Chapter 3

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

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Contributors

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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).

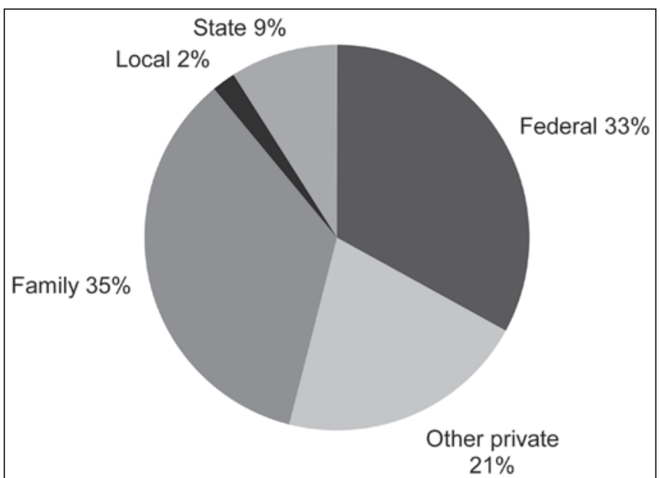

Figure 3.1—Forest ownership in the United States, 2006 (Butler 2008).

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.

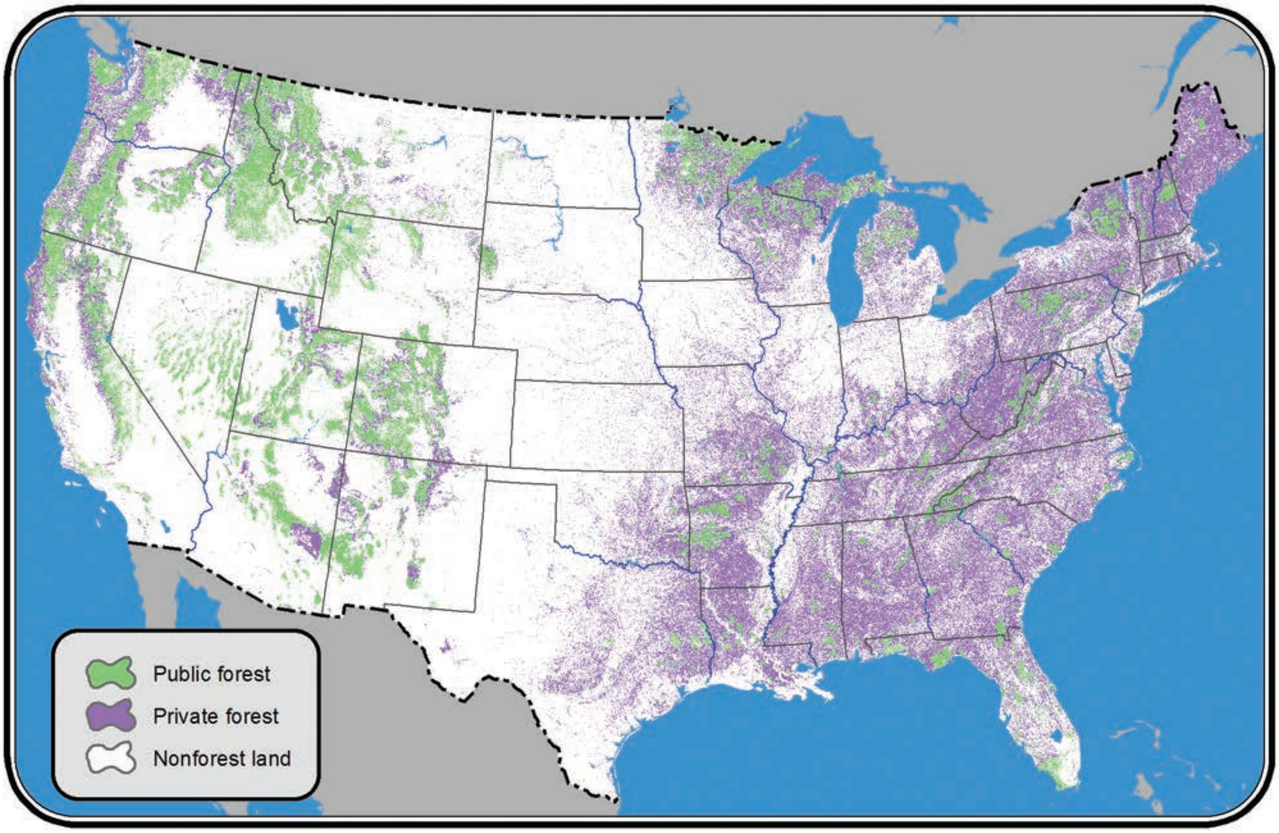

Figure 3.2—Distribution of public and private forest ownership in the United States.

