United States
Department of
Agriculture

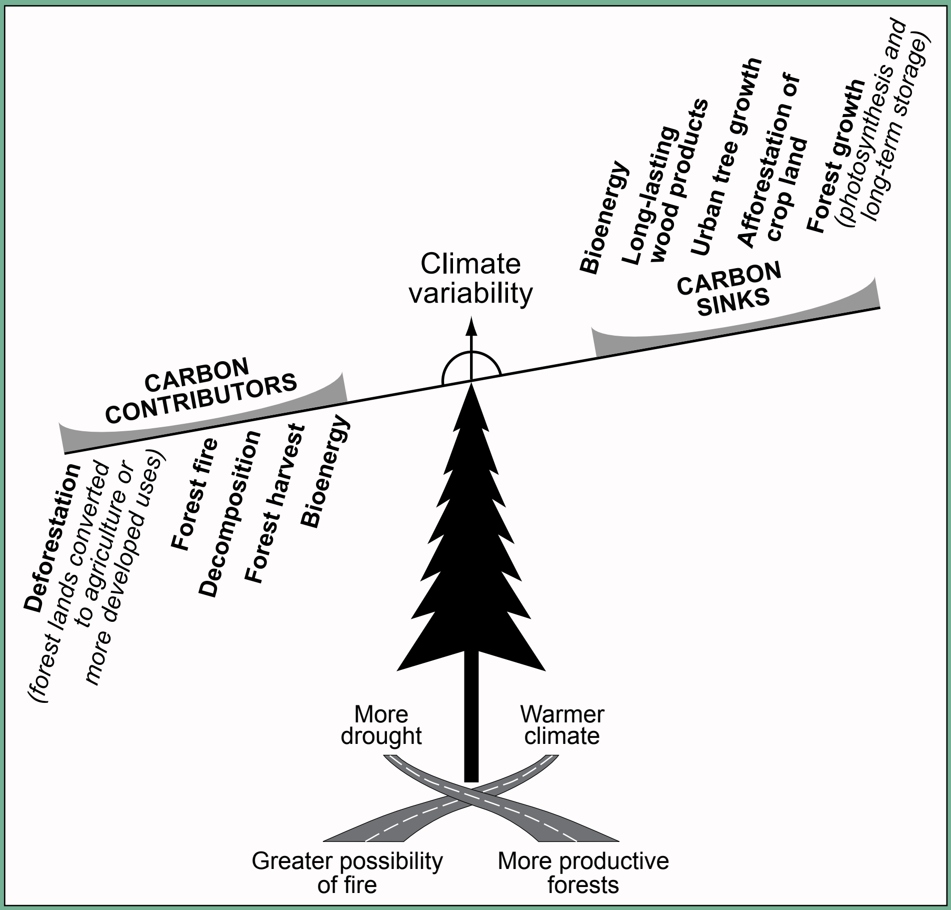
Forest Service

Pacific Northwest
Research Station

General Technical Report
PNW-GTR-833

October 2010

Economic Modeling of Effects of Climate Change on the Forest Sector and Mitigation Options: A Compendium of Briefing Papers



Authors

Ralph J. Alig is a research forester, Forestry Sciences Laboratory, 3200 SW Jefferson Way, Corvallis, OR 97331.

Economic Modeling of Effects of Climate Change on the Forest Sector and Mitigation Options: A Compendium of Briefing Papers

Ralph J. Alig
Technical Coordinator

Authors

Ralph J. Alig is a research forester, U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, 3200 SW Jefferson Way, Corvallis, OR 97331.

Judith Dempsey is a graduate research assistant, Department of Agricultural and Resource Economics, Oregon State University, Corvallis, OR 97331.

Robert G. Haight is a research forester, U.S. Department of Agriculture, Forest Service, Northern Research Station, St. Paul, MN 55108.

Taeyoung Kim is a graduate research assistant, Department of Agricultural and Resource Economics, Oregon State University, Corvallis, OR 97331.

Christian Langpap is an assistant professor, Department of Agricultural and Resource Economics, Oregon State University, Corvallis, OR 97331.

Andrew J. Plantinga is a professor, Department of Agricultural and Resource Economics, Oregon State University, Corvallis, OR 97331.

Brent Sohngen is an associate professor, AED Economics, Ohio State University, 2120 Fyffe Rd., Columbus, OH 43210.

Birger Solberg is a professor, Department of Ecology and Natural Resource Management, Norwegian University of Life Sciences, 1432 Aas, Norway.

Eric M. White is a research associate, Department of Forest Engineering, Resources and Management, College of Forestry, Oregon State University, Corvallis, OR 97331.

Abstract

Alig, Ralph J., tech. coord. 2010. Economic modeling of effects of climate change on the forest sector and mitigation options: a compendium of briefing papers. Gen. Tech. Rep. PNW-GTR-833. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 169 p.

This report is a compilation of six briefing papers based on literature reviews and syntheses, prepared for U.S. Department of Agriculture, Forest Service policy analysts and decisionmakers about specific questions pertaining to climate change. The main topics addressed here are economic effects on the forest sector at the national and global scales, costs of forest carbon sequestration as part of mitigation strategies, and mitigation aspects for nonindustrial private and public forest ownerships in the U.S. forest sector. Salient findings from the literature are summarized in the synthesis of the literature, along with identified research needs.

Keywords: Climate change, costs of forest carbon sequestration, nonindustrial private forests.

Contents

Overview

Global climate change from a buildup of greenhouse gases (GHG) poses physical, ecological, economic, and social issues. Forest ecosystems can transfer carbon from the air as part of the GHG complex and sequester it into plant tissue through the process of photosynthesis during the growth of trees and in other ecosystem components such as the understory and soil. Such forest sinks have a significant potential to help in mitigating climate change, and this report is a compilation of briefing papers prepared for U.S. Department of Agriculture, Forest Service policy analysts and decisionmakers about specific forest carbon sequestration topics. The briefing papers are part of a larger set prepared by agency scientists and cooperators, including a related one regarding forest bioenergy by White.[1] Given the large topic of climate change and forests, this report only touches on selected aspects and readers are referred to the growing literature for information on specific topics such as considerations when incorporating climate change considerations into specific natural resource management, e.g., Joyce et al.[2]

In the first chapter leading off the effects section, White et al. review the literature pertaining to **national-scale** economic modeling of effects of climate change on the U.S. forest sector. Across the globe and in the United States, forest resources are expected to be affected by changes in forest growing conditions induced by carbon dioxide "fertilization" and climate change. These changing forest conditions are expected to result in changes in forest management regimes, production practices, and, potentially, the uses of timberlands within the forest products sector. White et al. report studies that have used economic models to trace changes in forest growth resulting from climate change and changes in behavior within the forest products sector.

Our second chapter, by Sohngen et al., assesses research pertaining to a **global-scale** examination of the effects of climate change on the forest sector. In considering the global carbon cycle, this provides a synthesis of recent research that investigates how climate change may affect the global forest sector. The near-term (to 2020), medium-term (2020–2060), and long-run impacts (beyond 2060) are assessed. Understanding these important interactions between forests, climate

---

[1] White, E.M. 2010. Woody biomass for bioenergy in the United States—a briefing paper. Gen. Tech. Rep. PNW-GTR-825. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 45 p.

[2] Joyce, L.A.; Haynes, R.W.; White, R.; Barbour, R.J., tech. coords. 2006. Bringing climate change into natural resources management: proceedings. Gen. Tech. Rep. PNW-GTR-706. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 150 p.

change, and carbon flux remains an important research topic, not only for economists and ecologists separately, but more importantly for these scientists working together.

In the third chapter, leading into the mitigation section, Alig provides background information about U.S. land and forest resources and regional differences, along with potential interactions between the forestry and agricultural sectors regarding climate change mitigation. Selected examples of modeling studies involving the forestry and agricultural sectors are reviewed, centered on land-use changes.

The fourth chapter reviews alternative methods for estimating costs of future forest-based carbon sequestration as part of mitigation activities. This includes examining key assumptions in cost studies of carbon sequestration, e.g., assumed timber harvesting practices. Dempsey et al. review representative cost studies and point out difficulties at times in directly comparing cost estimates from different studies. Differences in the estimates may be due to the cost estimation method, but also to the many other assumptions that the analyst must make. They summarize reviews of the literature that develop comparable cost estimates from groups of studies, involving normalizing the data. Many implementation issues must be carefully addressed before a large-scale carbon sequestration program can effectively be put into operation.

The fifth chapter focuses on the large nonindustrial private forest ownership that contains many of the U.S. forest carbon sequestration opportunities. That ownership has some holdings of both forest and agricultural lands, and Langpap and Kim review what research indicates about how owners may respond to such incentives and how effective different policies might be in eliciting the desired forest management choices by nonindustrial private owners. They also review information from the literature about some implementation issues, such as dealing with additionality concerns.

The sixth chapter focuses on the connections between private and public forest ownerships in the climate change context. Forests in both ownerships provide a variety of goods and services to society, and under climate change and climate change policies, there are likely to be some disparate impacts to public and private forest land as well as important interactions between those ownerships. White et al. review studies that have examined the distinctions, linkages, and interactions between public and private forest land in the context of climate change and climate change policy. They identify key points from some of the existing literature on the interactions of public and private forest land within the context of climate change, to help in identifying research needs.

Chapter 1: The Forest Sector in a Climate-Changed Environment

by Eric M. White, Ralph J. Alig, and Robert G. Haight

Introduction

Policies adopted to address climate change will influence the extent, composition, and management of future forests. At the same time as policy is causing forest changes, climate change itself is expected to yield changes to environmental conditions, influencing the characteristics and growth of forests in rural and urban settings. In response to changes in forest productivity, timber producers will likely change management regimes, production practices, and, potentially, the uses of timberlands. Similarly, climate-induced changes in composition and health of urban forests may change the service flows that urban residents receive from those forests. The effects of climate policies on the future conditions of forests are discussed in other reports. The focus of this paper is how the forest sector may change in response to future altered climate conditions. To accomplish this, we rely on a number of U.S. studies, including those that use economic models to trace changes in forest conditions to responses in forest sector markets and timber production.

At the same time as policy is causing forest changes, climate change is expected to yield changes in forests.

A number of studies have focused on changes in ecological processes in forested ecosystems as a result of an altered climate (e.g., Iverson et al. 2008, Joyce 1995, Joyce and Birdsey 2000, Latta et al. 2010). Economic studies, including the ones discussed in this paper, extend that ecological research by explicitly including human activities (e.g., the demand for forest products by consumers, the desire to live in amenity-rich settings) in analyses of projected forest conditions and landscapes. In the United States, especially on private lands, landscape conditions reflect human activities (e.g., land-use change), forest management (e.g., silviculture systems), and the demands for products (e.g., timber) and services (e.g., recreation opportunities) from natural environments. The models described in this briefing paper focus on land-use changes involving forests, forest management activities, and the demand for timber products. The impacts of climate change on other forest products and services, such as recreation opportunities, have been discussed to a limited extent elsewhere (e.g., Irland et al. 2001).

Economic studies extend ecological research by explicitly including human activities.

We briefly describe the observed changes in climate and projections of future climate conditions as identified by the fourth assessment report of the Intergovernmental Panel on Climate Change (IPCC). Further description of these expectations are available in IPCC reports (Field et al. 2007, IPCC 2007) and a short briefing paper for policymakers developed by the Pew Center (Pew Center 2007). Although the literature on projected changes to forests from climate change is extensive and at times contradictory, we present a brief summary and rely on others (e.g. Field et al. 2007, Joyce and Birdsey 2000) to give this topic more comprehensive treatment. We describe the approaches that economists have adopted to trace expected changes in forest growth to impacts on the forest sector, including forest management; factors believed to be important in influencing future timber availability; and the modeled impacts on timber production, prices, forest management, and other factors as estimated from the existing models. We further treat the projected welfare impacts to wood product consumers and timber producers as result of climate change and examine some of the benefits received by society from urban forests and potential impacts of global change on those forests.

Observed Climate Changes and Projections for the Future

One challenge to projecting forest conditions is the uncertainty surrounding future climate. Published summaries of work being completed by the IPCC indicate that warming of the planet is now “unequivocal” (IPCC 2007, Pew Center 2007). Between 1955 and 2005, temperatures in North America increased moderately, with the increases becoming greater as one moves northwestward from the Southeast, toward Alaska and northwestern Canada (Field et al. 2007). Whether this warming is currently yielding changes in natural systems is less certain, with expert opinions indicating even likelihood and unlikelihood that some impacts of climate change on natural systems are now beginning to emerge. Looking forward, the IPCC has concluded that average global temperature increases over the 21st century are expected to range from 1.8 to 4.0 degrees Celsius (IPCC 2007, Pew Center 2007). In North America, average temperature increases are projected to be 1 to 3 degrees Celsius by 2040, increasing to 2 to 3 degrees Celsius late in the century (Field et al. 2007). For land areas, warming is anticipated to be greatest over the high northern latitudes. The IPCC has determined that such warming is “very likely” (> 90 percent probability) based on evidence that there will be an increase in the frequency of hot extremes, heat waves, and heavy precipitation (IPCC 2007). Uncertainty in these projections increases as anticipated global conditions are downscaled to regional conditions. The IPCC deemed it was “very likely” that precipitation will increase at high latitudes and “likely” (> 66 percent probability) that it will decrease in most subtropical regions. Areas in the Southern and Southwestern United States are considered to be in the subtropical zone. Precipitation in United States is projected to decrease in the Southwest but increase elsewhere (Field et al. 2007). Further, the IPCC reports “high confidence” (80 percent agreement) that areas such as the

Western United States will experience a decrease in water availability as a result of climate change (Pew Center 2007). In some areas, such as the West, precipitation may increasingly be in the form of rain rather than snow, decreasing snowmelt water availability (Field et al. 2007). The current IPCC climate scenarios each project continued increases in greenhouse gases (GHG) (including carbon dioxide [CO_2]) over the next several decades. However, GHG emissions in recent years have been below projections because of the global economic downturn.

Climate Change and Forest Impacts

Forest responses to climate change are expected to include changes in forest location, forest species, growth rate, and timber yield. There remains uncertainty about forest conditions in specific regions likely to result from climate change.

At the most general, forest responses to climate change are expected to involve changes in forest location, the combinations of forest species and classes on the landscape (i.e., forest compositions), and growth rate and timber yield (Shugart et al. 2003). These expected changes result largely from climate-change-induced changes in temperature, precipitation, and atmospheric CO_2 levels as well as a general lengthening of the growing season. Although these general relationships are fairly accepted, there remains a fair amount of uncertainty about the forest conditions in specific regions likely to result from climate change. Much of this uncertainty relates to ecosystem complexity, the variety of ecosystem conditions involved, and the ability of ecosystems to adapt (including through human intervention). In one example of uncertainty because of complexity, there is a lack of agreement on the extent to which increases in atmospheric CO_2 will have a fertilizer effect on plants (e.g., McKinley et al. 2009, Norby et al. 2005, Reich et al. 2006, Thornton et al.

Eric White

Changing forest conditions can affect the supply of timber available for use of wood products and storage of carbon in woody biomass.

