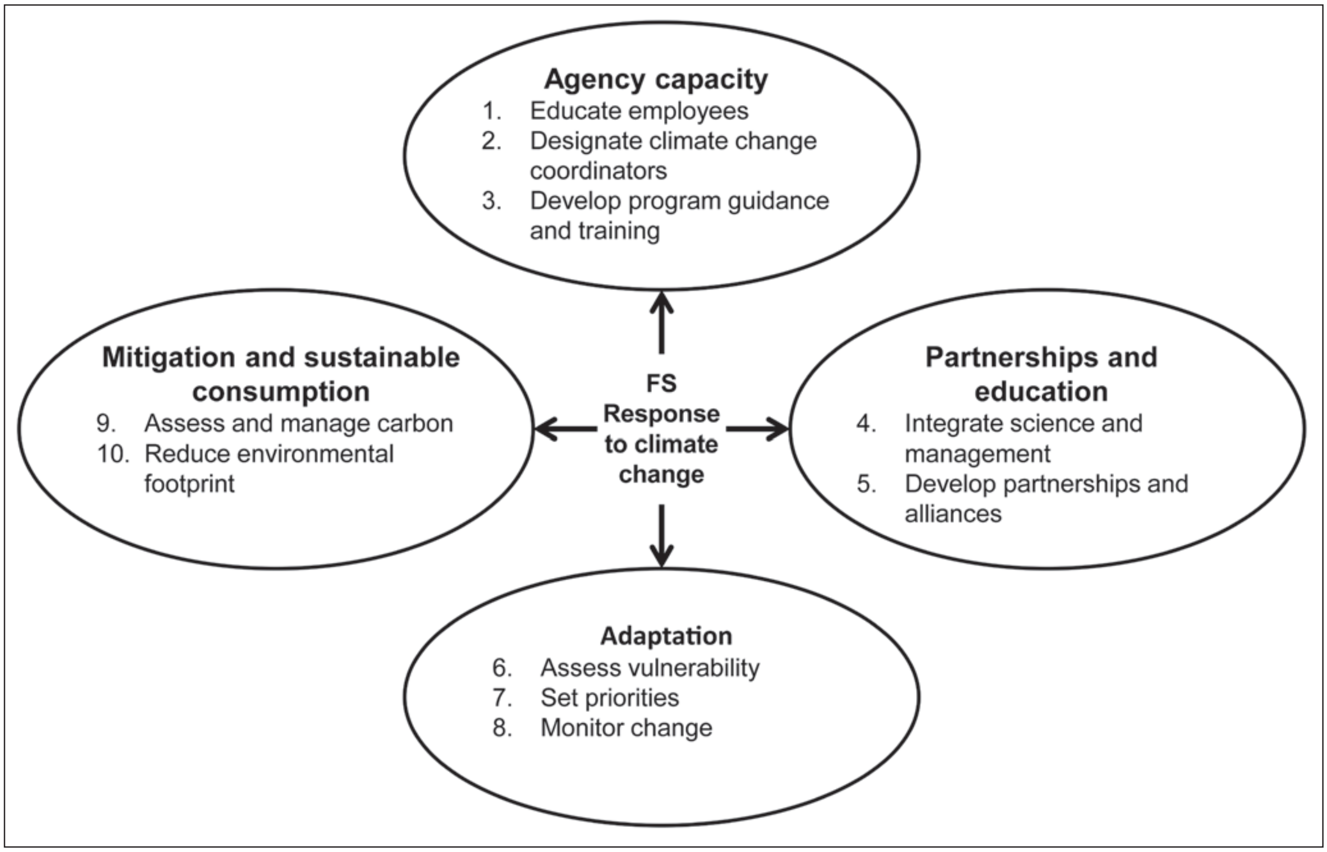

Figure 4.3—Four dimensions of action outlined by the U.S. Forest Service roadmap for responding to climate change. (From USDA FS 2011b.)

climate change in planning and project implementation. Climate change coordinators are designated for each Forest Service region. Current initiatives from research and management branches of the agency provide climate science, develop vulnerability assessments, prepare adaptive monitoring plans, and align planning, policy, and regulations with climate challenges (box 4.3).

The Performance Scorecard (USDA FS 2011c) (table 4.5) is used annually to document progress of national forests, regions, and research stations on adaptation plans and "climate smart" actions. The scorecard also identifies areas of weakness, knowledge gaps, and budgetary limitations, which the climate change advisor's office can subsequently highlight for attention.

U.S. Department of the Interior (DOI)

A U.S. Department of the Interior secretarial order (2009) provides a framework to coordinate climate change activities among DOI bureaus and to integrate science and management expertise with DOI partners. Climate Science Centers and Landscape Conservation Cooperatives form the cornerstones of the framework. Each has a distinct science and resource management role, but they share complementary capabilities in support of DOI resource managers and of integrated climate solutions with federal, state, local, tribal, and other stakeholders.

Climate science centers (CSC)—
Climate science centers are seven regional centers in development because of cooperative endeavors between DOI and universities to distill and make climate-adaptation information available to users. The CSCs fund the development of scientific information, tools, and techniques for resource

Box 4.3

U.S. Forest Service initiatives to promote progress toward achieving goals of the national roadmap for responding to climate change. (From USDA FS 2011b.)

Furnish predictive information on climate change and variability, both immediate and longer term, building on current research capacity and partnerships with the National Oceanic and Atmospheric Administration, National Aeronautics and Space Administration, U.S. Geological Survey, and other scientific agencies.

- Develop, interpret, and deliver spatially explicit scientific information on recent shifts in temperature and moisture regimes, including incidence and frequency of extreme events.
- Provide readily interpretable forecasts at regional and subregional scales.

Develop vulnerability assessments, working through research and management partnerships and collaboratively with partners.

- Assess the vulnerability of species, ecosystems, communities, and infrastructure and identify potential adaptation strategies.
- Assess the impacts of climate change and associated policies on tribes, rural communities, and other resource-dependent communities.
- Collaborate with the U.S. Fish and Wildlife Service and National Marine Fisheries Service to assess the vulnerability of threatened and endangered species and to develop potential adaptation measures.

Tailor monitoring to facilitate adaptive responses.

- Expand observation networks, intensify sampling in some cases, and integrate monitoring systems across jurisdictions (see, for example, the national climate tower network on the experimental forests and ranges).
- Monitor the status and trends of key ecosystem characteristics, focusing on threats and stressors that may affect the diversity of plant and animal communities and ecological sustainability. Link the results to adaptation and genetic conservation efforts.

Align Forest Service policy and direction with the Forest Service strategic response to climate change.

- Revise National Forest System land management plans using guidance established in the new Planning Rule, which requires consideration of climate change and the need to maintain and restore ecosystem and watershed health and resilience.
- Review Forest Service Manuals and other policy documents to assess their support for the agency's strategic climate change direction. Evaluate current policy direction for its ability to provide the flexibility and integration needed to deal with climate change.
- Develop proposals for addressing critical policy gaps.

Table 4.5—Performance scorecard used by the U.S. Forest Service for annual review of progress and compliance, and to identify deficit areas in implementation of the national roadmap for responding to climate change

Scorecard element	Questions to be addressed	Yes/no
Organizational capacity:		
Employee education	Are all employees provided with training on the basics of climate change, impacts on forests and grasslands, and the Forest Service response?	
	Are resource specialists made aware of the potential contribution of their own work to climate change response?	
Designated climate change coordinators	Is at least one employee assigned to coordinate climate change activities and be a resource for climate change questions and issues?	
	Is this employee provided with the time, training, and resources to make his/her assignment successful?	
Program guidance	Does the unit have written guidance for progressively integrating climate change considerations and activities into unit-level operations?	
Engagement:		
Science and management partnerships	Does the unit actively engage with scientific organizations to improve its ability to respond to climate change?	
Other partnerships	Have climate change-related considerations and activities been incorporated into existing or new partnerships (other than science partnerships)?	
Adaptation:		
Assessing vulnerability	Has the unit engaged in developing relevant information about the vulnerability of key resources, such as human communities and ecosystem elements, to the impacts of climate change?	
Adaptation actions	Does the unit conduct management actions that reduce the vulnerability of resources and places to climate change?	
Monitoring	Is monitoring being conducted to track climate change impacts and the effectiveness of adaptation activities?	
Mitigation and sustainable consumption:		
Carbon assessment and stewardship	Does the unit have a baseline assessment of carbon stocks and an assessment of the influence of disturbance and management activities on these stocks?	
	Is the unit integrating carbon stewardship with the management of other benefits being provided by the unit?	
Sustainable operations	Is progress being made toward achieving sustainable operations requirements to reduce the environmental footprint of the agency?	

Source: Adapted from USDA FS 2011b, 2011c.

managers to anticipate, monitor, and adapt to climate and to develop adaptation responses at multiple scales. Forest ecosystems are a primary focus of several CSCs.

Landscape conservation cooperatives (LCC)—
The LCCs complement existing science and conservation efforts of the CSCs and partners by leveraging resources and strategically targeting science topics to inform conservation decisions and actions (USDI FWS 2011). Each LCC operates within a specific landscape, with 21 geographic areas total (fig. 4.4). Partners include federal, state, and local governments, tribes, universities, and other stakeholders. The LCCs form a network of resource managers, scientists, and public and private organizations that share a common need for scientific information and interest in conservation. Land conservation cooperatives products include resource assessments, examples demonstrating the application of climate models, vulnerability assessments, inventory and monitoring protocols, and conservation plans and designs. Adaptation products include assessments of climate change effects and development of adaptation strategies for wildlife migration corridors, wildfire risk and fuel treatments, drought impacts and amelioration, detection and control of invasive species, and restoration of forest landscapes.

National Park Service (NPS)—
The NPS climate change response strategy (NPS 2010) provides direction for addressing effects of climate change in NPS-administered park units. The strategy directs NPS to adapt natural resources on its lands by using scenario exercises as a central approach, thereby creating flexible plans at park scales for dealing with climate effects. The broad goals of the strategy include developing effective natural-resource adaptation plans and promoting ecosystem resilience. Specifically the strategy requires that units (1) develop adaptive capacity for managing natural and cultural resources; (2) inventory resources at risk and conduct vulnerability assessments; (3) prioritize and implement actions and monitor the results; (4) explore scenarios, associated risks, and possible management options; and (5) integrate climate change effects in facilities management. The legacy dictum for NPS management has been to preserve and restore natural (usually interpreted as historical) conditions. Ecosystem dynamics associated with climate change have forced rethinking of this concept, and new paradigms are emerging in national park management for incorporating ecological change in adaptation philosophies and managing "beyond naturalness" (Cole and Yung 2010, Stephenson and Millar 2012). Emphasis on scenario exploration as a discussion focus is intended to promote solutions that address multiple feasible future outcomes.

Bureau of Land Management (BLM)—
The BLM focuses on a landscape approach to climate change adaptation, working within functional ecosystems at large scales and across agency boundaries, and assessing natural resource conditions and trends, natural and human influences, and opportunities for resource conservation and development. The landscape approach consists of (1) rapid ecoregional assessments (REA), which synthesize the information about resource conditions and trends within an ecoregion, with emphasis on areas of high ecological value (e.g., important wildlife habitats and corridors); (2) ecoregional direction, which uses the results of REAs to identify management priorities for public lands in an ecoregion and guide adaptation actions; (3) monitoring for adaptive management, which relies on monitoring and mapping programs to meet information needs and assessment, understand resource conditions and trends, and evaluate and refine implementation actions; and (4) science integration, which relies on participation with CSCs to provide science for management needs. To date, these have not yielded operational climate-change adaptation plans.

Regional Integrated Sciences and Assessments (RISA)

Funded by National Oceanographic and Atmospheric Administration's Climate Program Office, the RISA program supports research and stakeholder interaction to improve understanding of how climate affects various regions of the United States, and to facilitate the use of climate information in decisionmaking. The RISA teams analyze climate data; apply, provide, and interpret climatic information for resource managers and policymakers in the United States; and are a good source of information on climate change and regional effects of climate change.

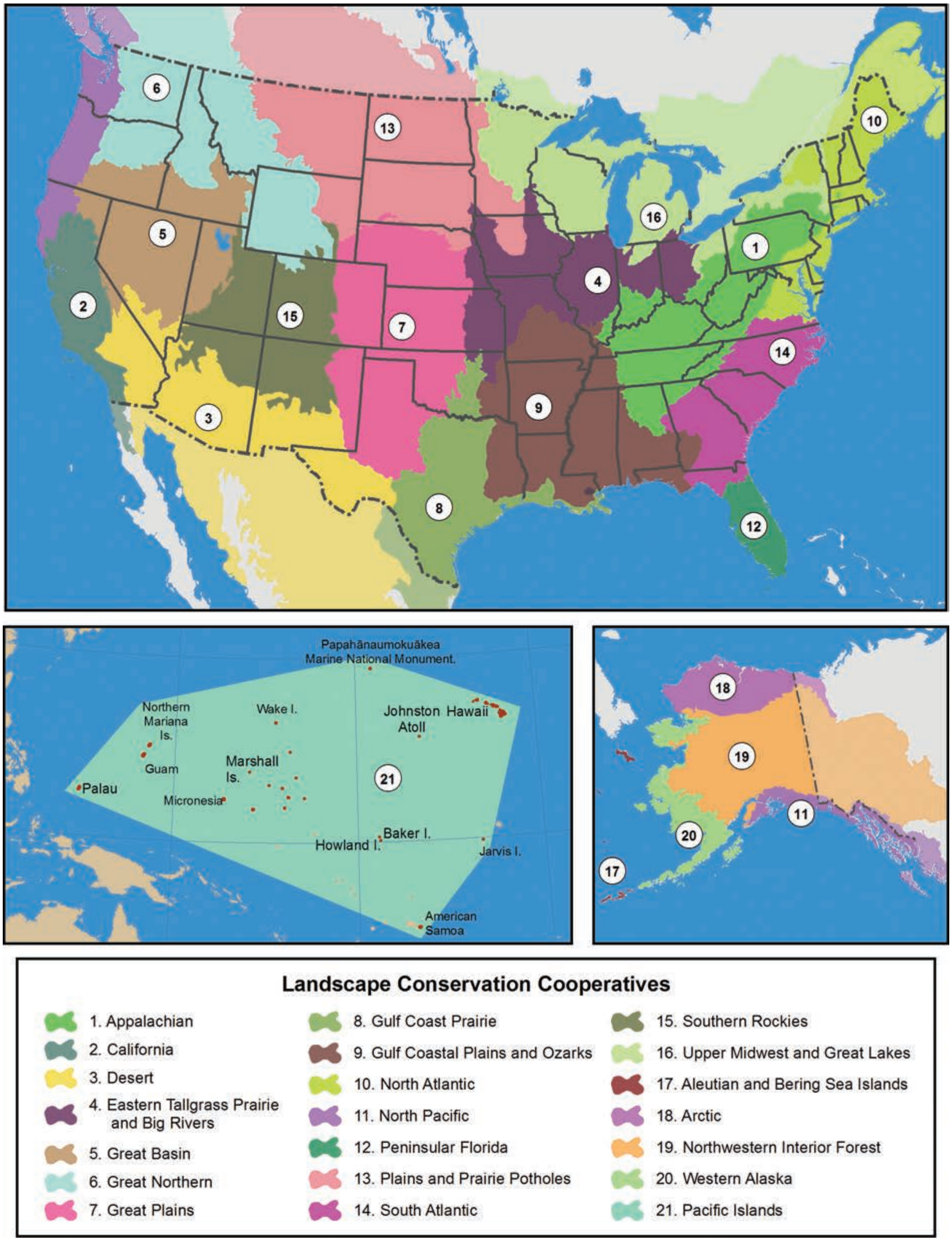

Figure 4.4—Twenty-one landscape conservation cooperatives of the U.S. Department of the Interior integrate climate-adaptation responses across federal, state, tribal, local, and private interests within geographically coherent regions. (From DOI FWS 2011.)

State and Local Institutions

Climate-adaptation responses of state and local institutions are diverse, ranging from minimal action to fully developed and formal programs. State responses that focus on forest-sector issues include the following.

Washington State climate response strategy—
Beginning in 2009, the Washington Department of Ecology in partnership with the departments of Agriculture, Commerce, Fish and Wildlife, Natural Resouces, and Transportation, began developing a strategy to prepare for the effects of climate change outlined in the Washington State Climate Change Impacts Assessment (McGuire et al. 2009, Washington State Department of Ecology 2012). This collaborative effort involving a variety of public and private stakeholders sought to develop recommendations for addressing the effects of climate change. The working groups emphasized the priority of forest resources in the strategy, and recommendations for climate-adaptation efforts in major forest ecological systems have been developed (Helbrecht et al. 2011) (box 4.4), including for fire management and genetic preservation (Jamison et al. 2011). These options emphasize research, assessing vulnerabilities, developing pilot projects, improving forest health, avoiding forest conversion, using prescribed fire, and using adaptive management in decisionmaking. Strategies consistent with adaptation on forest lands include (1) preserve and protect Washington's existing working forest, (2) assess how land management decisions help or hinder adaptation, (3) foster interagency collaboration, (4) promote sociocultural and economic relations between eastern and western Washington to improve collaboration, (5) develop options that address major disturbance events, and (6) incorporate state decisions with global and local factors when adapting to climate change (Washington State Department of Ecology 2012).

Box 4.4

Interim recommendations of the Washington State Climate Change Response Strategy's topic advisory group on species, habitats, and ecosystems. (From Helbrecht et al. 2011.)

Facilitate the resistance, resilience, and response of natural systems

1. Provide for habitat connectivity across a range of environmental gradients.
2. For each habitat type, protect and restore areas most likely to be resistant to climate change.
3. Increase ecosystem resilience to large-scale disturbances, including pathogens, invasive species, wildfire, flooding, and drought.
4. Address stressors contributing to increased vulnerability to climate change.
5. Incorporate climate change projections in plans for protecting sensitive species.

Build scientific and institutional readiness to support effective adaptation

6. Fill critical information gaps and focus monitoring on climate change.
7. Build climate change into land use planning.
8. Develop applied tools to assist land managers.
9. Strengthen collaboration and partnerships.
10. Conduct outreach on the values provided by natural systems at risk from climate change.

Western Governors' Association (WGA)—
A nonpartisan organization of governors from 19 Western States, two Pacific territories, and one commonwealth, the WGA addresses the effects of climate on forest health, wildfire, water and watersheds, recreation, and forest products. The WGA supports integration of climate adaptation science in Western States (WGA 2009) and published a report on priorities for climate response in the West (WGA 2010), including sharing climate-smart practices for adaptation, developing science to be used in decisionmaking, and coordinating with federal entities and other climate adaptation initiatives. The WGA is focusing on developing training to help states incorporate new protocols and strategies relative to climate change (box 4.5), and improving coordination of state and federal climate adaptation initiatives. The WGA recommends that new state-level programs be designed that are relevant for on-the-ground climate change issues and also comply with federal regulations.

Minnesota State climate response—
The Minnesota Department of Natural Resources is building intellectual and funding capacity to implement policies that address climate change and renewable energy issues,

Box 4.5

Goals of the Western Governors' Association (WGA) climate-adaptation initiative on training. (From WGA 2010.)

The WGA seeks to provide training to its 19 member states and collaborators with goals to:

1. Provide state resource planners with the tools, methods, and technical assistance needed to incorporate climate change into ongoing planning processes.
2. Create a forum to enhance communication and dialogue with climate adaptation researchers to help set priorities for investment in science and research that informs decisionmakers.
3. Identify multistate or cross-boundary climate adaptation needs, as well as regional data sharing needs, and consider how they may be addressed through regional collaboration.
4. Determine how state agencies can collaborate with federal and local governments and other partners.
5. Develop a clearinghouse of best practices that state agencies and managers may refer to when developing their state's adaptation efforts.

including vulnerability assessments that identify risks and adaptation strategies for forest ecosystems. These efforts complement climate adaptation efforts occurring in the state through the multi-institutional Northwoods climate change response framework (see "Regional Examples"). The Minnesota Forest Resources Council, which includes public and private stakeholders from the forestry sector, is developing recommendations to the governor and federal, state, county, and local governments on policies and practices that result in the sustainable management of forest resources. Regional landscape committees establish landscape plans that identify local issues, desired future forest conditions, and strategies to attain these goals (MFRC 2011). The regional landscape committees plan to integrate with the Northwoods climate change response framework to ensure that climate change is integrated in forest management and planning.

North Carolina State climate response—
The North Carolina Department of Environment and Natural Resources (DENR) is developing a comprehensive adaptation strategy to identify and address potential effects on natural resources, with emphasis on climate-sensitive ecosystems and land use planning and development. The North Carolina Natural Heritage Program is evaluating likely effects of climate change on state natural resources, including 14 forest ecosystems that are likely to respond to climate change in similar ways. The DENR co-hosted a statewide climate change adaptation workshop in 2010 and is now coordinating with other agencies on an integrated climate response and developing a climate change response plan and down-scaled climate assessments.

State university and academic responses—
In the Pacific Northwest, the University of Washington Climate Impacts Group (CIG) has a strong focus on climate science in the public interest. Besides conducting research and assessing climate effects on water, forests, salmon, and coasts, the CIG applies scientific information in regional decisions. The CIG works closely with stakeholders and has been a key coordinator for forest climate adaptation projects (e.g., Halofsky et al. 2011, Littell et al. 2011). An adaptation guidebook developed in collaboration with King County, Washington describes an approach for developing local, regional, and state action plans (Snover et al. 2007). In Alaska, the Alaska Coastal Rainforest Center, based at the University of Alaska Southeast, in partnership with the University of Alaska Fairbanks and other stakeholders, provides educational opportunities, facilitates research, and promotes learning about temperate rain forests. The center facilitates dialogue on interactions among forest ecosystems, communities, and social and economic systems and has developed a framework for integrating human and ecosystem adaptation. In Hawaii, the Center for Island Climate Adaptation and Policy, based at the University of Hawaii at Mānoa, promotes interdisciplinary research and solutions to public and private sectors, with a focus on science, planning, indigenous knowledge, and policy relative to climate adaptation. Recent projects focus on education, coordinating with state natural resource departments on adapting to climate change (CICAP 2009), and policy barriers and opportunities for adaptation. Forest-related climate issues include effects of invasive species, forest growth and decline, migration and loss of forest species, and threats to sustainability of water resources.

Industrial Forestry

The response from forest industries in the United States to climate change has to date focused mostly on carbon sequestration, energy conservation, the role of biomass, and other climate-mitigation issues. Detailed assessments and efforts to develop adaptation strategies for the forest-industry sector have mostly been at the global to national scale (Sedjo 2010; Seppälä et al. 2009a, 2009b). Many forestry corporations promote stewardship forestry focused on adaptability of forest ecosystems to environmental challenges, but most ongoing adaptation projects are small scale and nascent. For example, Sierra Pacific Industries (SPI) in California is evaluating the potential for giant sequoia (*Sequoiadendron giganteum* [Lindl.] J. Buchholz) plantations to serve as a safeguard against changing climates. Giant sequoia currently grows in small groves scattered in the Sierra Nevada. Germplasm would be collected by SPI from the native groves and planted in riparian corridors on productive industry land, then managed as reserves that would benefit from the resilience of giant sequoia to climatic variability and its ability to regenerate after disturbance.

In Australia, the forest industry has been more assertive in addressing climate change. For example, the National Association of Forest Industries of Australia is working to improve the ability of forest industry to reduce the harmful effects of, and exploit opportunities from, changing climate. A short-term objective is to promote general awareness of the extent and range of likely climatic impacts and vulnerabilities specific to key forest regions, together with practical options for adaptation and mitigation given available scientific knowledge. The longer term objective is to provide tools and mechanisms to promote incorporation of adaptation options in forest-based industries. The Australian Commonwealth Scientific and Industrial Research Organization developed an initial assessment of climate risks and adaptation strategies for plantation forestry, with specific recommendations for planting, germplasm selection, and silvicultural actions (Pinkard et al. 2010).

Native American Tribes and Nations

Many Native American tribes and nations have been actively developing detailed forest adaptation plans in response to climate change. Overall goals commonly relate to promoting ecosystem sustainability and resilience, restoration of forest ecosystems, and maintenance of biodiversity, especially of elements having historical and legacy significance to tribes. Maintenance of cultural tradition within the framework of changing times is also inherent in many projects.

An exceptional example of a tribal response is the climate change initiative of the Swinomish Tribe in Washington (SITC 2010) (box 4.6). The Swinomish Reservation (3900 ha) is located in northwestern Washington and includes 3000 ha of upland forest. The initiative focuses on building understanding among the tribal community about climate change effects, including support from tribal elders and external partners. A recent scientific assessment summarizes vulnerabilities of forest resources to climate change, and outlines potential adaptation options (Rose 2010). A report completed in 2009 provides a baseline for adaptation planning and states that the tribe's forest resources are at risk from wildfire, which will be addressed in an action plan on adaptive response (SITC 2010).

Tribes have been active partners in collaborative forest adaptation plans. An example is the Confederated Tribes and Bands of the Yakama Nation, whose reservation occupies 490 000 ha in south-central Washington. Tribal lands comprise forest, grazing, and farm lands in watersheds of the Cascade Range. The Yakama Nation has extensive experience in managing dry forest ecosystems and implementing forest action plans, and belongs to the Tapash Sustainable Forest Collaborative, in partnership with the U.S. Forest Service, Washington State Departments of Fish and Wildlife and of Natural Resources, and TNC. The collaborative encourages coordination among landowners to respond to common challenges to natural resources (Tapash Collaborative 2010). Climate change was ranked as a significant threat to forest productivity, leading to a proposal to incorporate specific adaptation strategies and tactics across the Tapash landscape, most of which relate to fire, fuels, and restoration in dry forest.

Box 4.6

Adaptation framework of the Swinomish Tribe Washington) climate change initiative. (From SITC 2010).

Phase 1 (2007–2009)

- Tribal buy-in leads to issuance of the 2007 Climate Change Proclamation
- Secure funding
- Identification of partners, development of advisory committee, and identification of roles and responsibilities
- Development of the impact assessment
  - Data review and analysis
  - Risk zone mapping and inventory
  - Vulnerability assessment
  - Risk analysis
- Policy and strategy scoping (intergovernmental)
- Community outreach
  - Formed tribal outreach group
  - Held public meetings
  - Conducted personal interviews of tribal members and elders
  - Conducted storytelling workshop with tribal members

Phase 2 (2010)

Development of the action plan

- Adaptation goals
- Strategy evaluation and priorities
- Action recommendations
- Coordination of funding
- Other implementation issues

Phase 3 (future work)

- Action plan implementation
- Monitoring and adaptive management
- Update of the impact assessment

Nongovernmental Organizations

Nongovernmental organizations and professional organizations serve a wide range of special interests, and thus respond to climate adaptation challenges in diverse ways.

Pacific Forest Trust (PFT)—

A nonprofit organization dedicated to conserving and sustaining America's productive forest landscapes, PFT provides support, knowledge, and coordination on private forest lands in the United States. Through its Working Forests, Winning Climate program, PFT has created policy and market frameworks to expand conservation stewardship of U.S. forests to help sustain ecosystem services (PFT 2011). The PFT also supports climate adaptation by working with private forest owners to promote stewardship forestry, whereby forests are managed to provide goods and services that society has come to expect. The PFT currently works with stakeholders on working forest lands to call on policymakers to safeguard U.S. forests for their value in adaptation and mitigation.

The Nature Conservancy (TNC)—

A science-based conservation organization, TNC has a mission to preserve plants, animals, and natural communities by protecting the lands and waters they need to survive. The TNC climate change adaptation program seeks to enhance the resilience of people and nature to climate change effects by protecting and maintaining ecosystems that support biodiversity and deliver ecosystem services. The program promotes ecosystem-based approaches for adaptation through partnerships, policy strategies for climate adaptation, tools to assist resource managers, and research. The Canyonlands Research Center (Monticello, Utah), a TNC initiative that conducts research and develops conservation applications for resource issues in the Colorado Plateau region, focuses on forest-climate concerns such as woodland ecosystem restoration, invasive species, and effects of drought on pinyon pine and juniper woodlands.

Trust for Public Land (TPL)—

A conservation organization that helps agencies and communities conserve land for public use and benefit, TPL uses vulnerability assessments, resilience and connectivity data, and other tools to realign its conservation planning at different spatial scales. The TPL is also designing and implementing restoration to enhance the climate resilience of protected tracts. As a member of the Northern Institute of Applied

Climate Science, TPL provides guidance to federal and nonfederal partners on strategic planning and on-the-ground management.

The Wilderness Society (TWS)—
The Wilderness Society leads efforts to fund natural resource adaptation and manage lands so they are more resilient under stresses of climate change, and is a leader in the Natural Resources Adaptation Coalition, which is focused on maintaining and restoring wildlands that include forest wilderness. Specific TWS goals relative to adaptation in forests include (1) restoring native landscapes to increase ecosystem resiliency, (2) protecting rural communities and providing flexibility in wildland fire management, (3) removing invasive species from ecosystems, and (4) repairing damaged watersheds.

Ski Industry

Although not a direct member of the forest sector, the ski industry relies on mountainous terrain, usually forested land leased from federal landowners, and is concerned about reduced snow, rising temperatures, extreme weather events, and other consequences of climate change that may affect the profitability of ski areas. Adaptation options used by the ski industry (Scott and McBoyle 2007) include (1) snowmaking to increase the duration of the ski season (Scott et al. 2006), (2) optimizing snow retention (slope development and operational practices such as slope contouring, vegetation management, and glacier protection), and (3) cloud seeding. Forest vigor and stand conditions within and adjacent to ski area boundaries are important to ski areas, because forests burned by wildfire or killed by insect outbreaks affect snow retention, wind patterns, and aesthetic value.

Examples of Regional and National Responses

Although general guidance and strategic plans about climate adaptation exist for many land management agencies, strategies for specific places and resource issues are in the early stages. Below we summarize examples for which the intent was to explore how forest adaptation strategies could be developed for specific locations.

Western United States

Olympic National Forest/Olympic National Park (ONFP), Washington—
This case study in the Northwest was undertaken to represent a large landscape within a geographic mosaic of lands managed by federal and state agencies, tribal groups, and private landowners (Littell et al. 2011). The ONFP supports a diverse set of ecosystem services, including recreation, timber, water supply to municipal watersheds, pristine air quality, and abundant fish and wildlife. Management of Olympic National Forest focuses on "restoration forestry," which emphasizes facilitation of late-successional characteristics, biodiversity, and watershed values in second-growth forest. Collaboration with adjoining Olympic National Park, which has a forest protection and preservation mission, is strong.

Development of the ONFP adaptation approach employed a science-management partnership, including scientific expertise from the CIG, to implement education, analysis, and recommendations for action. Analysis focused on hydrology and roads, vegetation, wildlife, and fish, which were the resources most valued by agency resource managers and most likely to be influenced by climate change. A vulnerability assessment workshop for each resource area was paired with a workshop to develop adaptation options based on the assessment, resulting in adaptation options for management issues within each disciplinary topic. Emphasis in adaptation was on conserving biodiversity while working to restore late-successional forest structure through active management. The overall process used in the case study has been adopted by local resource managers to incorporate climate change issues in forest plans and projects (Halofsky et al. 2011) and is currently being used to catalyze climate-change education, vulnerability assessment, and adaptation planning across 2.5 million ha in Washington state (North Cascadia Adaptation Partnership 2011).