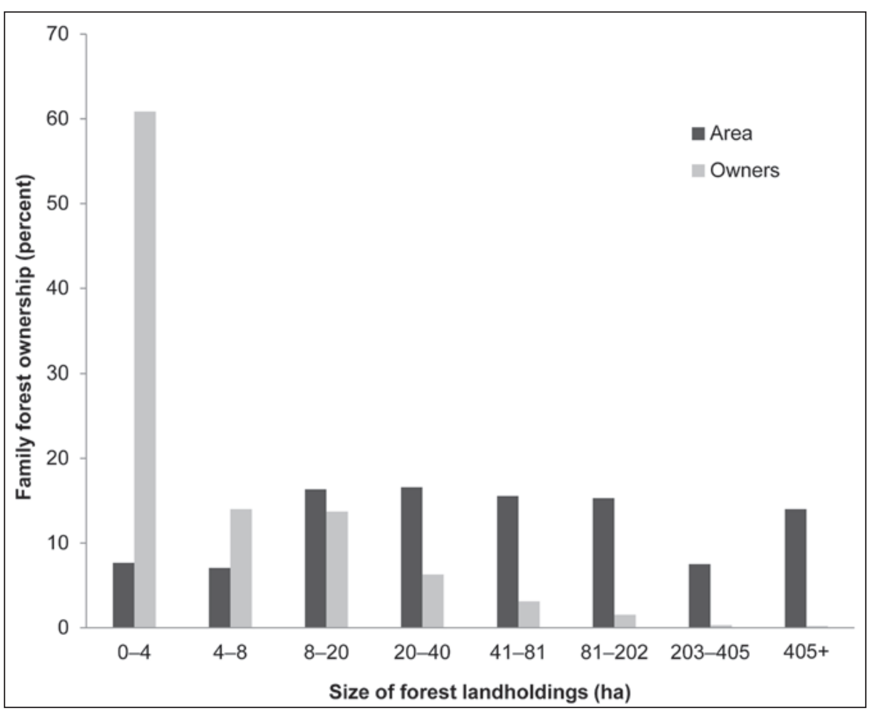

Figure 3.3—Family forest ownerships in the United States by size of forest holdings, 2006. (Butler 2008).

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

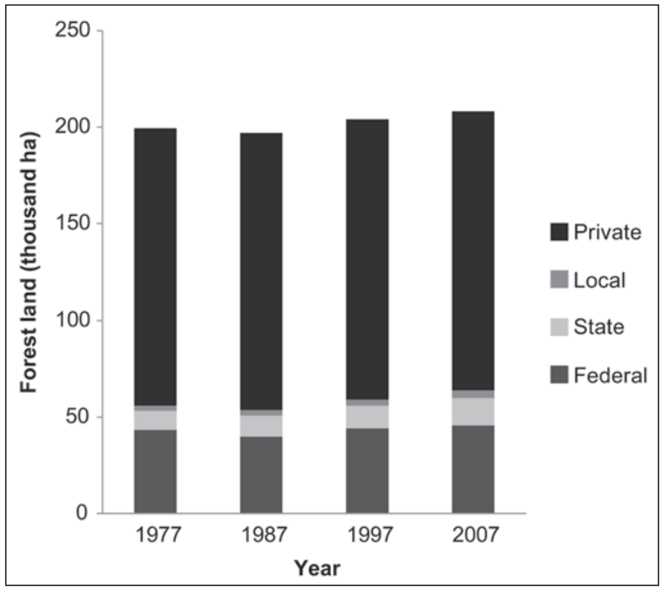

Figure 3.4—Trends in U.S. forest ownerships, 1977 through 2007 (Smith et al. 2009).

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.