2009). Some believe that although increased CO_2 promotes plant growth, over large areas, limitations in other inputs to plant growth (e.g., nitrogen or water availability) may reduce these increases. For example, nitrogen is required for tissue growth and, even in the presence of elevated CO_2, limited nitrogen may limit increased growth rates (e.g., Joyce 1995).

Uncertainty in ecosystem response also results from the range of environmental conditions projected from the global circulation models (GCMs) commonly relied on for economic studies. Generally, however, tree physiology and growth are expected to be altered by increases in atmospheric CO_2 concentration and changes in temperature and water availability conditions (Field et al. 2007). How these changes interact with the limiting factors and existing ecosystem processes and linkages will differ by region. Over large areas, it may take many decades for the impacts of climate changes on forests to be evident. However, localized impacts (e.g., warming in sensitive alpine ecosystems) may be evident much faster.

Climate change is expected to affect current forests through magnified disturbance regimes and future forests through changes in growth rates, mortality rates, and seed production.

Bosworth et al. (2008) described a number of uncertainties related to how forest conditions may be influenced by climate change. First, there is potential for forest condition changes to result in feedbacks that may mitigate or enhance forest changes. For example, increased forest growth or migration of forests into areas largely nonforested may reduce surface albedo—increasing the localized warming that influenced initial growth increases or migration (see Chapin et al. 2002 and Thompson et al. 2009 for discussions). Second, increases in ozone, which can damage tree leaves and slow growth, may offset any increases in growth as a result of increased atmospheric CO_2, water availability, or growing season. Third, the impacts that increased vigor of invasive weeds, responding to improved growing conditions, may have on stand productivity is uncertain. There is some evidence that invasive weeds, similar to trees, respond well initially to increased levels of CO_2 (Ziska 2003). The items identified by Bosworth et al. (2008) are not explicitly incorporated in the economic studies considered here.

Conceptual Linkages Between Climate Conditions and the Forest Sector

To project how future climate conditions may impact the forest sector, economists must link expected forest changes to the inputs and parameters used in forest sector economic models. Conceptually, climate change could affect both the existing timber stands (the existing "stocks" of resources) as well as the incremental growth rates of existing and new timber stands (the "flows") (Sohngen and Sedjo 2005). Mortality, as a result of fire or insect disturbances or long-term changes in environmental conditions, may result in losses in the existing stocks of forests. At the same time, the "flows" of incremental growth in forest stands may increase or decrease as a result of the changes in growing conditions (e.g., temperature, precipitation, CO_2) from climate change. Impacts on the stocks and flows of forest resources, along with other factors such as changing management practices, will combine to yield the future forest conditions. Stated differently, climate change is expected to affect current forests through magnified disturbance regimes (e.g., fire, insects, and disease) and future forests through changes in growth rates, mortality rates, and seed production (especially in unmanaged stands) (Alig et al. 2004a). Combined, these factors will influence future forest conditions and forest management activities.

Humans (as consumers) are connected to forested ecosystems through the market (e.g., timber) and nonmarket (e.g., recreation opportunities) products and services we receive from forests. Changing forest conditions affect the supply of timber available for use in the production of wood products. Changes in supply of wood products are reflected in short-term changes in the prices of wood products, all else being equal. Increases in the price of wood products may reduce the consumption of wood products generally and the substitution for some wood products (e.g., dimensional lumber) by others (e.g., engineered wood products) or by nonwood products (e.g., steel 2 by 4s). Conversely, declines in wood product price would likely increase the consumption of wood products, all else being equal, and possibly result in the substitution of wood products for nonwood products.

Timber producers may alter their management strategies to take advantage of changing growing conditions and timber markets.

In addition to changes in the behavior of consumers that result from changing forest conditions, producers of wood products and timber may change their practices in response to changing forest conditions. Timber producers may alter their management strategies to take advantage of changing growing conditions and timber markets. For example, if incremental growth rates experience a marked increase, timber producers may extend rotation lengths to take advantage of additional revenue from growth that could be generated, net of forest management costs. Manufacturers may alter the types of products they produce in response to changes in the forest supply (e.g., a greater reliance on producing dimension lumber relative to engineered wood products or vice versa). Over longer timeframes, the wood products sector may change timber processing infrastructure and material handling systems in response to changes in the flow and quality of supplied timber.

Model Operation

Two general approaches have been adopted to examine the potential impacts of climate change on the forest sector using economic models. The first general approach uses the output of GCMs to inform changes to the forest growth rates (the flows) and forest extent and disturbance (the stocks) used as inputs in the economic model.

Often an intermediate model is used to translate the changing environmental conditions identified in the GCM to changes in net primary productivity (or other parameters) that can be used to modify inputs in the economic model (e.g., Joyce 1995, Perez-Garcia et al. 2002). In all the cases described in this study, timber yields in the economic model are modified to reflect the changing growing conditions under climate change (i.e., the flows) (e.g., Alig et al. 2002, Irland et al. 2001, Joyce 1995, Perez-Garcia et al. 2002, Sohngen and Sedjo 2005, Sohngen et al. 2001). A limited number of studies have also changed input parameters for existing stocks to reflect dieback of existing stands as a result of climate change (e.g., Sohngen and Mendelsohn 1998, Sohngen et al. 2001). The timing of the expected climate conditions is important because most forest sector economic models project conditions for many decades into the future. Typically, in the studies considered here (e.g., Mills and Haynes 1995, Perez-Garcia et al. 2002, Sohngen and Mendelsohn 1998), the GCM has been used to depict climate conditions mid-21st century (e.g., 2065) for the equilibrium that would result with atmospheric CO_2 at some projected future increased level (e.g., 625 parts per million [ppm]) (Shugart et al. 2003). A linear trend between current conditions and estimated future conditions is then often used as input to the intermediate model or as direct changes in the growth and yield parameters. A limited number of studies have estimated decadal conditions from GCMs as inputs to an ecosystem model and economic model (e.g., Irland et al. 2001).

Several factors could affect the availability of timber, including land-use changes; shifts in species distribution; dieback associated with heat, drought, and increased disturbance; and land ownership patterns.

The second approach to examining the impacts of climate change on forests using economic models is to use sensitivity analysis (e.g., McCarl et al. 2000). Rather than using the output of GCMs and intermediate models to identify new values for inputs and parameters under a changed climate in the economic model, the sensitivity analysis approach examines a range of new values for model inputs and parameter values. The response to these changes in inputs and parameters is then used to identify likely responses of the forest sector to changes in forest characteristics such as growth and yield. If the range of input values considered is large enough, a sensitivity analysis approach is useful in that it can provide “sideboards” on the likely future changes in the forest sector in response to climate change. The sensitivity approach is accommodating to changing expectations on forest growing conditions because a range of future conditions have already been simulated.

Anticipated Timber Availability Factors Under Climate Change

A number of factors will influence the availability of timber in a climate-changed environment. Perhaps most recognized are the anticipated changes in growth and yield of timber (the flows described above) as a result of climate change. However, several other factors could affect the availability of timber, including land-use changes; shifts in species distribution; dieback associated with heat, drought, and increased disturbance; and land ownership patterns.

Changes in Growth and Yield

In the studies considered here, over broad geographic areas, climate change is expected to increase the growth and yield of timber (e.g., Alig et al. 2004a, Joyce 1995, Sohngen et al. 2001). In the study by Irland et al. (2001), timber growth in the United States was projected to increase by 1 to 3 percent per decade relative to the baseline projections under all the scenarios considered. In their global analysis, Sohngen et al. (2001) projected global forest productivity increases of 29 and 38 percent relative to the baseline by 2145, for the University of Illinois at Urbana-Champaign and Hamburg climate scenarios available at the time. The increases projected for North America (17 percent) by 2145 were less than the global increase because the higher latitudes are projected to suffer some losses to productivity from climate change. In addition to the productivity increase, Sohngen et al. (2001) projected an increase in yield of timber (accounting for shifts of productive southern species northward) in North America, of between 34 and 41 percent by 2145. Perez-Garcia et al. (2002) assumed a direct relationship between forest carbon and forest productivity, and they projected forest carbon increases in the U.S. west coast of between 2 and 15 percent and from less than 2 to 10 percent in the East by 2040, with increasing forest carbon under higher CO_2 emissions/hotter temperature scenarios. In Perez-Garcia et al. (2002), softwoods were projected to have a greater positive response to climate change than hardwoods.

Changes in growth and yield are projected to differ by U.S. region, with the Northeast generally projected to benefit. The South is projected to lose productivity under some scenarios because of a combination of warmer temperatures and decreased water availability. Alig et al. (2002) estimated that growth rates in the Northeast would increase, such as 0.3 percent annually for oak/hickory, but growth rates would decline across the South. Sohngen and Sedjo (2005) stated the South appears to be the most "vulnerable region" in terms of forest growth and the forest sector economy because of climate change. Additionally, those authors suggested that the northern conifer forests and those in the Mountain West are most susceptible to damage because of a changing growing environment and increased disturbance.

Land-Use Changes

Most studies thus far have assumed that the total area of land in agriculture and the total area of land in forests will remain about the same even as climate changes.

Although the total area of land in agriculture and forestry may indeed remain relatively constant over time, climate change could alter the distribution of land uses over time. Given that forest areas could be affected and the potential human and social consequences of these impacts, it is important to consider and assess the implications of climate change on the distribution of U.S. land uses.

Differential impacts of climate change in agriculture and forestry could lead to land-use shifts as one possible adaptation strategy by landowners. For example, if climate change results in relatively higher agricultural productivity, some land may be converted from forests to agricultural use. Such changes would alter the supply of products to national and international markets, changing the prices of forest products and the economic well-being of both producers and consumers. If climate change affects yields and costs of production for forest stands and agricultural crops, land could shift between forestry and agricultural uses as these two sectors adjust to climate changes. Given that the agricultural and forestry sectors sometimes compete for the same land, shifts in productivity of agricultural land could affect the ultimate distribution of forest land, and vice versa.

Using four climate change scenarios from the national assessment of climate change, Alig et al. (2002) found that less forested area was projected under four climate scenarios relative to the base case (no climate change). Furthermore, less cropland and more pasture land were projected to convert to forests under all scenarios. With their modeling of the forest and agricultural sector and exogenous estimates of productivity impacts at the time, Alig et al. (2002) provided regional results. Although climate change is likely to affect the margin between forestry and agriculture in specific locations, aggregate productivity changes in forestry appear to outstrip aggregate productivity changes in agriculture.

In addition to effects from adaptation, climate change policies may involve mitigation actions that can markedly impact the land-use distribution in a region.

Economic impacts of climate change on land-use distribution could also involve human migration patterns and affect the area of urban and developed areas by region. Regional patterns of growth and decline in the United States have shifted population and property value to more vulnerable areas (van der Vink et al. 1998), and concerns about climate change and severe weather events could alter coastal settlement patterns. A large amount of uncertainty surrounds any such movement of population and land-use impacts.

In addition to effects from adaptation, climate change policies may involve mitigation actions that can markedly impact the land-use distribution in a region. Although outside the scope of this chapter, Alig in chapter 3 in this volume summarizes recent results from studies that have examined possible implications of different climate change mitigation policies for land use in the United States that would promote reduced deforestation and active afforestation on former agricultural land to increase net carbon sequestration strategies.

Species Shifts

Changes in forest type and tree species distributions could have a number of ecological and forest sector consequences. Within regions of the United States, the forest sector has developed infrastructure and management systems based on current forest type distribution and dominant market species. Changes in forest type and species distribution may, over the long term, lead to changes in the equipment used in harvesting and processing timber and in forest management practices. Shifts in the distribution of species in response to changing habitat conditions may occur as a result of natural migration of species in response to changes in growing conditions or, perhaps more likely, as a result of changes in forest management and the tree species selected for reforestation or afforestation. Changes in tree species distribution could include land-use changes, if forests are established on land currently used for agriculture or other uses.

Shifts in the distribution of species may occur as a result of natural migration of species in response to changes in growing conditions or as a result of change in forest management and the tree species selected for reforestation or afforestation.

Modeling the projected shift in suitable habitat over the next century or more, Iverson et al. (2008) projected shifts northeastward of many eastern tree species. Some species were projected to move up to 800 kilometers (500 miles) under the highest emissions/hottest temperature scenario considered. In addition to shifting suitable habitat location, the area of suitable habitat for individual tree species may also change. About half of the species considered by Iverson et al. (2008) were projected to have an increase in habitat area. Both oak and pine species were projected to experience increases in habitat area, with a stretch toward the northeast. Losses in habitat area in the United States were projected for the northernmost forest types, including the maple-beech-birch, spruce-fir, and aspen-birch types (fig. 1-1). Those forest types are projected to have expansions in suitable habitat area in Canada. The types of shifts identified by Iverson et al. (2008) are reflected generally in the inputs in some of the economic models considered here (e.g., Sohngen et al. 2001). Iverson et al. (2008) noted that it is very unlikely that expanding tree species would widely colonize newly suitable habitat without human intervention.