Inyo National Forest and Devils Postpile National Monument, California—
Inyo National Forest (INF) in eastern California contains Mediterranean and dry forest ecosystems, grading from alpine through forest to shrub-steppe vegetation. Much of the

national forest is wilderness with a high degree of biodiversity. Water on the national forest is scarce, fire and insects are important issues, and recreation is the dominant use of public lands. Devils Postpile National Monument (DEPO) is a small national park unit surrounded by INF lands, and collaboration with INF is strong. Ongoing near- and mid-term projects of highest concern focus on vulnerability of INF resources to climate effects that might affect DEPO, and climate adaptation is a high priority in the DEPO general management plan. A science-management partnership facilitated sharing of knowledge about climate change and effects through targeted workshops (Peterson et al. 2011), and assessment reports developed by scientists (Morelli et al. 2011a) assisted managers to consider climate effects relevant to specific resource responsibilities. A scientific technical committee (Peterson et al. 2011) helped to meet science needs for managers of these units. For INF, the Climate Project Screening Tool (Morelli et al. 2011b) was developed, providing a screening process to rapidly assess if climate change would affect resources in the queue for current-year management implementation. Questions about climate-mediated quaking aspen (*Populus tremuloides* Michx.) decline spurred a review of aspen responses to climate and an aspen screening tool for the INF (Morelli and Carr 2011).

For DEPO, where ecosystem protection is prioritized, managing the monument as a climate refugium (Joyce et al. 2008, Peterson et al. 2011) is being evaluated. Because DEPO is at the bottom of a large canyon with cold-air drain age, it contains high biodiversity, and the potential for cold-air drainage to increase in the future may ameliorate the effects of a warmer climate (Daly et al. 2009). In anticipation of this, a network of temperature sensors in multiple-elevation transects and a climate monitoring station were recently installed to measure ongoing changes in temperature.

Shoshone National Forest, Wyoming—
Resource managers in Shoshone National Forest worked with Forest Service scientists to write a synthesis on climate change effects and a vulnerability assessment of key water and vegetation resources. The synthesis (Rice et al. 2012) describes what is currently understood about local climate and the surrounding Greater Yellowstone Ecosystem, including paleoclimate, and how future climate change may affect plants, animals, and ecosystems. The assessment highlights components of local ecosystems considered most vulnerable to projected changes in climate and will be integrated in resource-related decisionmaking processes of forest management through collaborative workshops to train managers.

The Strategic Framework for Science in Support of Management in the Southern Sierra Nevada, California (SFS)—
The SFS addresses collaborative climate adaptation for the southern Sierra Nevada bioregion of California (Nydick and Sydoriak 2011), including the southern and western slopes of the Sierra Nevada, three national parks, a national monument, three national forests, tribal lands, state and local public lands, forest industry, and other private lands. This landscape spans ecosystems from alpine through diverse conifer and hardwood forests to woodland and chaparral. The effort is coordinated by a coalition of federal resource managers and academic and agency scientists, and was launched with a public symposium to review the state of science on climate issues and adaptation options. The framework document (Exline et al. 2009) guides adaptation by asking (1) Which ecosystem changes are happening, why are they happening, and what does it mean? (2) What is a range of plausible futures? (3) What can we do about it?, and (4) How can relevant information be made available? Interactions among climate change and habitat fragmentation, encroaching urbanization, shifting fire regimes, invasive species, and increasing air pollution are also being considered. To date the SFS collaborative has generated a list of ideas to provide knowledge and tools regarding agents of change and potential responses (box 4.7). An information clearinghouse will be established, including data for vulnerability assessments, decision-support tools, and reports.

Southern United States

Uwharrie National Forest (UNF) represents a typical national forest context in the southeastern United States, containing 61 parcels mixed with private land and near

Box 4.7

Initiatives begun and proposed by the strategic framework for science in support of management in the Southern Sierra Nevada cooperative and their alignment with the goals of the strategic framework. (From Nydick and Sydoriak 2011.)

Goal 1: Detection and attribution

- Coordinate regional monitoring strategies—tree population dynamics and fisher (*Martes pennanti* Erxleben) populations

Goal 2: Forecasting future conditions

- Alternative fire management futures
- Comparison and integration of climate adaptation projects

Goal 3: Tools and actions

- Both projects under goal 2 also address goal 3
- Kaweah Watershed coordinated restoration initiative
- Enabling forest restoration goals via ecologically managed biomass generation, including a cost-benefit analysis

Goal 4: Communication

- Information clearinghouse for shared learning
- Education and outreach initiative

Integration across goals

- Reevaluation of invasive plant programs and practices under alternative climate futures
- Investigation of the vulnerability of blue oak woodlands to climate change and development of adaptive management guidelines

metropolitan areas (Joyce et al. 2008). Providing a wide range of ecosystem services, the region is undergoing a rapid increase in recreational demand. The UNF identified drought-related forest mortality, wildfire, insect outbreaks, soil erosion, stream sedimentation, and water shortages as key issues relative to climate effects. Revision of the forest land management plan explicitly considers climate change effects. Opportunities for adaptation in UNF focus on reestablishing longleaf pine (*Pinus palustris* Mill.) through selective forest management (Joyce et al. 2008). Replanting of drought-tolerant species could provide increased resistance to potential future drought and intense wildfire. Selective harvest and prescribed burns also could target restoration of longleaf pine savannas, mitigating water stress, fuel loads, and wildfire risk anticipated under warming conditions. Concerns about soil erosion and stream sedimentation focus on increasing the size of stream buffer zones where trees are not harvested. Collaboration with surrounding landowners to remove fuels in wildland-urban interfaces is a high priority.

Northern United States

The U.S. Forest Service in the northeast and upper Midwest is pursuing a comprehensive program of adaptation to climate change (fig. 4.5), including education and training, partnership building, vulnerability assessment and synthesis, planning and decision support, and implementation of demonstration projects. The Forest Service Northern Research Station, Northeastern Area State and Private Forestry, and Northern Institute of Applied Climate Science work collectively to respond to climate change needs. The Climate Change Response Framework (CCRF) developed by these entities augments the institutional capacity of national forests to adapt to climate change by providing a model for collaborative management and climate change response that can accommodate multiple locations, landscapes, and organizations (fig. 4.6). As of 2011, three projects were underway in the Northwoods, Central Hardwoods, and Central Appalachians (fig. 4.7).

The projects focus on building science-management partnerships, developing vulnerability assessments and synthesis of existing information, and establishing a

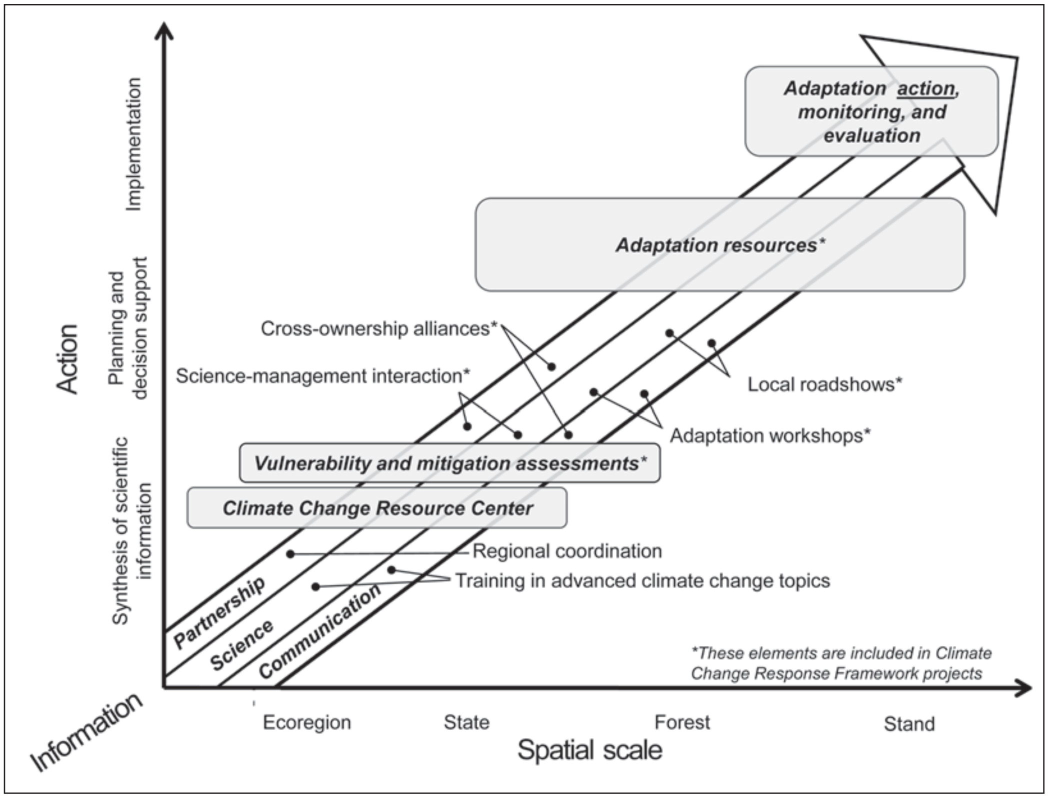

Figure 4.5—The U.S. Forest Service Eastern Region approach to climate change response works from ecoregional scales down to the stand scale by moving information to action through partnerships, science, and communication.

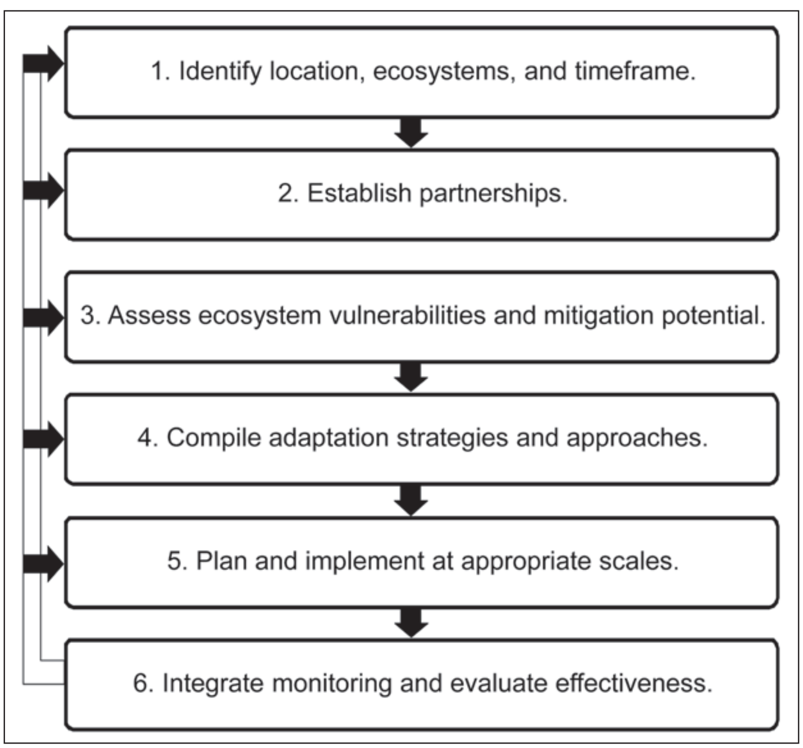

Figure 4.6—The Climate Change Response Framework uses an adaptive management approach to help land managers understand the potential effects of climate change on forest ecosystems and integrate climate change considerations into management. (From Swanston et al. 2012.)

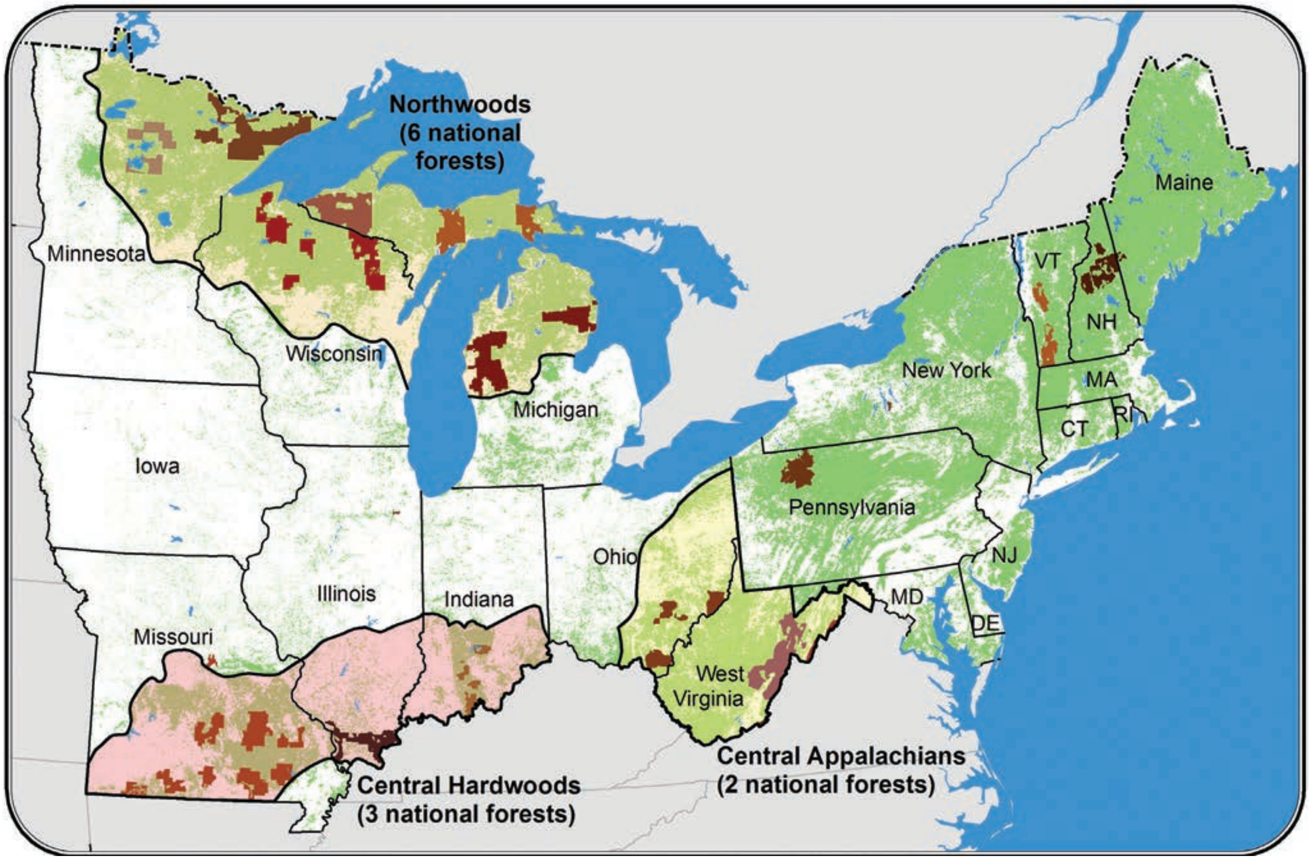

Figure 4.7—The U.S. Forest Service Eastern Region with Climate Change Response Framework (CCRF) projects, identified by shading. National forests are core partners in the CCRF projects, but the projects take an all-lands approach with numerous federal and nonfederal partners. The goal of the CCRF is to complement science-based management decisions made by multiple organizations, each with their own diverse goals, so that forest ecosystems managed by these organizations can become better adapted to a changing climate.

standardized process for considering management plans and activities in the context of the assessment. First, an ecosystem vulnerability assessment and synthesis evaluates ecosystem vulnerabilities and management implications under a range of plausible future climates. Second, a shared landscape initiative promotes dialogue among stakeholders and managers about climate change, ecosystem response, and management. Third, a science team encourages rapid dissemination of information. Fourth, an adaptation resources document includes relevant strategies and a process for managers to devise appropriate tactics. Fifth, demonstration projects incorporate project information and tools in adaptation activities. The CCRF emphasizes an all-lands approach, including national forests, other agencies, and other landowners and stakeholders.

The Northwoods CCRF Project covers 26 million ha of forest in Michigan, Minnesota, and Wisconsin, including six national forests, the Forest Service Northern Research Station, state resource agencies, universities, and other stakeholders. Products to date focus on northern Wisconsin, including a vulnerability assessment (Swanston et al. 2011), a forest adaptation resources document (Swanston and Janowiak 2012), and initiation of demonstration projects in Chequamegon-Nicolet National Forest, where each district was asked to integrate climate change considerations into forest activities. The Medford-Park Falls District identified two aspen stands where silvicultural prescriptions had already been written, but timber had not been marked for harvest, then used CCRF information to consider climate change effects and devised specific adaptation tactics.

A demonstration project started in 2011 convenes a variety of landowners, forest managers, and other stakeholders to discuss climate change effects on specific forest ecosystems, identify adaptation actions, and coordinate implementation of adaptation activities across multiple ownerships. Participants are using CCRF information and tools to devise adaptation tactics appropriate to their management goals. In addition, the Central Hardwoods CCRF, which covers 17 million ha of hardwood forest in Missouri, Illinois, and Indiana, has formed a regional coordinating team with partners from three national forests, the Northern Research Station, and other stakeholders. The Central Appalachians CCRF, which covers 11 million ha of central Appalachian forest in West Virginia and Ohio, includes partners from two national forests and state forestry agencies.

National Example

Watershed vulnerability assessment—
In 2010, a draft watershed vulnerability assessment process was tested in 11 national forests (Furniss et al. 2010), with the goal of quantifying current and projected future condition of watersheds as affected by climate change. A principal objective was to develop a general process that could be tailored to local data availability and resource investment (box 4.8). National forests were asked to include infrastructure, aquatic species, and water uses in the assessments, with analysis areas including at least one "river basin" watershed (hydrologic unit code [HUC] 4). Design of useful strategies for reducing the effects of climate change on ecosystem services requires the ability to (1) identify watersheds of highest priority for protecting amenity values, (2) identify watersheds in which climate-related risk to those values is greatest and least, (3) detect evidence of the magnitudes of change as early as possible, and (4) select actions appropriate for reducing effects in particular watersheds (Peterson et al. 2011).

Hydrologic specialists from participating forests developed an approach for quantifying watershed vulnerability within a relatively short period, and four national forests completed the process within 8 months. Acquiring suitable climate exposure data (the degree, duration, or extent of deviation in climate that a system experiences), which had not been previously used by the participants, was challenging. Threshold values for species and water use differed across the forests. For example, brook trout (*Salvelinus fontinalis* Mitchill) was viewed as a stressor in one forest and a valued resource in another. These differences suggest that, whereas information on processes and resource conditions can be shared among forests, local (forest- and watershed-scale) assessments have the greatest value.

Box 4.8

Steps defining the watershed vulnerability assessment process and the types of questions to be addressed. (From Furniss et al. 2010.)

Step 1—Set up the analysis and establish the scope and water resource values that will drive the assessment
Step 2—Assess exposure
Step 3—Assess sensitivity
Step 4—Evaluate and categorize vulnerability
Step 5—Recommend responses
Step 6—Critique the vulnerability assessment

Typical questions to be addressed in a watershed vulnerability assessment:

- Which places are vulnerable?
- Which places are resilient?
- Where are the potential refugia?
- Where will conflicts arise first, and worst?
- Which factors can exacerbate or ameliorate local vulnerability to climate change?
- What are the priorities for adaptive efforts?
- How can context-sensitive adaptations be designed?
- What needs tracking and monitoring?

Challenges and Opportunities

Assessing Adaptation Response

In recent years, federal agencies responsible for administering forest ecosystems have produced national climate change response strategies that define adaptation goals and describe a framework for action in field units. These strategies, intended to inform and guide consistent agency-wide responses, emphasize (1) staff training and education in climate sciences, (2) science-management partnerships, (3) assessment of vulnerabilities and risks, (4) maintenance of ecosystem sustainability and biodiversity conservation, (5) integration of climate challenges with other forest disturbance agents and stressors, (6) integration of adaptation with greenhouse-gas mitigation, (7) all-lands and collaborative approaches (working with whole ecosystems and across jurisdictional borders), (8) recognition of short- and long-term planning perspectives, (9) setting priorities, and (10) monitoring and adaptive management.

Adaptation strategies have been advanced unevenly by federal agencies at their regional levels and across local units (e.g., national forests and national parks). Implementation of these strategies is assisted by the presence of local motivated leaders, the support and flexibility provided by regional directors and supervisors, and the understanding and concurrence of constituencies. Some units have worked with local scientists to analyze regional climate projections, develop ecosystem vulnerability assessments, and develop intellectual capacity through staff and constituency education. Collaborative partnerships that extend across ownerships and jurisdictions have been developed as a foundation for some adaptation projects and an aid to communication across ownerships with different resource objectives. A few progressive units have implemented climate adaptation projects on the ground. Only a few site-specific adaptation projects, as described in this chapter, have been implemented across a range of resource issues and tiered to local and regional strategies. Responses of state governments have also been variable, with major forest-sector states in the Western and Northern United States taking leading adaptation strategies. Similar to the federal situation, concepts and frameworks for adaptation are sometimes available, but site-specific project implementation is rare. Education, vulnerability assessments, collaborative partnerships, biodiversity protection, and adaptive management have been key features in adaptation responses by tribes and nongovernmental organizations.

Among the groups that actively address forest adaptation, they commonly address climate change as a metadisturbance agent with other ecosystem stressors. Frequently, climate adaptation is not identified as the primary reason for planning; rather, climate response strategies are subordinate to ecosystem sustainability, forest and watershed restoration, and biodiversity conservation. Adaptation goals are thus commonly met through projects that address high-priority management goals, such as management of fuels and fire, invasive species, insects and pathogens, and watershed condition.

Implementation of site- and issue-specific adaptation plans has been uneven and often superficial across the forest sector, and there appears to be a tendency to rely on quantitative climate- and ecosystem-response models without corroboration to local ecosystems (Millar et al. 2007). A subtle danger in using complex, downscaled, spatially rendered models is that users (e.g., forest managers and planners) may accept model output as the single and likely future, rather than one among many possible outcomes. Models are better used to understand processes and cautiously project future climates and ecosystem responses on specific landscapes and definitive timescales, allowing adaptation treatments to be developed for those outcomes. Better understanding by practitioners of how models are built, and what they can and cannot do, would improve effective application of model output to adaptation.

Existing Constraints

Various organizations have made progress on adaptation in forest lands, but implementation has been slow, integration across the various sectors (e.g., multiple use, protected area, forest industry) unbalanced, awareness generally low, and site-specific projects few. Numerous barriers appear to impede development and implementation of plans that would promote widespread readiness for American forests to adapt to climate change.

Education, awareness, and empowerment—
Many natural resource science curricula now include courses on climate science, ecosystem responses to climate change, and implications for resource management. However, education on historical climatology is rare. Without a clear understanding of mechanisms of climatic dynamics,

use of concepts like "100-year floods" or "restoration to historic conditions," which rest on assumptions of stationary long-term conditions, may lead to inappropriate interpretations and management actions (Milly et al. 2008).

Lack of experience and understanding of climate science by resource managers can lead to low confidence in taking management action in response to climate threats; similar limitations through the chain of supervision and decisionmaking appear to constrain appropriate efforts. Inconsistent support for climate readiness and action extending from executive levels can impede regional planning, which in turn sets up barriers to local implementation. Even if resource managers are trained and competent in climate science, they may lack support from their superiors to implement adaptation strategies and projects.

Lack of public awareness of how climate change affects natural resources influences the level and nature of adaptation by public institutions. Despite widespread public engagement in land management over the past 30 years, pressure to act on climate change has not been as prominent as for other resource issues. On the one hand, little support exists for implementing projects directed to adaptation; on the other hand, there is often strong opposition to projects that address indirect effects of climate, such as forest thinning, postfire logging, herbicide treatments to encourage regeneration, and road improvements for watershed protection. Nonetheless, public pressure can result in climate issues being addressed in resource evaluations and plans.

In some cases, scientific expertise may be unavailable even when science-based strategies are recognized as essential. Scientific institutions have suffered budget reductions, and only some scientists have the interest and capacity to work in management contexts. Even experienced scientists may need to learn the culture, issues, expectations, and scientific focus of management organizations. The demand for scientific participation in on-the-ground adaptation will likely continue to exceed supply as more adaptation programs evolve.

Policy, planning, and regulatory constraints—Both public and private lands are subject to policy, planning, and regulatory direction. Federal agencies are constrained by hierarchies of laws and internal policy and direction, whereas private forest landowners have greater flexibility to determine actions and timing on their land, but remain bound by local, state, and federal laws. In federal agencies, site-specific projects are tiered to levels of planning at higher levels in the organization.

In national forests, site-specific projects tier to each forest's land management plan. These plans guide resource management activities on a forest to ensure that sustainable management considers the broader landscape and values for various resources. The U.S. Forest Service has developed procedures through a new national Planning Rule (Federal Register vol. 76, no. 30; 36 CFR Part 219) to amend, revise, and develop land management plans for 176 units in the National Forest System (NFS). The Planning Rule gives the Forest Service the ability to complete plan revisions more quickly and reduce costs, while using current science, collaboration, and an all-lands approach to produce better outcomes for federal lands and local communities. The Planning Rule addresses management in the context of climate change and changing environmental conditions and stressors, requiring plans to include components that address maintenance and restoration of ecosystem and watershed health and resilience, protection of key resources (e.g., water, air, and soil), and protection and restoration of water quality and riparian areas.

All forest management agencies face the challenge of working at spatial and temporal scales compatible with climate change. This demands integration of goals and projects from small to large scales, a reality that often clashes with the mix of ownerships and regulations, making collaboration across multiple organizations essential. As noted above, progress has been made by recent collaborative efforts that recognized that different regulatory and policy environments (e.g., federal versus private) were not necessarily a barrier. Even at small scales, such as a single national forest or national park, traditional planning approaches dissect lands into discrete units. Thus, harvest units, wilderness areas, developed recreation zones, and endangered species reserves are delimited, subject to standards and guidelines developed for the management zone. This area-constrained approach is static and inflexible, incompatible with the dynamism of climate and climate-related changes and responses.

Environmental laws developed over the past four decades were conceived primarily with an assumption of climatic stationarity, and thus many lack capacity (or legal authorization) to accommodate dynamics of climate-related changes. For example, endangered species laws are often interpreted as indicating native species ranges as they were in presettlement times (e.g., before 1900). Climate changes since then are catalyzing range shifts that sometimes define new native ranges. Enforced maintenance of species in the prior range could prove to be counter adaptive. The National Forest Management Act (NFMA) of 1976 implies maintenance of the status quo based on historical conditions, usually defined, like above, as presettlement (19th-century) ranges. For example, the NFMA "diversity clause" requires that a similar mix of species be reforested on national forest lands after harvest as was present before. Because regeneration is the most effective period for changing forest trajectories, planting nontraditional mixes of similar species or introducing new species might be a defensible adaptation response (Joyce et al. 2008).

Monitoring and adaptive management—
Future climates and environmental conditions will likely be nonanalog relative to the past. Compounding this situation is the imprint of human land use that has fragmented, restructured, and altered forest ecosystems over the past century. Forest adaptation practices must meet this challenge of novelty and surprise with equally innovative approaches informed by monitoring and adaptive management ("learn as you go"). However, adaptive management in public agencies and other institutions has had minimal success and been implemented slowly, owing in part to lack of funding commitment and lack of analyst capacity. Reorientation of programs, expectations, and interactions with public constituents will be required for monitoring and adaptive management to become a successful partner to adaptation.

Budget and fiscal barriers—
Significant additional funding will be needed for a full national response to forest climate adaptation. Education and training, development of science-management partnerships, vulnerability assessments, and development of adaptation strategies are critical components of the adaptation process and can be integrated with other aspects of management, but effective consideration of climate requires additional time and effort. Collaboration across management units and organizations, leveraging of institutional capacities, and other innovative solutions will be needed to address this budget challenge.

Vision for the Future

Vision—
Facilitating long-term sustainability of ecosystem function is the foundation of climate change adaptation. Just as there is no single approach to sustainable forestry, effective climate change adaptation will differ by ecosystem, management goals, human community, and regional climate. If adaptation is addressed in a piecemeal fashion (ecological, geographic, and social), large areas and numerous communities within the forest sector may suffer the consequences of poor preparation, slow response, and inefficiencies. The preceding sections describe principles, policies, approaches, and examples of addressing the challenges of climate adaptation. Here we offer a vision of successful adaptation across U.S. forests within the next 20 years: "A proactive forest sector makes the necessary investments to work across institutional and ownership boundaries to sustain ecosystem services by developing, sharing, and implementing effective adaptation approaches." This broad vision incorporates several critical concepts, each embodied by its own vision.

Investment—
Sufficient investment is allocated to successfully achieve visions of development, sharing, and implementation. This includes (1) investment in basic and applied research; (2) support of adequate staffing to accommodate increased planning, monitoring complexity, and interaction with partners; and (3) concerted effort to communicate to the general public the dynamic nature of climate and forests. Monitoring and data sharing are critical to adaptation and adaptive management, and are jointly supported across multiple agencies and land ownerships. Climate change resource centers, instructional courses, and professional meetings are supported to encourage rapid communication and amplify learning in

adaptation management and science. Planning, decisionmaking, and contracting processes that support implementation of ground-level activities are adequately funded so that lands in need of adaptive treatments can be reached before ecosystem function is jeopardized.

Development—

Research—Research into all aspects of forest ecosystem sciences continues to provide valuable insights into forest responses to climate change. Research into effectiveness of climate-adaptation strategies guides adaptive policy responses.

Assessment—Credible information is regularly produced and updated at scales relevant to management decisions that (1) assess vulnerability of ecosystem components, (2) incorporate a range of climate projections, (3) use multiple modeling approaches to project ecosystem response, and (4) incorporate skills and experience of scientists and land managers.

Learning—Active learning occurs through traditional research and other pathways: (1) formal adaptive management trials continually produce information to evaluate adaptation techniques; (2) working forests, especially national forests, serve as "living laboratories" with adequate support to pursue adaptive management including adaptation techniques; and (3) management on federal lands is sufficiently documented and monitored to identify broad landscape trends and efficacy of adaptation efforts.

Sharing—

Transparency—Management goals are clearly stated in forest planning documents, and explicit options for sustaining ecosystem function under a range of plausible future climates provide a preview of potential choices in meeting those goals.

Communication—Vulnerable ecosystems and ecosystem components are identified in vulnerability assessments and noted in management plans. Existing or conditional decisions to pursue different adaptation options are explicit in management plans, and associated risks to ecosystem services are addressed. This information is proactively shared and discussed with the general public.

Ownerships—Increased investment in local programs that facilitate forest stewardship assists small landowners in managing sustainably. Outreach to consulting foresters and professional associations creates an informed base of private landowners. Information about management activities is shared across boundaries of all public and private lands to enable the forest sector to take advantage of biological and management diversity across large landscapes. Collaboration to manage across administrative and ownership boundaries is commonplace.

Partnerships—Adaptation across landscapes is addressed by engaging in productive partnerships, spanning boundaries of agency, ownership, and discipline. Science-management partnerships provide critical information and perspective to members of each discipline and form strong communities of practice.

Implementation—

Planning—Climate change is incorporated in all planning activities, and on-the-ground prescriptions are adjusted to include adaptation where necessary. Planning is developed for explicit locations with attention to appropriate scale. Public land managers and forestry consultants are well versed in finding and interpreting climate and vegetation projections, and in adjusting plans to accommodate a range of plausible future climates. Open avenues of discussion provide the scientific community with feedback on the relevance and clarity of tools, information, and research directions.

Monitoring—Monitoring is integrated across multiple scales and coordinated across institutions. Monitored indicators are sensitive to changes in key ecosystem components. Monitoring data and summaries are freely available. Monitoring data, clear thresholds, and transparent processes for interpretation of data are incorporated in processes for decision-making and changes in management practices (the adaptive management cycle).

***Flexibility*—**Management plans acknowledge the increased potential of extreme events, novel climates, and unanticipated ecosystem responses. Decisionmaking structures have adequate flexibility to accommodate multiple potential futures, and to adopt alternative goals if prior goals are no longer feasible in vulnerable systems. Likewise, if adaptation approaches are ineffective, they are redirected. Lessons are shared with the management and scientific communities to encourage transparency and flexibility.

***Implementation*—**New information and lessons are rapidly incorporated in management activities. Active management is used to promote resistance in short terms and long-term resilience where appropriate, and the backlog from previous decades of lands in need of treatment is diminishing. Some forests are managed to "soften the landing" as they transition to new species assemblages and forest structures, such that ecosystem processes and ecosystem services are maintained. Forests affected by extreme events are rapidly restored, with due consideration of future climatic effects on species composition and the long-term function of the recovering forest.