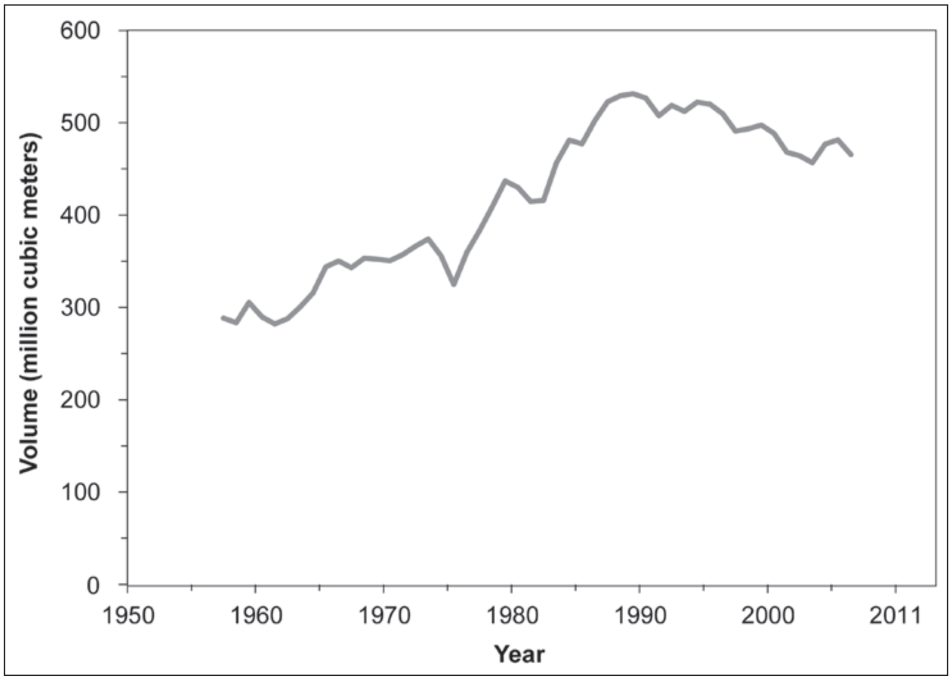

Figure 3.5—U.S. roundwood production, 1957 through 2006 (Howard et al. 2010a).

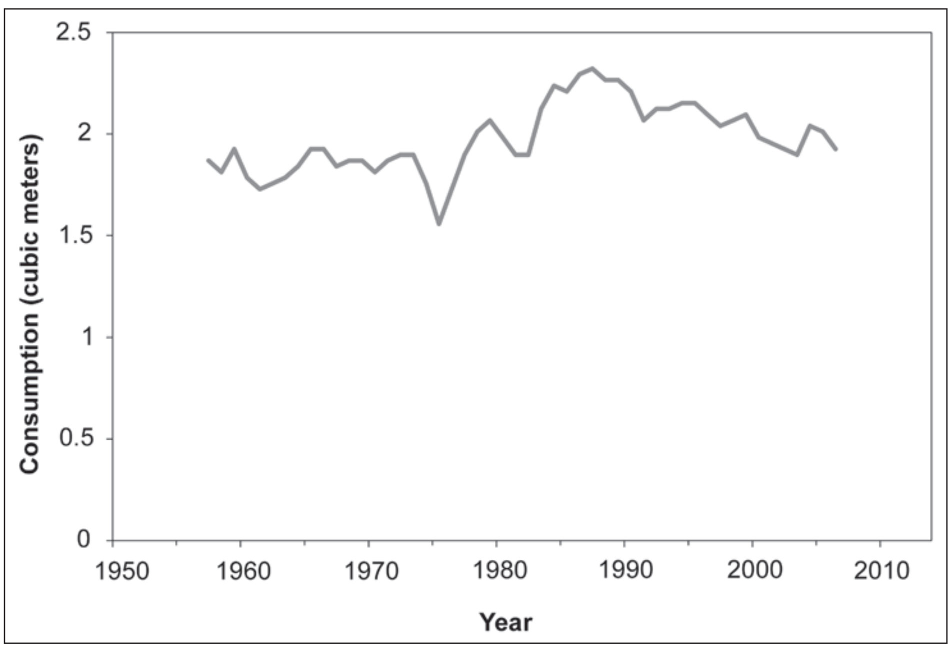

Figure 3.6—U.S. per capita apparent roundwood consumption, 1957 through 2006 (Howard et al. 2010b).