Dieback and Disturbance Regimes

In addition to the changes that may occur in the flow of resources, the stocks of current resources may be reduced through dieback that results from climate change. Dieback is typically modeled as a reduction in the suitability of growing conditions or as increased mortality from more frequent or more severe disturbance. Dieback from change in growing conditions is anticipated to result from increased heat and reduced water availability. The southern United States is thought to be the U.S. area with the greatest potential to experience limited water availability and heat-induced dieback in timber over the next several decades, although increased growth

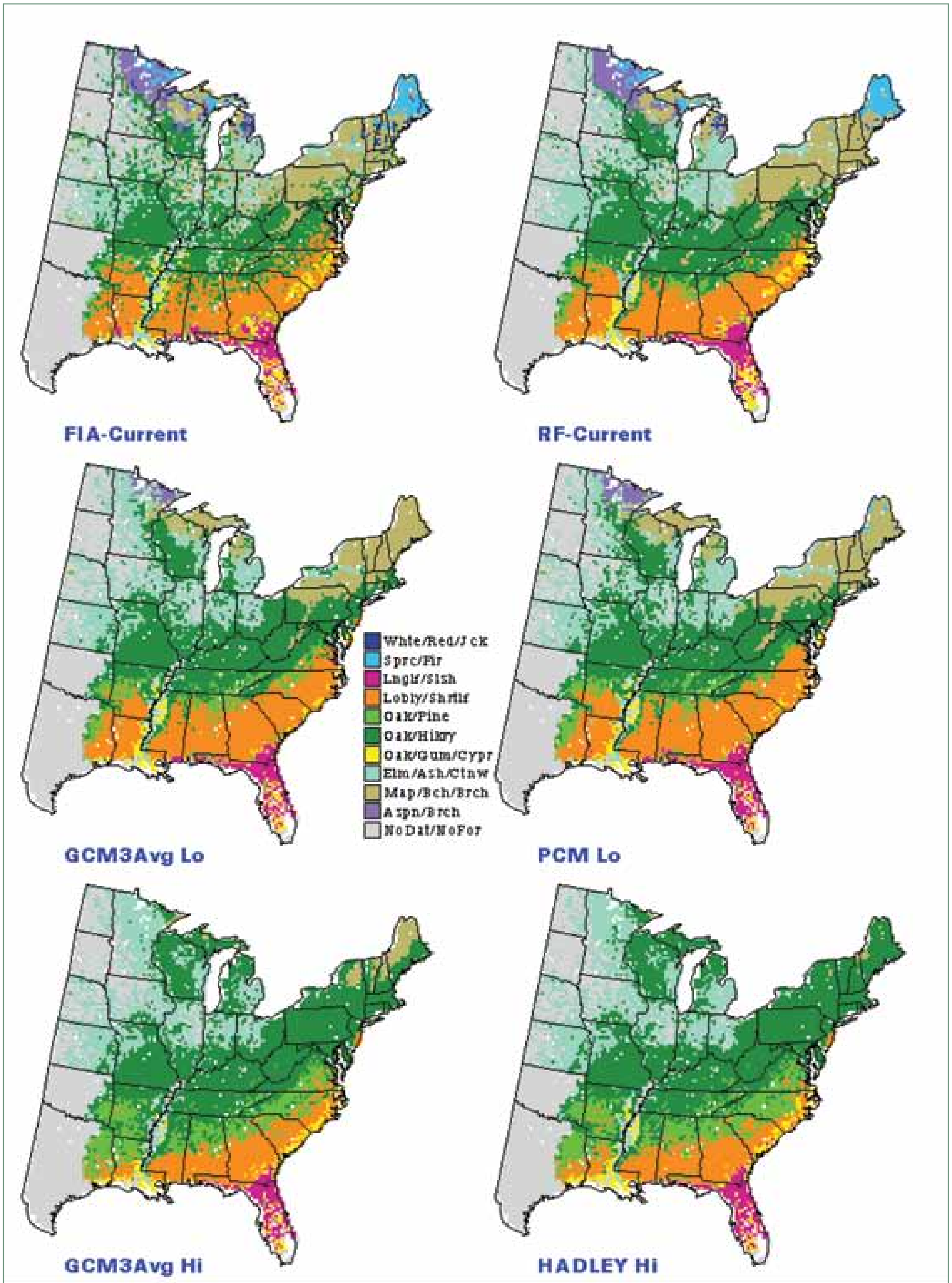


Figure 1-1—Current and projected forest type shifts in the Eastern United States as a result of climate change under low and high emissions scenarios. FIA = Forest Inventory and Analysis; GCM3_Avg = average of Hadley, PCM, and Geophysical Fluid Dynamics Laboratory global circulation models, PCM = Parallel Climate Model, RF = Random Forest Regression Model. Data source: Climate Change Tree Atlas: http://www.nrs.fs.fed.us/atlas/tree/ft_summary.html.

is projected later in the decade as managers respond to changing environmental conditions (e.g., Alig et al. 2004b).

The IPCC has expressed high confidence that the North American forest sector will likely be sensitive to changes in disturbance regimes from climate change (Field et al. 2007). Increases in disturbance are most frequently projected for the Western United States via more frequent or severe wildfires and insect and disease outbreaks (e.g., Bosworth et al. 2008). Citing the research of others, the IPCC suggested that the period of high risk for wildfire ignitions could increase by 10 to 30 percent and burned area could double under climate change (Field et al. 2007). Gan (2004) found that the infestation risk of the southern pine beetle (*Dendroctonus frontalis* Zimmermann) may increase by 2.5 to 5 times under changed climate conditions. If southern pine shifts northward, the risk might be 4 to 7.5 times the present risk. Research completed in the Pacific Northwest has found that increased winter temperature and spring precipitation have contributed to the occurrence of Swiss needle cast (*Phaeocryptopus gaumannii*) disease (Stone et al. 2008). Anticipated continued increases in temperature and precipitation from climate change are expected to lead to increasing spread and severity of that disease. Increased disturbance under climate change also includes the potential for increased frequency and magnitude of wind events (e.g., windthrow) and ice storms, which are more often associated with eastern forests.

Disturbance events can be modeled through changes in the existing stocks of forest resources. Historical dieback patterns are implicit in the growth and yield functions used to project future forest volumes. In the economic modeling literature, few studies account for dieback events that depart from historical patterns. However, the IPCC has (Field 2007) "very high confidence" (i.e., > 90 percent agreement in statement) that disturbances are currently increasing relative to historical patterns and will continue to increase. Currently, Sohngen (e.g., Sohngen and Mendelsohn 1998, Sohngen et al. 2001) has the most explicit inclusion of additional dieback anticipated from climate change. Under the assumptions adopted in Sohngen et al. (2001), 75 percent of the trees killed from dieback are available for timber salvage at their current volume. Future increases in value that would have occurred for the killed stocks are lost in the Sohngen et al. (2001) treatment.

One potential positive outcome of dieback is that timber producers can replant affected stands with species or varieties appropriate for the new growing conditions (e.g., Joyce 2007). Sedjo and Sohngen (1998) pointed out that although extensive dieback could result in "substantial damage" if forests were unable to provide their current levels of ecosystem services, natural systems tend to respond quickly to disturbance. Additionally, if dieback were to increase significantly from historical levels, we could expect increased human intervention to mitigate at least some of the impact of that disturbance. This pattern has been exhibited in the increased attention to fuels treatment and expansion in fire suppression capacity as a result of previous years' increases in wildfire costs and burned areas. In later sections, when possible, model results are discussed both with and without dieback outcomes.

Public and Private Timberland Ownership

If climate change were to have greater negative impacts on growth and yield on private lands, this could magnify the consequences for U.S. timber production.

The patterns of forest ownership may also factor into future timber availability under climate change conditions. Land ownership may influence timber availability because of differential impacts to growing conditions on lands in different ownership groups and because different ownership groups may respond in divergent ways to changes in forest growing conditions. Private timberlands provide the vast majority of the timber currently produced in the United States, with much of the timber coming from private lands in the South (Adams et al. 2006). If climate change were to have greater negative impacts on growth and yield on private lands, this could magnify the consequences for U.S. timber production. Currently, the economic studies considered here do not explicitly incorporate land ownership patterns into projections of climate change growth and yield; however, in general, most timber is assumed to come from privately owned timberlands (e.g., Irland et al. 2001, McCarl et al. 2000).

Differential impacts, by land ownership, from climate change in the United States are most likely in topographically rich landscapes, but could occur anywhere with systematic spatial patterns of land ownership. In many areas of the Western United States, private lands are concentrated at lower elevations, and public lands are concentrated at higher elevations. If climate change has a differential impact on the growing conditions of low-elevation forests, the private/public pattern of land ownership in the West could impact future timber production. In Washington and Oregon, Latta et al. (2010) have projected that private forest lands at low elevations would experience declines in productivity, while the higher elevation forests (in public ownership) would experience productivity increases. If such systematic differences are not adequately captured in economic models, gains in timber production, as estimated from the models, could be reduced because private timberlands provide most of the timber but suffer under climate change. However, because the West is most likely to experience systematic ownership differences in forest growing conditions from climate change and the East currently accounts for the majority of U.S. timberland and timber production, the impact of systematic ownership patterns on projected timber production is probably minor.

Private individuals and corporations own forest land for a variety of reasons. Individuals who own timberland tend to do so for reasons other than timber production, such as aesthetics, privacy, and recreation (Smith et al. 2009). Private individuals are much less likely to have written forest management plans. Corporations tend to manage land for financial returns, including returns from timber production. Because their ownership objectives and management capacities differ, the responses of private individuals to changes in forest conditions may be different from those of private corporations. It is possible, that private corporations may respond aggressively with mitigating activities to changing forest conditions and disturbance, whereas private individuals respond more passively. None of the modeling studies considered here differentiates growth and yield changes or dieback magnitude under climate change by type of private ownership. In considering mitigating behaviors and future conditions, it is useful to consider that different strategies for responding to climate change (e.g., planting newly suitable species, responding to increased disturbance regimes) may need to be tailored for the different private ownership groups.

Timber Management Under Climate Change

Where timber yields increase, timber production is generally projected to increase and stumpage prices to decline.

With climate change, forest management activities by producers, including choices of planting stock, thinning regimes, and harvesting practices, could be changed to take advantage of both new growing conditions and changes in forest sector markets (Alig et al. 2004a). For example, private timberland owners suffering production declines as a result of climate change (and wishing to maintain current production levels) would need to intensify management systems (e.g., planting improved stock or conducting more aggressive thinning or fertilizing activities) (Latta et al. 2010). Conversely, landowners facing increased productivity may need to change management regimes to accommodate increased growth or to take advantage of other market opportunities (e.g., carbon offset payments). In addition to forest management activities during the rotation, harvesting choices (e.g., rotation ages, silviculture systems) could also be modified in response to changing growing conditions or forest product markets (Alig et al. 2004a).

Timber Rotation Lengths

In general, the yields of forest communities in North America are expected to increase because of the fertilization effect of CO_2 and a longer growing season (Field et al. 2007). In those places where timber yields increase, timber production is generally projected to increase and stumpage prices are projected to decline. This results in lengthened timber rotations relative to current practice (Irland et al. 2001, McCarl et al. 2000). Timber rotation is the length of time producers allow timber to grow prior to harvest. If the opposite climate impact occurs and forest productivity declines, rotation lengths are expected to shorten, particularly over the short term. Rotation lengths shorten because timber supply is reduced (leading to increased stumpage values) and the annual growth of trees (representing the opportunity cost of forgoing future additional stumpage value) is less than under previous growing conditions. Additionally, timber rotation lengths could shorten if disturbance regimes increase markedly, reducing timber supply, or because producers choose to harvest sooner to avoid risk of timber losses to disturbance.

For the U.S. South, where productivity losses appear to be most likely to occur (although not certain), McCarl et al. (2000) projected that a 1 percent reduction in growth would shorten rotation length in that region by about 0.2 percent for the first 20 years after yield reduction, regardless of what happens to forest growth in other U.S. regions. Two decades post-change, McCarl et al. (2000) projected rotation lengths in the South would decrease further, with a mostly linear continued shortening in length as decades progress. If yields in southern U.S. forests were unchanged and northern forest yields increased, rotation lengths in the South were projected to remain largely unchanged (McCarl et al. 2000). In the North, rotation lengths were expected to increase by about 0.1 percent for the first 25 years under almost all scenarios (McCarl et al. 2000). However, when there is no change in timber growth in the South and the North experiences an increase in timber growth, rotation lengths were projected to remain largely unchanged in the North (and the South).

Harvest Levels

Under climate change, Perez-Garcia et al. (2002) projected that the global forest sector will increase harvest levels by 1.5 to 2.7 percent above the baseline by year 2040. The authors noted these global changes are very small, but some regional changes are greater. The Southern Hemisphere accounts for the greatest increases in harvest levels. For example, timber harvest in Chile was expected to increase by between 10 and 13 percent by 2040 relative to the baseline depending on the scenario (fig. 1-2). New Zealand was projected to increase harvest by 8 to 12 percent relative to the baseline. In the United States, the West was projected to increase harvest between 2 and 11 percent, with the greatest increases projected under the hottest/highest emission scenarios (Perez-Garcia et al. 2002). The U.S. South is projected to increase harvests under the moderate and high heat/emissions scenarios but reduce harvest under the lowest heat/emissions scenario. This projected reduction reflects reduced prices because other global regions are able to take greater advantage of changing growing conditions and increase timber production. Canada is projected to reduce harvest levels by up to 3 percent relative to the baseline under all scenarios (Perez-Garcia et al. 2002).

The results of the global study completed by Sohngen et al. (2001) are generally consistent with those of Perez-Garcia et al. (2002) for projected harvest levels over the next several decades. Sohngen et al. (2001) projected a 5 to 6 percent increase in harvests globally (relative to the baseline and depending on scenario) for the 1995 to 2145 period. Sohngen et al. (2001) also showed that most of the gains in timber harvest over the next several decades occur in the low mid-latitude forests, particularly in South America (10 to 19 percent) and India (14 to 22 percent). In the near decades, Sohngen et al. (2001) projected that North American harvests (Canada and United States combined) will decline by about 1 percent. This decline reflects some of the assumed dieback in the Sohngen et al. (2001) model and the general productivity losses projected for Canada. Global timber harvests in the latter half of the century are projected to have a more substantial increase, relative to the baseline, of between 18 and 21 percent. These later gains reflect the increased productivity of forests and increased demand for wood products in response to decreased prices. Most of this later-century increase is driven again by the low mid-latitude forests; however, North American harvests are projected to be about 14 percent above the baseline during that period.

In one of the initial studies examining the effect of climate change on the U.S. forest sector, Mills and Haynes (1995) projected that U.S. harvests would increase

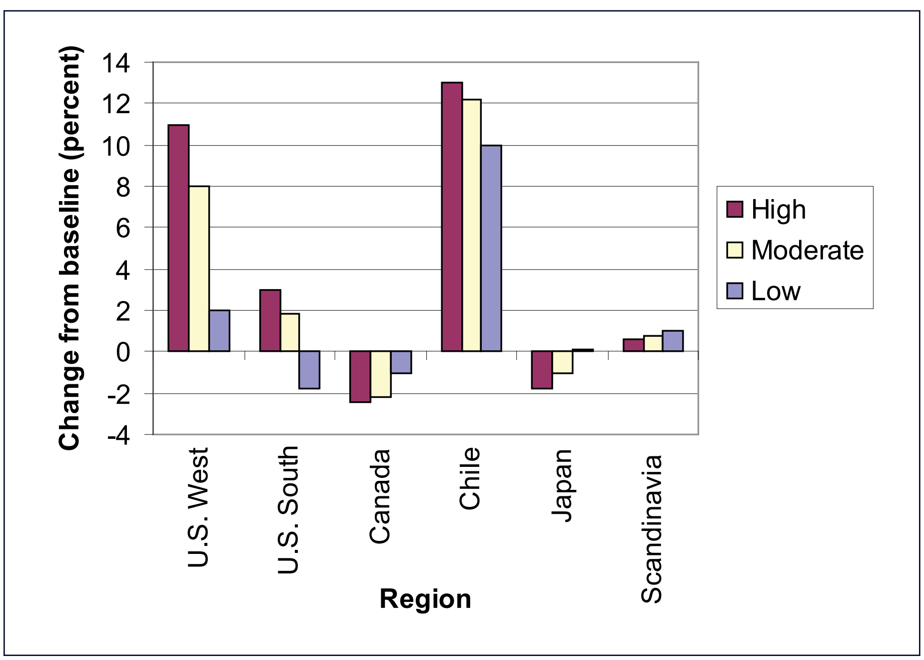


Figure 1-2—Approximation of projected harvest levels in 2040 under three carbon dioxide emission/temperature scenarios. Adapted from Perez-Garcia et al. 2002.

by 1 to 3 percent by 2040 relative to the baseline under climate change. The authors reported that most of the projected increase would take place several decades after climate-induced growth increases began. Regionally, Mills and Haynes (1995) projected that harvest levels would increase in the South and West (with slightly greater increases in the South owing to model assumptions) but decrease in the North. These projected increases in harvest levels in the South and West are consistent with the finding of Perez-Garcia et al. (2002) in their moderate and high emissions/temperature scenarios. In the Mills and Haynes (1995) study, harvest shifted toward regions with established production capacity and lower costs of production. Industry timberlands, relative to nonindustrial lands, experienced the greatest gains in timber harvest in the Mills and Haynes (1995) study. In the Pacific Coast States, harvest levels were projected to decline on nonindustrial lands, although industry timber harvest was projected to increase. Irland et al. (2001), using a dynamic optimization model, also found that total U.S. timber harvests would increase slightly under climate change, regardless of the climate change scenario. The South has the most consistent harvest gains across the climate scenarios modeled by Irland et al. (2001).