Path to the Vision

The U.S. forest sector can make significant progress toward a vision of sustained forest ecosystem function in the face of climate change by doing the following:

- **Embrace education**. Widespread understanding of the central role of climatic dynamics in ecosystem processes and services is fundamental; therefore, training and educational programs need to be deployed for resource professionals in agencies and for other organizations and the general public. Partnerships with universities can enhance scientific support to science-management partnerships for both adaptation and education.
- **Ensure accountability and infuse climate into all organizational efforts.** The responsibility for ensuring that resource management plans, projects, and decisions are "climate smart" rests on every professional within agencies and other organizations. Knowledge about climate is not an independent staff area but a context through which resource issues can be evaluated. Implementation of this knowledge is the responsibility of personnel across all resource disciplines.
- **Live the all-lands approach and make collaboration the norm.** Effective collaboration across administrative, political, and ownership boundaries, and across diverse cultural and social perspectives is difficult but necessary, often requiring focused effort over an extended period. "Early adapter" collaborations show how regulations, traditions, cultures, and organizational legacies can be navigated successfully. These collaborations need cross-agency and cross-sector support to catalyze effective partnerships.
- **Streamline planning and put projects on the ground.** Nimbleness and flexibility to implement changes are essential ingredients for successful adaptive responses to climatic challenges. Much of the current planning and project implementation process in public agencies contains bureaucratic requirements that detract from actual resource work. Planning processes that prioritize project implementation, including uncertainty, risk, and provisions for experimentation will have the most success. For resource managers, emphasizing education and resource projects rather than administrative tasks will expedite timely adaptation accomplishments.

The challenge of climate change adaptation will require creativity by future generations of forest resource managers. No one agency or organization can fully meet the challenge, but this task is within reach of the forest sector if willing partners work collaboratively toward sustainable management grounded in knowledge of climate science and dynamic ecosystems.

Carbon Management

Sequestering more carbon (C) in forests and offsetting C emissions with use of wood for energy and products are two of a range of objectives in managing forests. Increasing C storage and C offsets across a range of C pools and emission sources contributes to stabilizing atmospheric concentrations of carbon dioxide (CO_2). One time period of interest for the effect of an increase in C emissions on radiative forcing is 100 years (Pachauri and Reisinger 2007). Another, although less well defined, is the time period required to achieve "… stabilization of greenhouse gas (GHG) concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system" (United Nations 1992). In addition, management activities that would be most desirable would contribute co-benefits, avoid adverse impacts, and sustainably provide needed goods, services, and values.

The historical and current conditions of U.S. forests, forest management practices, and use of forest products have resulted in net C additions to forests and to harvested wood products stocks (tables 4.6, 4.7). However, recent forest sector projection scenarios for the 2010 Resource Planning Act assessment (USDA FS 2012a) suggest that annual C additions could decline more rapidly and U.S. forests could become a net C emitter of 10s to 100s of $Tg{\cdot}C{\cdot}yr^{-1}$ within a few decades. This possibility highlights the urgency in identifying the most effective C management strategies given the complexity of factors that drive broader trends on the forest C cycle, and the broad variety of goods, services, and values forests provide.

Understanding biophysical and social influences on the forest C cycle is critical to developing management strategies that can be used to effectively manage forest C stocks and offset C emissions with minimal risks of failure and adverse environmental effects (tradeoffs). With sufficient knowledge of social processes (e.g., landowner or wood-user response to incentives and markets), policies and incentives may be chosen to support strategies with maximum effect. For example, if forest C stocks are expected to decline owing to decreasing land area caused by land use change (e.g., exurban development), policies or incentives to avoid deforestation in those areas may be especially effective. Also, if forest C stocks are expected to decline owing to the effects of changing climate (e.g., prolonged periods of drought), thinning might be especially effective in those areas by protecting C stocks or ensuring some level of continued productivity. Thinning might also reduce impacts on water availability (mainly in arid and semiarid environments) and help increase forest resilience to various stressors (Jackson et al. 2005, Millar et al. 2007, Reinhardt et al. 2008). Protecting old-growth forests and other forests containing high C stocks may be more effective than strategies that would seek to attain C offsets associated with wood use, especially if those forests would recover C very slowly or would not recover in an altered climate.

Sometimes, even if harvest treatment strategies are effective, there may be tradeoffs (losses) judged to be greater than the offset benefits, such as loss of biodiversity in sensitive areas. Alternately, if climate change is expected to increase potential productivity on a given area over a long period of time, increasing forest C stocks through intensive management and forest products, including biomass energy, may be especially effective. Equally important, knowing which strategies to avoid for specific areas will prevent excessive risks and tradeoffs that could make strategies unsustainable. No widely accepted evaluation framework exists to aid decisionmaking on alternative C management strategies designed to maximize C storage while minimizing risks and tradeoffs.

This section discusses (1) current details and trends on where forest C is stored in the United States, (2) issues concerning how to measure progress and effectiveness in averting emissions, (3) current knowledge on the effectiveness of various management strategies in reducing atmospheric GHGs, and (4) effects of incentives, regulations, and institutional arrangements in implementing C management strategies.

Status and Trends in Forest-Related Carbon

Net annual C additions to forests and harvested wood products account for the vast majority of total annual GHG sequestration among all land uses in the United States (fig. 4.8). Within forests, the two largest C components are aboveground biomass and soil organic C (fig. 4.9). Because aboveground biomass accumulates, then shifts to dead wood, litter, or wood products in a matter of decades, there is an opportunity for forest management and land use activities to affect aboveground biomass accumulation and its disposition over decadal time (i.e., management modifications can result in higher C accumulation and emission offsets).

The change in forest C stocks over time is determined by change in forest area and the change in forest C per unit

Table 4.6—Net annual changes in carbon stocks in forest and harvested wood pools, 1990–2009

Carbon pool	1990	2000	2005	2009
	– – Teragrams of carbon per year – –			
Forest:				
Live, aboveground	-98.2	-78.3	-122.1	-122.1
Live, belowground	-19.3	-15.7	-24.1	-24.1
Dead wood	-8.6	-3.5	-8.4	-9.1
Litter	-8.8	7.5	-11.4	-11.4
Soil organic carbon	-14.9	17.6	-53.8	-53.8
Total forest	-149.8	-72.4	-219.9	-220.6
Harvested wood products:				
Products in use	-17.7	-12.8	-12.4	1.9
Products in solid waste disposal sites	-18.3	-18.0	-16.3	-16.7
Total harvested wood products:	-35.9	-30.8	-28.7	-14.8
Total net flux	-185.7	-103.2	-248.6	-235.4

Source: USEPA 2011.

Table 4.7—Carbon stocks in forest and harvested wood pools, 1990–2010

Carbon pool	1990	2000	2005	2010
	– – – – Teragrams of carbon – – – –			
Forest:				
Live, aboveground	15,072	16,024	16,536	17,147
Live, belowground	2,995	3,183	3,285	3,405
Dead wood	2,960	3,031	3,060	3,105
Litter	4,791	4,845	4,862	4,919
Soil organic carbon	16,965	17,025	17,143	17,412
Total forest	42,783	44,108	44,886	45,988
Harvested wood products:				
Products in use	1,231	1,382	1,436	1,474
Products in solid waste disposal sites	628	805	890	974
Total harvested wood products:	1,859	2,187	2,325	2,449
Total carbon stock	44,643	46,296	47,211	48,437

Source: USEPA 2011.

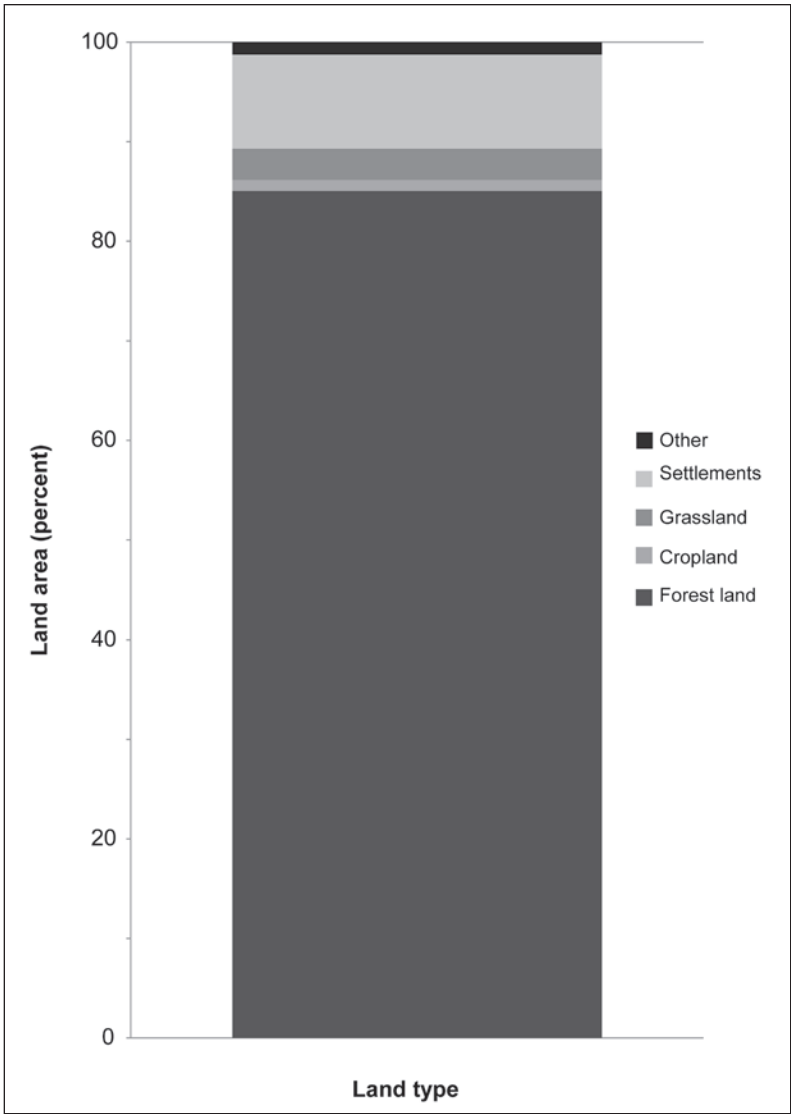

Figure 4.8—Contribution of land areas to net annual carbon sequestration, percentage by land type, 2009.

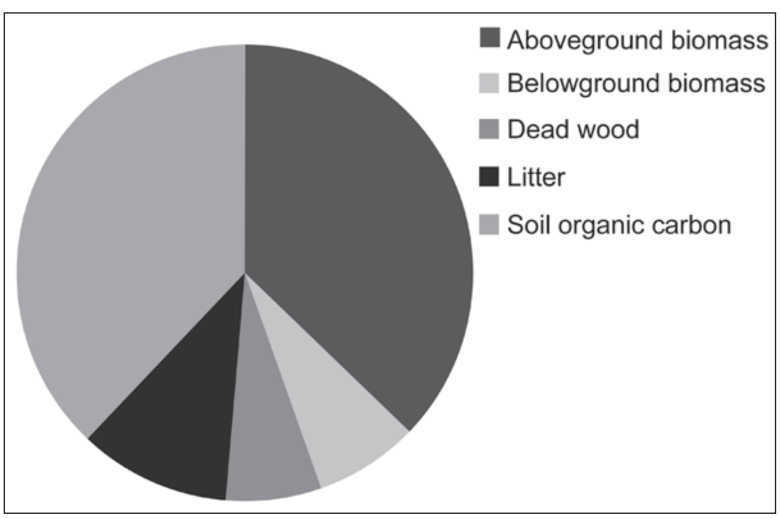

Figure 4.9—Forest carbon pools, share of carbon stored in 2009.

area (C density). Since the 1950s, timberland area nationwide has been stable (fig. 4.10) while the C per unit area has been increasing (i.e., increasing C density). In recent years, the annual increase per unit area has been increasing. The slow accumulation of forests is primarily the result of large-scale reforestation of the United States since the early 1900s. The increasing rate of annual sequestration is a result of gross growth per year continuing to increase, while mortality has increased slowly and harvest removals have stablized (fig. 4.10). Although there are national trends of stable forest area and increasing annual additions of C to forests, it is likely that there are local areas where mortality plus harvest exceeds growth.

Aboveground biomass C stocks are largely found in the Pacific coastal region, Appalachian Mountains, Rocky Mountains, Lake States, and central hardwoods (fig. 4.11). Despite the gradual net increase in forest land area and increased C stocks per unit area, there can be higher variation in net annual C sequestration at smaller spatial scales. A forest can easily become a net emitter of C on account of local disturbances such as wildfire. For most counties, it is estimated that C stocks have been increasing in recent years (fig. 4.12), although uncertainty in annual net sequestration estimates increase greatly as the scale decreases. Given the low density of forest plots that are remeasured each year, estimates of interannual variation in forest C stocks for a local area may only be detectable after major changes such as those occurring after large disturbance events (e.g., large wildfire).

Monitoring and Evaluating Effects of Carbon Management

Figure 4.13 shows C storage and emission processes that can be affected by management of C in forests and wood products. Carbon changes are evaluated by tracking C flows across the system boundaries over time. The boundary around the "forest sector" includes forest, wood products, and wood energy processes. The system boundary includes a defined forest area. A system can be defined to include only C fluxes to and from forests or wood products, or it may include C fluxes from equipment used to manage forests and make and transport wood products, nonwood products, and fossil fuel feedstocks. The effectiveness of C management activities for mitigating GHG emissions is based on forest removal (and retention) of CO_2 from the atmosphere.

Forest management can also affect GHG emissions beyond the "forest sector." System boundaries can be expanded to include processes to make energy from fossil fuels where wood energy can substitute, or to include GHG emitting processes to make nonwood products where wood products can substitute. System boundaries can also be expanded beyond the defined forest area to nonforest areas where actions may cause indirect land use change and associated GHG emissions. System boundaries also include a definition of the time period over which C storage or emissions are evaluated. The choice of system boundaries affects the overall assessment, and defining an objective to alter C management strategy, store C, or alter emissions cannot be done without clearly defining boundaries, processes, and time period. Currently, no standard approach exists for doing this to evaluate forest biomass as a replacement for fossil fuels.

Evaluation of C management strategies associated with forests requires (at a minimum) (1) monitoring C stock changes and emissions over time, and (2) evaluating the effects of altered activities that affect in-forest C (in situ) and associated C storage or emissions outside forests (ex situ). The first accounting framework (type A) determines how C fluxes in terrestrial systems and harvested wood products have actually changed for a current or past period because of management actions and other factors such as natural disturbances. The second accounting framework (type B) determines the degree to which a change in management under various mitigation strategies could increase C storage and decrease emissions.

This accounting compares mitigation activities to a baseline to determine the magnitude of additional C offsets compared to the baseline. A baseline is the level of C stock, C stock change, level of emissions or emissions change as the result of a given set of land conditions and activities (e.g., forest management, timber harvest, and disturbances) and off-land activities (e.g., substitution for fossil emissions, as defined by the accounting system and boundaries at a point in time or over a period of time). A baseline can be defined by a past set of conditions or an envisioned future set

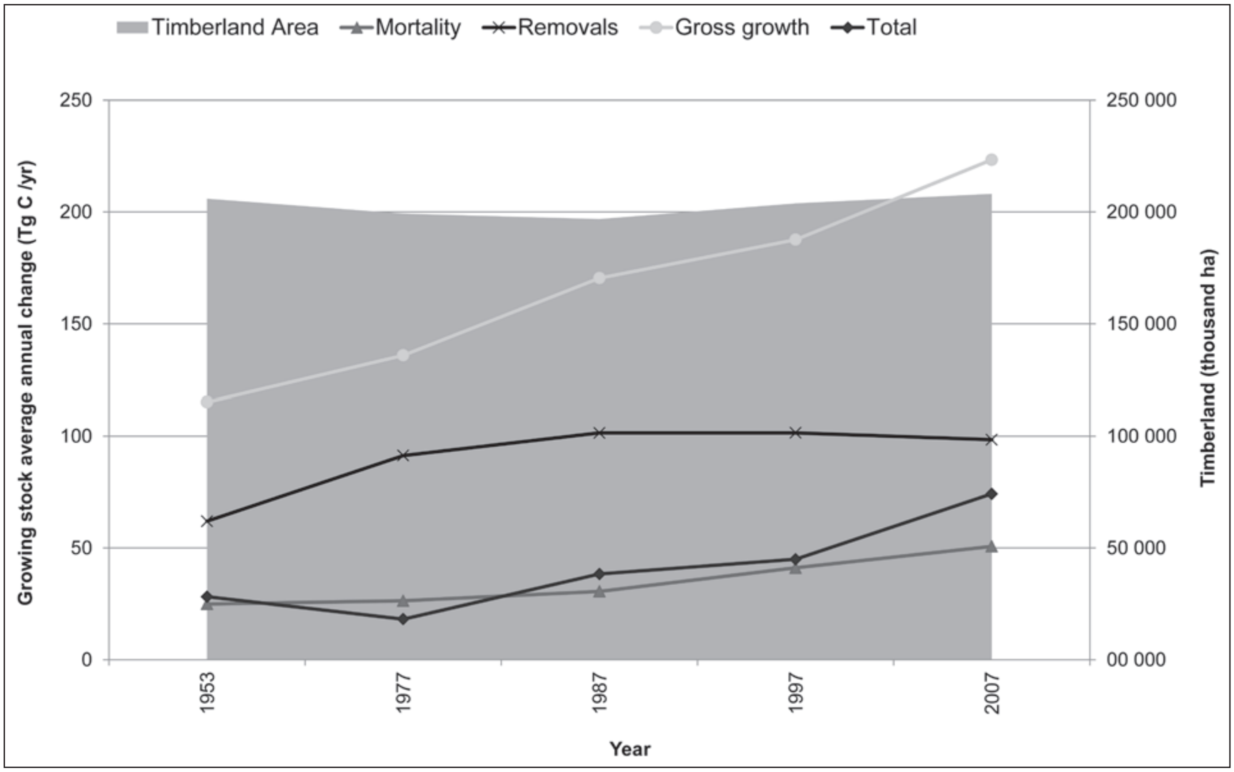

Figure 4.10—Growing stock carbon change owing to growth, mortality, and removals, along with timberland area, 1953–2007.

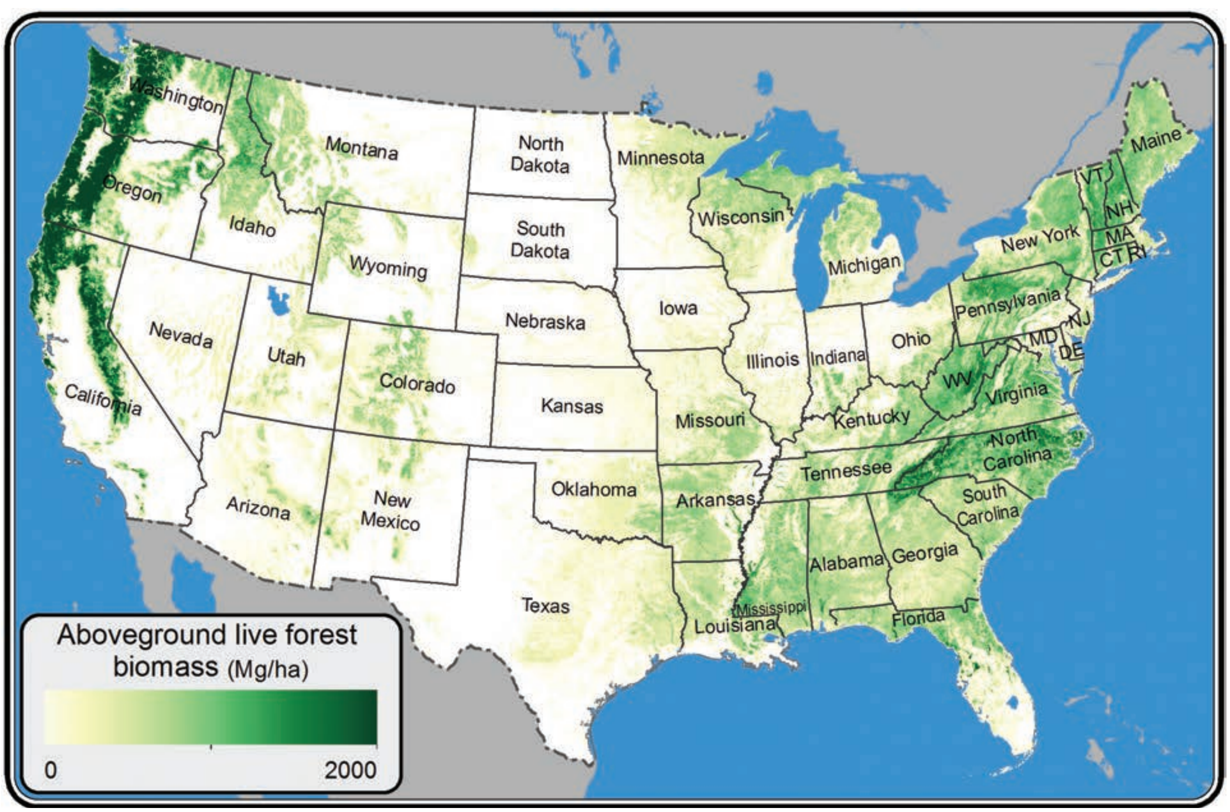

Figure 4.11—Aboveground live biomass in forests.

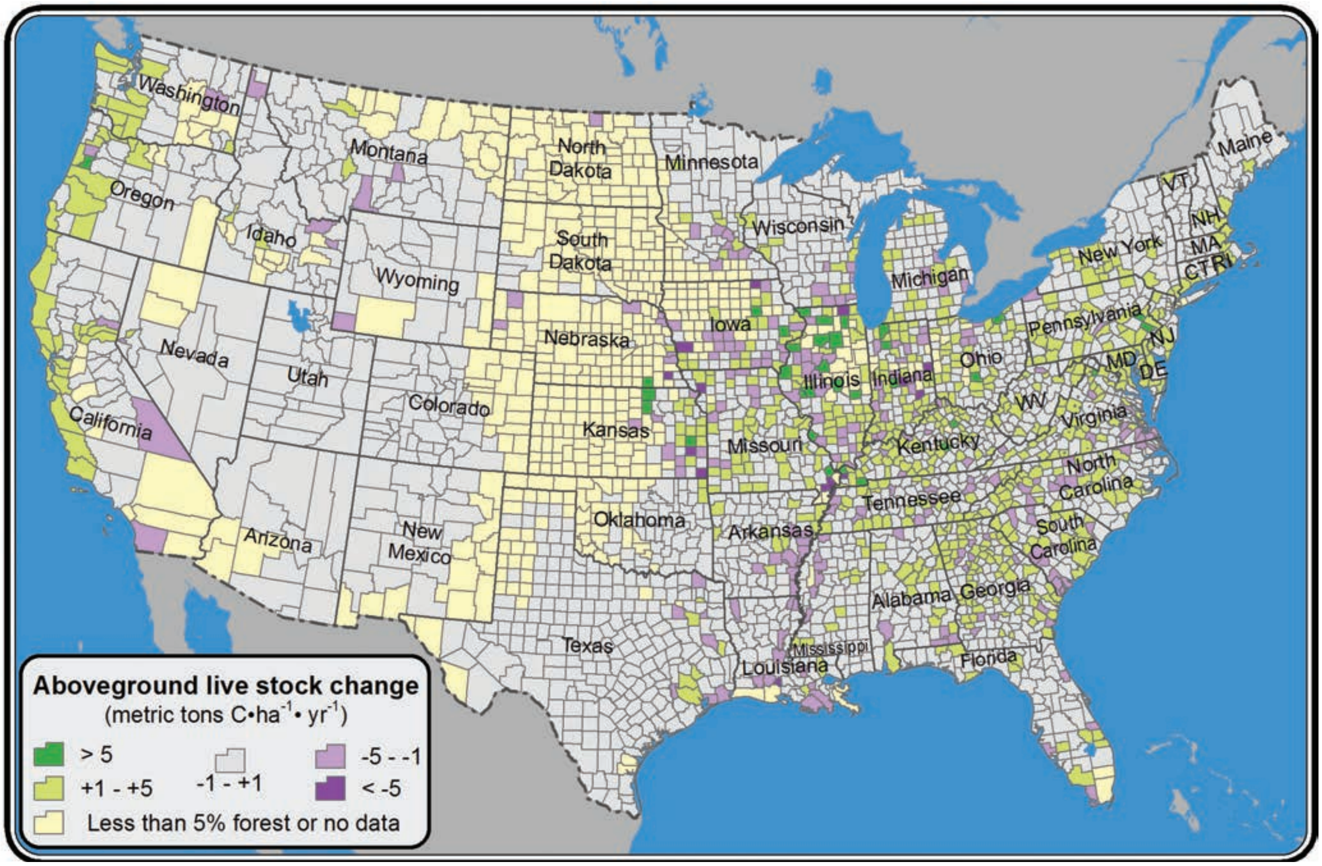

Figure 4.12—Aboveground live forest carbon (C) change.

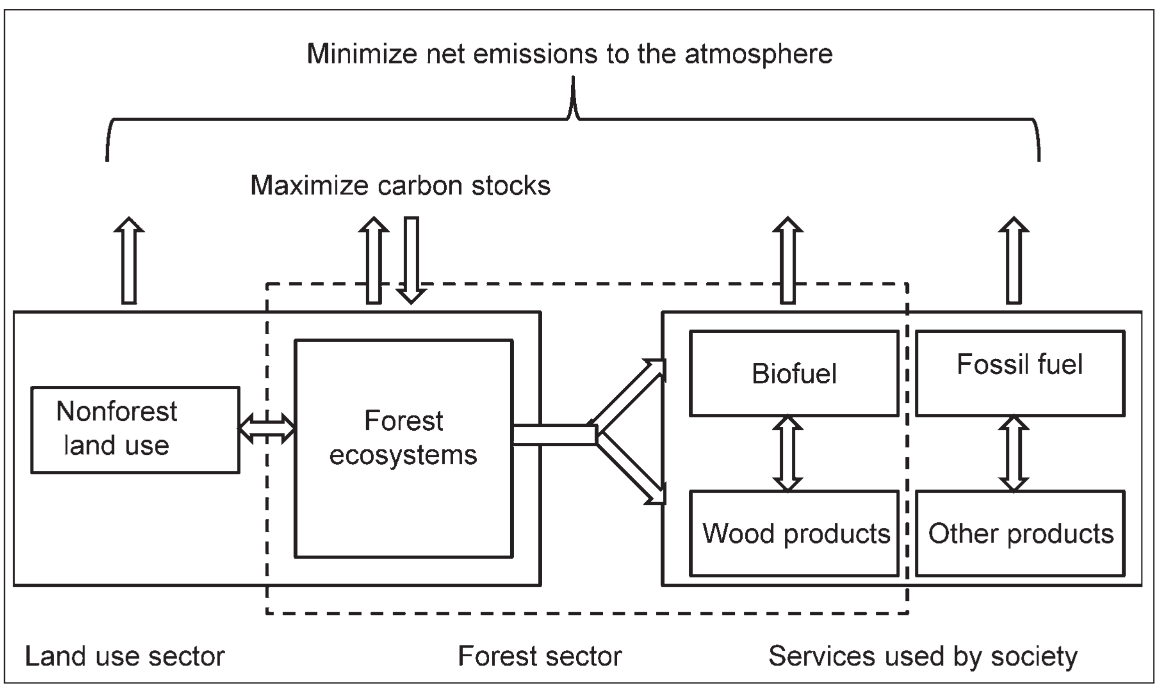

Figure 4.13—Forest sector and nonforest sector greenhouse gas emissions and stock changes that are influenced by forest management.

of conditions. The effectiveness of a new strategy, such as an incentive to increase wood use for energy, is determined by changes in landowner behavior. For example, with high energy use (high price) some landowners may convert nonforest land to wood plantations and accumulate more C as well as gain benefit from substituting wood for fossil fuel. In addition, an increase in wood prices could cause pulpwood to be used for energy and decrease oriented strandboard panel production and resultant C storage in panels.

A specific accounting framework for evaluating C management must include, explicitly or implicitly, a specification of the type of accounting framework (A or B) and of the system boundaries for the processes included (e.g., forest sector, service sector, nonforest land use, specific forest area, time period, wood C only, and other GHG emissions from processes).

A "common" type A accounting framework for monitoring is to define system boundaries to include current annual C exchange with the atmosphere from forest ecosystems at a given geographic scale, plus C additions and emissions for harvested wood products from those forests (fig. 4.13). This framework can be used to answer this management question for a given forest area: "Are forests and forest products continuing to (collectively) withdraw and store C from the atmosphere?" The framework is also the basis for reporting GHG emissions and sinks in many accounting systems such as that used in annual reports to the United Nations Framework Convention on Climate Change.

This framework is not intended for evaluating the full effects on atmospheric CO_2 of a change in strategy, which would require a system boundary that includes changes in nonwood C emissions and C emissions or storage outside the forest. Some excluded changes may include altered fossil fuel use, other land use emissions, and altered nonwood product emissions (fig. 4.13). Evaluating strategy changes requires a framework that includes all processes that significantly change atmospheric CO_2. If changes in emissions occur over many years, the framework must evaluate CO_2 fluxes over many years. For example, a strategy to increase use of wood for heat, electric power, or biofuels via incentives at a national level would change CO_2 flux estimates compared to a given baseline over an extended time from (1) wood for energy, (2) fossil fuels for energy, (3) land use change (crops to plantation, or forest to intensive plantation), and (4) flux from forests where wood is removed (including regrowth after removal). The accounting system needs to include all processes noted in fig 4.13.

"Leakage" is a term used to recognize certain C effects when evaluating the effects of a policy or management change by using a type B accounting framework. Leakage is the C effects of a program change that are outside the system boundaries defined by a limited set of processes, (e,g., C changes for a specific forest area). Leakage, which includes C changes on land outside of a system boundary (e.g., caused by changes in harvest or land use) (Gan and McCarl 2007, Murray et al. 2004, Pachauri and Reisinger 2007, Schwarze et al. 2002, Sohngen et al. 1999), differs depending on the mitigation activity and can be quite high (Gan and McCarl 2007, Murray et al. 2004). In the United States, leakage estimates associated with activities on a given land area range from less than 10 percent to greater than 90 percent (proportion of C benefit lost), depending on the activity and region (Murray et al. 2004). Globally, leakage estimates range between 42 and 95 percent (Gan and McCarl 2007).

Leakage tends to be highest where programs constrain the supply of forest products (e.g., no harvest is allowed) or constrain land use change (e.g., forest land conversion to agriculture) (Aukland et al. 2003; Depro et al. 2008; Murray et al. 2004; Sohngen et al. 1999, 2008; Sohngen and Brown 2008). In contrast, the indirect effects of a program can increase C benefits outside of a system boundary, a phenomenon termed "spillover" (Magnani et al. 2009). For example, spillover can occur if an increase in plantation forestry reduces C losses from established forests by increasing C flows in cheaper forest products (Magnani et al. 2009). Defining system boundaries to include indirect effects on C (e.g., multinational programs) or otherwise accounting for leakage ensures program integrity.

Carbon storage strategies may be ineffective because of flaws in incentive structures or policies, and not caused by the biophysical attributes of the strategy itself. For example, an incentive program might favor harvesting large trees that produce lumber, assuming that lumber would replace building materials that emit more C in manufacturing. If this incentive strategy were implemented, the lumber could go to nonbuilding uses, or an increase in harvest by one landowner could be offset by a decrease by another. This is a flaw of the incentive system, not of the underlying wood substitution strategy. If there were incentives for builders to use wood in structures rather than alternate materials, the strategy could be effective in reducing overall emissions from manufacturing; however, the effectiveness depends on the assumed changes in forest management.