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,

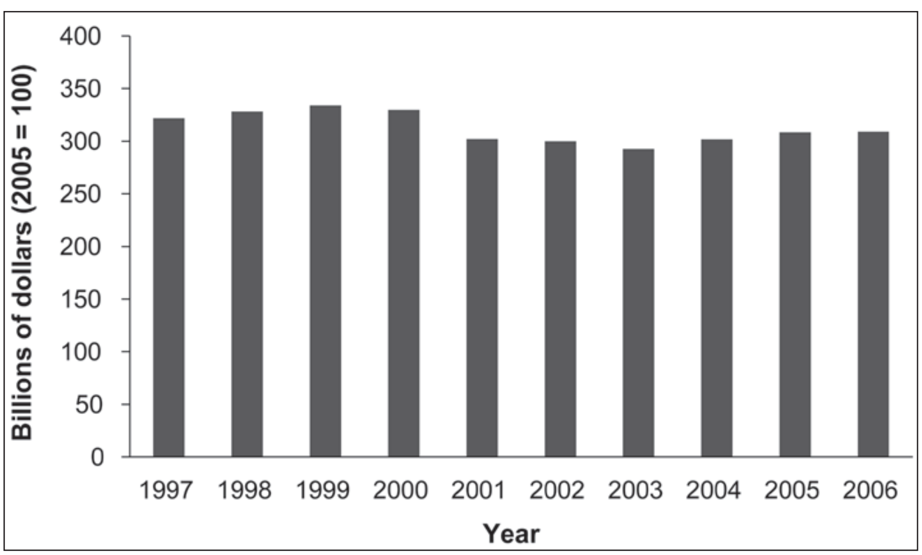

Figure 3.7—Value of shipments for forest products industries (NAICS codes 321, 322, and 337), 1997 through 2006 (Source: Howard et al. 2010a).

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.

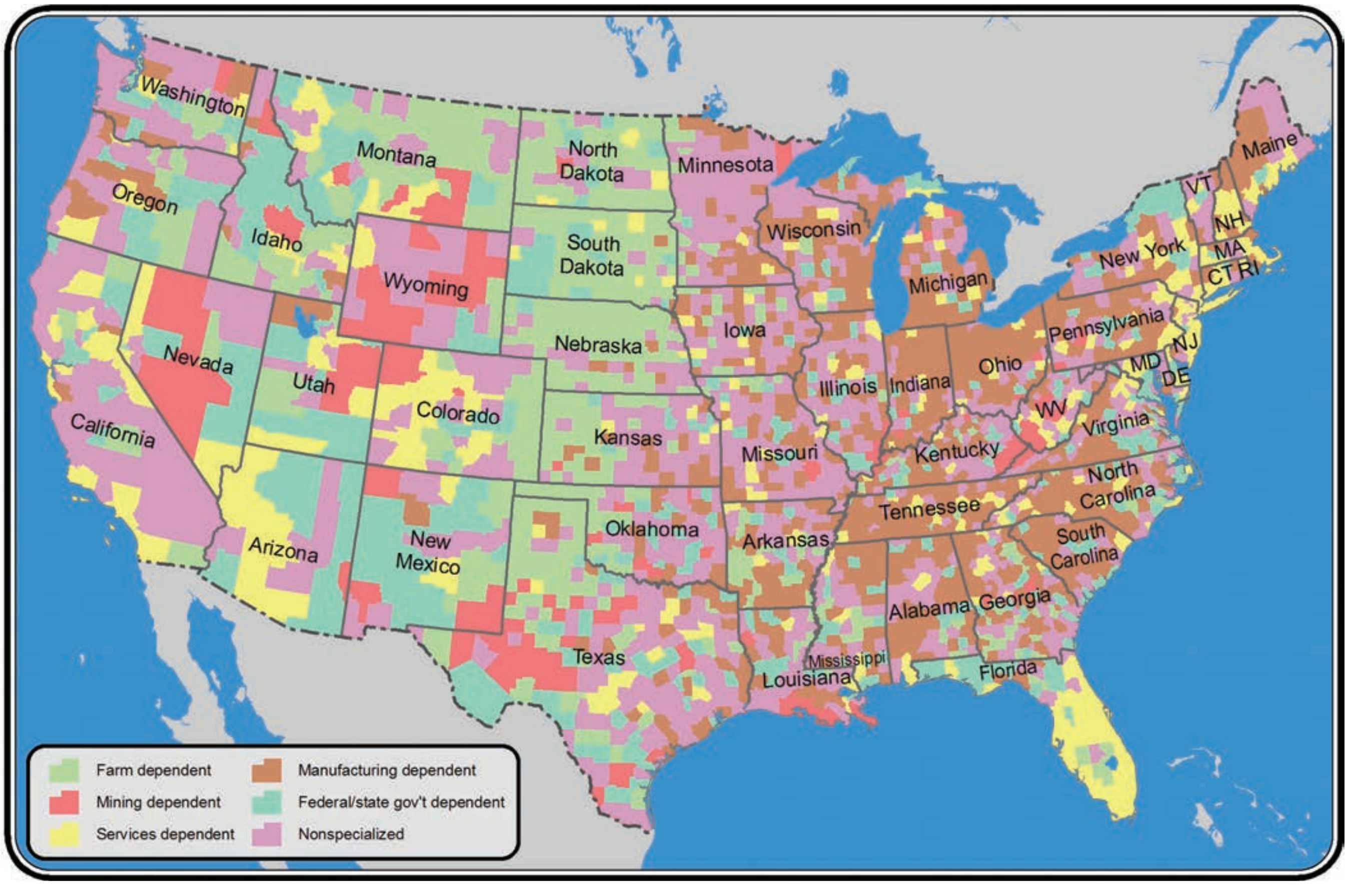

Figure 3.8—Economic dependence (USDA ERS 2011).