Although dieback mitigated the price changes to some extent, the modeled dieback did not dramatically change the general relationship between climate change and timber prices.

Timber Price Changes

In general, forest sector prices are expected to decline as a result of climate change (e.g., Irland et al. 2001, Perez-Garcia et al. 2002, Sedjo and Sohngen 1998, Sohngen et al. 2001). In the Sohngen et al. (2001) study, global timber prices, under all scenarios (including those with dieback) decline relative to the baseline (fig. 1-3). Prices are projected to depart from the baseline in a mostly linear fashion between the present time and 2050. In the presence of dieback, prices in Sohngen et al. (2001) are slightly closer to but still below the baseline. Although dieback mitigated the price changes to some extent, the modeled dieback did not dramatically change the general relationship between climate change and timber prices. In the Sohngen et al. (2001) model, the greatest departures from baseline prices were projected to occur post 2060. In the first 20 years of the Sohngen et al. (2001) simulation, prices remain close to the baseline projections, allowing timber producers in low mid-latitudes to take advantage of climate-change-induced yield increases and increase output.

In the Perez-Garcia et al. (2002) model, global timber prices under climate change are projected to be 0.8 to 3.1 percent below the baseline projection in 2040. These timber prices are projected to translate to wood product price decline. Perez-Garcia et al. (2002) pointed out, as with the projected harvest changes described above, projected global price changes are minimal, although there are larger

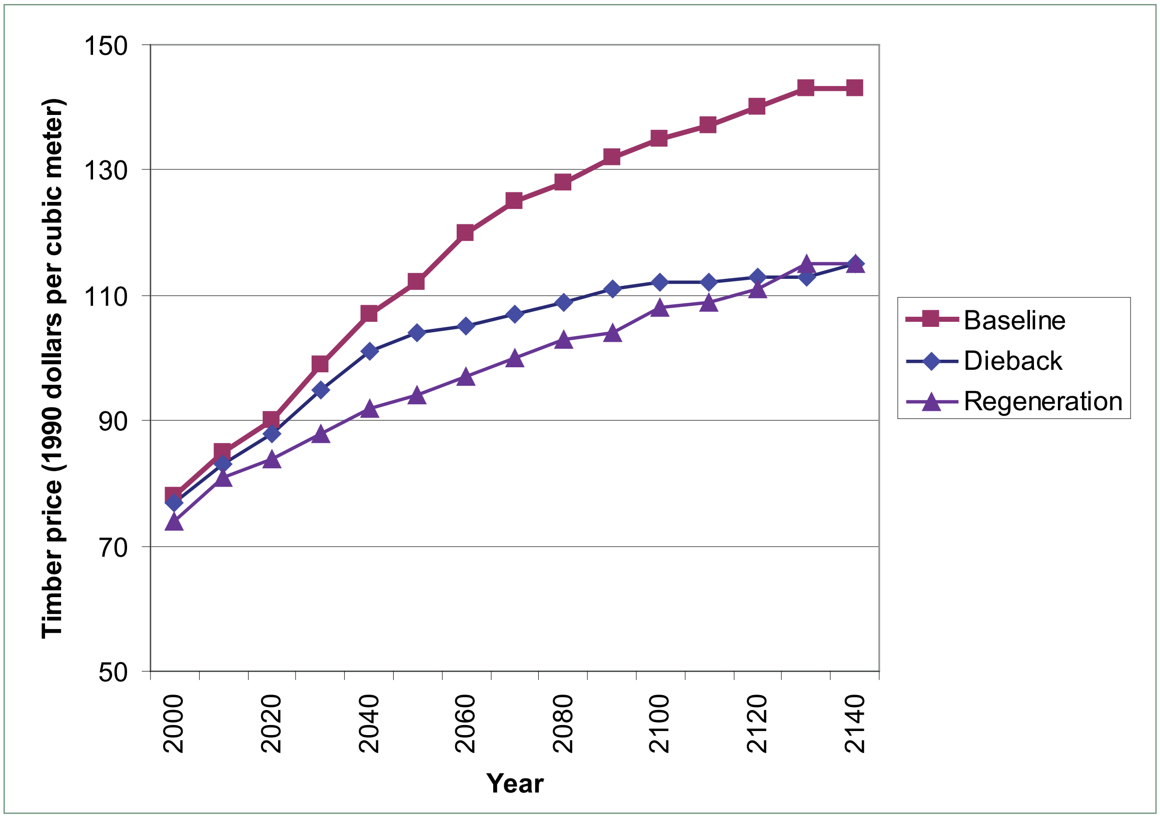


Figure 1-3—Projected timber prices under baseline and two climate change scenarios, including two with forest dieback. Adapted from Sohngen et al. 2001.

regional changes. As the climate scenarios represent even hotter temperatures/ higher emissions, prices are projected to decline farther below the baseline because timber productivity continues to increase in modeled scenarios.

For the United States, Perez-Garcia et al. (2002) projected price declines in the U.S. South and West ranging from less than 1 to about 3.5 percent (fig. 1-4). Under the smallest temperature changes, projected U.S. price declines are 0.5 percent or less. Similar patterns are expected for the U.S. North. In the Perez-Garcia et al. (2002) model, the greatest price declines were projected for Scandinavia and Western Europe (not shown). Studies completed for only the United States have found slightly greater projected price declines than those projected in the global models. In a study by Mills and Haynes (1995), stumpage prices are projected to decline by 6 to 35 percent under climate change scenarios (Joyce 1995). In a later study, Alig et al. (2002) projected prices for sawtimber to decline by 3 to 6 percent relative to the baseline for the period 2020 to 2050 under climate change.

The price impact results of the sensitivity analysis by McCarl et al. (2000) depart slightly from the other studies. For the United States, prices for timber are projected to slightly decline, relative to the baseline, in scenarios where the South

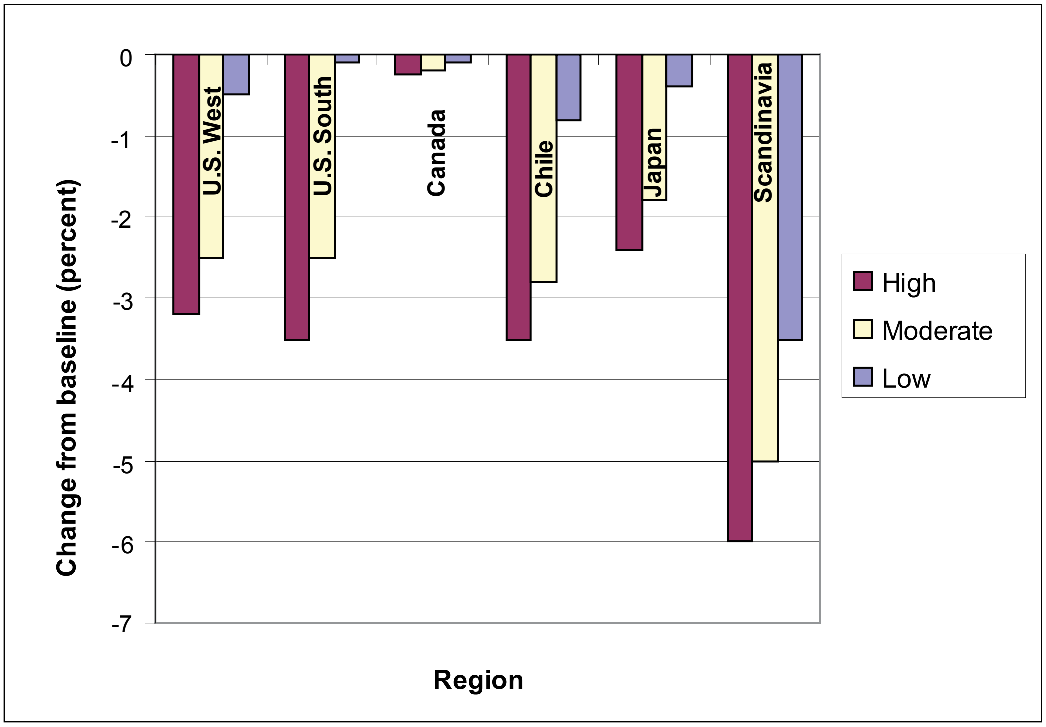


Figure 1-4—Approximation of projected log price changes in 2040 under three carbon dioxide emission/temperature scenarios. Adapted from Perez-Garcia et al. 2002.

experiences no change in growth and the North increased productivity. This pattern of growth change is one possible outcome of climate change. If the South experienced a 1 percent loss of productivity and the North experienced a corresponding gain in productivity, McCarl et al. (2000) projected that prices would increase slightly (less than 1 percent). In cases where both U.S. regions experienced a loss in productivity, greater price increases were projected. Note that the McCarl et al. (2000) study included forest sector imports only from Canada, potentially limiting the price-mitigating effect of forest sector imports. In the Sohngen et al. (2001) and Perez-Garcia et al. (2002) models, low mid-latitude timber producers would be the primary gainers (and timber exporters) under climate change and Canada would suffer production losses.

Because softwoods are projected to have greater increases in productivity than hardwoods (e.g., Perez-Garcia et al. 2002) in response to climate change, softwood prices may experience greater price declines (Mills and Haynes 1995). Regionally, Mills and Haynes (1995) projected that softwood stumpage prices might experience the greatest declines in the U.S. South and Pacific Coast regions. The other studies considered here do not report results separately for hardwoods and softwoods, often because the model results are similar for the two forest types (e.g., Irland et al. 2001, McCarl et al. 2000).

The Pacific Northwest has relatively more mitigation opportunities through altered forest management as compared to afforestation.

Climate change is projected to generally lead to faster growing trees that become larger in a shorter period of time.

Climate change is projected to generally lead to faster growing trees that become larger in a shorter period of time. Further, at least one study (Mills and Haynes 1995) suggested that softwoods may experience a disproportionately positive response. These factors combine to suggest some potential differences between sawtimber and pulpwood production in a climate-changed environment. Pulpwood is typically produced from trees that are too small or not of high enough quality to produce sawtimber. In the Northern United States, much of the pulpwood is produced from hardwood species. As climate change produces trees that can become larger more quickly, potentially displaces hardwoods north to Canada, and potentially yields feasible growing conditions in the Northern United States for productive southern pine species, the production of pulpwood is projected to decline. Irland et al. (2001) projected declines in pulpwood harvest of approximately 3 percent during the 2020 to 2050 period, regardless of which climate scenario is considered. Because of decreased production, pulpwood prices are projected to increase during this period (Irland et al. 2001). Irland et al. (2001) projected that in the 2000 to 2020 period, pulpwood prices would remain generally unchanged. Note that even with pulpwood price increases and sawtimber price declines in the 2020 to 2050 period, sawtimber prices are still high enough and timber growth rates fast enough to increase sawtimber production and decrease pulpwood production.

Consumer and Producer Welfare Under Climate Change

The basic relationships between changes in forest productivity and consumer and producer welfare are fairly straightforward and consistent among the studies considered. As Sohngen and Sedjo (2005) stated, "…if climate change makes forests more productive, then timber prices will fall, consumers will benefit (consumer welfare will rise relative to the baseline) and (forest product) producers will lose (producer welfare will decline from the baseline)." The opposite will occur if climate change makes forests less productive. In nearly all results, under a positive change to forest productivity, total welfare (the net combination of consumer and producer welfare) in the United States is projected to slightly increase from the baseline because the gains to consumer welfare are greater than the losses to producer welfare. Alig et al. (2002) projected U.S. total welfare increase of between 0.05 and 0.18 percent. Using a global economic model, Sohngen et al. (2001) estimated total welfare in North America would increase by $55 to $65 billion, depending on the climate scenario and not accounting for any potential forest dieback. The greater total welfare gains in Sohngen et al. (2001) can be traced to the projected global increase in timber supply, which leads to even larger consumer surplus gains relative to the baseline.

In nearly all cases of positive change to forest productivity, total welfare in the United States is projected to slightly increase from the baseline because gains to consumer welfare are greater than losses to producer welfare.

Although welfare gains are projected in aggregate for the United States as a whole, some locations and groups may suffer losses. Within the United States, the South and West regions and timber and wood product producers are projected to suffer welfare losses under some climate scenarios. In the Perez-Garcia et al. (2002) model, total welfare is positive under climate change relative to the baseline in the U.S. North and West (table 1-1). Both regions are major consumers of wood products, and the gains to those individuals overcome losses to producers. However, the U.S. South is currently the major timber-producing region in the United States (Adams et al. 2006), and because timber producers suffer losses relative to the baseline, that region is projected to experience total welfare losses under a climate scenario of 592 ppm CO_2 in 2100 and a temperature rise of 1.6 degrees Celsius. Under higher CO_2 and temperature levels, Perez-Garcia et al. (2002) projected greater gains in consumer surplus in the South, offsetting producer losses and yielding small gains in total welfare. However, in scenarios where the South experiences net gains in total welfare, that region's gains are about one-third to one-half the gains experienced in the North or West.

In the Perez-Garcia et al. (2002) model, consumers in all U.S. regions experienced a gain in welfare regardless of climate scenario (table 1-1). Conversely, log producers in each U.S. region were projected to experience losses in producers' surplus, relative to the baseline, because of declines in timber prices as a result of gains in yield. Wood product producers in the West were projected to experience

Table 1-1—Projected net present welfare changes in U.S. regions under three climate emissions/temperature scenarios, 1985 to 2040

Regions	High	Moderate	Low
	Million dollars (1993)		
North:			
Log producers	-922.1	-741.6	-225.4
Product producers	-104.6	-125.3	-144.1
Consumers	8,467.5	6,279.4	806.9
Total	7,440.8	5,412.5	437.3
South:			
Log producers	-5,062.5	-4,946.6	-4,234.4
Product producers	-136.9	-511.1	-1,285.1
Consumers	9,605.8	7,219.1	1,238.0
Total	4,406.4	1,761.3	-4,281.5
West:			
Log producers	-1,524.5	-1,030.1	-267.0
Product producers	8,261.0	6,141.3	829.6
Consumers	241.8	154.8	340.6
Total	6,978.4	5,266.0	903.3

Adapted from Perez-Garcia et al. 2002.

slight gains in surplus relative to the baseline. In the other regions, wood product producers are projected to have surplus losses under nearly all scenarios. The only exception to that pattern is a projected slight gain in wood product producer surplus in the Southern United States under the highest CO_2 scenario.