As described above, the focus of evaluating C management strategies was on understanding how altered management influenced C on a given land area. It is also possible to evaluate strategies by focusing on the change in C storage or emissions associated with producing one unit of wood energy or one unit of wood product, by using life cycle assessment (LCA). An "attributional LCA" is similar to a type A accounting framework and includes specification of processes that include forest growth, harvest, manufacturing, end use, disposal, and reuse, with the objective of estimating storage and emissions over the life cycle of one unit of product. Attributional LCAs are used to monitor inputs and emissions associated with production and do not include all process that would be affected by a change in production or in processes. A "consequential LCA" also specifies a unit of product and system boundary, but is similar to the type B framework noted above because the objective is to estimate the change in emissions associated with a one-unit change in product production or some change in processes over the life cycle. Consequential LCAs are typically used to analyze the potential response of a change to a system, such as a change in policy, and can include the effect of changing demand levels for products on production and emissions from other products across many sectors.

Different C management strategies are often evaluated by using different system boundaries, accounting frameworks, models, assumptions, functional units (land area vs. product units), and assumed incentives. Therefore, it can be difficult to compare the effectiveness of different strategies. However, it is possible to describe the effects of strategies on changing particular processes, uncertainty in attaining the effects, and timing of the effects.

Carbon Mitigation Strategies

Carbon mitigation through forest management focuses on (1) land use change to increase forest area (afforestation) or to avoid deforestation, or both; (2) C management in existing forests; and (3) use of wood as biomass energy, in place of fossil fuel or in wood products for C storage and in place of other building materials. Estimates of the amount of the Nation's CO_2 emissions offset by forests and forest products (using the type A framework) vary with assumptions and accounting methods (e.g., from 10 to 20 percent) (McKinley et al. 2011), with 13 percent (about 221 Tg·C·yr $^{-1}$) being the most recent estimate for the United States (USEPA 2011). The first two strategies aim to maintain or increase forest C stocks (using the type B framework with a boundary around forest area and other land capable of growing forests). The last strategy focuses on increasing C storage or reducing fossil fuel emissions, including C fluxes associated with forests and products removed from the forest (using the type B framework with a boundary around the forest sector, services, and nonforest land processes [fig. 4.13]. The mitigation potential of these strategies differs in timing and magnitude (table 4.8).

Land use change: afforestation, avoiding deforestation, and urban forestry–

Afforestation—In the United States, estimates of the potential for afforestation (active establishment or planting of forests) to sequester C vary from 1 to 225 Tg·C·yr $^{-1}$ for 2010 to 2110 (U.S. Climate Change Science Program 2007, USEPA 2005). Afforestation can be done on land that has not been forested for some time (usually more than 20 years), such as some agricultural lands, or on lands than have not historically supported forests, such as grasslands. Reforestation refers to establishing forests on land that has been in nonforest use for less than the specified time period. Mitigation potentials, cobenefits, and environmental tradeoffs depend on where afforestation and reforestation efforts are implemented (table 4.8).

The mitigation potential of afforestation and reforestation on former forest land is significant and generally has the greatest cobenefits, lowest risk, and fewest tradeoffs. Forest regrowth on abandoned cropland comprises about half of the

Table 4.8—Mitigation strategies, timing of impacts, uncertainty in attaining carbon (C) effects, cobenefits and tradeoffs[a]

Mitigation strategy	Timing of maximum impact	Uncertainty about strategy (biophysical risks)	Uncertainty about strategy (structural risks)[b]	Cobenefits	Tradeoffs
Land use change:					
Afforestation (on former forest land)	Delayed	Low	Leakage	Erosion control; improved water quality; increased biodiversity and wildlife habitat	Lost revenue from agriculture
Afforestation (on nonforest land)	Delayed	Moderate	Leakage	Biodiversity	Erosion; lower streamflow; decreased biodiversity and wildlife habitat; increased nitrous oxide emissions; competition for agricultural water; local warming from lower albedo
Avoided deforestation	Immediate	Low	Leakage	Watershed protection; maintain biodiversity and wildlife habitat, some recreational activities	Lost economic opportunities affecting farmers or developer directly
Urban forestry	Delayed	High		Reduced energy use for cooling; increased wildlife habitat; possible recreational opportunities	High maintenance might be required in terms of water, energy, and nutrients; possible damage to infrastructure
In situ forest C management:					
Decreasing C outputs	Immediate	Moderate	Leakage	Increased old-growth seral stage; increased structural and species diversity, and wildlife habitat; effects on benefit depend on landscape condition	Displaced economic opportunities affecting forest owners, forest industry, and employees
Increasing forest growth	Delayed	Low	Leakage	Higher wood production, potential for quicker adaptation to climate change	Lower streamflow, loss of biodiversity, release of nitrous oxide, greater impact of disturbance on C storage
Fuel treatments	Delayed	High		Lower risk from fire and insects; increased economic activity; possible additional offsets from use of wood; climate change adaptation tool	Lost economic opportunities to firefighting business and employees; lower C on site; site damage caused by treatment
Ex situ forest C management:					
Product substitution	Part immediate, part delayed	Moderate	Leakage	Increased economic activity in forest product industries	Active forest management on larger area; low C storage in forests
Biomass energy	Immediate to delayed depending on source	Moderate/high	Leakage	Increased economic activity in forest product industries; could reduce cost of forest restoration efforts	Intensive management on large area; lower C storage in forests

[a] Uncertainty as defined here is the extent to which an outcome is not known. All mitigation strategies have a risk of leakage and reversal, which could compromise C benefits. Timing of maximum impact is adapted from IPCC (2007) and uncertainty, cobenefits, and tradeoffs from McKinley et al. (2011).

[b] The potential degree of leakage or other structural risk for a strategy depends on the incentives, regulations, or policy used to implement it. For example, if an incentive program to increase forest growth is in only one region, then growth may be decreased in other regions. If the incentive is nationwide, there is little leakage within the United States, but may be leakage to other countries. Other structural risks can come from improper selection of locations to implement the strategy. For example, fire hazard reduction treatments could be done on land areas where fire-risk offsets are insufficiently long lasting or emission mitigating in comparison to expected avoided emissions from fire. There can also be risk in selecting wood for fuel (e.g., from older forests) where C recovery will be very slow.

U.S. C sink (Pacala et al. 2001). One study estimated that sequestering the equivalent of 10 percent of U.S. fossil fuel emissions (160 Tg of C) would require that 44 million ha, or one-third of U.S. croplands, be converted to tree plantations (Jackson and Schlesinger 2004). Another report estimates that 262 000 to 1 133 000 ha are needed to sequester 1 Tg of C annually (USEPA 2005). Given potential global food shortages and high value of many crops, forest establishment on productive croplands is not likely tenable and may cause project leakage (Murray et al. 2004). However, establishing forest plantations on marginal agricultural land or abandoned agricultural land is more feasible, because potential interference with food production is lower. Where climatic and soil conditions favor forest growth (over crops), irrigation and fertilization inputs would be low relative to gains in C storage. Cobenefits may include erosion control, improved water quality, higher species diversity, and wildlife habitat. The cobenefits of afforestation are enhanced where native species comprise a substantial proportion of the regenerated forest. Monocultures of nonnative or native improved-growth species may yield high C storage rates and have a low risk for unintended results, but may also provide fewer cobenefits.

Afforestation on lands that do not naturally support forests may require more human intervention and environmental tradeoffs. Carbon storage in tree and shrub encroachment into grasslands, rangelands, and savannas is estimated to be 120 Tg·C·yr $^{-1}$, a C sink that could be equivalent to more than half of what existing U.S. forests sequester annually, although this estimate is highly uncertain (U.S. Climate Change Science Program 2007). This C sequestration shows the potential (unintentional) effects of land use change and other human activities (Van Auken 2000). Planting trees where they were not present historically can sometimes alter species diversity, lower the water table, cause soil erosion on hill slopes, and absorb more solar energy compared with the native ecosystem (Farley et al. 2008, Jackson et al. 2008, Jobbagy and Jackson 2004, McKinley and Blair 2008, Schwaiger and Bird 2010). Irrigation and fertilization would likely be needed in many areas, particularly in arid and semiarid regions, which might compete with agricultural water supply and other uses. Afforestation also has the potential to reduce streamflow because some species of trees use more water than grasses or crops (Farley et al. 2005, Jackson et al. 2005). Use of nitrogen (N) fertilizers may increase nitrous oxide emissions, a GHG with roughly 300 times more global warming potential than CO_2. This type of afforestation has more risks compared with afforestation on lands that naturally support forests.

Avoiding deforestation—Avoiding the loss of forested land can prevent a significant loss of C to the atmosphere. Currently, global deforestation results in the gross annual loss of nearly 90 000 km^2, or 0.2 percent of all forests (FAO 2007, Pachauri and Reisinger 2007), which is estimated to release 1400 to 2000 Tg·C·yr $^{-1}$, with about two-thirds of the deforestation occurring in tropical forests in South America, Africa, and Southeast Asia (Houghton 2005, Pachauri and Reisinger 2007). Over a recent 150-year period, global land use change released 156 000 Tg of C to the atmosphere, mostly from deforestation (Houghton 2005). In contrast, forested area in the United States increased at a net rate of about 340 000 ha·yr $^{-1}$ in a recent 5-year period (2002 to 2007). Increases in forested area and forest regrowth are largely responsible for the current U.S. forest C sink of 211 Tg·C·yr $^{-1}$ (USEPA 2011). However, these dynamics will change, with future land use expected to decrease total forested area by more than 9 million ha by 2050 (Alig et al. 2003). Development and conversion of forest to pasture or agricultural land are responsible for much of the current and expected loss of forests. In addition, increased area burned by fire may result in the conversion of some forests to shrublands and meadows (Westerling et al. 2011), or a permanent reduction in C stocks on existing forests if fire-return intervals are reduced (Balshi et al. 2009, Harden et al. 2000). Potential C mitigation estimates through avoided deforestation are not available for the United States.

Avoided deforestation protects existing forest C stocks and has many cobenefits and low risk (table 4.8). Cobenefits include maintaining ecosystem properties and processes, such as watersheds, biodiversity, wildlife habitat, and some recreational activities (McKinley et al. 2011). Risks include incentives to avoid deforestation in one area that may increase removal of forest in other areas, with little net lowering of atmospheric CO_2. Avoided deforestation may decrease economic opportunities for timber, agriculture, pasture, or urban development (Meyfroidt et al. 2010). Leakage can be large for avoided deforestation, particularly if harvest is not allowed (Murray et al. 2004). Regenerating forests after severe wildfires may be important for avoiding conversion of forest to meadow or shrubland (Donato et al. 2009, Keyser et al. 2008).

***Urban forestry*—**Urban forestry, the planting and management of trees in and around human settlements, offers limited potential to store additional C, but urban trees provide some indirect ways to reduce fossil fuel emissions and have many cobenefits. Although U.S. urban C stocks are surprisingly large (Churkina et al. 2010), the potential for urban forestry to help offset GHG emissions is limited for two reasons: (1) urban areas make up only a small fraction of the U.S. landscape (3.5 percent) (Nowak and Crane 2002), and (2) urban trees generally require intensive management. Urban forests have important indirect effects on climate by cooling with shading and transpiration, potentially reducing fossil fuel emissions associated with air conditioning (Akbari 2002). When urban forests are planted over very large regions, the climate effects are less certain, because trees have both warming effects (low albedo) and cooling effects, and may result in complex patterns of convection that can alter air circulation and cloud formation (Jackson et al. 2008). However, urban trees can have high mortality rates in all regions (Nowak et al. 2004), and they require ongoing maintenance, particularly in cities that are in arid regions; risks increase when irrigation, fertilization, and other forms of maintenance are necessary (Pataki et al. 2006).

In Situ Forest Carbon Management

Carbon mitigation through forest management focuses on efforts to increase forest C stock by either decreasing C outputs in the form of harvest and disturbance, or increasing C inputs through active management. Potential C mitigation for a combined effort including increased harvest intervals, increased growth, and preserved establishment could remove 105 $Tg{\cdot}C{\cdot}yr^{-1}$, although achieving these results would require large land areas. It is estimated that between 479 000 and 707 000 ha of manageable forest land is needed to store 1 $Tg{\cdot}C{\cdot}yr^{-1}$ (USEPA 2005).

Increasing forest carbon by decreasing harvest and protecting large carbon stocks—
Forest management can increase the average forest C stock by increasing the interval between harvests or decreasing harvest intensity (Balboa-Murias et al. 2006, Harmon and Marks 2002, Harmon et al. 2009, Jiang et al. 2002, Kaipainen et al. 2004, Liski et al. 2001, Schroeder 1992, Seely et al. 2002, Thornley and Cannell 2000). Increasing harvest intervals would have the biggest effect on forests harvested at ages before peak rates of growth begin to decline (culmination of mean annual increment [CMAI]), such as some Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco) forests in the northwestern United States. Increasing rotation age for forests with low CMAI, such as southern pine species that are already harvested near CMAI, would yield a decreasing benefit per year of extended rotation.

Harvesting forests with high biomass and planting a new forest reduces overall C stocks more in the near term than if the forest were retained, even counting the C storage in harvested wood products (Harmon et al. 1996, 2009). For example, some old-growth forests in Oregon store as much as 0.0011 $Tg{\cdot}C{\cdot}ha^{-1}$ (Smithwick et al. 2002), which would require centuries to regain if these stocks were liquidated and replaced, even with fast-growing trees (McKinley et al. 2011). Low intensity or partial harvests, including leaving dead wood on site, maintain higher C stocks compared to clearcuts (Harmon et al. 2009), while possibly reducing the risk of disturbance, such as fire and damaging storms, and concurrently allowing forests to be used for wood products or biomass energy. However, although thinning increases the size and vigor of residual trees, it generally reduces net C storage rates and C storage at the stand scale (Dore et al. 2010, Schonau and Coetzee 1989). Studies evaluating the harvest effects on soil C provide mixed conclusions (Johnson and Curtis 2001, Nave et al. 2010). Decreasing removal of C from forests through longer harvest intervals or less intense harvests increases forest C stocks. Benefits of decreased outputs include an increase in structural and species diversity (table 4.8). Risks include C loss owing to disturbance and reduced substitution of wood for materials that emit more C in manufacturing.

Managing forest carbon with fuel treatments—
Since 1990, CO_2 emissions from wildland forest fires in the conterminous United States have averaged 67 $Tg{\cdot}C{\cdot}yr^{-1}$ (USEPA 2009a, 2010). The possibility that fuel treatments, although reducing onsite C stocks, may contribute to mitigation by providing a source for biomass energy and avoiding future wildfire emissions, is attractive, especially because fuel treatments may play an important role in climate change adaptation. Fuel treatments have other important benefits, including their potential to protect property and restore forest conditions more resilient to periodic wildfire. It is unlikely that fuel treatments would be implemented solely to manage C stocks.

Fuel treatments are a widespread forest management practice in the Western United States. (Battaglia et al. 2010), and are designed to alter fuel conditions to reduce wildfire intensity, crown fires, tree mortality, and suppression difficulty (Reinhardt et al. 2008, Scott and Reinhardt 2001). Fuel treatment to reduce crown fire hazard can be done by reducing surface fuels, ladder fuels (small trees), and canopy fuels (Peterson et al. 2005). All of these remove C from the site, whether through harvest or prescribed fire (Reinhardt et al. 2010, Stephens et al. 2009), and alter subsequent forest C dynamics by modifying the residual stand.

Crown fires often result in near-total tree mortality, whereas many trees can survive surface fires. This contrast in survival has led to the notion that fuel treatments may offer a C benefit by removing some C from the forest to protect the remaining C (Dore et al. 2010, Finkral and Evans 2008, Hurteau et al. 2008, Mitchell et al. 2009, Stephens et al. 2009). Thinned stands that burn in a surface fire typically have much higher tree survival and lower C losses than similar, unthinned stands that burn in a crown fire (e.g., Finkral and Evans 2008, Hurteau and North 2009, Hurteau et al. 2008, Stephens et al. 2009), although the net effect of fuel treatment C removal and surface fire emissions may exceed that from crown fire alone, even when materials from fuel treatments are used for wood products (Reinhardt et al. 2010). Because fuel treatment benefits are transient, they may lapse before a wildfire occurs, in which case the C removed by the fuel treatment is not offset by reduced wildfire emissions.

Modeling studies suggest that fuel treatments in most landscapes will result in a net decrease in landscape C over time (Ager et al. 2010, Harmon et al. 2009, Mitchell et al. 2009), because the savings in wildfire emissions is gained only on the small fraction of the landscape where fire occurs each year. For treatments to yield a substantial C benefit, the following conditions would be required: (1) relatively light C removal would substantially reduce emissions, (2) fire occurrence is high in the near term (while fuel treatments are still effective), and (3) thinnings can provide wood for energy or long-lived products that yield substitution benefits. If fuel treatments are implemented, it is advantageous from a C management standpoint to use removed fuels for energy production or wood products, rather than burning them onsite (Coleman et al. 2010, Jones et al. 2010). Feasibility and energy implications depend in part on hauling distance (Jones et al. 2010). An intriguing alternative to hauling bulky biomass to conversion facilities is in situ pyrolysis to produce energy-dense liquid fuel and biochar which can remain onsite to enhance soil productivity and sequester C (Coleman et al. 2010).

Increasing forest carbon stocks by increasing forest growth—
Increasing growth rates in existing or new forests could increase C storage on the landscape and increase the supply of forest products or biomass energy. Practices that increase forest growth include fertilization, irrigation, use of fast-growing planting stock, and control of weeds, pathogens, and insects (Albaugh et al. 1998, 2003, 2004; Allen 2008; Amishev and Fox 2006; Borders et al. 2004; Nilsson and Allen 2003). The potential associated with increasing forest growth differs by site and depends on the specific climate, soil, tree species, and management.

Increased yields from these practices can be impressive. In pine forests in the Southern United States, tree breeding has improved wood growth by 10 to 30 percent (Fox et al. 2007b), and has increased insect and stress resistance (McKeand et al. 2006.) In this region, pine plantations using improved seedlings, control of competing vegetation, and fertilization grow wood four times faster than naturally regenerated second-growth pine forests without competition control (Carter and Foster 2006). Tree breeding and intensive management could also provide an opportunity to plant species and genotypes that are better adapted to future climates.

Many U.S. forests are N limited and would likely respond to fertilization (Reich et al. 1997). Nitrogen and phosphorus fertilizers have been used in about 6.5 million ha of managed forests in the southeast to increase wood production (Albaugh et al. 2007, Fox et al. 2007a, Liski et al. 2001, Seely et al. 2002). Fertilization can produce 100 percent gains for wood growth (Albaugh et al. 1998, 2004), although the benefits of fertilization for growth and C increase would need to be balanced by the high emissions associated with fertilizer production and potential emissions from eutrophication in aquatic systems (Carpenter et al. 1998) (table 4.8). Other risks include reduced water yield (faster growth uses more water), which is more pronounced in arid and semiarid forests, and a loss of biodiversity if faster growth is done by replacing multispecies forests with monocultures (limited diversity can make some forests vulnerable to insects and pathogens). In some areas, increasing the genetic and species diversity of trees and increasing C stocks could be compatible goals (Woodall et al. 2011).

Markets for current or new forest products can provide revenue to invest in growth-enhancing forest management. For example, expectation of revenue from the eventual sale of high-value timber products would support investment in treatments or tree planting to increase growth rate. Taxation or other government incentives may also support growth-enhancing management. To the extent that incentives to alter growth also alter timber harvest and wood product use, evaluation will require type B accounting with system boundaries that include forest sector, services sector, and possibly nonforest land.

Ex Situ Forest Carbon Management

Carbon is removed from the forest for a variety of uses, and those uses can have different effects on C balances. Depending on the forest product stream, C can be stored in wood products for a variable length of time, oxidized to produce heat or electrical energy, or converted to liquid transportation fuels and chemicals that would otherwise come from fossil fuels (fig. 4.14). In addition, there can be a substitution effect when wood products are used in place of other products that emit more GHG in manufacturing (Lippke et al. 2011).

Strategies that would add to storage in long-lived wood products, increase use life, and increase use of wood products in place of higher emitting alternate products can complement strategies aimed at increasing forest C stocks. Risk and uncertainty in attaining benefits need to be considered when comparing strategies for increasing forest C with strategies for attaining wood product C offsets. Strategies need to ensure energy offsets are attained in an acceptable period of time and that substitution effects are attained.

Carbon in forest products—
Wood and paper continue to store C when in use and also in landfills (fig. 4.14). Rates of net C accumulation depend on rates of additions, disposal, combustion, and landfill decay. The half-life for single-family homes made of wood built after 1920 is about 80 years (Skog 2008, USEPA 2008), whereas the half-life of paper and paperboard products is less than 3 years (Skog 2008). About two-thirds of discarded wood and one-third of discarded paper go into landfills (Skog 2008). Decay in landfills is typically anaerobic and very slow (Barlaz 1998), and 77 percent of the C in solid wood products and 44 percent in paper products remain in landfills for decades (Chen et al. 2008, Skog 2008). However, current rates of methane release and capture can eliminate this C storage benefit for certain low lignin paper products (Skog 2008). About 2,500 Tg of C was accumulated in wood products and landfills in the United States from 1910 to 2005 (Skog 2008), with about 700 Tg of C (in 2001) in single- and multifamily homes (Skog 2008). In 2007, net additions to products in use and those in landfills combined were 27 $Tg{\cdot}C{\cdot}yr^{-1}$ (USEPA 2009b), with about 19 $Tg{\cdot}C{\cdot}yr^{-1}$ from products in use (Skog 2008).

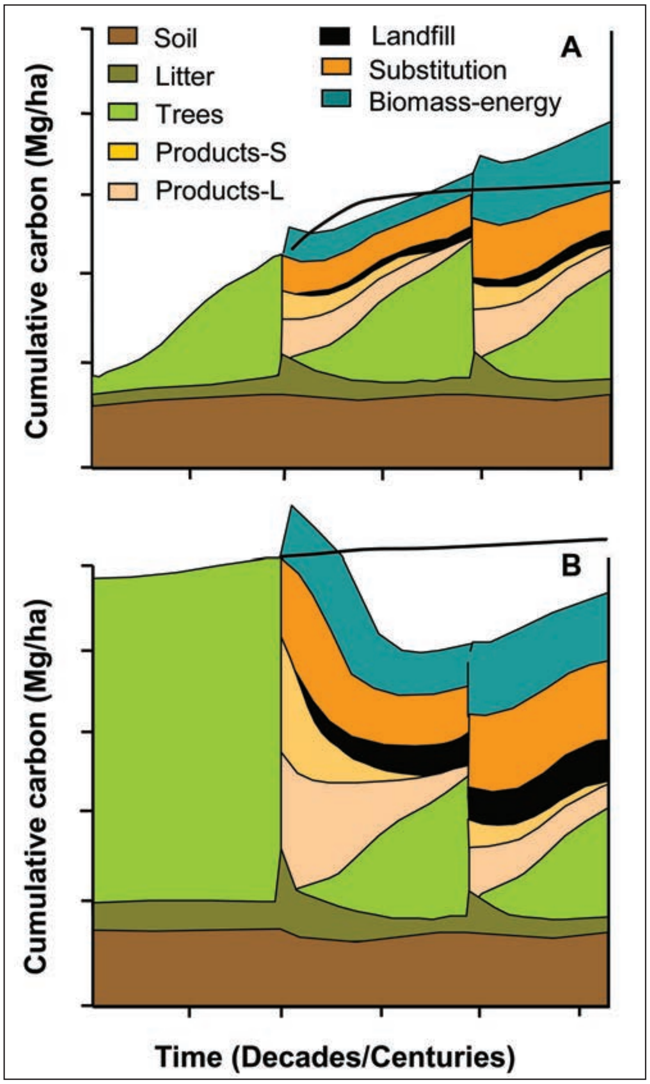

Figure 4.14—Carbon (C) balance from two hypothetical management projects with different initial ecosystem C stocks and growth rates. Cumulative C stocks in forest, C removed from forest for use in wood projects (long [L]- and short-lived [S]), substitution, and biomass energy are shown on land that (A) has been replanted or afforested, or (B) has an established forest with high C stocks. The heavy black line represents the trajectory of forest C stocks if no harvest occurred. Actual C pathways vary by project. Carbon stocks for trees, litter, and soils are net C stocks only. The scenario is harvested in x-year intervals, which in the United States could be as short as 15 years or longer than 100 years. This diagram assumes that all harvested biomass will be used and does not account for logging emissions. Carbon is sequestered by (1) increasing the average ecosystem C stock (tree biomass) by afforestation, or (2) accounting for C stored in wood products in use and in landfills, as well as preventing the release of fossil fuel C through product substitution or biomass energy. The product-substitution effect is assumed to be 2:1 on average. Biomass is assumed to be a 1:1 substitute for fossil fuels in terms of C, but this is not likely for many wood-to-energy options. This scenario represents a theoretical maximum C benefit, given this composition of forest products and management practices. Carbon "debt" is any period of time at which the composition of forest products and remaining forest C stocks after harvest is lower than estimated C stocks under a no-harvest scenario. (Adapted from McKinley et al. [2011], Pachauri and Reisinger [2007], and Solomon et al. [2007]).

Product substitution—

Net C emissions associated with production and use of forest products can be substantially less than those associated with steel and concrete. Use of 1 Mg of C in wood materials in construction in place of steel or concrete can result in 2 Mg of lower C emission (Sathre and O'Connor 2008, Schlamadinger and Marland 1996). Sometimes, using wood from faster-growing forests for substitution can be more effective in lowering atmospheric CO_2 than storing C in the forest where increased wood production is sustainable (Baral and Guha 2004, Marland and Marland 1992, Marland et al. 1997) (fig. 4.14a). On the other hand, harvesting forests with very high C stocks that have accumulated over many decades may result in a large deficit of biological C storage that could take many decades to more than a century to restore (McKinley et al. 2011) (fig. 4.14b). Opportunities for substitution in the United States are largely in nonresidential buildings (McKeever et al. 2006, Upton et al. 2008) because most houses are already built with wood, although opportunities to increase the substitution effect in residential buildings exist, for example, by using wood for walls in houses (Lippke and Edmonds 2006). Attaining the substitution effect requires incentives that avoid or reduce type 1 risks by encouraging increased use of wood (box 4.9). Incentives focused on landowners to harvest wood for products may not provide as many substitution effects because of leakage. In

Box 4.9

Each strategy has risks and uncertainties in attaining carbon (C) impacts as well as non-C impacts—cobenefits and tradeoffs. In this section, we describe two general sources of risk that may prevent a strategy from attaining its C mitigating potential in terms of magnitude or timing, or both, or possibly resulting in reversal. Type 1 risk refers to the failure of incentives or regulations. This risk stems from the constructs of the policy or incentive structure, which might not have the intended effect on human behaviors. This might include, for example, lower than expected participation in markets or unintended negative economic distortions, such as supply-side diversions, that alter forest management or forest product use.

Type 1 risks are "structural risk." One "structural risk" if not accounted for is, for example, "leakage" in the form of shifting of harvest or land use. Type 2 risk or "biophysical risk" refers to failures caused by unpredictable or greater than expected biophysical events, such as natural disturbance (e.g., wildfire, insects). Disturbance can be a cause of "risk of reversal" or failure to attain "permanence." Type 3 risk or "tradeoffs" is the intensification of negative non-C impacts. Uncertainty may also result in greater than expected mitigation. For example, changing climate and atmospheric chemistry (e.g., increasing carbon dioxide or nitrogen deposition) may result in faster accumulation of C in forests than expected for some period of time.

addition, incentives would help avoid type 1 risks in which wood may come from forest conditions where C recovery is slow (fig. 4.14a) and instead comes from forest conditions where C recovery is fast (e.g., fig. 4.14b).

Biomass energy—

Biomass energy could prevent the release of an estimated 130 to 190 $Tg \cdot C \cdot yr^{-1}$ from fossil fuels (Perlack et al. 2005, Zerbe 2006). Biomass energy comprises 28 percent of renewable energy supply and 2 percent of total energy use in the United States; the latter amount has the potential to increase to 10 percent (Zerbe 2006). Currently, wood is used in the form of chips, pellets, and briquettes to produce heat or combined heat and generation of electricity (Saracoglu and Gunduz 2009). These basic energy carriers can be further transformed, using advanced conversion technologies, into liquid transportation fuels, and gases (e.g., methane and hydrogen) (Bessou et al. 2011, Demirbas 2007). Conversion processes for these fuels are still largely experimental and require further development to improve efficiency and commercial viability. The GHG balances for simple energy carriers (e.g., wood chips and pellets) for producing heat and electricity are more certain than for advanced energy carriers. In addition, the potential exists to create high-value chemicals and other bioproducts from wood that would otherwise be made from fossil fuels, resulting in reduced emissions compared to use of fossil fuels (Hajny 1981, USDOE 2004).

Most biomass for energy is a byproduct of conventional forest product streams, such as milling residues (Gan and Smith 2006a), with some use of trees killed by insects, disease, and natural disturbance (Peng et al. 2010, Tumuluru et al. 2010). However, most of these residues, mainly sawdust and bark, are already used for direct heating in milling operations or used for other wood products, such as particle board (Ackom et al. 2010, Mälkki and Virtanen 2003, Nilsson et al. 2011); obtaining higher quantities of biomass feedstock would require using other residues. A number of currently unused residues have been identified, including residues from logging, hazardous fuel reduction treatments, precommercial thinning, urban areas, insect kill, and other sources (Ackom et al. 2010; Gan 2007, Gan and Smith 2006b; Mälkki and Virtanen 2003; Perlack et al. 2005, 2011; Repo et al. 2011; Smeets and Faaij 2007).

If forest harvesting is expanded to meet the demand for biomass energy, roundwood from standing trees will increasingly be used for energy. For example, short-rotation plantations devoted to biomass feedstock production have been proposed (Fantozzi and Buratti 2010, Tuskan 1998). If prices for biomass energy increase, short-rotation forest crops such as poplars could become a significant feedstock source (Solomon et al. 2007). Carbon emissions from increased use of roundwood for energy may be offset over time by a subsequent increase in forest C. This can be done through increased forest growth on land where the roundwood is harvested. The amount and speed of the offset are influenced by the time period considered, forest growth rate, initial stand C density, and the efficiency with which wood offsets fossil fuel emissions (Schlamadinger et al. 1995). The offset can also be done through increased landowner investment in forestry. The investment can include converting nonforest land to forest, retaining land in forest that would otherwise be converted to nonforest, or planting land in faster growing pulpwood or short-rotation plantations. Forest inventory and C projections for the United States indicate that for scenarios with higher wood energy use (versus those with lower wood energy use) there will be more land retained in forest and more land in plantations for the Southern United States (USDA FS 2012b). The effect on forest C of retaining land in forest is greater than the effect of increasing plantation area. Landowner investment in revenue for biomass is expected to be low for most of the United States.

Reductions in GHG emissions from wood-to-energy pathways depend, in part, on how efficiently wood substitutes for fossil fuels. The energy value of wood (energy content per unit mass) is lower than for fossil fuels (Demirbas 2005, Patzek and Pimentel 2005), a difference that is most pronounced when wood substitutes for fossil fuels with high energy values (e.g., natural gas). The risk of not attaining various levels of offset from use of wood for energy differs, depending on whether biomass is from residues or from greater use of roundwood (Schlamadinger et al. 1995, Zanchi et al. 2010). Risks for using residues are relatively small, especially if forests and supply chains are well managed. Risks associated with using roundwood differ by forest conditions, treatments, and degree of landowner response by investment in more intensive forest management. Large increases in demand could cause loss of C if natural forest with high C density were converted to forest plantations or agricultural biomass plantations with lower C density.