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

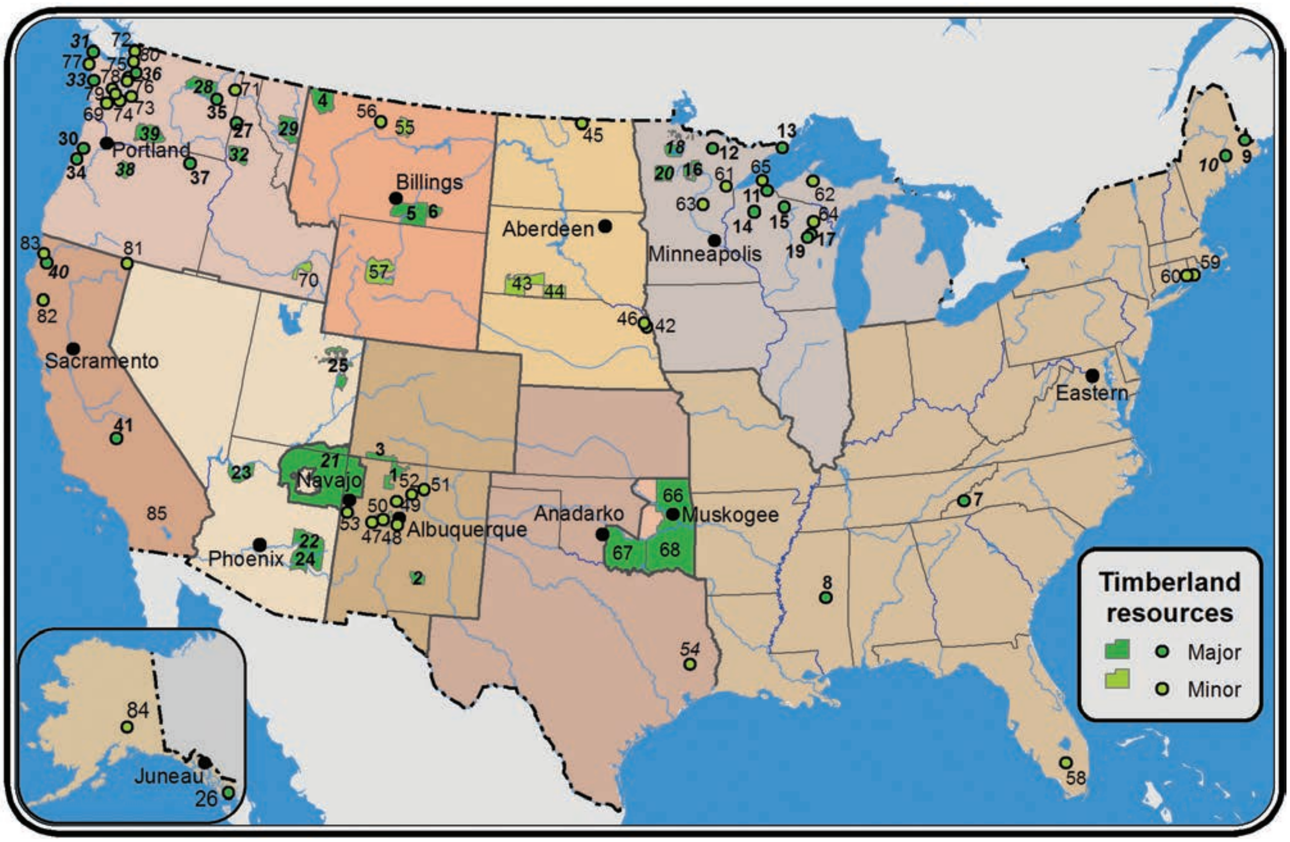

Figure 3.9—Reservations with significant timberland resources. Numbers 1 through 41 have over 4047 ha of commercial timberland, or over 2360 m^3 allowable cut, per reservation. Numbers 41 through 83 have less area in timberland, but what they have is economically viable. (1) Jicrailla, (2) Mescalero 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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