Because, in part, consumers can change their purchasing behavior and choose substitute goods, the welfare of producers is about 10 times as sensitive to changes in growth and yield (such as in response to climate change) as consumer welfare (McCarl et al. 2000). If existing stands suffer mortality because of changes in climate conditions and increased disturbance, producers experience greater losses. Sohngen and Sedjo (2005) projected that producers' surplus in North America could decline by $1.4 to $2.1 billion per year relative to the baseline in a scenario where existing stocks are subject to dieback. In a scenario without dieback, producer losses are about 30 percent less (Sohngen and Sedjo 2005).

Globally, Perez-Garcia et al. (2002) projected small total welfare changes of between 0.4 percent ($1.8 billion) and 0.44 percent ($15.8 billion) in response to climate change. Sohngen et al. (2001) found slightly larger changes in global total welfare, relative to baseline projections, of 3 percent ($113 billion) if dieback occurs to 6.7 percent ($251 billion) without dieback. In both cases, as found in the U.S.-only models, global welfare gains result from the lower cost of wood products to consumers that overcome welfare losses to producers. Sohngen et al. (2001) projected the largest gains in consumer welfare will accrue to North America, Europe, and the former Soviet Union (table 1-2). Across all the scenarios considered, producers in South America and China are projected to experience the greatest gains in welfare relative to the baseline. Producers in the United States are projected to experience the largest nominal losses relative to the baseline (however the authors do not report the percentage change).

Timing of Welfare Changes

The welfare measures reported above are in net present value terms. That is, they are the sum of welfare changes from the baseline for all the future periods in the economic simulation, discounted to the present day. Through the discounting, changes that occur in earlier decades have more weight than changes in later decades. In any one period in the future, the welfare changes experienced by producers and consumers may differ from that projected for the entire simulation period. For example, although producers are projected to suffer losses when the whole simulation period is considered, they might experience gains in welfare in some decades. The expected temporal patterns of welfare changes are uncertain.

Potential impacts of climate change on urban forests are important because about 80 percent of the U.S. population lives in urban areas, and this proportion will increase.

Climate Change and Urban Forests

In addition to the possible effects of climate change on the U.S. forest products sector, climate change may have significant impacts on urban forests. Although the projected effects of climate change on rural forests have been discussed extensively elsewhere, fewer studies have summarized the potential impacts to urban forests from climate change. The potential impacts on urban forests are important because approximately 80 percent of the U.S. population lives in urban areas, and

Table 1-2—Projected changes in the net present value of welfare for consumers and producers under climate scenarios with and without forest dieback, relative to a baseline, 1995 to 2145

Scenarios	North America	Europe	Former USSR	China	South America	India	Asia-Pacific
				Billion dollars (1990)			
Without dieback:							
Consumer welfare	80.3	44.5	37	17.2	17.5	4.2	26.2
Producer welfare	-24.7	5.6	-0.2	5.5	2.3	1.6	-7.5
Total welfare	55.5	50.1	36.8	22.7	19.8	5.7	18.7
With dieback:							
Consumer welfare	35	19.5	16.2	7.7	7.8	1.9	11.8
Producer welfare	-39.3	25.8	-24.6	8.6	14.7	3.8	3.3
Total welfare	-4.3	45.3	-8.4	16.4	22.6	5.7	15.1

Adapted from Sohngen et al. 2001.

this proportion will increase in the decades ahead. This section briefly reviews the extent and value of urban forests, documents threats to urban forests, and suggests how climate change may affect those threats.

The urban forest (i.e., all trees and associated natural resources within urban areas) of the United States is an extensive and valuable natural resource. Nowak et al. (2001) estimated 3.8 billion trees grew in 281 000 km^2 (108,500 mi^2) of urban area in the coterminous United States in the 1990s. Urban and developed areas are projected to exceed 500 000 km^2 (193,000 mi^2) by the year 2030, with almost half of this urban growth taking place in forested areas (Alig et al. 2004b, Nowak and Walton 2005, Nowak et al. 2010). Urban forests provide a wide range of benefits, including protection against soil erosion, provision of habitat for wildlife, improvement in local air quality, reductions in the urban heat island effect, energy savings through building shading (Donovan and Butry 2009) and insulation, carbon sequestration, and reductions in stormwater runoff (e.g., Dwyer et al. 1992, McPherson et al. 2005). Urban tree cover also provides cultural benefits that lead to improved quality of urban life as trees may improve the scenic quality of a city neighborhood, provide privacy, reduce stress, and shelter residents from the negative effects of undesirable land uses (e.g., Dwyer et al. 1991, Westphal 2003).

It is difficult to put an economic value on the environmental and cultural benefits of urban forests because most of those services are not traded in markets. Nevertheless, some of the benefits of urban forests may be capitalized into the values of residential property, and hedonic property price models document those values based on property characteristics and home sale prices (Donovan and Butry 2010). Summarizing studies of home sales in several U.S. cities, Sander et al. (in press) concluded that increasing proximity to forested areas and increasing tree cover are associated with increasing home sale price. For example, in Grand Rapids, Michigan, housing lots that directly bordered a forest preserve sold for 19 to 35 percent higher prices than other lots (Thorsnes 2002). A North Carolina study found that increasing forest cover by 10 percent on a forest parcel increased home sale price by an average of $800 (Mansfield et al. 2005). A Minnesota study found that a 10 percent increase in tree cover within approximately 100 m of a home increased home sale price by $1,371 and within 250 m increased home sale price by $836 (Sander et al., in press). The positive impact of trees on home sale price together with the size of the U.S. housing stock (115 million housing units in 2000) (Radeloff et al., in press) suggest that the total impact of trees on residential property value in the United States is very large.

Invasive insects and pathogens are among the greatest threats to urban forests and can have substantial economic effects. In a comprehensive study of the economic impacts of biological invasions of forests in the continental United States, Aukema et al. (2010) concluded that local governments and homeowners are the sectors sustaining the greatest economic damage, which includes expenditures for treatment, removal, and replacement of infested trees and reductions in property value associated with tree mortality. These governmental and residential expenditures represent transfers of wealth between sectors (such as from homeowners to tree removal firms), and impacts on residential property values represent wealth that is lost from the economy. For example, emerald ash borer (EAB) (*Agrilus planipennis* Fairmaire), a phloem-feeding beetle native to Asia and introduced in the United States in the 1990s, is projected to cause average annual expenditures of more than $1 billion for treatment and replacement of trees by local governments and homeowners in the Eastern United States from 2009 to 2018 (Kovacs et al. 2010a). Residential property value losses associated with EAB damage are projected to exceed $340 million annually (Aukema et al. 2010). Sudden oak death (*Phytophthora ramorum*), a nonindigenous forest pathogen that causes substantial mortality in coast live oak (*Quercus agrifolia* Née) and several other oak tree species on the Pacific Coast of the United States, is projected to cost $6 million per year in treatment, removal, and replacement costs and $105 million per year in property value losses to single-family homes (Kovacs et al. 2010b).

Wildfire is another significant threat to urban forests with substantial economic effects. The wildland-urban interface (WUI) is the area where houses meet or intermingle with wildland vegetation, including trees, shrubs, and grass (Stewart et al. 2007). According to recent estimates, the WUI encompassed 11 percent of the land area (715 100 km^2 or 276,100 mi^2) and 38 percent of the housing units (44.3 million) in the contiguous United States in 2000 (Radeloff et al. 2005). In Western and Southeastern States, where wildfires burn the most area, 45 percent of the housing units are in the WUI (11.1 and 4.1 million units, respectively) (Hammer et al. 2009). Although wildfire risk varies widely, the presence of homes in fire-prone vegetation increases the risk of loss of life and property and increases fire prevention and suppression costs. For example, wildfires in the WUI of southern California destroyed 3,079 structures in 2007, and suppression costs to the state totaled nearly $300 million (Hammer et al. 2009).

Conclusions

Although there remains uncertainty in the physiological and disturbance responses of U.S. forests to climate change, there is general agreement in results from currently available studies examining the impact of climate change on the forest sector. Broadly, the model results included here indicate that the forest sector (both globally and in the United States) is fairly resilient to changes in forest stocks and growing conditions resulting from the modeled climate change scenarios to date. Although there are projected to be impacts to forest production, forest sector prices, and consumer and producer welfare, changes are generally projected to be small. Currently the forest sector (globally and in the United States) is operating in a manner that reflects a diverse arrangement of resources, processing capacity, and consumer demand. Climate change would likely impact those arrangements and, over time, economic theory and the output of economic models suggest the forest sector would adapt accordingly.

The Northern United States are generally projected to experience productivity increases with climate change. However, some of this increase may coincide with a displacement of some currently important northern species (northern hardwoods and spruce/fir forest types) north to Canada because of changing growing conditions. Concurrently, changing growing conditions may make way for some productive southern pine species to be planted in portions of the U.S. North. The Western United States is generally expected to see gains, particularly in the timber-important Pacific Coast States. However, the West is also generally the focus of concerns related to increased disturbance, in the form of increased wildfire or insect and disease outbreaks, because of climate change. Whether these disturbances will be mitigated by human intervention or depart significantly from the general long-term historical levels remains to be seen. Within the United States, the South is generally considered the region most likely to suffer growth losses because of changing climate conditions. Dieback and increased disturbance mortality may also impact existing forest stands in that region.

The South is generally considered the region most likely to suffer growth losses because of changing climate. . . Studies are generally consistent in projecting that climate change will lead to aggregate yield increases globally and for portions of the United States.

The economic studies considering the impact to the forest sector from climate change are generally consistent in projecting, in model inputs, that climate change will lead to aggregate yield increases globally and for portions of the United States. These yield increases will lead to increased timber production, which will result in price declines. Timber harvest increases in the United States are most consistently projected for the northern and western regions. The U.S. South is projected to have increased harvest under some scenarios and decreased harvest in others. In one study, the South is projected to remain near baseline harvest levels only in a scenario when that region's productivity remains stable and the North increases productivity. In another study, the South is projected to increase harvest under the hottest temperature/highest emission scenarios but reduce harvest under the lowest temperature/lowest emissions scenario. The global models indicate that much of the increased global timber productivity will come from producers in the low mid-latitudes who are able to respond quickly to changing growing conditions and are expected to experience some of the largest growing condition improvements. An increase in timber harvest is projected to occur fairly quickly in the decades post growth change, followed by a small slowdown and then a long, sustained increase.

Climate change is projected to result in welfare changes for consumers of wood products and producers of timber. The studies considered here are generally consistent in projecting that total welfare, net of forest product consumers and timber producers, will increase relative to the baseline. However, the South and West United States are projected to suffer total welfare losses under some scenarios, and timber producers are projected to experience welfare losses in most scenarios. Wood product consumers are projected to gain in nearly all the scenarios considered. The temporal pattern of welfare changes by decade, post climate change, is largely unknown.

The current projections for the forest sector under climate change are based on existing studies that were completed using the information and models available at the time. One important uncertainty in considering the existing model results is the impact that unaccounted-for dieback or increased levels of disturbance may have on the expected responses of the forest sector to climate change. Historical levels of dieback and disturbance are represented in the growth and yield functions used in the models. Additional dieback (including that possibly from disturbance) was included in the Sohngen et al. (2001) study. In that analysis, increased dieback did change model output but did not change the general relationships between climate change and forest sector outcomes. For example, under the dieback scenarios, timber prices were still projected to decline with climate change, although this change was mitigated slightly by the modeled dieback. If the dieback or disturbance experienced under climate change is greater than that captured in the models, actual impacts to the forest sector may differ from model results.

Recent trends in land use and housing growth not only create stresses on natural ecosystems, they also increase society's vulnerability to natural hazards.

Although the total area of land in agriculture and forestry may remain relatively constant over time, climate change could alter the distribution of rural land uses and affect forest area if climate change affects yields and costs of production for different land-use alternatives. Given the potential human and social consequences of these impacts, it is important to extend and enhance modeling tools to assess the implications of climate change on the distribution of land uses in the United States. Another uncertainty is how climate change may affect human migration patterns and thereby areas of urban and rural land uses such as forest that may be converted to developed uses, including those of coastal areas. Another research area is the relationship between human settlement patterns and vulnerabilities to natural disasters. In terms of risks and hazards, natural disasters have many varied consequences, including damage to forest ecosystems and human communities. Recent trends in land use and housing growth not only create stresses on natural ecosystems, they also increase society's vulnerability to natural hazards. Global climate change has also been indicted in recent catastrophic weather events, and although scientific opinion is mixed regarding its role in current patterns, scientists agree that there is potential for significant change in the future. In the short run, i.e., over the past 50 years, the likelihood of natural hazards has been relatively stable, but losses in the United States have increased because our vulnerability to these hazards has increased (Alig et al. 2010). More houses and more wealth concentrated in regions of the country facing significant hazard levels describe the trend in the United States over the past 50 years.

Another uncertainty is how timber producers and private and public landowners will respond to changing forest growing conditions. Currently, the U.S. timber industry is experiencing a general reduction in capacity (e.g., milling infrastructure), a move away from vertically integrated companies (i.e., a forest product company owns the processing mill as well as the timberland) to a business model where timber is obtained from lands owned by other corporations and private entities and production is regionally concentrated. Combined, these factors may make it difficult for the timber industry to adapt to and mitigate climate change impacts on forests, particularly in the short term. Private landowners own forests for a variety of reasons, and those owners may not adopt adaptation and mitigation activities that promote continued or improved timber production. However, it is possible that expanded programs by land agencies (e.g., the State and Private Forestry branch of the USDA Forest Service) and conservation organizations could improve the implementation of adaptation and mitigation activities for timber production by private landowners. The results of current studies assume that producers will adopt forward-looking, optimal responses, and significant departures from this assumption may yield unanticipated impacts on the forest sector.

Another uncertainty is how timber producers and private and public landowners will respond to changing forest growing conditions.

Given the extent and value of urban forests, an area of research that deserves more attention is projecting the effects of climate change on threats to urban forests, including invasive forest insects and diseases and wildfires. Climate change will likely increase the frequency and intensity of these disturbances, and we need to quantify the associated costs and losses, including government and homeowner expenditures for prevention and mitigation activities and homeowner losses in property value. Documenting these potential costs and losses associated with disturbances to urban forests will add to our growing understanding of the overall effects of climate change on forests.

There are a number of opportunities for further research examining the impacts of climate change on the forest sector. First, because most of the existing studies were completed several years ago, it would be useful to update those analyses using

In future studies, it will be important to examine how the forest sector responds to concurrent changes in climate conditions and comprehensive climate policies.

the most recent climate projection and economic models. Second, additional studies that quantify how increased disturbance and dieback and suboptimal responses by timber producers and landowners affect model projections would help to identify how these uncertainties might impact the forest sector under climate change. Third, the expectation is that global trade will increasingly be important and economic models that better account for the dynamics of global trade will be useful as climate change is projected to have diverse positive and negative effects on timber growth and yield in different portions of the globe. Finally, the existing studies have examined the forest sector impacts from climate change in isolation. In future studies, it will be important to examine how the forest sector responds to concurrent changes in climate conditions and comprehensive climate policies (e.g., a carbon cap-and-trade system, increased demand for woody biomass for biofuels). For example, the climate scenarios considered here suggest that timber harvest will increase in response to improved yields. However, it is not known how that relationship would be affected if forest carbon offsets are also valued. It is probable that the combination of improved forest yields and a carbon value would have an impact on timber harvest levels (and ultimately consumer prices) not represented in current modeling.