Recent research has provided contrasting conclusions regarding the potential C mitigation benefits from using wood for energy. A number of studies report that using biomass instead of fossil fuels can significantly reduce net C emissions (Boman and Turnbull 1997, Cherubini et al. 2009, Jones et al. 2010, Malmsheimer et al. 2011, Mann and Spath 2001, Spath and Mann 2000). Other studies report that the postharvest regrowth period during which forest C is initially low negates the benefits of wood energy (Bracmort 2011, Cardellichio and Walker 2010, Fargione et al. 2008, Manomet Center for Conservation Sciences 2010, Mathews and Tan 2009, McKechnie et al. 2011, Melamu and von Blottnitz 2011, Melillo et al. 2009, Pimentel et al. 2008, Repo et al. 2011, Schlamadinger et al. 1995, Searchinger et al. 2009). Studies that used life cycle assessments (LCAs) with both biomass pathways and forest C dynamics over time calculated lower reductions in CO_2 emissions than similar LCAs without forest C dynamics. For some cases and time periods, LCAs with biomass pathways and forest C dynamics indicate biomass emissions can be higher than fossil emissions (Johnson 2009, Manomet Center for Conservation Sciences 2010, McKechnie et al. 2011, Pimentel et al. 2008, Searchinger et al. 2008).

These conflicting conclusions are caused by differing assumptions and methods used in the LCAs (Cherubini et al. 2009, 2012; Matthews and Tan 2009). Emerging C accounting methods are increasingly focused on the effect of emissions on the atmosphere and climate over an extended time period, rather than assuming C neutrality (Cherubini et al. 2012). Continuing efforts are needed to provide evaluation frameworks that are adequate to evaluate the overall C and climate effects of specific combinations of forest management and wood energy use.

Mitigation Strategies: Markets, Regulations, Taxes, and Incentives

Forests currently comprise about a third of the land area in the United States, but fragmentation and conversion of forest to other land uses is increasing, especially in the East (Drummond and Loveland 2010). Various mechanisms exist at national, regional, and local scales that can enhance mitigation efforts while providing incentives to keep forests intact. National forests are not eligible for incentive programs or market-based payments for C sequestration or other ecosystem services, but markets and incentive programs can potentially play a role in ecosystem-enhancing mitigation on private and nonfederally owned land. Markets and incentive programs can provide a means for landowners to be financially compensated for voluntary restoration activities that improve ecosystem services. Some of these mechanisms, such as C markets, are designed to encourage mitigation, while other mechanisms help maintain or augment C stores as an ancillary benefit.

Markets, registries, and protocols for forest-based carbon projects—

Carbon markets are an emissions trading mechanism and are typically designed to create a multisector approach that encourages reductions and often (but not always) enhances sequestration of GHG emissions (measured in megagrams of CO_2 equivalent, or CO_2e) in an economically efficient manner. Registries exist to track and account for the C, and protocols outline the specific methodologies that are a prerequisite to creating legitimate C offsets.

The United States does not have a national-level regulatory market, but several mandatory regional efforts and voluntary over-the-counter markets provide limited opportunities for mitigation through forest-based C projects. Offsets generated from these projects can compensate for emissions generated elsewhere. Forest C projects generally take the following form:

Avoided emissions

- Avoided deforestation (or avoided conversion): projects that avoid emissions by keeping forests threatened with conversion to nonforest intact.

Enhanced sequestration

- Afforestation/reforestation: projects that reforest areas that are currently nonforested, but may have been forested historically.
- Improved forest management: projects that offer enhanced C mitigation through better or more sustainable management techniques. These projects are compatible with sustainable levels of timber harvest.
- Urban forestry: projects that plant trees in urban areas. Only sequestered C is eligible (avoided C emissions that result from energy savings are not eligible for credit).

The Regional Greenhouse Gas Initiative (RGGI) is a mandatory multistate effort in New England and the Mid-Atlantic that allows offset credits to be generated through afforestation projects within RGGI member states. The Climate Action Reserve is another mandatory initiative that is based in California but accepts forest projects from throughout the country. In addition, protocols created by the American Carbon Registry, Verified Carbon Standard provide quality assurance to domestic and international forest C projects that may be sold on the voluntary market (Kollmuss et al. 2010, Peters-Stanley et al. 2011). In 2009, 5.1 Mg of CO_2e, or 38 percent of the global share of forest-based C offsets, was generated in North America (Hamilton et al. 2010). However, factors such as substantial startup and transaction costs and restrictions on the long-term use and stewardship of forest land enrolled in C projects often serve as barriers to engagement for many private forest landowners in the United States (Diaz et al. 2009).

Tax and incentive programs—

Some states offer reduced taxes on forest land, as long as certain requirements are met. These tax incentives may be crafted to maintain a viable timber industry and achieve open space objectives, but have the added benefit of helping to maintain or enhance forest C stores. For example, private forest landowners enrolled in Wisconsin's Managed Forest Law Program receive an 80 to 95 percent tax reduction on land that is at least 80 percent forested and is managed for the sustainable production of timber resources. Vermont's Use Value Appraisal Program is similar. Carbon benefits from these programs must be evaluated based on specific circumstances; younger, rapidly growing forests have higher rates of C uptake, whereas older stands may have lower C uptake but higher overall storage (Harmon 2001, Malmsheimer et al. 2008). A "no harvest" unmanaged forest scenario may produce more or less C benefit than a sustainably managed forest, but much depends on current C stocks, the likelihood of disturbance, and whether and how the harvested timber products are used (Ingerson 2007, Nunnery and Keeton 2010). The timeframe of expected C benefits therefore depends on both forest management regimes and forest product pathways (long-term vs. short-term products) (McKinley et al. 2011).

Several federal programs administered by the USDA Natural Resources Conservation Service, Forest Service, and Farm Service Agency provide cost-share and rental payment incentives for good farm, forest, watershed, and wildlife habitat stewardship. As an ancillary benefit, these programs may also help maintain or enhance C stores, but this is currently not an explicit goal of any of these programs. The area enrolled in each program fluctuates annually and depends on commodity prices, program funding, and authorization levels, as sanctioned in the Farm Bill. In 2010, 13 million ha of United States farmland were enrolled in the Conservation Reserve Program, down from 15 million ha in 2005 (Claassen et al. 2008, USDA Farm Service Agency 2010). A brief description of relevant programs is shown in table 4.9.

If policy favored land management that would decrease the buildup of atmospheric CO_2, it might be possible to either fine tune existing incentive programs to more explicitly support C mitigation strategies, and develop an alternative incentive program that prioritizes C management. In the case of the former, the explicit objective of the program could remain as is (to determine general eligibility), but the financial incentives for enrollment could be related to estimated average C benefit per hectare, rather than being calculated based only on hectares enrolled. Carbon benefit per hectare could be estimated at a county or regional scale based on a combination of factors, including geographic location, land use, species planted, and overall landscape connectivity. This may help to ensure that priority lands for C management receive the highest potential benefits. Alternatively, a specific forest C incentive program could complement current incentive programs by targeting small family forest owners and providing financial incentives that may be sufficient to ensure that forests remain as forests. Best management practices could be made available (e.g., for artificial regeneration, thinning, and insect control) (table 4.10), and financial incentives could be based on estimated C benefits (Pinchot Institute for Conservation 2011). These estimated

Table 4.9—Programs that influence carbon mitigation

Program	Agency	Land area	Purpose
		Millions of hectares	
Conservation Reserve Program and Continuous Conservation Reserve Program	Farm Service Agency	~13	Reduce erosion, increase wildlife habitat, improve water quality, and increase forested acres
Environmental Quality Incentives Program (EQIP)	Natural Resources Conservation Service (NRCS)	~6.9	Forest management practices including timber stand improvement, site preparation for planting, culverts, stream crossings, water bars, planting, prescribed burns, hazard reduction, fire breaks, silvopasture, fence, grade stabilization, plan preparation
Conservation Stewardship Program (CSP)	NRCS	n/a	Incentives for sustainable forest management and conservation activities
Wildlife Habitat Incentive Program (WHIP)	NRCS	0.26	Assistance/incentives to develop or improve fish and wildlife habitat, including prairie and savanna restoration, in-stream fish structures, livestock exclusion, and tree planting
Forest Legacy Program	Forest Service (FS)	~0.8	Incentives to preserve privately-owned working forest land through conservation easements and fee acquisitions
Stewardship Program	FS	~14	Encourages private landowners to create and implement stewardship plans on their land

n/a = not applicable.

Table 4.10—Tools and processes to inform forest management

Organization	Relevant content	Internet site
U.S. Forest Service Forest Inventory and Analysis	Forest statistics by state, including carbon (C) estimates	http://fia.fs.fed.us
	Sample plot and tree data	
	Forest inventory methods and basic definitions	
U.S. Forest Service Forest Health Monitoring	Forest health status	http://www.fhm.fs.fed.us
	Regional data on soils, dead wood stocks	
	Forest health monitoring methods	
U.S. Department of Agriculture (USDA) Greenhouse Gas Inventory	State-by-state forest C estimates	http://www.usda.gov/oce/global_change/gg_inventory.htm
United Nations Framework Convention on Climate Change and Intergovernmental Panel on Climate Change	International guidance on C accounting and estimation	http://unfccc.int http://www.ipcc.ch
USDA Natural Resources Conservation Service	Soil Data Mart—access to a variety of soil data	http://soildatamart.nrcs.usda.gov
U.S. Forest Service, Northern Research Station	Accounting, reporting procedures, and software tools for C estimation	http://www.nrs.fs.fed.us/carbon/tools
U.S. Energy Information Administration, Voluntary GHG Reporting	Methods and information for calculating sequestration and emissions from forestry	http://www.eia.gov/oiaf/1605/gdlins.html
U.S. Environmental Protection Agency	Methods and estimates for GHG emissions and sequestration	http://www.epa.gov/climatechange/emissions/usinventoryreport.html

benefits would require only a statistically robust verification of practices rather than annual site monitoring.

The Role of Public Lands in Mitigation

Public lands encompass large areas of forests and rangelands, about 37 percent of the land area of the United States, with federally managed lands occupying 76 percent of the total area managed by all public entities. A decision to manage these lands for C benefits would involve a complex set of interacting forces and multiple jurisdictions, and would be governed by laws mandating multiple uses of land in the public domain. The Council on Environmental Quality (CEQ) has the responsibility of overseeing environmental policy across the federal government. The CEQ has developed draft guidelines for all agencies describing how federal agencies can improve their consideration of the effects of GHG emissions and climate change when evaluating proposals for federal actions under NEPA (Sutley 2010). Another recent policy that affects all federal agencies is Executive Order 13,514 (2009), which requires agencies to set targets that focus on sustainability, energy efficiency, reduced fossil fuel use, and increased water efficiency. In addition, the order requires agencies to measure, report, and reduce GHG emissions from direct and indirect activities, including federal land management practices. The CEQ guidance and these orders are being considered by land management agencies, but it is unclear how effective they will be in reducing GHGs, given the many other uses of federal lands. It should be noted that large areas of forest land protected by conservation organizations (e.g., The Nature Conservancy) across the United States are being managed for public benefits but may not be subject to some of the regulatory issues cited above.

Managing Forests in Response to Climate Change

Managing forests in response to climate change is just one component of the broad and complex task of sustainable natural resource management. Climatic variability (year to year, and decade to decade differences in climate) has always been a factor in forest management, but now resource managers must begin to address directional trends in human-caused climate change in the context of increased variability and movement away from historical averages. Climate change provides a context to be considered in management, but it is rarely appropriate to focus on climate change exclusive of other issues that affect forest resources. An increasing number of potential strategies and forest management options are now available for addressing climate change. However, these strategies and options are rarely institutionalized. Implementing these approaches, or at least a thorough consideration thereof, through planning and management processes on public and private lands is a major organizational and social challenge.

If projected changes in temperature, precipitation, and extreme weather events are realized, management activities that facilitate adaptation to climate change can be realistically viewed as providing additional time until biological and social systems adjust to a new climate. The sooner action is implemented, the more options will be available to prepare forest systems for a new climate. Two major institutional shifts are needed for successful adaptation in U.S. forests. First, scientists and resource managers need to agree that static and equilibrium concepts relative to ecosystem function and management (historical range of variation, restoration of "presettlement" conditions, climax vegetation, etc.) will be less relevant in the future. Ecosystems that exist in nonanalog climates with increased disturbance, new species, and invasive species will rarely be in equilibrium with climate or other environmental factors, and it will not be possible to preserve them intact in a specific location over time. Second, natural resource management organizations will need to consider climate effects as part of normal business operations. If ongoing management protocols and projects include the role of climatic variability and change, then accomplishment targets and on-the-ground practices can be adjusted as needed. This will minimize surprises and lead to realistic long-term planning objectives. If climate effects are not considered, rapid changes in ecosystem dynamics will challenge their ability to manage forest resources sustainably.

As noted in the adaptation section above, adapting to climate change is a viable option for most natural resource management organizations if viewed as adaptive management in the context of climatic variability and change. Currently, most public and private institutions need considerable input from the scientific community to help interpret climate science and model output, and to project the effects of climate change on natural resources at different spatial and temporal scales. Successful science-management partnerships have typically required 2 to 5 years to make substantial progress on science-based solutions to climate challenges. To sustainably manage the Nation's forests, natural resource management organizations will need to make climate change a mainstream issue (much as "ecosystem management," and "ecological restoration" did previously) that can be addressed without continuous high-level, external scientific input.

Multi-institutional collaboration is required, both now and in the future, to apply consistent strategies and tactics across large landscapes. Cooperation among agencies and other organizations in addressing natural resource issues has often been challenging. However, recent efforts between the U.S. Forest Service and National Park Service to collaborate on climate change adaptation, and nascent efforts by U.S. Fish and Wildlife Service Landscape Conservation Cooperatives to instill an all-lands approach in conservation issues, including climate change, provide hope that collaboration will become more common. Perhaps more challenging are the barriers of "paralysis by analysis" within agencies, external litigation, and appeals, which delay timely implementation of projects that can facilitate adaptation. It will be difficult to break the gridlock that seems to envelop public land management in some regions until engagement of stakeholders and consistent, open communication of climate science with the public, policymakers, and land managers becomes commonplace. Climate change is at the forefront of many policy and management discussions on private lands as well (e.g., the Southern Forest Futures Project, http://www.srs.fs.usda.gov/futures), with similar concerns about the effects of climate change on forest lands and potential management options for adaptation and mitigation. In regions dominated by private lands such as the southeast and northeast, dealing with complex ownership patterns and a wide range of management objectives will be critical for successful climate-smart management across large landscapes. Given multiple management objectives and limited funding and staff for implementation, it will be necessary to optimize long-term strategies on a regional to subregional basis by considering where the most benefit can be gained.

Projections of climate change effects are relatively certain for some components of forest ecosystems, and less certain for others, especially beyond the mid-21st century. Developing effective management options to address uncertain, dynamic, and novel conditions will require ongoing monitoring to identify ecosystems at risk, detect change, and evaluate the success or failure of management activities. Now more than ever, land managers will need detailed information on forest conditions to inform management decisions and help adapt to changing conditions. The U.S. Forest Service Forest Inventory and Analysis program and Forest Service Forest Health Monitoring network provide information on changes in forest growth and condition over most of the Nation. In addition, the Forest Service operates 80 experimental forests and ranges that are critical assets for change detection, climate-change experiments, and management demonstrations. Combined with other large networks such as the National Ecological Observatory Network (20 core sites to be established in representative ecoclimate domains), the National Science Foundation Long-Term Ecological Research Program (27 sites located across the United States, Puerto Rico, and Antarctica), National Weather Service weather stations, and numerous U.S. Geological Survey gauging stations, many variables are monitored across a broad geographic area. In most cases, these networks operate independently, and although some lack central data storage, data management protocols, and easy access, efforts are underway to increase data access for many core data sets. These monitoring networks can help detect changes in climate and forest condition, but they are not a substitute for on-the-ground monitoring that will be required to assess the effectiveness of specific management activities. This will require a larger investment by land management agencies, although improved efficiency and coordination can, in some cases, compensate for insufficient funding.

In the near term, it is logical to pursue management strategies that are relatively low cost, have few barriers, and will produce near-term results. For adaptation, this would include reducing co-occurring stressors in forests (e.g., air pollution, exotic pathogens), implementing fuel reduction where feasible and effective, and reducing stand densities where feasible and appropriate (resistance and resilience strategies). For C management, this would include reducing deforestation, increasing afforestation, reducing wildfire severity where feasible, increasing growth, and increasing use of wood-based bioenergy where economically justified.

In the long term, specific adaptation strategies will need to be considered in light of emerging scientific evidence on climate change effects and assessments of the effectiveness of various management actions on the ground. Resilience strategies in the face of increasing large-scale disturbances often include standard management practices (e.g., forest thinning). Specific strategies for C management will need to be guided by emerging scientific evidence on how to concurrently manage forests in situ for products, energy, and other ecosystem services. Strategies will differ by location, inherent forest productivity, and local management objectives. The mandate for productivity on commercial private lands contrasts with objectives on public lands, but both private and public perspectives need to be accommodated in order to manage C across broad spatial scales, meet multiple management objectives, and benefit local economies.

It will also be important to consider how adaptation and mitigation can be coordinated to optimize implementation across specific landscapes. For example, fuel reduction treatments can reduce the prevalence of crown fires in dry forests, while also providing material for local bioenergy use (the long-term effect on C dynamics is site-specific based on current evidence). The interaction of adaptation and mitigation has been poorly assessed to date, and successful models of both strategic and tactical approaches to this interaction are needed. This topic may provide opportunities for coalitions among partners who would not normally collaborate on other natural resource issues. In the near term, we anticipate that federal land management agencies will continue to lead the development of science-management partnerships and collaborative approaches to adaptation and C management across public lands. Successful adaptation and C management will accelerate across larger landscapes if and when community-based partnerships become more engaged with climate change as a component of sustainable resource stewardship.

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Chapter 5
Improving Scientific Knowledge

James M. Vose and David L. Peterson[1]

Scientific literature on the effects of climatic variability and change on forest ecosystems has increased significantly over the past decade, providing a foundation for establishing forest-climate relationships and projecting the effects of continued warming on a wide range of forest resources and ecosystem services. In addition, certainty about the nature of some of these effects and understanding of risk to biosocial values has increased as more evidence has been accrued.

The recent expansion in scientific analysis of the effects of climate on ecological disturbance has provided empirical data on how wildfire and insects respond to warmer climatic periods. However, more information is needed on the interaction of ecological disturbances and other environmental stressors, especially for large spatial and temporal scales. Thresholds for climatic triggers of environmental change are generally poorly understood relative to fire, insects, interactions, and functionality of forest ecosystems. Moreover, simulation modeling can suggest how and when those thresholds might be exceeded, additional empirical data on thresholds will be more definitive, and more process-level research is required to improve current or the next generation of predictive models. In general, our understanding of stress complexes in forest ecosystems needs to be expanded to more ecosystems and transitioned from qualitative to quantitative descriptions.

Despite a century of ecological research on human-altered landscapes, our ability to interpret ecological change in the context of human land use and social values is far from complete. We especially need to improve our ability to quantify climate-ecosystem relationships in the context of land use change at larger spatial and temporal scales. Inferences about climate change effects will be more relevant if various land uses, including evaluation of future alternatives, are considered in a context that incorporates humans, rather than excluding them or considering their actions to be "unnatural" or negative. This leads to the broader need to develop a framework for quantifying ecosystem services that is transportable across different institutions and that will include a wide range of biosocial values.

Some general scientific issues need additional focus. First, the value and interpretation of empirical (statistical) models versus process (mechanistic) models warrants a rich discussion within the scientific community. Conceived from different first principles (e.g., assumed equilibrium [empirical] vs. dynamic [process] climate-species relationships), the output from these models often differs considerably or is difficult to reconcile because of different assumptions, spatial resolution, and hierarchical levels (e.g., species vs. life form) between the models. This disparity needs to be resolved so that resource managers can understand and apply model output appropriately. Second, the direct effects of elevated carbon dioxide (CO_2) on forest ecosystems need to be clarified. Most existing evidence is based on experimental treatments on seedlings and small trees, and on simulation models that assume certain types of growth responses. Assuming CO_2 stimulation (or not) can drive the output of vegetation effects models to such an extent that it greatly modifies simulated response to climate. A unified effort by scientists to resolve the significant challenges in scaling and interpreting data on CO_2 effects is needed to provide accurate projections of vegetation change. Third, effects models that can explore multicentennial patterns of vegetation distribution, disturbance, and biogeochemical cycling dynamics would provide more realistic scenarios for planning and policy decisions. Most projections of climate change effects extend to only 2100, the limit of projections for most global climate models, and a relatively short time for robust evaluations of ecosystem dynamics.

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Some specific research priorities for forest ecosystems include:

- Develop and implement long-term studies on the effects of elevated CO_2 in mature forests. This may involve whole forest stands or physiological measurements of individual trees within stands. Studies in disparate forest ecosystems would provide a broad perspective on this topic.
- Develop a standard approach for tracking carbon dynamics in different forest ecosystems over space and time. This will improve ecological knowledge, as well as input to carbon accounting systems. It will be especially useful if it can be applied in a straightforward manner by resource managers.
- Identify the appropriate uses and limitations of remote sensing imagery for detecting the effects of climatic variability and change in forest ecosystems. A great deal of remote sensing data are available, but they are accessible to only a few specialists. If resource managers were provided tools to access, analyze, and help interpret the most reliable and relevant data, it would provide timely feedback on forest stress and other characteristics on a routine basis.
- Determine which ongoing and long-term forest measurements are useful or could be modified for tracking the effects of climate change. Building on existing infrastructure for monitoring will be efficient and extend time series of measurements taken with established protocols.
- Identify standard approaches for evaluating uncertainty and risk in vulnerability assessments and adaptation planning. Straightforward qualitative and quantitative frameworks will advance the decisionmaking process on both public and private lands.
- Evaluate recently developed processes and tools for vulnerability assessment and adaptation planning to identify which ones are most effective for "climate smart" management on public and private lands. The availability of straightforward social and logistic protocols for eliciting and reviewing scientific information and stakeholder input will make climate change engagement more effective and timely.

It will be especially important to frame the above topics at the appropriate spatial and temporal scales in order to provide relevant input for different climate change issues. In addition, climatic data at different spatial scales needs to be matched with applications at different spatial scales to be relevant for climate smart management. Despite the urgency to provide downscaled climatic and effects data, the appropriate grain and extent of these data differ by resource (hydrology vs. vegetation vs. wildlife) and resource use (timber management vs. water supply vs. access for recreation). Sharing of information and experience within and among organizations involved in climate change will accelerate the incorporation of proven methods and applications across any particular landscape.

Chapter 6

Future Assessment Activities

Toral Patel-Weynand[1]

Introduction

Climate change science has progressed significantly since the first National Climate Assessment (NCA) was produced (National Assessment Synthesis Team 2001). The ability to project climatic regimes and effects on forest ecosystems has increased through improved models and scaling techniques, as well as a combination of experimental studies and field observations that have either validated expected responses or challenged conventional thinking. However, as noted in previous chapters, critical information gaps exist in our ability to project how forest ecosystems will respond to the direct and indirect effects of climate change. Ongoing research is addressing many of these knowledge gaps, although the complexity of some scientific issues makes it clear that management and policy decisions over the next several years will continue to be made based on imperfect information.

By as early as the mid-21st century, a warmer and more variable climate, along with interacting stressors, will challenge the ability of public and private land managers to manage forest resources. In response, large-scale adaptation and mitigation strategies and tactics will need to be developed and applied across the United States to ensure the sustainability of ecosystem services in a changing climate. Engagement will be required in both biophysical and socioeconomic research to make viable options available to manage re-sources that are being affected by climatic variability and change at various spatial and temporal scales.

The 2013 NCA and the technical products that federal agencies and others are providing to the NCA are taking the first steps to help improve nationwide climate assessment capabilities in an integrated fashion. In the forest sector, a number of issues have emerged that need attention from national stakeholders and federal and state agencies, as well as from a resource management perspective. The current NCA approach is more focused than past climate assessments in supporting the Nation's activities in adaptation and mitigation, and in evaluating the current state of scientific knowledge relative to climatic effects and trends. It advocates a long-term, consistent process for evaluating climatic risks and opportunities and for providing information to support decisionmaking processes. The U.S. Global Change Research Program and NCA are working toward establishing a permanent assessment capacity both inside and outside of the federal government.

The NCA plans to have assessment activities draw upon the work of stakeholders and scientists across the country as an ongoing and continuous process. Assessment activities will support the capacity to conduct ongoing evaluations of vulnerability to climate stressors, observe and project effects of climate change within regions and sectors, allow for the production of a set of reports and Web-based products that are relevant for decisionmaking at multiple levels, and develop consistent indicators of progress in adaptation and

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mitigation. The NCA is working to make products of this process useful within management and policy contexts. It is expected that an ongoing NCA process will be established and sustained through a cooperative community-wide effort that incorporates federal, state, and local governmental agencies, nongovernmental organizations, academia, and tribal and private interests. The long-term objective of the process is to enhance coordination of climate assessment efforts and facilitate communication between stakeholders and data providers.

Regional Issues

Several large-scale emerging issues identified by the forest sector will need attention beyond the 2013 NCA effort. Ecological disturbance, invasive species, urban forests, forest conversion to other uses, and fragmentation will need ongoing research and monitoring so adaptation options can be developed and evaluated. Additional effort is also needed to better understand how climate change will affect ecosystem services, human health, water and watersheds, energy and bioenergy, carbon (C) sequestration, and forest industry viability. Many organizations are working to identify potential vulnerabilities and effects, along with adaptation options to address them, but few analyses and interventions are set within a risk-based framework. Developing a risk-based framework to assess climate-related changes is therefore a critical need for the future.

Developing a Risk-Based Framework

The NCA provided a simple set of guidelines on how to use a risk-based framework for technical input products for the 2013 report (Yohe and Leichenko 2009). The guidelines are based on the risk and uncertainty framework developed by the Intergovernmental Panel on Climate Change (Moss and Schneider 2000). Risk assessment is also being incorporated in other national and state climate change management efforts. For example, all four National Research Council panel reports of “America’s Climate Choices” incorporate this framework, as does the draft Adaptation Plan for the United States. Although incorporating risk throughout a technical input product is challenging, the NCA recommends that this framework be incorporated at least for key vulnerabilities. Key vulnerabilities are those with a large magnitude, early timing, high persistence and irreversibility, wide distributional aspects, high likelihood, and high importance (based on human perceptions) (Schneider et al. 2007). All characteristics do not necessarily apply to a key vulnerability, and findings that may have a low likelihood but high consequence are still of interest to the NCA audience because of their high risk.

A risk management framework for natural resources identifies risks and quantifies the magnitude and likelihood of environmental and other effects to the extent possible. Although risk management frameworks have been used (often informally) in natural resource management for many years, it is a new approach for projecting climate-change effects, and some time may be needed for both scientists and resource managers to feel comfortable with this approach. Risk assessment for climate change should be specific to a particular region and time period, and needs to be modified by an estimate of the confidence in the projections being made. Further work is needed to refine and expand existing risk management frameworks to better address climate change vulnerabilities and potential effects.

Social Issues

The complexities of human behavior and social vulnerability, value and significance of forests, and their joint sensitivity to climate change argue for social science research and community involvement when planning for, managing, and communicating about climate change in the forest sector. Research indicates that place matters, the planning or decisionmaking process matters, and original, specific, and local solutions may be best. However, a consistent and logical framework is needed to quantify ecosystem services across different forested landscapes, communities, and management institutions and to incorporate a wide range of biophysical and social values.

Management Options

The scientific literature is growing, and on-the-ground activities are underway, but no standard evaluation framework exists to aid decisions about effective management approaches—encompassing both biophysical and social processes—for adapting to or mitigating climate change. A framework is needed that will incorporate elements such as local forest productivity, management objectives, and socioeconomic conditions.

Identifying areas where forests are most vulnerable to change (i.e., have low resistance and resilience) and where the effects of change on ecosystem services will be greatest is a significant challenge for resource managers. One would expect forest ecosystems and species near the limits of their biophysical requirements to be vulnerable, but the complexities of fragmented landscapes and multiple stressors are likely to change response thresholds in many forest ecosystems. Under these conditions, traditional approaches to forest management are likely to fail. Management approaches that anticipate and respond to change by guiding development and adaptation of forest ecosystem structure and function will be needed to sustain desired ecosystem services and values across large landscapes and multiple decades. Land managers who are currently managing forest ecosystems in a sustainable manner are probably already using "climate smart" practices, and implementation of climate smart management at all spatial scales and by a variety of organizations (federal agencies, private land owners, conservation groups) can affect long-term resilience and sustainability. However, a systematic effort to communicate and implement these experiences more extensively is needed.

Ecological Disturbances and Extreme Events

Climatic variability is a driver of regionally episodic fires and endemic insect outbreaks; therefore, "new" science on climate and ecological disturbances is principally concerned with quantifying the mechanisms and variability in relationships between climate and ecological disturbance. Relationships between pathogens and climate change are not as well understood, but it is plausible that higher stress in tree species will reduce forest vigor and increase mortality. From an ecosystem perspective, thresholds can be reached either through cumulative effects of individual disturbances over time or one large event, and can lead to new forest composition, land cover, and landscape patterns. However, more information is needed on the interaction of ecological disturbances and other environmental stressors, especially for large spatial and temporal scales.

Climate affects forests through extreme events (e.g., hurricanes, wildfire, etc.) and through enabling conditions (e.g., long warm or cool periods, and long wet or dry periods). These events or enabling conditions can have short-term effects on forests, after which there is a transition or recovery, followed by long-term outcomes. Management can primarily respond to the enabling conditions by building resilience, as well as facilitating the transition or recovery. Are we prepared to confront and respond to climate-related forest changes within the context of forest management? The answer lies in our ability to recognize potential loss, quantify risk, examine options, identify tradeoffs, anticipate rare but high-consequence events, and invest commensurate with risk. The challenge before us will require new tools, information, and technology, as well as the experience of resource managers.

Coordination With Other Assessment Activities

Evaluating the future of forests requires understanding of human behavior in the context of a changing climate. Recently, the U.S. Forest Service conducted an assessment of current and future forests and rangelands, as required by the Forest and Rangeland Renewable Resources Planning Act (RPA) of 1974, which mandates that current conditions, trends, and forecasts for the next 50 years be assessed. Recent changes to the assessment mandated by the RPA include (1) presenting conditions, trends, and forecasts in a global context, (2) utilizing global climate models (three were used) and emission scenarios (e.g., A1B, A2, and B2), and (3) integrating the analysis with socioeconomic factors (e.g., wood product markets and the price of timber, and agricultural markets and the future of crop prices). The RPA assessment indicates that forest area in the United States peaked in 2010 at 253 million ha and will likely decline through 2060 to between 243 and 247 million ha. Product markets, population, income, and climate all interact to determine future forest area, biomass, and forest C. Climate will influence the outcomes, and although significant variation exists across potential climate futures, it is still small relative to human factors in the short run.

Effects on Tribal Lands

American Indian and Alaska Native tribes face disproportionate risks from climate change. Tribes have unique rights, cultures, economies, and vulnerabilities to climate change effects. For indigenous peoples, the effects of climate change and the proposed solutions may affect tribal subsistence, land rights, cultural survivability, and financial resources. Tribes recognized a critical need for coordination among public agencies and organizations in accessing climate change resources and information, and in 2009, the Tribal Climate Change Project (University of Oregon 2012) was established to determine the needs, lessons learned, and opportunities American Indian tribes have in planning for the effects of climate change. Key research areas for the project are (1) increased understanding of tribal adaptation and mitigation planning for the physical effects of climate change, (2) increased understanding of management of off-reservation resources, and (3) government-to-government relationships in addressing climate change through consultation, cross-landscape assessments, and tribal involvement in federal and state climate change plans. These types of research efforts will continue to be important for filling critical knowledge gaps.