Acknowledgments

The final report benefited from the helpful review comments of Lloyd Irland, Greg Latta, John Perez-Garcia, Andrew Plantinga, and Brent Sohngen.

English Equivalents

When you know:	Multiply by:	To get:
Degrees Celsius (C)	1.8C + 32	Degrees Fahrenheit
Meters (m)	3.28	Feet (ft)
Kilometers (km)	0.621	Miles (mi)
Square kilometers (km^2)	0.386	Square miles (mi^2)

References

Adams, D.; Haynes, R.W.; Daigneault, A.J. 2006. Estimated timber harvest by U.S. region and ownership, 1950–2002. Gen. Tech. Rep. PNW-GTR-659. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 64 p.

Alig, R.J.; Adams, D.M.; Joyce, L.A.; Sohngen, B. 2004a. Climate change impacts and adaptation in forestry: responses by trees and markets. Choices: The Magazine of Food, Farm and Resource Issues. Fall: 7–11.

Alig, R.J.; Adams, D.; McCarl, B. 2002. Projecting impacts of global climate change on the U.S. forest and agriculture sectors and carbon budgets. Forest Ecology and Management. 169 (2002): 3–14.

Alig, R.J.; Kline, J.D.; Lichtenstein, M. 2004b. Urbanization on the U.S. landscape: looking ahead in the 21st century. Landscape and Urban Planning. 69: 219–234.

Alig, R.J.; Stewart, S.; Wear, D.; Stein, S.; Nowak, D. 2010. Conversions of forest land: trends, determinants, projections, and policy considerations. In: Pye, J.M.; Rauscher, H.M.; Sands, Y.; Lee, D.C.; Beatty, J.S., tech. eds. Advances in threat assessment and their application to forest and rangeland management. Gen. Tech. Rep. PNW-GTR-802. Portland, OR: U.S. Department of Agriculture, Forest Service Pacific Northwest Research Station: 1–25.

Aukema, J.; Leung, B.; Britton, K. [et al.]. 2010. Buy now, pay later: the unexpected costs of importing forest insects in the U.S. Working paper. On file with: Robert Haight, USDA Forest Service, Northern Research Station, 1992 Folwell Ave., St. Paul, MN 55108.

Bosworth, D.; Birdsey, R.; Joyce, L.; Millar, C. 2008. Climate change and the Nation's forests: challenges and opportunities. Journal of Forestry. 106(4): 214–221.

Chapin, F.S.; Sturm, M.; Serreze, M.C. [et al.]. 2002. Role of land-surface changes in Arctic summer warming. Science. 310(5748): 657–660.

Donovan, G.H.; Butry, D. 2009. The value of shade: estimating the effect of urban trees on summertime electricity use. Energy and Buildings. 41(6): 662–668.

Donovan, G.H.; Butry, D. 2010. Trees in the city: valuing street trees in Portland, Oregon. Landscape and Urban Planning. 94: 77–83.

Dwyer, J.F.; McPherson, E.G.; Schroeder, H.W.; Rowntree, R.A. 1992. Assessing the benefits and costs of the urban forest. Journal of Arboriculture. 18(5): 227–234.

Dwyer, J.F.; Schroeder, H.W.; Gobster, P.H. 1991. The significance of urban trees and forests: toward a deeper understanding of values. Journal of Arboriculture. 17(10): 276–284.

Field, C.B.; Mortsch, L.D.; Brklacich, M. [et al.]. 2007. Climate change 2007: impacts adaptation and vulnerability. In: Parry, M.L.; Canziani, O.F.; Palutikof, J.P. [et al.], eds. Fourth assessment report of the Intergovernmental Panel on Climate Change. Cambridge, United Kingdom: Cambridge University Press: 617–652.

Gan, J. 2004. Risk and damage of southern pine beetle outbreaks under global climate change. Forest Ecology and Management. 191: 61–71.

Hammer, R.B.; Stewart, S.I.; Radeloff, V.C. 2009. Demographic trends, the wildland-urban interface, and wildfire management. Society and Natural Resources. 22: 777–782.

Intergovernmental Panel on Climate Change [IPCC]. 2007. Climate change 2007: synthesis report, summary for policymakers. http://www.ipcc.ch/pdf/assessment-report/ar4/syr/ar4_syr_spm.pdf. (11/30/2009).

Irland, L.C.; Adams, D.; Alig, R. [et al.]. 2001. Assessing socioeconomic impacts of climate change on U.S. forests, wood product markets, and forest recreation. BioScience. 51(9): 753–764.

Iverson, L.R.; Prasad, A.M.; Matthews, S.N.; Peters, M. 2008. Estimating potential habitat for 134 eastern U.S. tree species under six climate scenarios. Forest Ecology and Management. 254: 390–406.

Joyce, L.A. 1995. Productivity of America's forest and climate change, executive summary. In: Joyce, L.A., ed. Productivity of America's forests and climate change. Gen. Tech. Rep. RM-GTR-271. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 1–2.

Joyce, L.A. 2007. The impacts of climate change on forestry. In: Adams, D.M.; Haynes, R.W., eds. Resource and market projections for forest policy development: twenty-five years of experience with the US RPA timber assessment. Dordrecht, The Netherlands: Springer: 449–488.

Joyce, L.A.; Birdsey, R., tech. eds. 2000. The impact of climate change on America's forests: a technical document supporting the 2000 USDA Forest Service RPA assessment. Gen. Tech. Rep. RMRS-GTR-59. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 133 p.

Joyce, L.A.; Haynes, R.W.; White, R.; Barbour, R.J., tech. coords. 2006. Bringing climate change into natural resources management: proceedings. Gen. Tech. Rep. PNW-GTR-706. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 150 p.

Kovacs, K.F.; Haight, R.G.; McCullough, D.G. [et al.]. 2010a. Cost of potential emerald ash borer damage in U.S. communities, 2009–2019. Ecological Economics. 69: 569–578.

Kovacs, K.F.; Vaclavic, T.; Meentemeyer, R.; Haight, R.G.; Pang, Y-W. 2010b. Estimating the economic costs and property value losses of sudden oak death damage in California from 2010–2020. Working paper. On file with: Robert Haight, USDA Forest Service, Northern Research Station, 1992 Folwell Ave., St. Paul, MN 55108.

Latta, G.; Temesgen, H.; Adams, D.; Barrett, T. 2010. Analysis of potential impacts of climate change on forests of the United States. Forest Ecology and Management. 259(4/5): 720–729.

Mansfield, C.; Pattanayak, S.K.; McDow, W.; McDonald, R.; Halpin, P. 2005. Shades of green: measuring the value of urban forests in the housing market. Journal of Forest Economics. 11: 177–199.

McCarl, B.; Adams, D.; Alig, R.J. [et al.]. 2000. Effects of global climate change on the U.S. forest sector: response functions derived from a dynamic resource and market simulator. Climate Research. 15: 195–205.

McKinley, D.C.; Romero, J.C.; Hungate, B.A. [et al.]. 2009. Does deep soil N availability sustain long-term ecosystem responses to elevated CO_2? Global Change Biology. 15: 2035–2048.

McPherson, G.; Simpson, J.R.; Peper, P.J.; Maco, S.E.; Xiao, Q.F. 2005. Municipal forest benefits and costs in five U.S. cities. Journal of Forestry. 103(8): 411–416.

Mills, J.R.; Haynes, R.W. 1995. Influence of climate change on supply and demand for timber. In: Joyce, L.A., ed. Productivity of America's forests and climate change. Gen. Tech. Rep. RM-GTR-271. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 46–55.

Norby, R.J.; DeLucia, E.H.; Glelen, B. [et al.]. 2005. Forest response to elevated CO_2 is conserved across a broad range of productivity. Proceedings of the National Academy of Science. 102(50): 18052–18056.

Nowak, D.J.; Noble, M.H.; Sisinni, S.M.; Dwyer, J.F. 2001. Assessing the U.S. urban forest resource. Journal of Forestry. 99(3): 37–42.

Nowak, D.J.; Stein, S.M.; Randler, P.B.; Greenfield, E.J.; Comas, S.J.; Carr, M.A.; Alig, R.J. 2010. Sustaining America's urban trees and forests: a Forests on the Edge report. Gen. Tech. Rep. NRS-62. Newton Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 27 p.

Nowak, D.J.; Walton, J.T. 2005. Projected urban growth (2000–2050) and its estimated impact on the U.S. forest resource. Journal of Forestry. 103(8): 383–389.

Perez-Garcia, J.; Joyce, L.A.; McGuire, A.D.; Xiao, X. 2002. Impacts of climate change on the global forest sector. Climatic Change. 54: 439–461.

Pew Center on Global Climate Change [Pew Center]. 2007. Highlights from “Climate Change 2007 Synthesis Report of the IPCC Fourth Assessment Report Summary for Policymakers.” http://www.pewclimate.org/docUploads/PewSummary_AR4.pdf. (11/30/2009).

Radeloff, V.C.; Hammer, R.B.; Stewart, S.I. [et al.]. 2005. The wildland-urban interface in the United States. Ecological Applications. 15: 799–805.

Radeloff, V.C.; Stewart, S.I.; Hawbaker, T.J. [et al.]. [In press]. Housing growth in and near United States protected areas limits their conservation value. Proceedings of the National Academy of Sciences.

Reich, P.B.; Hobbie, S.E.; Lee, T. [et al.]. 2006. Nitrogen limitation constrains sustainability of ecosystem response to CO_2. Nature. 440(13): 922–925.

Sander, H.; Polasky, S.; Haight, R.G. [In press]. The value of urban tree cover: a hedonic property price model in Ramsey and Dakota County, Minnesota, USA. Ecological Economics.

Sedjo, R.; Sohngen, B. 1998. Impacts of climate change on forests (Internet edition). Washington, DC: Resources for the Future. 9 p. http://www.rff.org/rff/Documents/RFF-CCIB-09.pdf. (7/6/2010).

Shugart, H.; Sedjo, R.; Sohngen, B. 2003. Forests and global climate change: potential impacts on U.S. forest resources. Washington, DC: Pew Center on Global Climate Change. 52 p.

Smith, W.B.; Miles, P.D.; Perry, C.H.; Pugh, S.A. 2009. Forest resources of the United States, 2007. Gen. Tech. Rep. WO-78. Washington, DC: U.S. Department of Agriculture, Forest Service. 336 p.

Sohngen, B.; Mendelsohn, R. 1998. Valuing the market impact of large-scale ecological change in a market: the effect of climate change on U.S. timber. American Economic Review. 88: 686–710.

Sohngen, B.; Mendelsohn, R.; Sedjo, R. 2001. A global model of climate change impacts on timber markets. Journal of Agriculture and Resource Economics. 26(2): 326–343.

Sohngen, B.; Sedjo, R. 2005. Impacts of climate change on forest product markets: implications for North American producers. The Forestry Chronicle. 81(5): 669–674.

Stewart, S.I.; Radeloff, V.C.; Hammer, R.B. 2007. Defining the wildland urban interface. Journal of Forestry. 105(4): 201–207.

Stone, J.K.; Coop, L.B.; Manter, D.K. 2008. Predicting effects of climate change on Swiss needle cast disease severity in Pacific Northwest forests. Canadian Journal of Plant Pathology. 30: 169–176.

Thompson, M.; Adams, D.; Johnson, K.N. 2009. The albedo effect and forest carbon offset design. Journal of Forestry. 107(8): 425–431.

Thornton, P.E.; Doney, S.C.; Lindsay, K. [et al.]. 2009. Carbon-nitrogen interactions regulate climate-carbon cycle feedbacks: results from an atmosphere-ocean general circulation model. Biogeosciences. 6: 2099–2120.

Thorsnes, P. 2002. The value of a suburban forest preserve: estimates from sales of vacant residential building lots. Land Economics. 78: 426–441.

van der Vink, G.; Allen, R.M.; Chapin, J. [et al.]. 1998. Why the United States is becoming more vulnerable to natural disasters. Eos, Transactions, American Geophysical Union. 79(44): 533–537.

Westphal, L.M. 2003. Urban greening and social benefits: a study of empowerment outcomes. Journal of Arboriculture. 29(3): 137–147.

White, E.M. 2010. Woody biomass for bioenergy in the United States—a briefing paper. Gen. Tech. Rep. PNW-GTR-825. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 45 p.

Ziska, L.H. 2003. Evaluation of the growth response of six invasive species to past, present, and future carbon dioxide concentrations. Journal of Experimental Botany. 54: 395–404.

Chapter 2: The Forest Sector, Climate Change, and the Global Carbon Cycle—Environmental and Economic Implications

by Brent Sohngen, Ralph J. Alig, and Birger Solberg

Introduction

The forest sector (i.e., forestry and forest industry, including the use of forest land) plays an important role in the global climate change debate—partly because the sector influences the global carbon cycle, and partly because the sector is influenced by possible global climate change caused by increased concentrations of greenhouse gases, among which carbon dioxide (CO_2) is the most important. This paper assesses literature examining the impacts of climate change on the forest sector, focusing on studies that have considered economic impacts and market adaptation. The report also considers how activities in the forest sector—such as mitigation through afforestation, reduced deforestation, and forest management—may affect the global carbon cycle. For the most part, researchers have not considered how mitigation efforts may be influenced by climate change (and vice versa), so we only briefly discuss the interaction between these two effects in the concluding sections of the paper.

Studies of climate change impacts in the forest sector and studies of adaptation generally link estimates from ecological models to timber models. A number of different types of ecological and timber models have been developed over the years, ranging from local to regional and global. The ecological models provide insights into a host of potential effects that climate change may have on forests, including tree growth, carbon fertilization, disturbances and dieback, and other effects (Alig et al. 2004). We draw on this set of results to enhance our understanding of potential impacts on the forest sector in different regions. This paper focuses on a discussion of economic implications in markets. Other environmental issues such as effects on biodiversity, water catchments, wildlife, and recreation are not discussed here, as the uncertainty on these issues seems to be rather high (e.g., Gitay et al. 2001, Kauppi and Solberg 1999).

The paper addresses short-, medium-, and long-run implications, and it considers implications in separate geographical regions, including boreal, temperate, and tropical regions. Institutional factors are also important for forest sector responses (Solberg et al. 1996) but have not been widely addressed in the climate change impacts literature in forestry to date. Thus, it is difficult to make generalizations relating to how institutional factors may affect adaptation and economic responses.

The structure of this paper is as follows. We begin with a discussion about the baseline for future global timber market activity by describing results from several studies that have projected future market conditions. Second, we examine the potential ecological and market effects of climate change impacts in forests, including market-based adaptation to climate change. Finally, we examine the potential economic implications of mitigation activities in forests, including a discussion about the interactions between climate change impacts and mitigation.

To understand the potential impacts of climate change on the forest sector, it is important to have an understanding of the baseline.