Carbon Estimation

Through the U.S. Forest Service Forest Inventory and Analysis (FIA) program, C accounting is accomplished by first estimating land use and then estimating forest biomass. Forest C flux is approximated as a change in forest stocks over time. The FIA is primarily a large-scale inventory, and climate change would need to have a significant effect on net forest growth for FIA data to detect it. Accounting for C and managing ecosystems raises significant questions because of the uncertainty in how C pools will change with climate. Thus, management will require an integrated approach to mitigation and adaptation at large spatial scales. Avoiding deforestation and increasing afforestation can be recommended in the near term, whereas application of improved forest management across various regions may cause C losses in some locations and gains in others. Applying climate smart management at the stand scale is important but will be more effective in a broad landscape context.

Carbon mitigation can also be assessed through life cycle analysis**.** A recent synthesis of findings about the mitigation effectiveness of alternate forest management and wood use options concluded: “In the long term, sustainable forest management strategy aimed at maintaining or increasing forest carbon stocks, while producing an annual yield of timber, fiber, or energy from the forest, will generate the largest sustained mitigation benefit” (Metz et al. 2007). This raises the questions: “Which forest management and wood use strategies yield the greatest offset, in the near term and long term?” and “How confident are we in gains from those actions?” Afforestation and avoided deforestation are approaches with the highest confidence (lowest uncertainty) for providing C mitigation. Other approaches for which moderate uncertainty exists about effective mitigation include decreasing harvest, increasing forest growth, reducing hazardous fuels, using wood for energy, and substituting wood for nonwood products. Additional investigation is needed for all these topics. Life cycle evaluations of management and wood use options suggest more intensive approaches to wood production, harvest, and use to maximize C mitigation. The nature of future C markets, especially a regulated C market versus a voluntary market, will affect participation and influence wood product markets. Participation by private land owners may depend on management objectives and type of ownership (e.g., small vs. large properties). The motivation of private corporate entities relative to wood and C management, if surveyed accurately, will also provide important insights.

Conclusions

Abundant data exist on the climatic, physical, ecological, and social aspects of how forests and forestry may respond to climate change, but the synthesis and integration of these for identifying adaptation options and making decisions are limited by (1) an inability to respond rapidly to new information such as projections of future climate; (2) social, political, and economic forces that affect the structure and function of forest ecosystems and their management; and (3) inadequate resources for synthesis and integration, particularly for adaptation and mitigation options and consequences. Stakeholders provided diverse recommendations for preparation of the forest sector technical report for the NCA, and most stakeholders emphasized that connections among various biophysical and social factors are not well understood or easily modeled. For example, the effects of climate change on wildfire can be partially mediated by fuels management, which in turn has a set of cascading effects on other forest processes and values, depending on the efficacy and intensity of management.

A number of periodic assessments by the Forest Sector are relevant to the NCA request for delivery of interim products between the 2013 and the 2017 NCA reports. For example, required sustained efforts such as the RPA assessment and periodic efforts such as the National Sustainability Report (USDA FS 2011) can provide integrated national-scale information pertinent to the issues discussed here. However, a concerted effort will be necessary to identify these products and make them available to resource managers and decisionmakers.

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Chapter 7
Conclusions

David L. Peterson and James M. Vose[1]

Introduction

Forest ecosystems in the United States in the year 2100 will differ from those of today as a result of a changing climate. Those differences will be superimposed on the human imprint of forest management and the legacies of other land use activities, stressors, and disturbances of the 19th and 20th centuries. Future changes in forest ecosystems will occur across both public and private lands and will challenge our ability to manage forests sustainably, especially as the human population continues to grow, demands for ecosystem services increase, and fossil fuel supplies decrease. We summarize below the most important inferences from the preceding chapters, with emphasis on issues most relevant to land managers.

Forest Disturbance

Although increases in temperature, changes in precipitation magnitude and seasonality, higher atmospheric carbon dioxide (CO_2) concentrations, and higher nitrogen (N) deposition may over time modify ecosystem structure and function, the fastest and most significant effects on forest ecosystems will be caused by altered disturbance regimes. A warmer climate will increase the area burned by wildfire and the area affected by bark beetles and other insects. These two factors, individually, in combination, and as components of broader stress complexes, may lead to permanently altered species composition, distribution of forest age and structure, and spatial patterns across large landscapes.

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An increase in wildfire throughout the United States, which will likely include at least a doubling of area burned by the mid-21st century, will challenge government agencies and social institutions. Fire directly affects human communities near wildlands, but it is also stretching the ability of federal and state agencies to pay for fire suppression. Expanded efforts to reduce hazardous fuels can reduce the severity of wildfire on a local basis, but if the current investment in reducing stand densities and fuels does not increase significantly, it will be impossible to mitigate the effects of increasing crown fires.

The current advance of bark beetles in forests throughout the Western United States and Canada is unprecedented and often affects more land area than wildfire on an annual basis. Similar to wildfire, insects cause a rapid change in forest structure and function but with a slower return of carbon (C) to the atmosphere. Insects appear to affect fire severity in some cases and are a component of stress complexes that include prolonged drought. The prospect of bark beetles affecting higher elevations and different tree species than in the past portends major changes in forest ecosystems previously considered unaffected by beetles. Reducing stand densities and improving stand vigor can reduce impacts at the local scale, but it will be challenging to implement effective mitigation across large landscapes. A strategy of modifying the spatial pattern of age and structure in forests affected by beetles may provide some hope for controlling the spread of insects in the long term.

Invasive plants are another important component of stress complexes throughout the United States, and although the exact trajectory of this stressor in forest ecosystems is difficult to project, invasive plant species will likely become more numerous and widespread in the future. Many invasive species are more competitive in a warmer environment with elevated CO_2. However, increased disturbance from fire, insects, and land use change are among the most important factors facilitating their dispersal and population growth. This risk may be highest in mountain ecosystems, where cooler temperatures have historically limited the spread of invasives.

Geomorphic disturbance will also increase if storms become more intense as is projected by many climate models. Concentrating precipitation in shorter periods of time increases erosion and mass wasting during and following storms, and increases the duration and intensity of low soil moisture (drought) during the rest of the year. Pulses of erosion and movement of sediment into streams are difficult to predict, but if they do indeed increase, they will affect decisions about management of roads and other infrastructure, as well as access for users of forest land. Increased drought will exacerbate stress complexes with insects and fire, leading to increased tree mortality, slow regeneration, and changes in species assemblages at some forest ecotones.

Forest Processes

Stabilizing greenhouse gas concentrations in the atmosphere is a major goal for slowing global warming and buying time for implementation of alternative energy strategies and adaptation of forest ecosystems and human institutions to climate change. Forest growth and afforestation in the United States currently account for a net gain in C storage and offset approximately 13 percent of the Nation's fossil fuel CO_2 production. Overall, forest area has been stable since 1950, while C density (C per unit area) has increased. This assimilation of C is a function primarily of forest regrowth following timber harvest and land clearing in the previous two centuries and is projected to continue to around 2040, at which point U.S. forests could become a net emitter of C. The majority of this C is in live aboveground biomass and soil organic C, so anything that affects these two components will significantly affect total C storage. During the next few decades, Eastern forests are expected to continue to sequester C through favorable response to elevated CO_2 and higher temperature, while Western forests may begin to emit C through expanded fire and insect disturbance. At large spatial and temporal scales, reduced in forest land cover may offset some of the C gains expected in Eastern forests.

No standard evaluation framework exists to aid decisions about which management approaches—encompassing both biological and social processes—would be most effective in maximizing C storage (reducing emissions) while minimizing risks. However, five approaches guide strategic and tactical management of forest C: (1) increase forest area and avoid deforestation, (2) manage C in existing forests, (3) use wood as biomass energy, (4) use wood in place of other building materials, and (5) use wood products for C storage. These approaches differ considerably based on local forest productivity, management objectives, and economic conditions.

No-regrets strategies for enhancing C storage include preventing conversion of forest land to other uses and extending the life cycle of wood products. Avoided deforestation protects existing forest C stocks with low risk and has many co-benefits, although incentives to avoid deforestation in one area may increase removal of forest in other areas, as well as decrease economic opportunities for timber, agriculture, and urban development. Evidence for the benefits of fuel treatments (thinning plus surface fuel treatment) for C storage is equivocal, and the value of C offsets would be higher if thinning material had higher commercial value as long-lived products that yield substitution benefits and not just as bioenergy. The benefit of stored C in wood products is multiplied when wood is used in place of materials that require much higher C emissions to produce (e.g., concrete and steel). Careful management of forest products has potential for C mitigation that accrues over time and complements strategies for increasing forest C stocks, but effective strategies need to ensure that energy offsets are attained in an acceptable period of time and that substitution effects are attained.

The effects of climate change on water resources and biogeochemical cycling will differ by forest ecosystem and local climatic conditions, as mediated by local management actions. Large-scale disturbances such as fire, bark beetle outbreaks, and defoliating insects will reduce water uptake, causing a near-term increase in runoff and potentially erosion. In systems with a long regeneration time, as in low-elevation forests and woodlands of the Southwest, erosion may be high for years to decades following disturbance. Increased temperature during the past few decades has decreased snow cover depth, duration, and extent, a trend that will likely continue with further warming. Decreased snow cover will alter the seasonal timing of runoff and exacerbate soil moisture deficit in some forests, which may decrease tree vigor and increase susceptibility to insects and pathogens. In addition, fuels may remain dry and flammable for a longer period of time, leading to higher fire hazard and a longer period of time during which wildfires will burn. Less snow and drier fuels may also extend the time during which prescribed burning can be conducted, a potential benefit to resource managers.

Elevated CO_2 may increase the water use efficiency of some tree species, thus reducing evapotranspiration, but the effect on hydrologic dynamics will likely be modest. Warmer temperature may also modify tree phenology, although the effects on evapotranspiration are uncertain. If species and genotypes that grow fast are widely planted in the future, their demand for soil water could reduce streamflow in some locations. Warmer temperature may also accelerate the rate of nutrient cycling in some systems, promoting increased forest growth and elevated N levels in streams.

Species Distributions

It has been difficult to infer if changes in forest species distribution and abundance are occurring in response to climate change, partly because of the lack of long, high-quality time series on species distribution, and partly because the legacy of widespread land use actions are so persistent in most landscapes. Most models predict that suitable habitat for many species will move upward in elevation and northward in latitude and be reduced or disappear from current habitats in lower elevations and lower latitudes. This is supported by both process-based and empirical modeling, although the different assumptions and resolutions of the models lead to rather different spatial and temporal inferences about habitats and species. It is possible that new climatic conditions will "move" faster in some locations than tree species can disperse, creating uncertainty about the future vegetation composition of these new habitats. It is also possible that topographic diversity, and thus microclimatic diversity, in mountainous regions will be sufficient to support most current species but with different spatial distributions and abundances. Despite the uncertainty of current modeling, the paleoecological literature suggests that major changes in species distribution and abundance, often mediated by disturbance, are possible with small but persistent changes in temperature and precipitation.

Risk and Social Context

The Intergovernmental Panel on Climate Change (IPCC) and U.S. Global Change Research Program (USGCRP) currently emphasize that risk and uncertainty should be clearly articulated in order to provide a realistic context for interpreting scientific data and inferences. Risk assessment considers both the magnitude of a particular climate-change effect and the likelihood that it will occur. A risk management framework for natural resources means that risks are identified and that magnitude and likelihood of effects are quantified to the extent possible. Although risk management has been used (often informally) in natural resource management for many years, it is a new approach for projecting climate-change effects, and some time may be needed for both scientists and resource managers to feel comfortable with it. Risk assessment for climate change should be specific to a particular region and time period, and needs to be modified by an estimate of confidence in the projections being made.

The IPCC and USGCRP also emphasize that climate-change effects need to be considered in light of ecosystem services provided to local communities and human enterprises. Climate-change effects in forests are likely to reduce ecosystem services in some areas and increase them in others. Some areas may be particularly vulnerable because current infrastructure and resource production are based on past climate and steady-state conditions. For example, increased fire and insect attacks will, at least temporarily, reduce productivity, economic benefits from timber harvest, and C storage, and, in some cases, will increase surface runoff and erosion. In this case, potential losses of resource value and economic value are large, exclusive of the huge economic cost of fire suppression that may be required. Any change in forest ecosystems that affects water resources will result in a significant loss of ecosystem services.

Preparing for Climate Change

Federal agencies, and the U.S. Forest Service in particular, have made significant progress in developing scientifically based principles and tools for adapting to climate change. Adaptation builds on a sequence of activities that starts with education, continues with an assessment of vulnerability of natural resources to climate change, and culminates in development of adaptation strategies and tactics. This process is most effectively conducted through a science-management partnership in which scientists lead the education and vulnerability assessment phases, and resource managers provide most of the input for adaptation. Tools and techniques available to facilitate this process are readily available in recent materials developed by the Forest Service, and can be applied to both public and private lands. In addition, several case studies of adaptation for national forests and national parks, individually and in collaboration with other stakeholders, are now available and can be emulated by other land management organizations. Collaboration across multiple land ownerships over large landscapes will ultimately lead to the most effective adaptation strategies and plans.

Although uncertainty exists about the magnitude and timing of climate change effects on forest ecosystems, sufficient scientific information is available to begin taking action now. However, on-the-ground implementation of adaptation plans and carbon management are rare in both public and private forest sectors. This is due to a perceived lack of urgency, a limited number of personnel trained in climate change science, inadequate guidelines and protocols, and inadequate resources to implement another "unfunded mandate."

Fortunately, land managers who are currently managing forest ecosystems in a sustainable manner are often already using "climate smart" practices. For example, thinning and fuel treatments implemented to reduce fire hazard also reduce intertree competition and increase resilience in a warmer climate. Increasing culvert size under roads reduces the risk of damage to roads and downstream resources that may occur in response to higher flood frequency and magnitude. Building on practices compatible with adapting to climate change will provide early successes and experience for resource managers who may want to start the adaptation process but do not have sufficient money, time, or personnel to initiate a major effort. We anticipate that climate change will be a standard component of sustainable resource management by the end of the decade, and that C management and adaptation will be fully embraced by forest management organizations. Building the foundation for this new management context as soon as possible will ensure that a broad range of options will be available for managing forest resources sustainably.

Appendix 1: Regional Summaries

Alaska

Jane M. Wolken and Teresa N. Hollingsworth[1]

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).

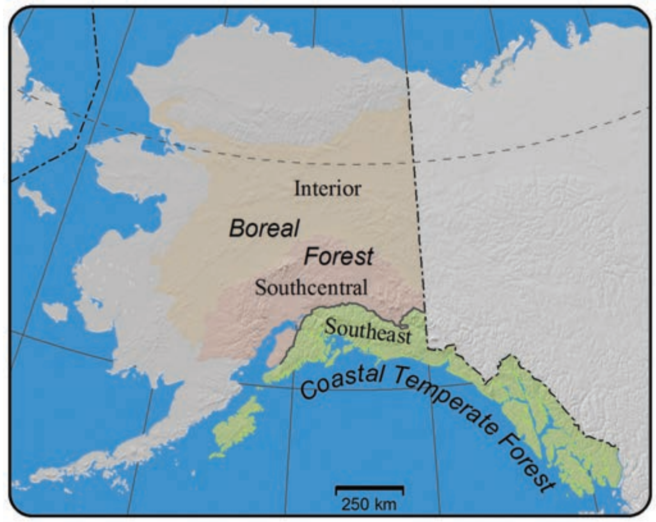

Figure A1-1—The boreal (interior and south-central) and coastal-temperate (southeast) forest regions of Alaska. The boreal forest ecoregions include the Alaska Range transition, coastal mountains transition, Pacific mountains transition, Kenai boreal, and intermontane boreal; the coastal-temperate forest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

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 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

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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.

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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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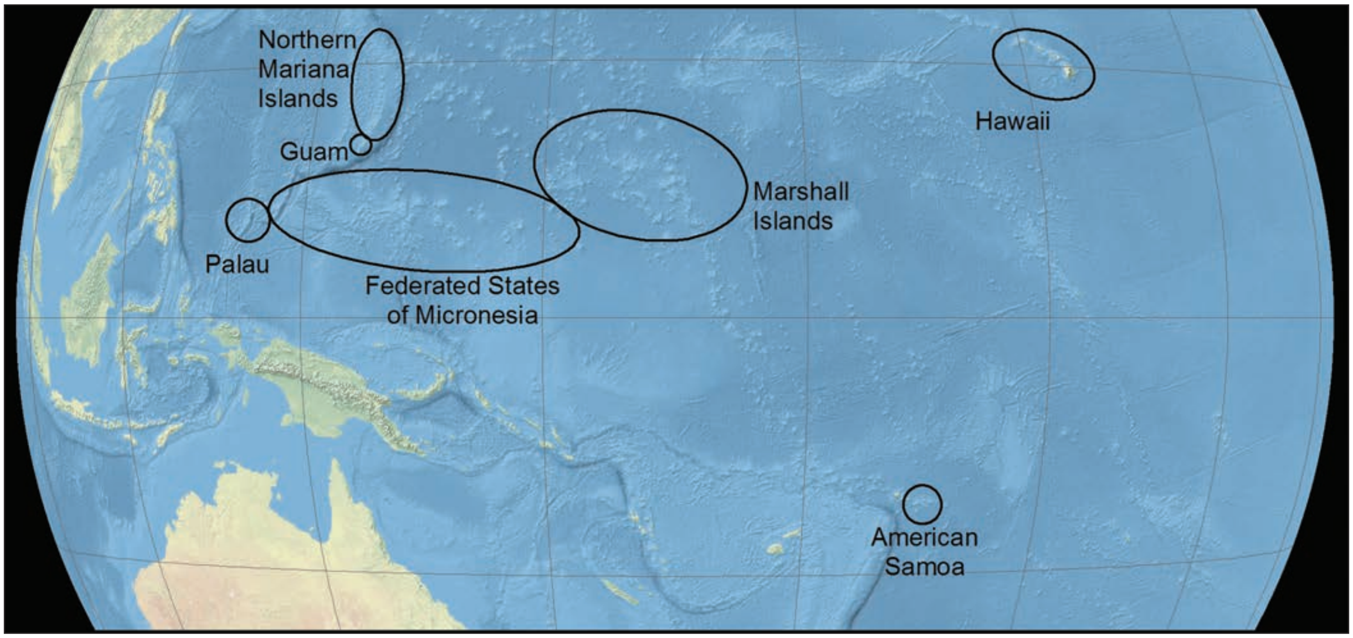

Figure A1-3—Hawaii and the U.S.-affiliated Pacific islands.

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 $mm \cdot yr^{-1}$) (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.

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.

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.

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.

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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 (Schafer 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 (*Dendroctonous 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.

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Southwest

David L. Peterson[1]

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).

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

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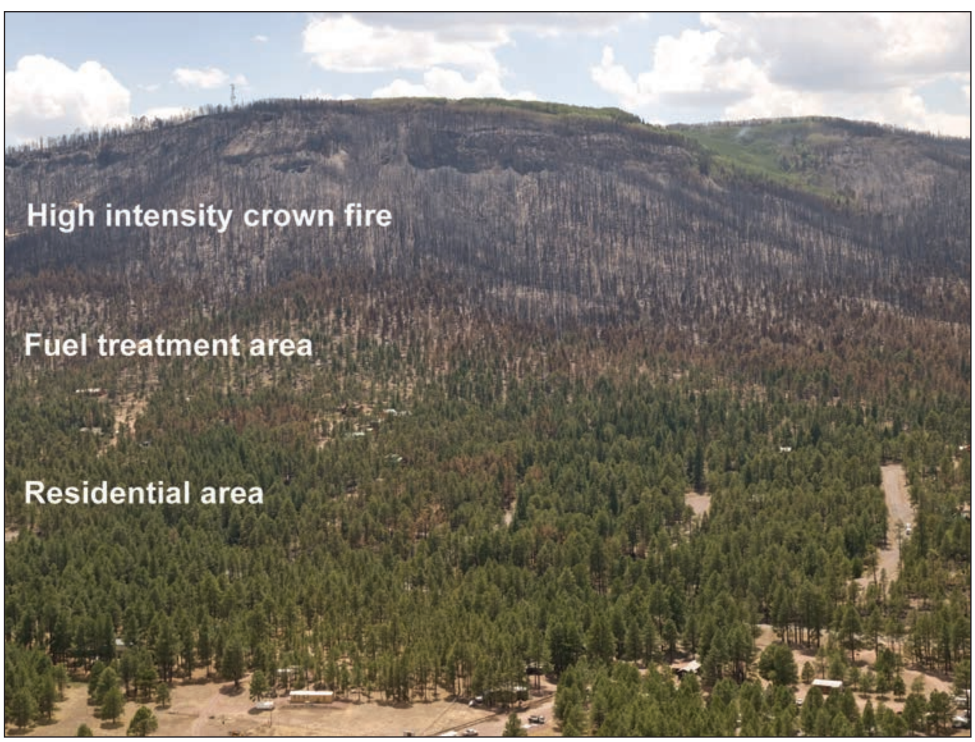
[2] Littell, J.S. Relationships between area burned and climate in the Western United States: vegetation-specific historical and future fire. Manuscript in preparation. On file with: U.S. Geological Survey, Alaska Climate Science Center, 4210 University Drive, Anchorage, AK 99508.

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.

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Great Plains

Linda A. Joyce[1]

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
				Thousands of hectares				
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 industial. 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 (CO_2)on grassland, Wyckoff and Bowers (2010) analyzed the relationship between historical climate and tree growth and suggest that the interaction of climate change and elevated CO_2 could be a potential factor in the expansion of forests from the Eastern United States into the Great Plains.

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Midwest

Christopher W. Swanston and Stephen D. Handler[1]

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.

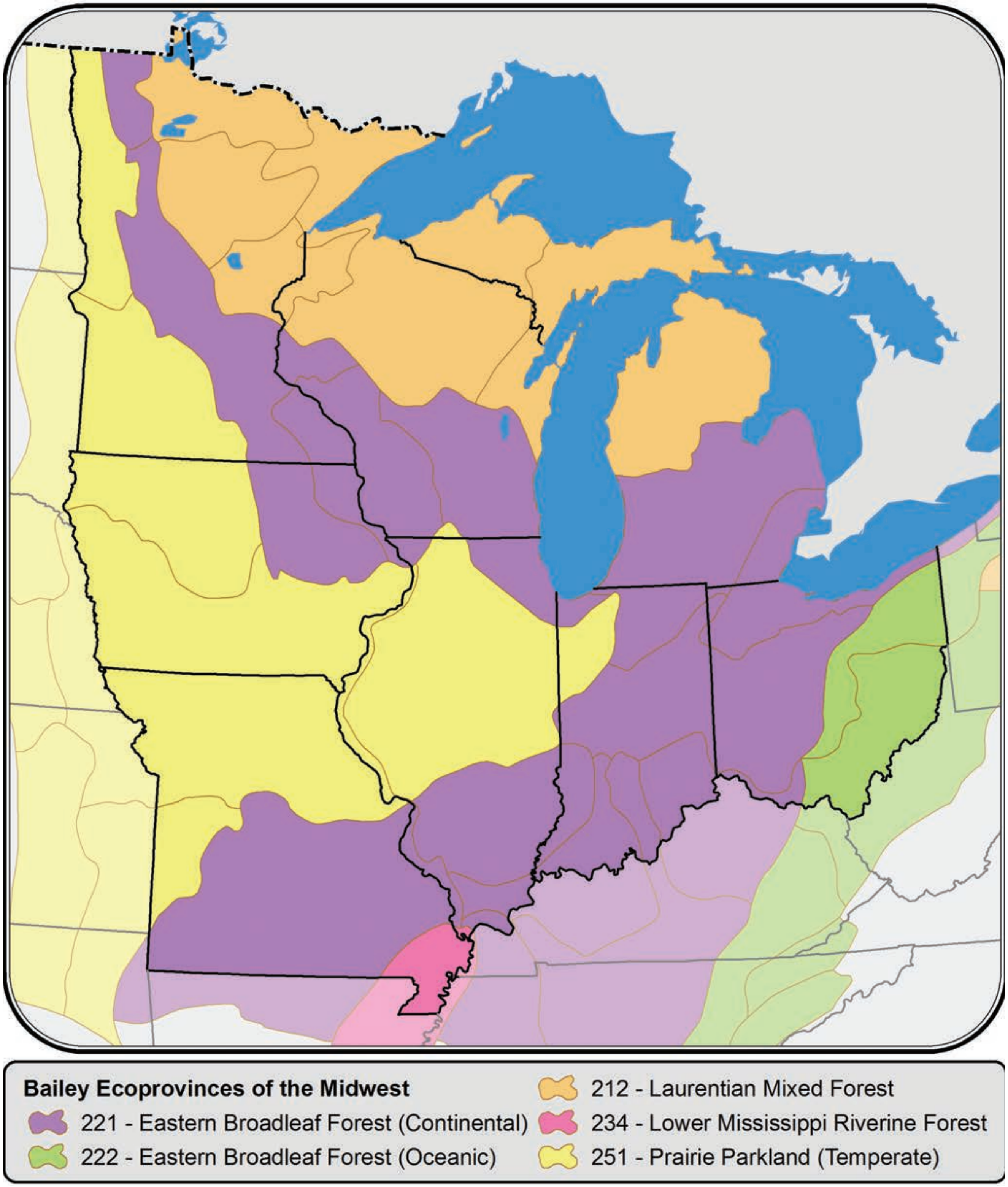

Figure A1-6—Ecoregions within the Midwest, according to Bailey et al. (1995).

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, Mehmood and Zhang 2001). Fostering climate preparedness as a component of sustainable land stewardship will require significantly increased outreach and coordination to communicate relevant and credible information to private forest landowners. Conversely, inadequate attention to land stewardship will place this forest sector at greater risk of avoidable impacts of climate change.

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Northeast

Lindsey E. Rustad[1]

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. Paleoecological 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).

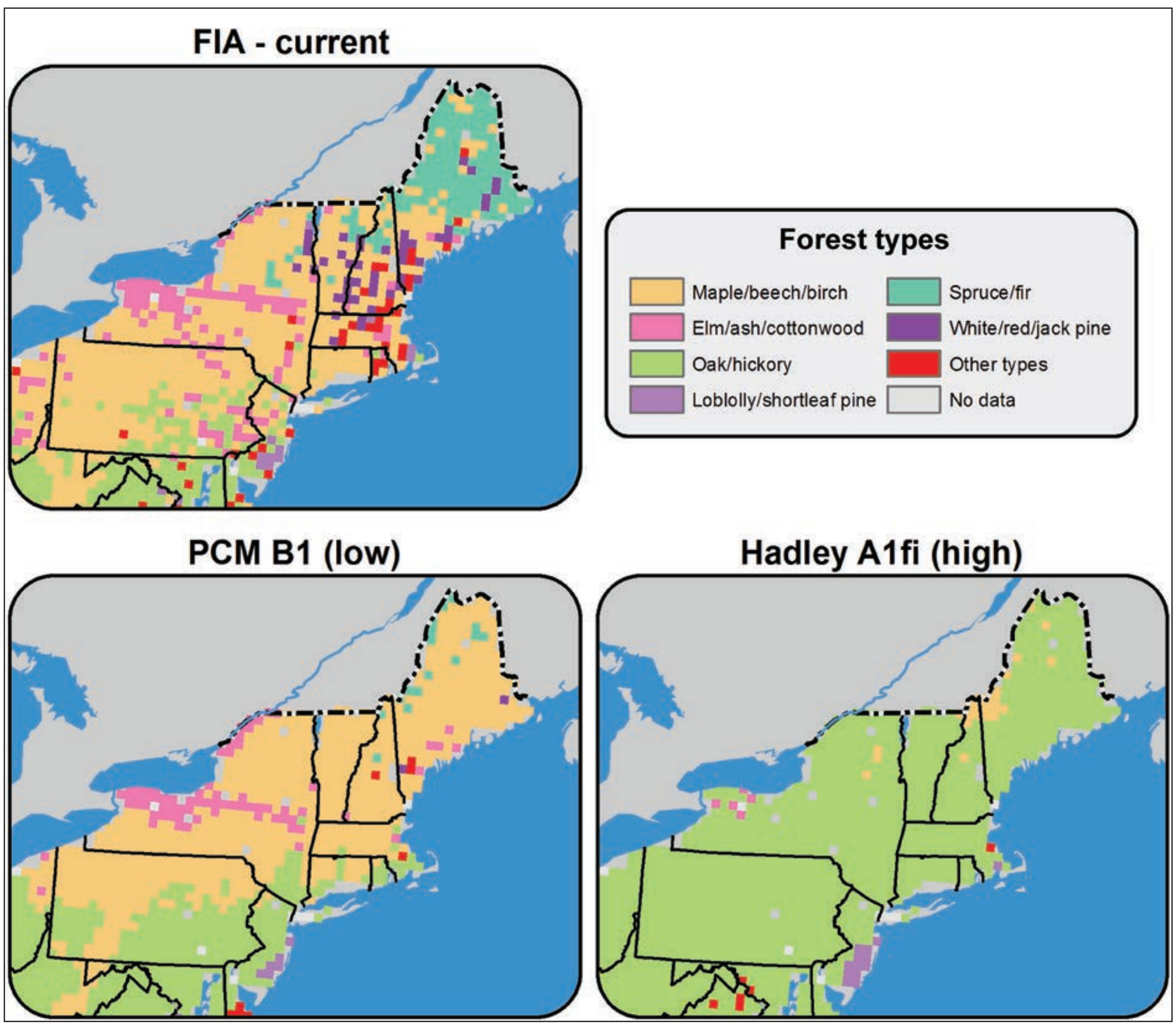

Figure A1-7—Suitable habitat for forest vegetation in New England is expected to shift with changes in climate associated with different emissions scenarios. From Mohan et al. (2009).

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 (CO_2). For example, Ollinger et al. (2008) used the model PnET-CN to project that net primary productivity in deciduous forests would increase by 52 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 CO_2 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* Viellot) (-42 to -73 percent), and rose-breasted grosbeak (*Pheucticus ludovicianus* L.) (-23 to -71 percent). Species such as Bicknell's thrush

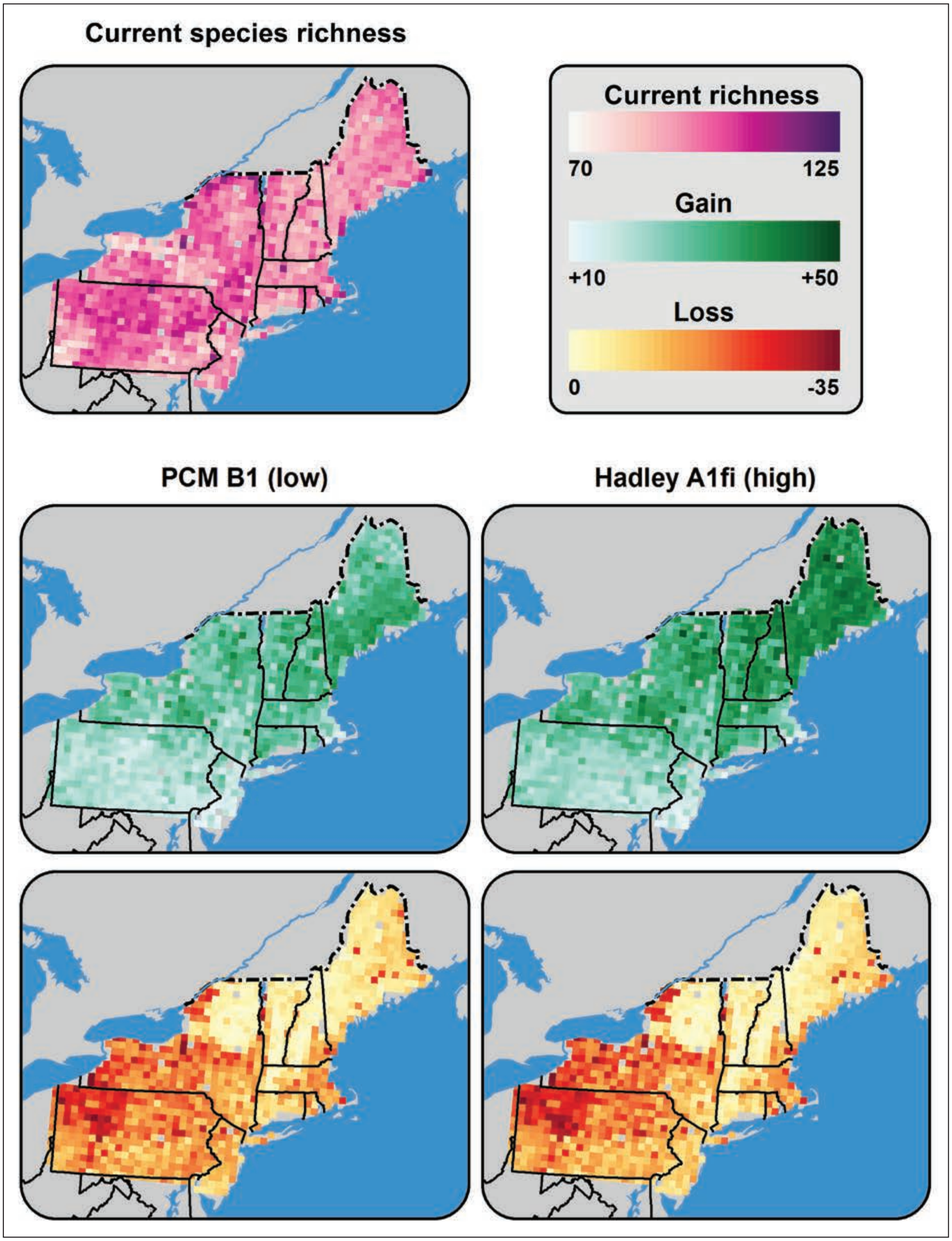

Figure A1-8—Climate change is expected to affect bird species richness more intensely in some areas of the Northeastern United States than in others. From Rodenhouse et al. (2008).