Baseline

To understand the potential impacts of climate change on the forest sector, it is important to have an understanding of the baseline, that is, projections of the forest sector without climate change. Many studies that assess economic impacts first estimate baseline conditions, and then compare climate conditions to these baseline conditions to assess potential adjustments and adaptations. Estimates of potential climate change impacts upon the forest sector are best understood in light of how they could alter future market conditions relative to "no climate change" baseline conditions.

Current global timber harvests are approximately 1.6 billion m^3 of industrial roundwood per year (UNFAO 2005). An assessment of timber market studies suggests that this could rise to 1.9 to 3.1 billion m^3 by 2050, depending on timber demand growth and relative price changes (Solberg et al. 1996). These changes would represent an increase in annual timber harvests of 0.5 percent to nearly 2.0 percent. Prices are predicted to increase from 0 to 0.5 percent per year in real terms. Under all of these scenarios, timber harvest intensities (m^3/ha per year) in different forests increase, and global harvest intensity is predicted to increase 63 percent relative to the baseline (table 2-1).

An alternative set of scenarios based on the global timber market model described in Sohngen et al. (1999) and updated in Sohngen and Mendelsohn (2007) suggests similar results. The scenarios assume that worldwide population increases from 6.4 to 9.8 billion people over the next 100 years, and that global gross domestic product increases by 1.7 percent per year on average. In addition, the scenario assumes that technology improves by 1.5 percent per year in forest products production, and that plantation yields rise at 2.5 percent per decade. Under these assumptions, timber harvests are projected to increase to approximately 2.3 billion m^3 by 2105. Prices are predicted to rise at 0.2 percent per year. Most of the new timber harvests from this model are predicted for subtropical plantation regions, where

Table 2-1—Actual timber harvest in 1995, base timber harvest intensity for different land ownership types, and assumed percentage change in timber harvest intensity by 2020 under two alternative scenarios

Harvest categories	Temperate industry	Tropical plantations	Temperate NIPF	Temperate public	Rest of world	Global
1995 timber harvest (million m^3/year)	196	65	419	307	592	1579[a]
1995 timber harvest intensity (m^3/ha per year)	4.56	2.83	1.93	1.41	0.25	0.54
Base scenario (percent)	21	112	30	42	40	50
Optimistic plantation scenario (percent)	32	112	55	60	40	63

NIPF = nonindustrial private forest.
[a] Represents total global timber harvests, not average.
Adapted from Solberg et al. 1996.

technology improvements are increasing the yield of forests substantially over long periods. In contrast, declining timber harvest intensities are projected for currently inaccessible forests in tropical and boreal regions.

For tropical and subtropical regions, the studies project increases in timber harvests from fast-growing plantations. Most differences between the studies discussed above relate to different predictions of timber harvests from plantations in subtropical regions. The results from the global timber model of Sohngen et al. (1999) project a potentially stronger movement toward subtropical plantation establishment and harvests. For temperate regions, both studies predict increases in timber harvests in the short term. Solberg et al. (1996) did not provide harvest projections beyond 2050. Sohngen et al. (1999) suggested that temperate regions will not increase timber harvests substantially in the long run. In boreal regions, the global timber model predicts a decline in harvests over time. One reason for this decline in harvesting activity in boreal regions is that prices are projected to stabilize over time. If prices remain constant, incentives to expand infrastructure for harvesting timber in the boreal region are smaller. Solberg et al. (1996) suggested an increase in harvests in boreal regions in the short term in part owing to price increases but also to the fairly large stocks and low costs of accessing stocks in many boreal regions.

These two studies are broadly consistent by suggesting an increasing role for subtropical plantations in global timber supply. They differ on the extent to which this new wood supply will offer alternatives to harvesting natural forests in temperate and boreal regions. However, some of these differences can be explained by the relatively short outlook period for the Solberg et al. (1996) study relative to the longer term projections provided by Sohngen et al. (1999). Solberg et al. (1996) also offered important insights into institutional factors, such as ownership, rights of use of forest land, and international agreements that may affect future supply of wood from particular tropical countries.

Ecological and Timber Market Implications of Climate Change

It is widely recognized that climate change is likely to have strong influences on the structure and function of forests (IPCC 2007b, Watson et al. 2000). When considering how the ecological effects of climate change translate into economic effects (the interest of this paper), it is convenient to categorize the response into three general areas: forest productivity changes (e.g., Latta et al. 2010), ecosystem disturbances, and changes in forest species distribution. Productivity changes are adjustments in the productivity of forests that alter the growth rates of timber species (in either a positive or negative way). Changes in disturbance influence the standing stock of trees, and include pest infestations, forest fires, windthrow, and ice damage. Finally, changes in species distribution result from shifts in climate, which ultimately alter the optimal geographic location of different timber species.

It has long been recognized that there is potential for additional carbon in the atmosphere to enhance the growth of trees (the so-called carbon fertilization effect).

Emily Jederlinich

Timber harvest is a major disturbance on private forest lands.

Results from modeling studies suggest that carbon fertilization can, in turn, have a large impact upon the predicted effects of climate change on forest structure and growth (see Cramer et al. 2001, VEMAP 1995). Although earlier model results suggested that CO_2 could enhance global growth rates in forests (e.g., Melillo et al. 1993), more recent results suggest that inter-annual variation in temperature and precipitation could have positive or negative effects on annual growth, depending on the direction of change (Schimel et al. 2000, Tian et al. 1998). Thus, carbon fertilization effects may be limited by changes in annual weather or by other limiting nutrients (Melillo et al. 1993). The so-called carbon fertilization effect could reach a saturation point for particular species and for ecosystems (Gitay et al. 2001). A recent comparison of CO_2 experiments across a number of sites and timber ages, however, indicates that carbon has a relatively consistent, and positive, effect on net primary productivity (Norby et al. 2005). A recent study may illustrate this evidence. Boisvenue and Running (2006) reviewed historical trends in net primary productivity in forests and found that over the last 50 years, most studies have reported increasing growth trends in forests where water is not a limiting factor.

Although tree growth and productivity effects will have clear long-run implications for forests, one of the more important near-term effects could be dieback. Some authors have suggested that climate change could lead to dieback in existing (or future) forests because of water stress, insect infestations, or fires (Bachelet et al. 2003, 2004; King and Neilson 1992; Scholze et al. 2006; Shugart et al. 1986; Smith and Shugart 1993; Solomon and Kirilenko 1997). Two causes of dieback appear in the literature. The first is that changes in climate (drying or warming) could make forests more susceptible to insects, fire, and other disturbance agents. Current evidence suggests that climate change may already be causing more intense fires in some regions of the world (Westerling et al. 2006). Climate change could also shift the distribution of climatic variability and climatic extremes (Houghton et al. 1996, Watson et al. 2000). Predictions of the size and scope of changes in climate or extreme events depend heavily on climate predictions made by climate modelers. The distribution of the climate effects geographically (e.g., where changes in precipitation and temperature occur) and over time is one of the most uncertain aspects of climate modeling, suggesting high uncertainty surrounding the regional distribution of forest dieback effects.

The distribution of the climate effects geographically and over time is one of the most uncertain aspects of climate modeling, suggesting high uncertainty surrounding the regional distribution of forest dieback effects.

Beyond the direct effects of climate on forests, a related issue is the influence of climate change on the productivity of competing land uses (Alig et al. 2002), such as agricultural crop and livestock production. Large changes in productivity of farmland could lead to an expansion, or contraction, of agricultural land. Given the historical interrelationship between forests and agriculture, shifts in productivity of agricultural land in particular could have large effects on the ultimate distribution of forest land. Current research does not indicate that climate change will lead to large-scale increases in agricultural land at the expense of forests in most temperate regions in the short run (Alcamo et al. 1997, Alig et al. 2002, Gitay et al. 2001, Reilly et al. 2003, Watson et al. 1996). In recent times, most expansion of agricultural land has occurred in tropical forests (Houghton 2003, UNFAO 2005), and these trends are likely to continue over the next 20 years (Watson et al. 2000).

In the next three subsections of this paper, we examine the implications of these broad effects for timber markets. We consider both geographic and temporal factors, e.g., where and when the impacts may occur. Although considerable uncertainty still exists with respect to projections associated with climate change, a number of the economic results are fairly robust across the models, and therefore provide some reasonable assurances about the capacity of markets to adapt to change. Our general findings for different regions and time periods are summarized in table 2-2.

Short-Term Climate Impacts (2005–2025)

The recent study by Scholze et al. (2006) is perhaps the most comprehensive global assessment to date. The researchers examined potential climate impacts on ecosystems across 16 climate models and 52 climate scenarios, providing information on the average potential effect of climate change as well as uncertainty. Uncertainty is inferred by assessing results across the range of climate models and scenarios analyzed. They do not incorporate humans, so their predictions are based on what could happen to forests if humans were not already affecting forests. Their results show that in the short term (next 25 years), forests will likely be a net sink for carbon globally. The risk of forests becoming a source for carbon in the next 25 years, however, is inversely related to global temperature change over the century. For example, in the short term under a number of the climate scenarios analyzed in the Scholze et al. (2006) paper, the size of the carbon sink in the biosphere becomes larger on average across the climate scenarios if the global average temperature change over the century is >2 °C. An increasing sink implies that forests either are expanding in area, or otherwise increasing their stock of carbon. For global average temperature changes <2 °C over the century, the biosphere becomes a net source of carbon under some of the climate scenarios. If forests become a source for carbon, then emissions of carbon to the atmosphere from dieback and decay processes are larger on net than forest growth. The short-term results in the paper by Scholze et al. (2006) contrast with their longer run results that suggest there is greater probability that forests become a carbon source under larger temperature changes (see below).

Table 2-2—Ecological and economic implications of climate change on the forest sector

Area	Short term (2005–2025)	Medium term (2025–2065)	Long term (2065–2105)
Boreal	• ↑ Productivity • ↑ Risk of fire/natural disturbance • ↑ Salvage; ↑ Timber supply	• ↕ Productivity • ↑↑ Risk of fire/natural disturbance • ↑ Expansion of species northward • ↑ Southern range displaced by more southerly forest types • ↑ Salvage; ↑ Timber supply	• ↕ Productivity • ↑↑ Risk of fire/natural disturbance • ↑↑ Expansion of species northward • ↑↑ Southern range displaced by more southerly forest types • ↑ Salvage; ↑ Timber supply
Temperate	• ↕ Productivity • ↕ Timber supply; ↓ Timber prices	• ↕ Productivity • ↑ Risk of fire/natural disturbance • ↑ Movement of species northward • ↑ Salvage; ↑ Timber supply	• ↕ Productivity • ↑↑ Risk of fire/natural disturbance • ↑ Movement of species northward • ↑ Salvage; ↑ Timber supply
Tropical	• ↕ Productivity • ↑ Plantation establishment • ↑ Timber supply to world market	• ↕ Productivity • ↑ Risk of fire/natural disturbance • ↑ Risks to plantations and natural forests • ↑ Salvage; ↑ Timber supply	• ↕ Productivity • ↑↑ Risk of fire/natural disturbance • ↑ Risks to plantations and natural forests • ↑ Salvage; ↑ Timber supply
World market effect	• ↑ Supply from rising productivity and the possibility of salvage • ↓ World timber prices • ↕ Producer welfare • ↑ Consumer welfare	• ↑ Supply from rising productivity and the possibility of salvage • ↓ World timber prices • ↕ Producer welfare • ↑ Consumer welfare	• ↑ Supply from rising productivity and the possibility of salvage • ↓ World timber prices • ↕ Producer welfare • ↑ Consumer welfare

↑ = increases in indicator; ↓ = decreases in indicator; ↕ = both increases and decreases in indicator likely; double arrows indicate stronger effects likely.

Where humans influence regeneration processes, they can speed the movement of tree species across the landscape, and where humans have smaller impacts, forest adjustment processes will be slower.

Evidence suggests that climate change could have relatively larger near-term effects in boreal regions (Kirschbaum and Fischlin 1996, Watson et al. 2000). Boreal forests are already characterized by long-term, historical shifts in natural fire frequency that have large effects on forest and carbon stocks (Kurz and Apps 1999). If climate change alters the natural fire frequency (e.g., Bachelet et al. 2004, Westerling et al. 2006), then there could be fairly substantial impacts on boreal forests in the near term. In addition to potential changes in fire or other disturbance frequency, many ecological models also project a movement of species north with a warmer climate (Solomon and Kirilenko 1997, Watson et al. 2000). Because boreal regions, except for the Nordic countries and Western Russia, tend to be unmanaged, humans are less likely to be part of the adaptation process, thus slowing the movement of species. Where humans influence regeneration processes, they can speed the movement of tree species across the landscape, and where humans have smaller impacts, forest adjustment processes will be slower (e.g., Sohngen et al. 1998). Slower adaptation could have negative implications for carbon stocks (Nilsson and Shvidenko 2000, Solomon and Kirilenko 1997).

Many of the near-term effects of climate change in boreal forests are likely to occur mostly beyond the accessible margin, so that global markets experience few significant impacts. One reason for this is that native forests in boreal regions are expected to play a smaller proportional role in wood supply over the next 20 years (see Solberg et al. 1996, Sohngen et al. 2000). Sohngen and Sedjo (2000) suggested that over the period 2005–2025, timber harvest levels are not projected to change substantially in boreal forests of North America, Europe, or Russia. Despite the likely small timber market impacts globally, there could be locally important implications for boreal communities that are dependent on forest resources.

It takes a long time for changes in timber growth rates to have a marked effect on timber inventories and timber supply.

The dynamics of timber markets in temperate regions have been examined more thoroughly than for other regions, particularly in the United States (Sohngen and Alig 2000). A range of ecological scenarios has been explored for these forests, including changes in annual timber growth, potential dieback, and changes in species distribution. The results indicate that timber supplies from temperate regions would not be dramatically affected in the short run if the primary effects of climate change are changes in the rate of growth of timber. It takes a long time for changes in timber growth rates to have a marked effect on timber inventories and timber supply (Alig et al. 2002; Joyce et al. 1995, 2001; Mills and Haynes 1995; Perez-Garcia et al. 1997).

Similar results have been found in studies conducted in Europe. Trømborg et al. (2000) used a regional partial equilibrium forest sector model to analyze the market impacts (in a 15 to 20 year perspective) of possible accelerating forest growth in Europe. Three scenarios were studied: a base scenario that assumed a 1.4 percent per year increase in standing stock (this reflects the actual average situation in Europe in 1994), a medium scenario assuming 2.0 percent per year increase in growth (i.e., about 43 percent higher growth than the base scenario), and a high scenario of 2.7 percent per year increase in standing stock (i.e., 93 percent higher growth than the base). The projected impacts of accelerating growth in timber production were found to be fairly small over the next 10 to 20 years for saw log and sawn wood markets, whereas pulpwood prices were found to decrease substantially. Solberg et al. (2003) showed similar results, applying a more detailed partial equilibrium model with respect to forestry, international trade, and forest industry technologies.