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

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Southeast

Steven G. McNulty[1]

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).

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 (CO_2) than wildfires, at least in the short term (Wiedinmyer and Hurteau 2010). Historically, longleaf pine was a dominant species across the region. It is well adapted to drought, with thick bark and fast seedling growth, allowing it to thrive in habitats subjected to periodic wildfire (Brockway 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

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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

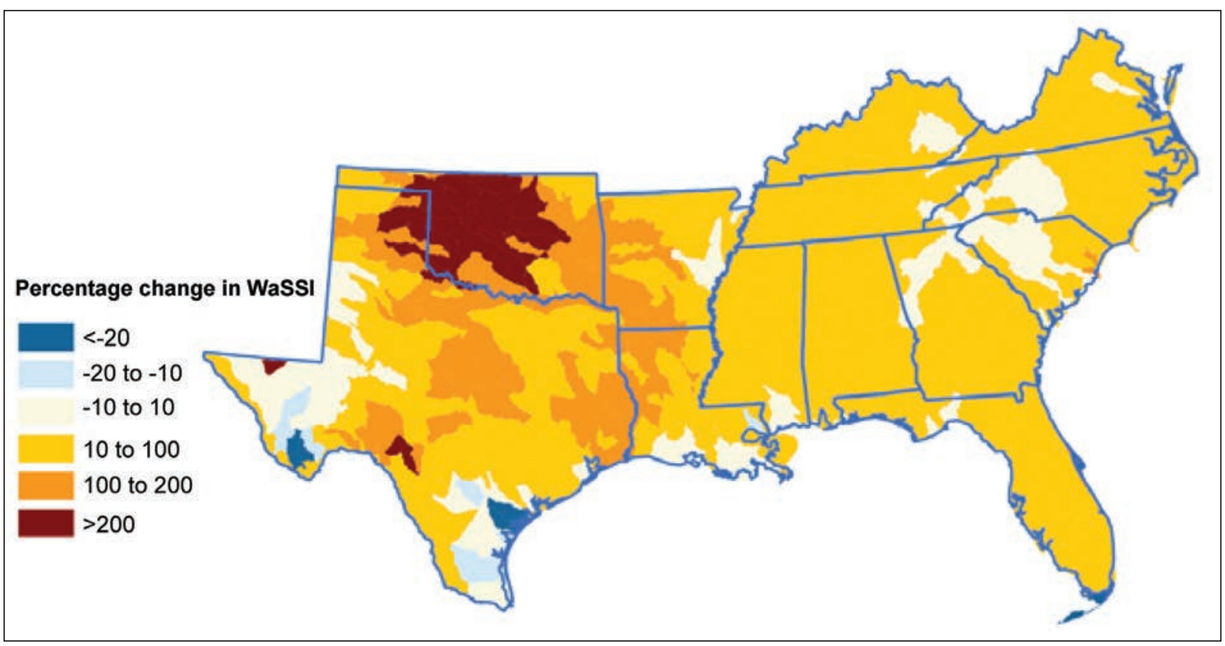

Figure A1-9—Percentage change in water supply stress owing to climate change, as defined by the water supply stress index (WaSSI) for 2050 using the CSIROMK2 B2 climate scenario. WaSSI is calculated by dividing water demand by supply, where higher values indicate higher stress on watersheds and water 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 (Sohngen et al. 2001).

Carbon sequestration is an increasingly valued component of forest productivity. Globally, forests sequester up to 16 percent of all the CO_2 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 CO_2 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.

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Appendix 2: Risk-Based Framework and Risk Case Studies

Risk-Based Framework for Evaluating Changes in Response Thresholds and Vulnerabilities

Dennis S. Ojima, Louis R. Iverson, and Brent L. Sohngen[1]

What is "risk," and how can a "risk-based framework" help plan for climate change? Risk is described by the likelihood of an impact occurring and the magnitude of the consequences of the impact (Yohe 2010) (fig. A2-1). High-magnitude impacts are always risky, even if their probability of occurring is low; low-magnitude impacts are not very risky, even if their probability of occurring is high. Applying this approach to forest management is difficult because both the likelihood of occurrence and the magnitude of the effects may be difficult to estimate (especially at local scales) and often depend on past and current land use, and the timing, frequency, duration, and intensity of multiple chronic and acute climatic disturbances.

Despite these challenges, there is much that we do know and it is possible to begin thinking about approaches for developing a risk-based framework for forests in the context of climate change. A risk management framework simply means that risks are identified and estimates are made for their probability of occurrence and their impact. Where we have sufficient knowledge, this framework provides a means to quantify what is known, identify where uncertainties exist, and help managers and decisionmakers develop strategies with better knowledge of risks.

Climate change is likely to affect forest ecosystems, and the risk of negative consequences to forests and associated socioecological systems will probably increase (Ryan and Archer 2008). However, predicting these risks is difficult because of uncertainty in almost all aspects of the problem.

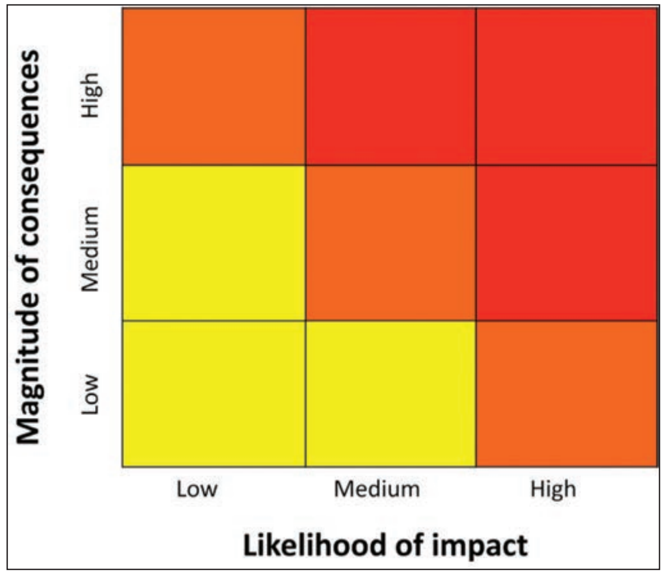

Figure A2-1—A conceptual risk framework used to help identify risks associated with climate change and prioritize management decisions (Yohe and Leichenko 2010). Colors represent varying degrees of risk (red = highest; yellow = lowest). In a qualitative definition of consequence, low = climate change is unlikely to have a measurable effect on structure, function, or processes within a specified timeframe (e.g., 2030s, 2050s, 2090s); medium = climate change will cause at least one measurable effect on structure, function, or processes within a specified timeframe; and high = climate change will cause multiple or irreversible effects on structure, function, or processes within a specified timeframe. In a qualitative definition of likelihood, low = climate change impacts are unlikely to be measurable within the specified timeframe, medium = climate change impacts are likely to be measurable within the specified timeframe, and high = climate change impacts are very likely (or have already been observed) within or before the specified timeframe.

How can we incorporate uncertainty into an analysis of risks and subsequent management decisions?

Regional and local projections of climate change are uncertain (Baron et al. 2008, Fagre et al. 2009, Joyce et al. 2008). Despite these uncertainties, climate science has advanced to provide a set of robust climate change projections: the climate is warming, the probability of large precipitation events is increasing, seasonal patterns will be altered, and

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extreme events are more likely (Solomon et al. 2007). These tendencies are becoming more apparent in observations across the United States and will affect forest resources nationwide (Karl et al. 2009). A key challenge is to determine how climate change may alter local socioecological systems, trigger threshold-dependent events, and create nonlinear interactions across interconnected stressors on forest resources (Allen et al. 2010, Fagre et al. 2009), and further, how climate change effects can be addressed by local management actions. Forest managers have extensive experience adapting forest management practices to climate variability and a wide range of disturbance regimes. For example, conifer plantations are often managed in short rotations, which limits exposure to risks from insects, wildfires, and windstorms. In mixed-age hardwood forests where management is often less intensive (e.g., where partial harvests are the norm), managers simultaneously choose trees to remove and trees in the understory to release for the next generation of growth. Hence, by using silvicultural techniques to select the species, density, and age class distribution of the next generation of forest, managers can influence susceptibility to a range of future threats.

Given what we do know about climate change, a robust decisionmaking approach is needed that acknowledges sources of uncertainty, incorporates what is known of system vulnerabilities, and evaluates assets critical for making sound forest management decisions (Australian Government 2005, Baron et al. 2008, Fagre et al. 2009, Joyce et al. 2008, Ranger and Garbett-Shiels 2011). A risk management approach provides a robust framework for planning management options for climate change, where uncertainties are recognized and key elements relative to various management objectives and priorities are explicitly addressed (Dessai and Wilby 2011, McInerney and Keller 2008, Ranger and Garbett-Shiels, 2011, Yohe and Leichenko 2010). This approach incorporates aspects of vulnerability assessments, identifies priority actions relative to multiple management goals, identifies critical information needs, and provides a vision of short- and long-term strategies to enhance the flexibility of management decisions and reduce the probability of poor decisions (Australian Government 2005, Peterson et al. 2011). This approach also promotes a shift from reactive adaptation to proactive adaptation and coping management (Ranger and Garbett-Shiels 2011), including the following general strategy:

- Identify actions to avoid, that is, avoid choices that lead to less flexibility to adjust to changing conditions in the future.
- Implement "no regrets" management to cope with stresses now and increase resilience to anticipated climate-related stresses.
- Make decisions that integrate across landscapes and governance and that include all concerned and affected stakeholders.
- Develop activities that have strong links among observations, research, and management to understand how ecosystems and social systems are changing, help make decisions, understand thresholds, and help adjust future management and research.

The risk framework must consider the socioecological context of the system being evaluated, reflecting the contribution of forest ecosystem services to different communities and the capability of forest systems to withstand different climate effects. Providing a more thorough consideration of sources of uncertainty allows for improved development of management strategies, which include key socioeconomic properties. This integrated and multisectoral approach will incorporate an improved assessment of risk and current management capacity, and will identify critical uncertainties that may exist under future scenarios if novel consequences emerge.

Case studies using a risk-based framework and concepts are discussed in the following sections on carbon, fire, forests, and birds. They are intended as examples, using different approaches to convey risk assessment, and will hopefully create interest by scientists and land managers in refining risk assessments for the effects of climate change on a wide range of forest resources.

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Risk Case Study: A Framework for Assessing Climate Change Risks to Forest Carbon Stocks

Christopher W. Woodall and Grant M. Domke[1]

Forest ecosystems have the ability to reduce the effects of climate change through the sequestration of carbon (C) (Pan et al. 2011) as well as contribute to net emissions through disturbance events such as wildfires and widespread tree mortality (Kurz et al. 2008). A conceptual framework for assessing climate-change risks to forest ecosystem C stocks facilitates efficient allocation of efforts to monitor and mitigate climate-change effects, and the U.S. National Greenhouse Gas Inventory (NGHGI) of forest C stocks (Heath et al. 2011) may be used as a basis for development of a climate-change risk framework for forest C stocks.[2]

A forest C stock risk framework incorporates two components of risk: consequence and likelihood (fig. A2-2). One of the most critical future consequences of climate change on forest C stocks is the shift from C sink (net annual sequestration) to C source (net annual emission). Although global forests currently sequester more carbon than they emit on an annual basis (Pan et al. 2011), the ability of forests to continue this trend in the future has been questioned (Birdsey et al. 2006, Reich 2011). If the strength of the C sink decreases and forests became net emitters of C and other greenhouse gasses (GHG) (e.g., methane) a positive feedback loop may be created whereby negative climate change effects may further exacerbate forest C emissions. Likelihood can be phrased as the probability of a C stock becoming a net emitter of C. For individual C stocks that are least affected over short timespans (e.g., 50 to 100 years), likelihoods would be minimal. Taken together, the C risk framework hinges on the concepts of both a "status change," in which forest C stocks transition between C source or sink, and a "tipping point", when forest systems might collapse with concomitant emission of C and potential positive feedbacks that may exacerbate climate change.

The consequences of a C stock becoming a net emitter of C is postulated as being directly related to its population estimate over a region of interest. In this case study, it is the C stocks of individual forest pools for the entire United States as reported to the Intergovernmental Panel on Climate Change to meet United Nations Framework Convention on Climate Change requirements (USEPA 2011a, 2011b). If a pool is the largest in the United States, then that pool has the largest consequence on global climate change if it is entirely emitted. All current U.S. forest C stocks represent nearly 25 years of U.S. GHG emissions at current emission rates (Woodall et al. 2011). The pools and estimates (Tg of C) of C stocks in 2008 (Heath et al. 2011) are ordered as: soil organic carbon (17,136 Tg of C), aboveground live biomass (16,854 Tg of C), forest floor (4,925 Tg of C), belowground biomass (3,348 Tg of C), and dead wood (3,073 Tg of C).

The likelihood of any individual C stock becoming a net emitter of C is an emerging area of research. For the purposes of this risk framework (fig. A2-2), it is proposed that the likelihood of a C stock becoming a net emitter is related to the empirical variation in the stock across the diverse ecosystems and climates of the United States. If climate change occurs such that a mesic boreal forest ecosystem becomes a xeric mixed-hardwood shrubland, then the contemporary range in variation in C stocks between those systems should indicate likelihood of C emission. For example, if forest floor C stocks change minimally regardless of climate, then in turn climate change should least affect these stocks. As an initial appraisal of empirical variation in C stocks across the United States, the coefficients of variation (percentage) of individual plot-scale measurements of C stocks (Forest Inventory and Analysis; Heath et al. 2011) across the United

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States are ordered as dead wood (126.9 Tg of C), belowground biomass (107.8 Tg of C), aboveground live biomass (104.5 Tg of C), forest floor (73.7 Tg of C), and soil organic carbon (67.6 Tg of C). Although climate change events can alter natural variation in C stocks, when compared to contemporary levels, these estimates of variation can provide a starting point for a risk framework.

When the consequences and likelihoods of forest C stocks becoming net emitters of C are viewed together, a cohesive approach to monitoring and managing risk emerges. Given the magnitude of potential emissions coupled with the natural variability in these stocks at the continental scale, annual monitoring of dead wood and aboveground live biomass C stocks are needed. In addition, strategies to mitigate negative climate change events (e.g., droughts) can be undertaken. The major research gap in such an approach is how far a pool would move within the risk framework after a climate change event (the length and direction of the negative/positive arrows [fig. A2-2]). For example, if forest lands convert to grasslands as a result of reduced precipitation and lack of tree regeneration, how would the aboveground biomass pool align itself within the risk framework? Despite the qualitative nature and research gaps within the forest C stock risk framework, this approach provides a conceptual means of identifying priority research needs and a decision system for mitigating climate change events.

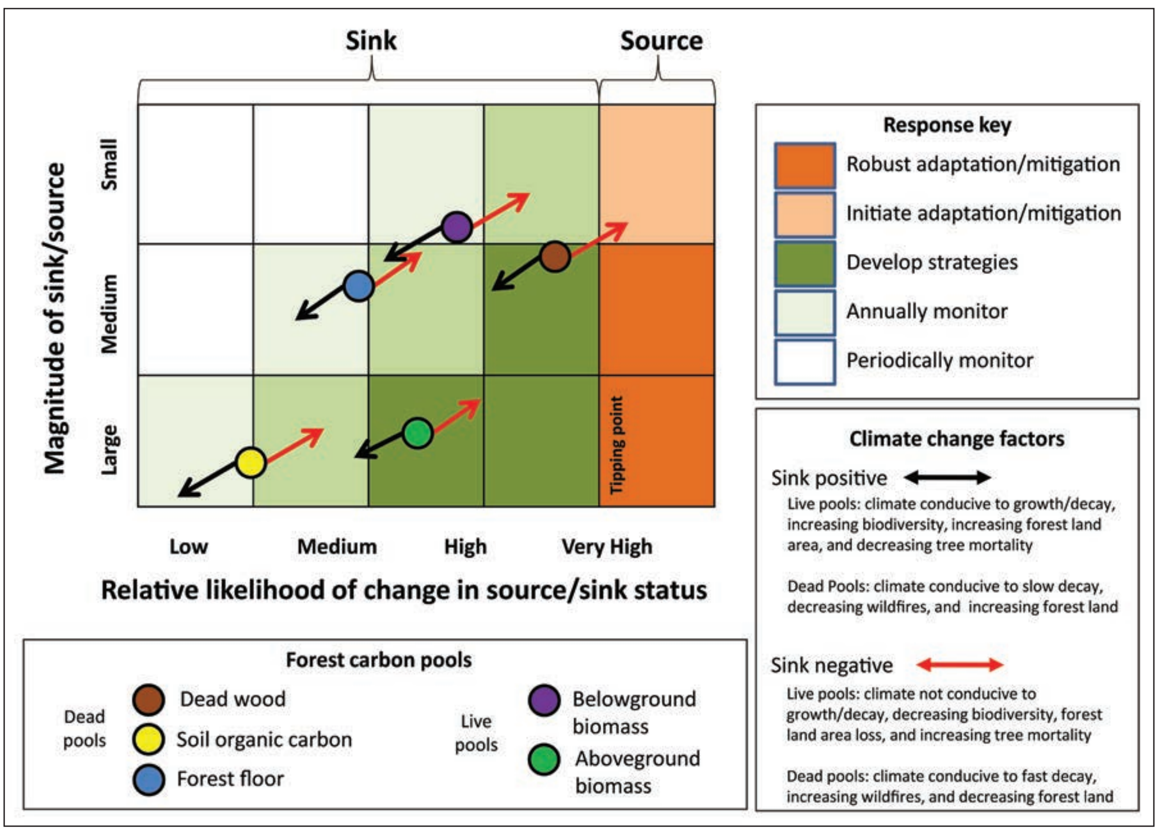

Figure A2-2—Climate change risk matrix for forest ecosystem carbon (C) pools in the United States, in which climate change may cause C pools to move in a positive (sink = net annual sequestration) or negative (source = net annual emission) direction. Likelihood of change in C stocks is based on the coefficient of variation across the national Forest Inventory and Analysis plot network (x-axis). Size of C stocks is based on the U.S. National Greenhouse Gas Inventory (y-axis). Societal response (e.g., immediate adaptive response or periodic monitoring) to climate change events depends on the size and relative likelihood of change in stocks. The dead wood pool, a relatively small stock, exhibits increasingly high variability across the landscape and therefore may be affected by climate change and disturbance events such as wildfire. In contrast, the forest floor is also a relatively small C stock, but has low variability. Potential future climate-change effects are not incorporated in the matrix, because they represent many complex feedbacks both between C stocks (e.g., live aboveground biomass transitioning to the dead wood pool) and the atmosphere (e.g., forest floor decay).

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Risk Assessment for Wildfire in the Western United States

David L. Peterson and Jeremy S. Littell[1]

Introduction

Wildfire is one of the two most significant disturbance agents (the other being insects) in forest ecosystems of the Western United States, and in a warmer climate, will drive changes in forest composition, structure, and function (Dale et al. 2001, McKenzie et al. 2004). Although wildfire is highly stochastic in space and time, sufficient data exist to establish clear relationships between some fire characteristics and some climatic parameters. An assessment of wildfire risk in response to climate change requires brief definitions of the terms "fire hazard" and "fire risk," which are often confused in the scientific literature and other applications (Hardy 2005). Fire hazard is the potential for the structure, condition, and arrangement of a fuelbed to affect its flammability and energy release. Fire risk is the probability that a fire will ignite, spread, and potentially affect one or more resources valued by people. The most common means of expressing wildfire risk are (1) frequency, (2) a combination of intensity (energy release) and severity (effects on forests, structures, and other values), and (3) area burned.

Fire Frequency

Fire frequency, which is the number of fires for a particular location and period of time, differs by region as a function of both lightning and human ignitions, with the requirement that fuels are sufficiently dry and abundant to burn. Lightning ignitions dominate mountainous regions with convective weather patterns (e.g., most of the Rocky Mountains), whereas human ignitions dominate regions with little lightning and high human populations (e.g., southern California). Modeling studies (+4.2 ºC scenario) (Price and Rind 1994) and empirical studies (+1.0 ºC scenario) (Reeve and Toumi 1999) suggest that lightning frequency will increase up to 40 percent globally in a warmer climate. Although no evidence exists to suggest that recent climate change has yet caused an increase in lightning or fire frequency in the West, lightning may increase as the temperature continues to rise (Price and Rind 1994, Reeve and Toumi 1999). Assuming that human population will increase throughout the West, it is reasonable to infer that human ignitions will also increase in most regions. Even if the sources and numbers of potential ignitions do not change, a warmer climate may facilitate increased drying of fine surface fuels (less than 8 cm in diameter) over a longer period (on a daily and seasonal basis) than currently exists (Littell and Gwozdz 2011), allowing more potential ignitions to become actual ignitions that will become wildfires.

Fire Intensity

Fire intensity, or energy released during active burning, is directly proportional to **fire severity** in most forests, and can be expressed as effects on vegetation, habitat, and, in some cases, human infrastructure. Results of modeling based on a doubled carbon dioxide (CO_2) emission scenario suggest that fire intensity will increase significantly by 2070 in the northern Rocky Mountains, Great Basin, and Southwest (Brown et al. 2004). Fire severity and biomass consumption have increased in boreal forests of Alaska during the past 10 years (Turetsky et al. 2010), and large, intense fires have become more common in California (Miller et al. 2008) and the Southwestern United States during the past 20 years. However, interannual and longer term variability in climate-fire relationships can affect trends, making it difficult to infer whether climate change is responsible. Longer time series of fire occurrence, when available, will allow better quantification of the influence of multidecadal climatic variability (e.g., the Pacific Decadal Oscillation or Atlantic

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Multidecadal Oscillation). Fire intensity and severity are a function of both climate and land use history, especially the effects of fire exclusion on elevated fuel loads, and forests with high fuel loading will continue to be susceptible to crown fire in the absence of active management (see below).

Fire Area

Fire area has a stronger relationship with climate in the Western United States than does either fire frequency or severity/intensity. An empirical analysis of annual area burned (1916 to 2003) for federal lands in the West projected that, for a temperature increase of 1.6 °C, area burned will increase two to three times in most states (McKenzie et al. 2004). In contrast, a complex, mechanistic model projected that, for the same temperature increase, area burned will increase by only 10 percent in California (Lenihan et al. 2003). Using the 1977 to 2003 portion of the same data set used by McKenzie et al. (2004), Littell et al. (2009) stratified fire area data by Bailey's ecoprovinces (Bailey 1995) to account for fire-climate sensitivities. On average, the model explained 66 percent of the variability in historical area burned by combinations of seasonal temperature, precipitation, and Palmer Drought Severity Index. In most forest ecosystems and some woodlands, fire area was primarily associated with drought conditions, specifically, increased temperature and decreased precipitation in the year of fire and seasons before the fire season. In contrast, in arid forests and woodlands in the Southwest, fire area was influenced primarily by the production of fuels in the year prior to fire and secondarily by drought in the year of the fire.

Littell et al.[2] projected the statistical models of Littell et al. (2009) forward for a 1 °C temperature increase, calculating median area burned and probabilities that annual fire area would exceed the maximum annual area burned in the historical record (1950 to 2003). Fire area is projected to increase significantly in most ecoprovinces (fig. A2-3); probability of exceeding the historical maximum annual burn area varied greatly by ecoprovince (range 0 to 0.44). For the Pacific Northwest, the projected increases in area burned from Littell et al. (see footnote 2) are consistent with those found by Rogers et al. (2011) using the MC1 simulation model. A weakness of the statistical models is that, if the projected increased area burned were sustained over several decades, then at some point the large areas burned and decreasing fuel loads would result in less area burned than projected by the models. Neither statistical nor process-based models can satisfactorily account for the effects of extreme fire years and biophysical thresholds that may be exceeded in a much warmer climate.

Conclusions

Based on information summarized above and on expert judgment of the authors, the effects of climate change on fire risk are summarized for fire regimes that occur in forests of the Western United States (table A2-1). We estimate risk for a 2 °C increase, which is more likely by mid-21st century than the more conservative temperature scenarios used by McKenzie et al. (2004) and Littell et al. (see footnote 2). All fire regimes in forest ecosystems would experience some increase in fire risk. Low-severity and mixed-severity fire regimes dominate dry forest ecosystems of the West and

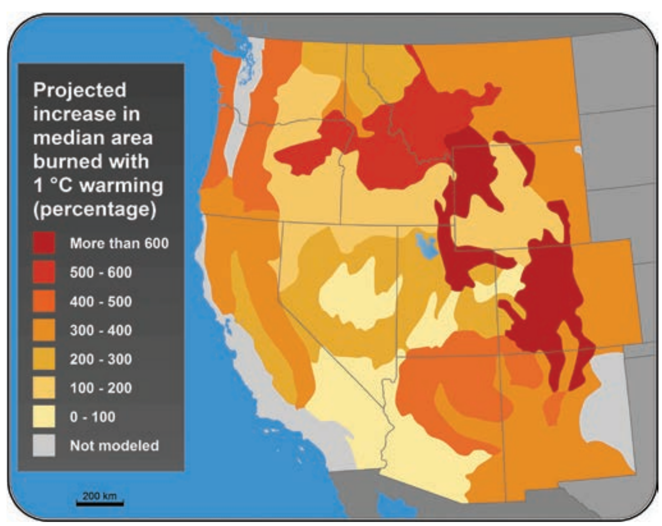

Figure A2-3—Percentage of increase (relative to 1950 to 2003) in median area burned for Western United States ecoprovinces for a 1 °C temperature increase. Color intensity is proportional to the magnitude of the projected increase in area burned.

[2] Littell, J.S. [N.d.]. 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. Department of the Interior, Geological Survey, Alaska Climate Science Center, 4210 University Drive, Anchorage, AK 99508.

Table A2-1—Likelihood and magnitude of increased wildfire risk for fire regimes in forests of the Western United States, based on a temperature increase of 2 °C [a] [b]

	Fire regime			
Risk parameter	**Low severity**	**Moderate severity**	**High severity**	**Rationale for risk ratings**
Frequency:				
Likelihood	Moderate	Moderate	Moderate	More fires will occur in all forests because of longer fire seasons and higher human population. In low-severity systems with low fuel loads, more fires will maintain resilience to fire and climate change; in low-severity systems with high fuel loads, more fires will cause more crown fires. In moderate-severity systems, more fires could convert them to low-severity systems. In high-severity systems, even a small increase in fire frequency will have a large effect on forest structure, function, and carbon dynamics.
Magnitude	Low	Moderate	High	
Overall risk and potential action	Low; no action recommended	Moderate; encourage fire prevention in high population areas	Moderate; encourage fire prevention in high population areas	
Intensity/severity:				
Likelihood	Moderate[c]	Moderate	Low	In low-severity and high-severity systems, fire intensity and severity will probably be higher because of more extreme fire weather and elevated fuel loads for the next few decades. In high-severity systems, fuel moisture, not quantity, is limiting, so intensity and severity will not change much; crown fires are always intense and kill much of the overstory.
Magnitude	Moderate[c]	Moderate	Low	
Overall risk and potential action	Moderate; increase fuel treatment area and fuel removal	Moderate; increase fuel treatment area and fuel removal	Low; no action recommended	
Area burned:				
Likelihood	High	High	Moderate	All fire regimes will experience more area burned. This will be especially prominent in drier, low-severity and moderate-severity systems. In high-severity systems, more area will burn, but the percentage increase will be less than in other systems; this will have significant local ecological effects.
Magnitude	High	Moderate	Moderate	
Overall risk and potential action	High; greatly increase fuel treatment area, allow some fires to burn	Moderate; increase fuel treatment area, allow some fires to burn	Moderate; no action recommended	

[a] Risk ratings are qualitative estimates based on information summarized above and on expert judgment of the authors.
[b] Fire regimes are defined as (1) low severity: 5- to 30-year frequency, less than 20 percent overstory mortality (dry mixed-conifer forests and woodlands); (2) mixed severity: 30- to 100-year frequency, patchy and variable overstory mortality (mesic mixed-conifer and drier high-elevation forests); and (3) high severity: more than 100-year frequency, more than 80 percent overstory mortality (low-elevation conifer and wetter subalpine forests).
[c] Fire intensity/severity are expected to increase in the next few decades, but they may decrease if fuel loadings are sufficiently reduced over time.

would incur the greatest overall risk in terms of land area. High-severity regimes cover less land area, so they would have less influence on large-scale ecological changes; however, local effects could be significant, particularly where high-severity fire regimes occur close to large population centers, where socioeconomic exposure could be high even if probability of an event were low.

Management of fire risk is a standard component of fire management in the Western United States. Fire suppression has traditionally been used on both public and private lands to reduce fire area and fire severity. Increasing area burned will provide significant challenges for federal agencies and other organizations that fight fire because of the high cost of suppression and difficulty of deploying firefighters to multiple large fires that may burn concurrently and over a longer fire season. Fuel treatments in dry forest ecosystems of the West can greatly reduce the severity of wildfires (Johnson et al. 2011), although funding is available to treat only a small percentage of the total area with elevated fuel loadings. Fuel treatments that include mechanical thinning and surface fuel removal are expensive, especially in the wildland-urban interface, and in a warmer climate, more fuel may need to be removed to attain the same level of reduction in fire severity as is achieved under current prescriptions (Peterson et al. 2011). Allowing more wildfires to burn unsuppressed is one way to achieve resource benefits while reducing risk, although this approach is often politically unacceptable, especially when fire could threaten human infrastructure and other values. Managing fire risk will be one of the greatest challenges for forest resource managers in the West during the next several decades.

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Risk Assessment for Forested Habitats in Northern Wisconsin

Louis R. Iverson, Stephen N. Matthews, Anantha M. Prasad, Matthew P. Peters, and Gary W. Yohe[1]

We used a risk matrix to assess risk from climate change for multiple forest species by discussing an example that depicts a range of risk for three tree species of northern Wisconsin. Risk is defined here as the product of the likelihood of an event occurring and the consequences or effects of that event. In the context of species habitats, likelihood is related to potential changes in suitable habitat at various times in the future. Consequences are related to the adaptability of a species to cope with the changes, especially the increasing intensity or frequency of future disturbance events. Data were generated from an atlas of climate change for 134 tree species of the Eastern United States (USDA FS 2011).

A risk matrix allows managers to determine which species need adaptation strategies, further evaluation, or monitoring programs. For example, a two-dimensional framework of likelihood versus consequence was used to assess the risk of future flooding on infrastructure in New York City (Yohe 2010, Yohe and Leichenko 2010), providing qualitative judgments about the magnitude of vulnerability and the likelihood of flooding exposure at specific points in time. This matrix illustrated changes in risk (e.g., potential number of buildings damaged or destroyed) over time, generated by the implications of sea level rise on the return times of what are now considered a 100-year storm and a 10-year storm. This matrix was not intended as a basis for policy decisions, but rather to help organize individual and governmental thinking about near- and long-term risk around likelihood and consequence.

We adopted the same matrix structure to assess the likelihood of exposure and magnitude of vulnerability (or consequences) for three tree species in northern Wisconsin (fig. A2-4). Much of the climate change literature focuses on potential decreases in forest species ("losers"), but increases may also pose management challenges, so the matrix was modified to include species or forest assemblages that are projected to increase in suitable habitat in the future ("gainers") (fig. A2-4). The risk matrix is demonstrated for black ash (*Fraxinus nigra* Marsh.) (loser), white oak (*Quercus alba* L.) (gainer), and yellow poplar (*Lireodendron tulipifera* L.) (new migrant).

Black ash carries more risk because, among other disadvantageous traits, it has low resistance to the emerald ash borer (*Agrilus planipennis* Fairmaire), which currently threatens all ash species in North America (Prasad et al. 2010). White oak is expected to gain habitat in northern Wisconsin, because it is well adapted to drier conditions and increased disturbance. Relative to other species, projected risk over time for this species is relatively low. Yellow poplar is not now recorded in northern Wisconsin, according to forest inventory information of the U.S. Forest Service. As a potential new migrant into the region, this species may provide new opportunities for habitat or wood products.

Using methods described in the DISTRIB system (Iverson et al. 2008, 2011; Prasad et al. 2009), data for the likelihood (x-axis) are based on a series of species distribution models to assess habitat suitability for 134 tree species in the Eastern United States, for current and future (2040, 2070, and 2100) climatic conditions. "Likelihood" in this context is, for any point in time, the potential that a section of forest within a specified region will have suitable habitat for a given species relative to its current suitable habitat. In this example, we use emission scenarios of modeled climate change, PCMlo and Hadhi, to elicit a range of possible risks, from low to high, associated with future climates. The

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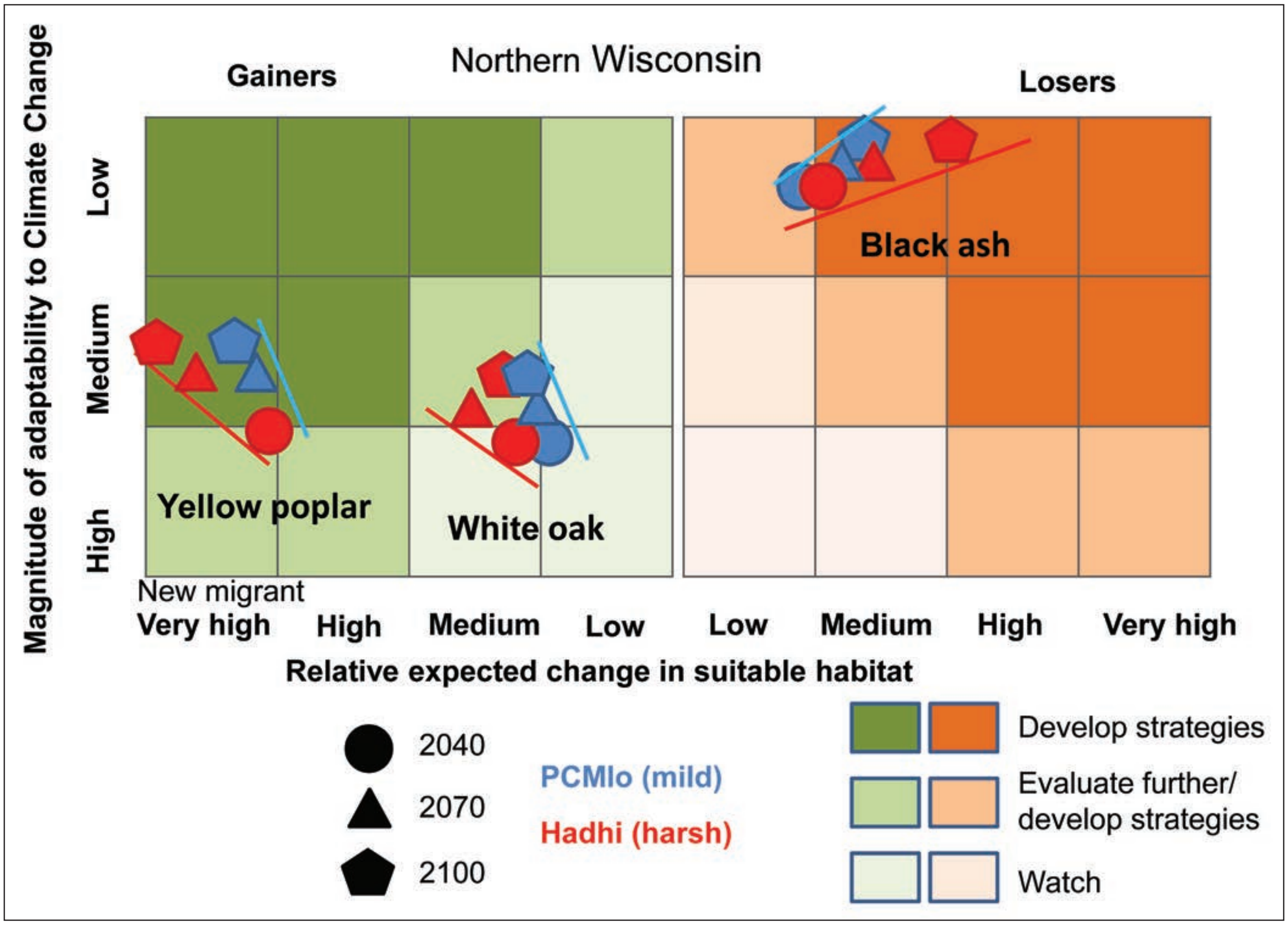

Figure A2-4—Risk matrix of potential change in suitable habitat for three tree species in northern Wisconsin that are expected to either lose habitat (black ash), gain habitat (white oak), or become a potential new migrant because of newly appearing habitat (yellow poplar).

matrix shows large variation between the emissions scenarios, with Hadhi causing larger changes in suitable habitat for all species. For black ash, which loses habitat, the x-axis ranges from 0 (complete loss of habitat over time) to +1 (no change in habitat over time). For white oak, which gains habitat, the x-axis ranges from +1 to +8. For yellow poplar, a species entering new habitat, the range is confined to the leftmost column of the graph. These numbers themselves are not directly the scale of "likelihood," but rather are scales of future:current importance values, and are plotted only to show the quantitative linkages.

Consequences in this context are related to the adaptability of a species or forest assemblage under climate change, based on a literature assessment of species biological traits and capacity to respond to disturbances that are likely to occur within the 21st century, including how those disturbances will be affected by climate change. Data for this axis comes from a literature-based scoring system, called "modification factors," to capture species response to climate change (Matthews et al. 2011). This approach was used to assess the capacity for each species to adapt to

12 disturbance types and to assess nine biological characteristics related to species adaptability. Each character was scored individually from -3 to +3 as an indication of the adaptability of the species to climate change. The mean, scaled values for biological and disturbance characteristics were each rescaled to 0 to 6 and combined as a hypotenuse of a right triangle; the resulting metric (ranging from 0 to 8.5) was used for the y-axis of the risk matrix (fig. A2-4). Because several disturbances (e.g., floods, droughts, insect attacks) are expected to increase over time, we also used a formula based on modification factors to enhance relevance for certain factors from 2040 to 2100. This analysis does not include socioeconomic consequences, such as the effects of decreased black ash on local basket-making economies of Native Americans.

The risk matrix has a number of useful applications. It provides a visual tool for comparing species risks relative to changing habitats associated with climate change. Trajectories displayed in the matrix reveal insights about species response to climate change and can be considered in the development of potential adaptation strategies, although they cannot account for nonlinear responses to extreme climate and altered disturbance regimes. The risk matrix can also help organize "climate change thinking" on a resource management team and communicate information to stakeholder groups and the general public. Finally, the risk matrix can be used to assess climate change risk for a variety of resource disciplines, and although the metrics may not be derived from the same methodologies, the capacity to rate one species against another, or one location against another, will promote a consistent approach to climate change risk management.

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Risk Assessment for Two Bird Species in Northern Wisconsin

Megan M. Friggens and Stephen N. Matthews[1]

Species distribution models for 147 bird species have been derived using climate, elevation, and distribution of current tree species as potential predictors (Matthews et al. 2011). In this case study, a risk matrix was developed for two bird species (fig. A2-5), with projected change in bird habitat (the x axis) based on models of changing suitable habitat resulting from changing climate and tree species habitat. Risk was evaluated for three time steps (2040, 2070, 2100) and based on two climate models and two emission scenarios (Hadhi vs. PCMlo).

To assess the y-axis of the matrix (fig. A2-5), we used the System for Assessing Vulnerability of Species (SAVS) (Bagne et al. 2011, Davison et al. 2011) to estimate species adaptability to future changes, including disturbances. The SAVS tool is based on 22 traits that represent potential areas of vulnerability or resilience with respect to future climate change. Each trait forms the basis of a question that is scored according to predicted effect (reduced, neutral, or increased population). By selecting responses for each question, a user creates a score that represents relative vulnerability to climate change effects, with higher positive values indicating higher vulnerability. Scores are calculated considering all 22 traits and divided among 4 categories: habitat, physiology, phenology, and biotic interactions. To calculate a baseline that could be used to compare current versus future vulnerability, we zeroed out individual questions for traits relating to exposure to future conditions and calculated a score based on the intrinsic characteristics of a species that reflect its sensitivity to population declines as a result of stochastic or other events.

Northern Wisconsin is near the edge of the distribution of the northern cardinal (*Cardinalis cardinalis* L.) and offers relatively limited habitat opportunities owing to the effects of current winter climatic conditions. However, with projected increases in temperatures for northern Wisconsin, the habitat for the northern cardinal is projected to double by the end of the century (future:current habitat ratio of 2.2). The northern cardinal uses habitats ranging from shrublands to forests, has a broad diet, and has been shown to be positively associated within an urbanizing landscape (Rodewald and Shustack 2008).The SAVS baseline scores indicate less vulnerability (-0.91) and that the species does not show increased vulnerability risk under climate change (-1.82). Characteristics such as adaptability of nesting locations and flexibility in reproductive time contribute to the less vulnerable score.

In contrast, the mourning warbler (*Oporornis philadelphia* A. Wilson) shows higher risk based on its more specialist nature, specificity to breeding habitats, and Neotropical migration life history. These innate traits make the mourning warbler more susceptible under current conditions (SAVS +3.64) and is also considered at an increased risk of exposure to negative effects of climate change (+5.45). The mourning warbler is primarily a boreal species and despite its use of early successional habitats and a positive response to some human disturbances such as timber harvest (Hobson and Schieck 1999), its occurrence in northern Wisconsin declined over a recent 16-year interval (Howe and Roberts 2005). Moving beyond contemporary changes, its habitat is projected to decrease by two-thirds of its current status by the end of the century (future:current ratio as low as 0.13 or 0.33, depending on climate model). These potential changes in habitat are attributed to higher temperatures and loss of boreal forest habitat (Iverson et al. 2008). In addition, the premontane and montane tropical life zones inhabited by the mourning warbler during winter are predicted to be highly sensitive to climatic affects (Enquist 2002). Therefore, when viewed together, the likelihood and magnitude of projected climate change suggest high risk for this species, and an increased opportunity for the northern cardinal, whose habitat will expand into northern Wisconsin.

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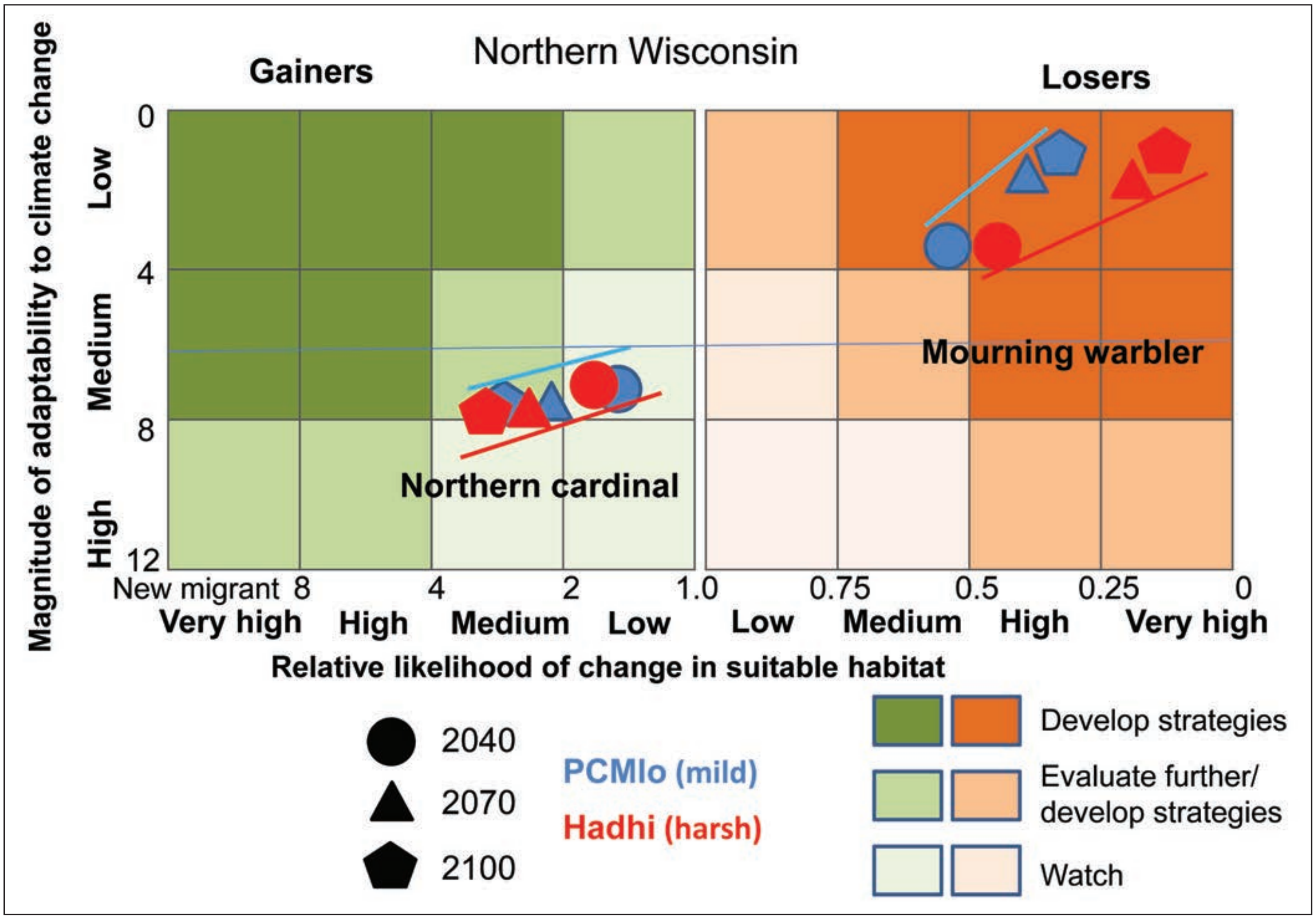

Figure A2-5—Risk of the effects of climate change on the northern cardinal and mourning warbler, expressed as a combination of likelihood of habitat change (x-axis) and magnitude of adaptability (y-axis). Values are rescaled from calculations that used the approach in the System for Assessing Vulnerability of Species (Bagne et al. 2011, Davison et al. 2011).

In this case study, we focused on two species with contrasting responses to climate change, but the general approach can be applied to a wide range of species, using either quantitative information or qualitative logic. The empirical statistical models used here provide insights on the broad-scale determinants of species distributions, but with some limiting assumptions. Models derived from mechanistic relationships that explore processes regulating population dynamics also demonstrate the importance of local climatic conditions on avian populations (Anders and Post 2006, Rodenhouse 1992), but they are available only for a limited number of species. The detailed parameterizations of mechanistic models also have important assumptions and can be difficult to apply across a broad array of species. Thus, more refined inferences on how climate change may affect avian populations will require careful consideration of both empirical and mechanistic approaches to modeling species distributions, especially the influence of ecological disturbances on habitat, as well as threshold values for minimum habitat quantity and quality.

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Appendix 3: Western Mountain Initiative Synthesis

Response of Western Mountain Ecosystems to Climatic Variability and Change: A Synthesis From the Western Mountain Initiative

Crystal L. Raymond[1]

Introduction

The Western Mountain Initiative (WMI), a consortium of research groups in the Western United States, focuses on understanding and predicting responses—especially sensitivities, thresholds, resistance, and resilience—of mountain ecosystems to climatic variability and change (Peterson et al. 2012). The WMI addresses how climatic variability and change influence forest processes, disturbance dynamics, hydrologic changes, and hydroecological interactions in five bioregions: Pacific Northwest, Sierra Nevada, Northern Rocky Mountains, Central Rocky Mountains, and Southern Rocky Mountains. A guiding theme of WMI research is understanding the linkages among these processes. This focus on linkages (e.g., climate change affects disturbance regimes, hence vegetation, hence erosion) and the depth and breadth of place-based knowledge represented by this work contribute to multisite regional comparisons.

Research addresses four key questions: (1) How are climatic variability and change likely to affect disturbance regimes? (2) How are changing climate and disturbance regimes likely to affect the composition, structure, and productivity of vegetation? (3) How will climatic variability and change affect hydrologic processes in the mountainous West? and (4) Which mountain resources and ecosystems are likely to be most sensitive to future climatic change, and what are possible management responses? Results to date have documented how climatic variability and change affect several trends: long-term patterns of snow, glaciers, and water geochemistry; forest productivity, vigor, and demography; and changing patterns of treeline dynamics and forest disturbances. Empirical and simulation modeling indicates that major changes in hydrologic function and ecological disturbance will occur as the climate continues to warm.

The WMI research has documented trends in temperature, precipitation, and snowpack, and the exceedance of biological and ecologically meaningful thresholds of these variables in the mountainous West. In the northern Rocky Mountains, trends show that extremely cold days (≤ -18 °C) end on average 20 days earlier and have declined in number, and the number of extremely hot days (≥ 32 °C) has increased over the last 100 years (Pederson et al. 2011). Trends in snowpack observations in the northern Rocky Mountains indicate declines in snowpack and earlier arrival and melt of peak snow water equivalent over the last 40 years (Pederson et al. 2011). Although much of this change in snowpack is attributed to climatic variability, an extension of this analysis to the whole Rocky Mountains region and to the last 800 years using tree-ring based reconstructions of snowpack shows only two periods of sustained low snowpack comparable to those observed in the 20th century (Pederson et al. 2011).

Here we focus on WMI results in three areas related to forest ecosystem response to climatic variability and change: (1) trends in the structure and function of western forest ecosystems, (2) the effects of exceeding critical thresholds, and (3) the potential for future changes in these systems. Changes have been documented in forest demography, treeline dynamics, and ecological disturbances and interactions. Although it is not possible to definitively attribute recent changes as being caused by climate change, the effects of dominant modes of climatic variability can be used

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with some confidence to infer expected changes in ecosystems in a warmer climate. For example, relationships have been documented between the Pacific Decadal Oscillation (20- to 30-year cool and warm phases) and temporal variation in regional-scale tree growth (Peterson and Peterson 2001, Peterson et al. 2002), wildfire (Hessl et al. 2004), and long-term drought (Gedalof et al. 2004). These relationships suggest that significant changes in ecosystem processes will occur as temperature, and extremes in temperature, continue to increase.

Forest Die-Off and Demography

Recent broad-scale syntheses have documented climate-induced forest mortality in some locations (Allen 2009, Allen et al. 2010). These syntheses reveal diverse patterns in forest die-off attributed to drought and heat, including localized increases in background mortality and regional-scale forest die-off linked to biotic agents (Allen et al. 2010). Drought-induced die-off is commonly observed at the elevation and geographic margins of species ranges and is often associated with prolonged periods of moisture stress. Recent and ongoing WMI research continues to address key uncertainties in forest mortality processes (Breshears et al. 2009; McDowell et al. 2008, 2010).

Based on demographic trends in long-term plot data, mortality rates in old forests have increased in each of three subregions (Northwest, California, and interior West) and across elevation zones and tree size classes (van Mantgem et al. 2009), and tree recruitment rates have not changed, contributing to lower stem density and basal area of old forests. Since 1955, both temperature (particularly at the higher elevations occupied by forests) and climatic water deficit (evaporative demand that is not met by available water) increased, and both were positively correlated with tree mortality rates (van Mantgem et al. 2009). Warming-induced tree mortality is consistent with the apparent role of warming in recent forest dieback in western North America (Adams et al. 2009) and the positive correlation between short-term fluctuations in background tree mortality and water deficits in California and Colorado (Bigler et al. 2007, van Mantgem and Stephenson 2007).

Treeline Dynamics

One of the most distinctive features of mountain environments is the transition from subalpine forest to alpine tundra (alpine forest-tundra ecotone, AFTE), and climate change is likely to affect these environments in the mountainous West. Advance of trees into tundra can alter cycling of water, carbon, and nutrients and the maintenance of biodiversity. Treeline advance and changes in treeline patterns have the potential to alter snow retention and hydrology, with implications for local soil moisture and nutrient transport.

Studies of treeline phenomena and limiting factors among the five WMI bioregions have demonstrated that although ecological dynamics of the AFTE are influenced by climate, mechanistic processes that shape the ecotone—seed rain, seed germination, seedling establishment, and subsequent tree growth form—also depend on microsite patterns (Malanson et al. 2007). In the West, these mechanistic processes are similar among AFTEs, but other processes—prior climate, geomorphology, genetics, and historical grazing practices—create geographic differences in responses of ecotones to climate change. Climate change may affect successful seed dispersal, germination, and survival by modifying the biophysical environment. The three-dimensional pattern at treeline is typically patchy, including krummholz and dwarf trees, with expansion often facilitated by other plants (Resler 2006). The formation of vegetation structures that add wind protection, snow collection, and soil development allows subalpine forest species to initiate patches and expand in the upper treeline (Smith et al. 2003). Climate and variation in geomorphology, geology, and disturbances (e.g., snow avalanches) also control mortality at treeline and limit treeline elevation.

At regional to continental scales, control of the AFTE by temperature is locally modified by moisture (Malanson and Butler 2002). Evidence for AFTE response to climatic variability suggests upslope advance during warmer conditions, but advance in many sites may be limited by moisture. For example, in the 19th century, Glacier National Park experienced upward expansion of the ecotone (Bekker 2005), although in the 20th century, density of existing patches increased but advance was limited (Klasner and Fagre 2002).

Treeline advances in the latter half of the 20th century coincided with the cool (wet) phase of the Pacific Decadal Oscillation, and advance stopped during the warm (dry) phase of the 1980s and 1990s (Alftine et al. 2003).

A warmer, wetter climate could alter the structure of treeline in all WMI bioregions. In the Pacific Northwest, high snowpack inhibits establishment and growth of trees, so at the highest treeline elevations, increased snow plus increased winter precipitation could limit expansion of krummholz higher in the ecotone. The other WMI bioregions are drier, so higher precipitation could improve conditions for tree establishment and growth. For example, in Sierra Nevada treelines, tree growth increased during warmer, wetter periods in the 20th century (Millar et al. 2004). At all sites, effects are likely to be greatest in the lower AFTE, where deeper soils and root zones can use increased water (Malanson et al. 2007).

A warmer, drier climate could reduce tree establishment in the ecotone of all WMI treeline sites except the Pacific Northwest. In the Pacific Northwest, less moisture is unlikely to reduce tree establishment and growth in the AFTE. A warmer climate could increase tree growth in the AFTE, and less snow could facilitate expansion in the upper ecotone and encroachment in meadows in the lower ecotone. Conversely for the other WMI bioregions, a drier climate could further limit tree establishment, growth, and species diversity in the AFTE (Malanson et al. 2007).

Ecological Disturbance and Interaction of Stressors

Empirical and process-based models have been used to estimate the extent and magnitude of future disturbances across the West. The indirect effect of climate change on forests through changing disturbance regimes is likely to cause more rapid changes than the direct effects of higher temperatures on trees (fig. A3-1), and accelerated species turnover will occur after severe disturbance because seedlings are less resistant to changing climate than are mature individuals (McKenzie et al. 2009). In light of the importance of disturbance in Western forests, WMI research has emphasized quantifying the effects of climatic variability and change on the areal extent and broad-scale spatial patterns of wildfire and insect outbreaks (Littell et al. 2010).

Of particular concern are increases in fire area in a warming climate and the effects of extreme wildfire events on ecosystems (Gedalof et al. 2005, Littell et al. 2009, McKenzie and Littell 2011). Strong climatic controls exist on area burned by wildfire across the West at the spatial scales of entire states (McKenzie et al. 2004), ecoprovinces (Littell et al. 2009), and sections within ecoprovinces (Littell et al. 2010). In forests across the Northwestern United States, climate during the fire season appears to control area burned, whereas in arid mountains and shrublands, antecedent climate (e.g., wetter, cooler summers or winters preceding the fire season) can increase area burned during the fire season by increasing fuel abundance and continuity (Littell et al. 2009, 2010). Research has also quantified sediment yields after wildfire in different rainfall regimes across the Western United States (Moody and Martin 2008), providing a key context for potential climate-mediated watershed changes in postfire runoff and erosion relationships.

Mountain pine beetle (*Dendroctonus ponderosae* Hopkins) infestations have historically occurred frequently and extensively throughout western North America (Logan and Powell 2001). Warming and drought affect development rates of beetle life stages, winter mortality, and host tree susceptibility (Carroll et al. 2004). Across the West, current stand structural conditions make host species susceptible to beetle attack (Hicke and Jenkins 2008), and as warming continues, we might expect that forests will become susceptible to insect attack more frequently (Raffa et al. 2008). Mountain pine beetle outbreaks are facilitated when the insect's reproductive cycle is very close to one year and when larvae emerge at an optimal time for feeding, dispersal, and survival of cold seasons (Logan and Powell 2001). As temperatures increase, the life cycle shortens; therefore, a warmer climate is projected to reduce the area of climatic suitability for the beetle at low elevations but increase suitability at higher elevations (Hicke et al. 2006, Littell et al. 2010). Increases in mountain pine beetle outbreaks with climate

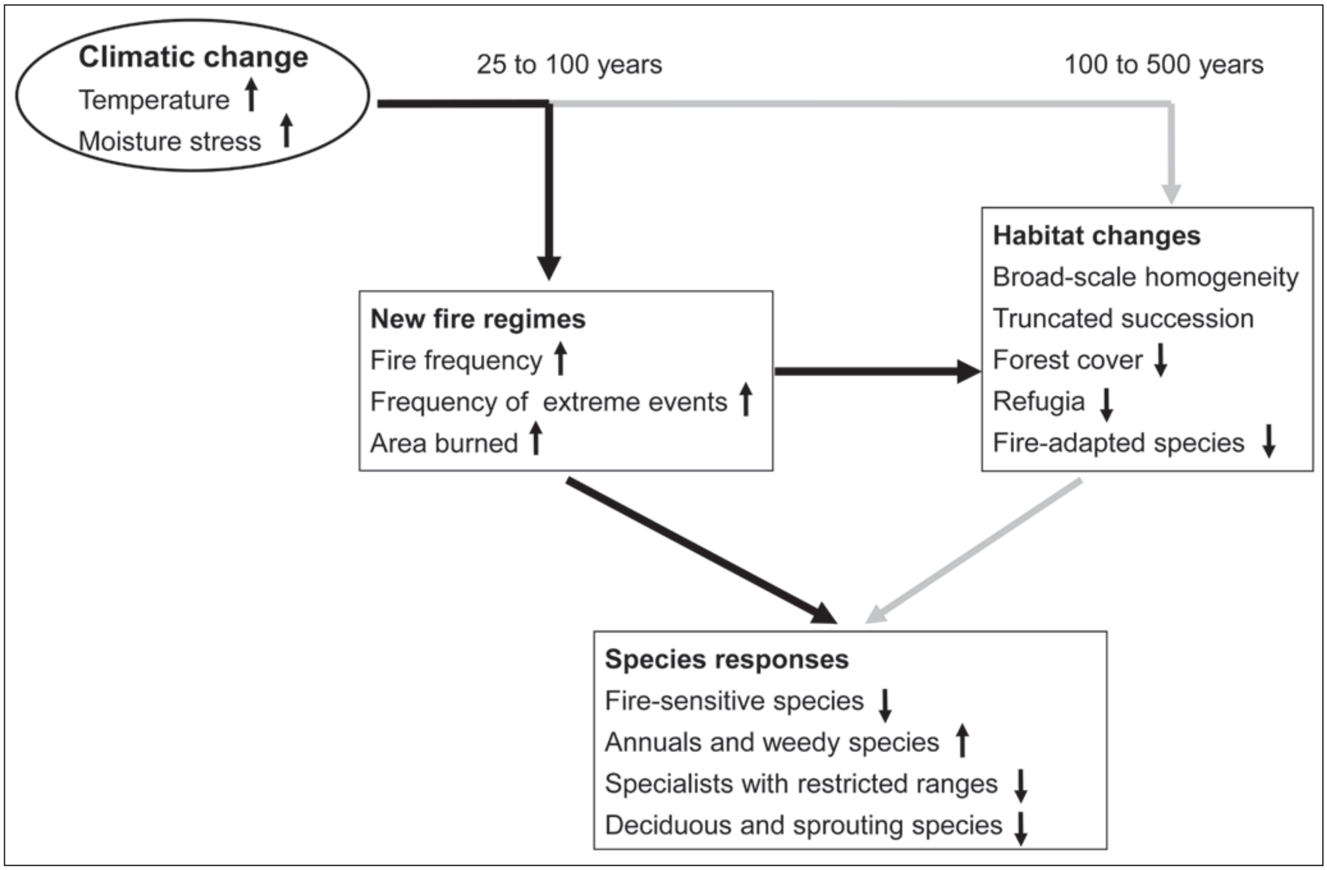

Figure A3-1—Conceptual model of the relative time scales for disturbance versus climate change alone to alter ecosystems. Times are approximate. The focus here is on fire, but much of the same logic applies to insect outbreaks. Adapted from McKenzie et al. (2004).

change will affect the carbon cycle in forest ecosystems and have feedbacks to the climate system (Hicke et al. 2012a).

Research on disturbance interactions and their effects on ecosystem processes (Allen 2007, McKenzie and Littell 2011, McKenzie et al. 2009) suggests that synergistic interactions between disturbances produce larger effects than would occur from an individual disturbance. For example, bark beetle outbreaks have been linked to increased likelihood of stand-replacing fire and changes in fire behavior, with the nature of the effect depending on the time since infestation (Jenkins et al. 2008, Lynch et al. 2006), although there is conflicting evidence about whether fire hazard is higher following bark beetle outbreaks (Hicke et al. 2012b). Combined with increasing climatic stress on tree populations and growth, disturbance interactions can alter forest structure and function faster than could be expected from species redistribution or disturbance alone (fig. A3-1). Simultaneous climatically driven shifts in the locations of species optima, ecosystem productivity, disturbance regimes, and interactions between them could reset forest succession over large areas and short timeframes.

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