More dramatic scenarios have been examined where climate changes substantially over the next 20 years, causing dieback and changes in the distribution of important commercial tree species. Under these fairly dramatic scenarios, climate change could have substantial effects on timber supply in the short term. Specifically, widespread dieback, when combined with salvage logging, is projected to increase short-term timber supplies and reduce prices (Sohngen and Mendelsohn 1998). One important uncertainty regarding the effects of dieback in boreal regions relates to global trade. Sohngen et al. (2000) found that timber production in boreal and temperate regions could decline in the near term if climate change causes dieback in boreal and temperate zones, but could be enhanced in subtropical and tropical regions. Both studies suggest that producers' economic welfare would be reduced by potential dieback, but that they can actively participate in mitigating

Emily Jederlinich

Most forests after timber harvest regenerate back to forest status either naturally or through tree planting.

Short-rotation plantation species are expected to be particularly suitable for adaptation during climate change, so that tropical and subtropical countries could potentially benefit from climate change.

those effects through salvage, and by changing the tree species they regenerate to those that are better suited to a new climate. Adaptation through regeneration can have important implications for the economic viability of particular forest stands (Lindner 1999, 2000). The importance of trade was also shown by Kallio et al. (2006), who studied the market impacts of a relatively large decrease in European timber supply caused by increased biodiversity protection. The impacts are rather modest because increased imports from Russia offset to a large degree the decline in domestic roundwood supply.

Tropical regions are not expected to experience large immediate impacts from climate change. Currently, natural tropical forests contribute only a small portion of the world's timber harvest, and climate change over the next 20 years is not expected to change this. Many of these same countries, however, are providing increasingly large shares of the world's timber from their plantations. Short-rotation plantation species are expected to be particularly suitable for adaptation during climate change, so that tropical and subtropical countries could potentially benefit from climate change. Sohngen et al. (2000) found that if climate change increases forest productivity in plantations, South American timber harvests could increase by more than 20 percent over the next 20 years (relative to the baseline) with climate change. The effects in subtropical and tropical plantations are directly linked to the size of the change in net growth implied by climate change.

Ecological models do not suggest large near-term additional disturbances in natural tropical forests, and the largest impacts in the near term on these forests are likely to result from deforestation rather than from climate change (Gitay et al. 2001). Deforestation, although slowing in recent years (Houghton 2003; UNFAO 1999, 2005), is predicted to continue to cause conversion of tropical forests to agriculture (Palo et al. 2000). For example, annual net deforestation rates in tropical areas of Africa and South America are (annual percentage loss in parentheses) 4.0 million ha/yr (0.7 percent per year) and 4.2 million ha/yr (0.7 percent per year), respectively (UNFAO 2005).

Medium-Term Climate Impacts (2025–2065)

Left unabated, climate change is expected to intensify during the middle of this century (IPCC 2007b). The most important impacts on forests and timber markets are likely to occur in the medium to long term. The Intergovernmental Panel on Climate Change suggests that approximately 1/7 to 2/3 of all temperate and boreal forests are likely to undergo some type of ecological change over the century (Gitay et al. 2001, Watson et al. 1998). Those changes could include dieback of existing species (Bachelet et al. 2004, King and Neilson 1992, Scholze et al. 2006, Smith and Shugart 1993, Solomon and Kirilenko 1997, for example), movement of tree species from one region to another region, and accelerating impacts of climate change and CO_2 concentrations on forest growth.

In boreal regions, climate change is generally expected to cause an increase in forest growth and an increase in forest area over the coming century (Cramer et al. 2001, Scholze et al. 2006). Most of the expansion of forests, however, is far to the north, in regions that currently are tundra, and generally considered to be inaccessible. Scholze et al. (2006) suggested that if the average global temperature change is expected to rise above 3 °C over the century, then boreal forests will be at a risk of losses owing to dieback and disturbance, among other factors. Losses at higher temperatures appear to be driven largely by increases in forest fire activity associated with larger temperature changes (Scholze et al. 2006).

Perez-Garcia et al. (1997) relied on ecological studies that suggested rising growth rates in boreal forests. Not surprisingly, their economic model showed an increasing supply of timber from boreal regions over the next 40 years. Heavier timber harvests in boreal forests in turn were found to reduce timber prices, and negatively affect producers' economic welfare in temperate regions. More recent results by Perez-Garcia et al. (2002) also assumed that biomass of boreal forests increases, but they found the opposite result for the timber harvest in boreal regions. For example, their study suggests that lower worldwide prices for timber cause a reduction in timber harvests in Canadian boreal forests. Thus, their results show that economic impacts of lower prices outweigh the benefits of rising forest productivity in boreal regions. The results in Perez-Garcia et al. (2002) are qualitatively similar to those in Sohngen et al. (2000), who suggested that boreal regions become less important over time both in the baseline and during climate change, as global timber harvests shift toward subtropical plantation regions. In combination, these results suggest that, regarding timber harvests, boreal forests will continue to become relatively less important globally over the medium term, and that climate change is likely to exacerbate the situation.

Ecological studies suggest a wide range of potential impacts in temperate forests in the medium term. Bachelet et al. (2004) examined impacts in the coterminous United States with a single ecological model and two climate scenarios. Their results suggested that total forest biomass could expand under a wetter climate and could contract in scenarios with less moisture. Cramer et al. (2001) considered only a single climate scenario, but a range of ecological models. Their results imply increases in net ecosystem productivity over the century projected by most of the ecological models. The results by Scholze et al. (2006) suggested both increases and decreases in forest area, depending on the climate scenario and the region, with

Studies that focus only on changes in productivity of forests, and not on stock effects, show that climate change has larger implications for supply in the medium term than in the short term.

higher temperature changes in the temperate regions. As temperatures increase above 3 °C, their results suggest an expansion in forest area, and an expansion in wildfire activity in temperate zones. More wildfires occur partly because there are more forests to burn.

It takes some time for forest inventories to reflect the influence of climate change on timber growth; thus economic studies that focus only on changes in productivity of forests, and not on stock effects, show that climate change has larger implications for supply in the medium term than in the short term (Joyce et al. 1995, Perez-Garcia et al. 1997). McCarl et al. (2000) also showed losses accelerating over time if growth effects are negative. Sohngen and Mendelsohn (1998) combined changes in timber growth, dieback from disturbance, and shifts in species range based on Vegetation/Ecosystem Modeling and Analysis Project (VEMAP 1995). For all of the ecological and climate scenarios examined, the average effects imply that dieback would occur on an additional 0.7 million ha per year (in the United States only) over a 70-year period, forest growth would increase by 5 percent by 2070, and forest area would increase by 14 percent by 2150. If forest fires occurred on all areas where dieback occurred, the scenarios suggested a 41 percent increase in fire activity on average over the current situation. Despite the fairly substantial losses of timber projected during this century owing to dieback, salvage was found to reduce the economic losses, and timber supply was found to increase during the medium term.

One critical question for timber markets in the medium term lies with regeneration, e.g., which species should be replanted to thrive under new climate conditions. If climate conditions change substantially, landowners in temperate regions will be looking for signals to alter the tree species they replant. Whether the signal is strong enough to perceive will have only small effects during the medium term, but will have notable effects on timber supply in the long run. If the more dramatic ecological scenarios involving dieback and tree species change are accurate, getting the answer to this question right will determine the long-run outlook for timber supply from temperate regions.

Scholze et al. (2006) suggested relatively smaller effects in tropical forests than for boreal and temperate regions in the medium and long run, but their results suggest that risks of biome shifts and wildfire disturbance in natural tropical forests are nonetheless substantial. Any changes that do occur in native tropical forests will have relatively small effects on timber markets because these regions do not provide a large supply of industrial timber for markets, and they are not projected to become large suppliers in the future (e.g., Daigneault et al. 2008).

Plantations in subtropical regions—Chile, Argentina, Brazil, South Africa, Australia, and New Zealand—are projected to provide more than 30 percent of market share in the middle of the century (Daigneault et al. 2008). If climate change drastically alters productivity in these plantations, there could be large timber market impacts. Most of the ecological models consider impacts only in native forests, whereas subtropical plantations tend to be cultivated with nonindigenous tree species. Thus, it is difficult to know exactly how climate change will influence their potential growth under the new climates in which they have been introduced. However, most subtropical plantations focus on very-short-rotation species (rotation lengths are often less than 20 years, and frequently less than 10 years), so that timberland managers can adjust and adapt rapidly if climate change has dramatic effects. For example, if losses of forests in traditional industrial supply regions of the temperate zone (e.g., United States, Canada, Europe) become substantial, subtropical plantation species may benefit (see Sohngen et al. 2000).

Long-Term Climate Impacts (Beyond 2065)

The long-run effect of climate change on ecosystems will be heavily influenced by the amount of climate change. Scholze et al. (2006) found that for global average temperature changes above 3 °C, the natural sink potential in forests declines over the century, with a substantial probability of forests becoming a large source of carbon beyond 2065. One reason for this is the increase in wildfire activity they model, and another reason is the potential shift in biome type. For example, under 38 percent of the climate scenarios investigated, they predict biome shifts in 10 percent of existing forests in tropical areas when global average temperature change exceeds 3 °C over the century. Under 88 percent of the climate scenarios, they predict biome shifts in 10 percent of existing boreal forests when global average temperature change exceeds 3 °C over the century. For climate change of less than 2 °C, 19 percent and 44 percent of climate change scenarios were found to cause biome shifts in 10 percent of existing tropical and boreal forests, respectively. Even for the smaller changes in temperature, potentially substantial shifts could occur in tropical and boreal regions.

The results in Bachelet et al. (2004) illustrate the uncertainty in long-term analysis. They examined only two climate scenarios, one was warmer and wetter over time, and the other was warmer and drier. In the warmer and wetter scenario, forests were found to become a stronger sink over the century, whereas in the warmer and drier scenario, forests were found to become a strong source by the end of the century. There were strong regional differences within the country in both scenarios. For example, under the more pessimistic climate scenario, the Northeast

One important aspect that has been ignored in most of the literature to date is that the market response will likely influence the ultimate effects that ecosystems experience.

and Southeast and West are projected to become strong sources of carbon emissions toward the end of the century. For the more optimistic climate scenario, most regions become strong sinks for carbon over the century, although the Northwest becomes a source.

At an aggregate level, different ecological models agree on the overall response in forests (e.g., Cramer et al. 2001). Warming with plenty of additional precipitation will enhance forest productivity, whereas drying of forests leads to potential losses. These potential losses become more pronounced when global average temperatures exceed increases of 2.5 to 3.0 °C. Models do find specific differences in specific regions, and this limits our understanding of where impacts are likely to occur.

For timber markets, the long-term story is one of adaptation. Specifically, one question is whether landowners and land managers will be able to respond to climate, ecological, and market signals adequately during this century. According to the ecological studies, landowners and managers will face a host of hurdles, including changes in forest fire activity, changes in the potential for land to sustain forests, and changes in the rates of growth of tree species. Beyond adaptation on the landscape, the entire forest products industry will need to adapt, for example, by learning to use new tree species in forest production processes. The extent to which the industry responds to climate change will drive signals during the century and will influence the extent of market effects at the end of the century.

One important aspect that has been ignored in most of the literature to date is that the market response will likely influence the ultimate effects that ecosystems experience. Nearly all ecological studies are built on potential vegetation, and none

Ralph Alig

Reduction in deforestation is a significant mitigation option.

of the models incorporate human management of forests (e.g., Bachelet et al. 2004, Scholze et al. 2006). Forest ecosystems, however, already are heavily influenced by human management. It has long been recognized that foresters respond to changes in disturbance by altering forest management. For example, Reed (1984) and Haight et al. (1995) showed how timber rotation ages are adjusted in response to disturbances. Recent economic studies show that there are many opportunities to efficiently manage (not eliminate) forest fires by adjusting timberland management (Amacher et al. 2005) and by adjusting fire suppression activities (Calkin et al. 2005). In the last 30 to 50 years, foresters have substantially altered the landscape by shifting forest species types toward more favored market species. For example, in the U.S. South, they have expanded the area of southern pine through planting efforts (Alig and Butler 2004, USDA FS 1988), and globally, foresters have expanded nonindigenous plantations by around 2.8 million ha per year (ABARE-Jaako Poyry 1999, UNFAO 2005). To develop a better understanding of both ecological and economic effects, it would seem prudent to build modeling systems that capture both systems and their interactions.

The area of forest land in all regions will ultimately depend not only on climate impacts on forests, but also on climate impacts on agricultural productivity.

The area of forest land in all regions (boreal, temperate, and tropical) will ultimately depend not only on climate impacts in forests, but also on climate impacts on agricultural productivity. If agricultural productivity declines (increases) as a result of climate change, the area of land devoted to agriculture is likely to increase (decrease) in the long run, inducing additional (fewer) pressures on forests. One global study that includes agricultural and forest impacts predicts that climate change will reduce net deforestation rates over the next century (Alcamo et al. 1997). For example, Alcamo et al. predicted that the net rates of global deforestation will decline from 17 million ha per year between 2000 and 2050 without climate change to 14 million ha per year with climate change. They predicted that beyond 2050, climate change could cause net afforestation of 6 million ha per year compared to net deforestation of 0.2 million ha per year without climate change during the same period. Their projection of a gain in forest area arises mainly because agriculture demands less land during climate change.

The Role of Mitigation

A different way that climate change could have large effects on forests is through the policies that stimulate mitigation, such as afforestation, reduced deforestation, and forest management. There has been considerable research on the potential for mitigation to help reduce the costs of climate impacts. Metz et al. (2001) suggested that 60 to 87 Pg C (1 Pg C = 1 billion metric tonnes carbon, or 1×10^{15} g C) could be sequestered in forests over the coming century, and Sohngen and Mendelsohn

(2003) suggested that this amount of carbon could cost up to $187/t C. Such large levels of sequestration would have large effects on land use, potentially increasing the area of forests at the end of the century by 1 billion ha. Large-scale changes in forest management are also possible.

Studies that examine climate change impacts on the forest and agricultural sectors have not considered the influence of mitigation, and mitigation studies have typically not considered climate change impacts.

Studies that examine climate change impacts on the forest and agricultural sectors have not considered the influence of mitigation, and mitigation studies have typically not considered climate change impacts. However, many interactions are likely between mitigation activities and climate change. First, many integrated assessment models of the climate and economic systems suggest that mitigation efforts in forestry can increase the benefits and reduce the costs of climate policy (e.g., Manne and Richels 2006, Sohngen and Mendelsohn 2003). To the extent that forestry mitigation reduces the overall costs of mitigating climate change, policies may be adopted that limit total warming. As shown in Scholze et al. (2006), less warming suggests that the impacts in forests will be reduced in the long run.

Second, if mitigation in forestry becomes an important component of overall climate change policy, future land uses will change substantially. Within the range of carbon prices of $60 to more than $200/t C, Sohngen and Mendelsohn (2003), Sathaye et al. (2006), and Sohngen and Sedjo (2006) suggested that there could be as many as 1 billion more hectares of land in forests by 2100 (or an increase of around 30 percent). These carbon prices are well within the range of current estimates of the costs of stabilizing future climate (Weyant et al. 2006), suggesting that if forestry is ultimately included as a creditable opportunity, then large land-use changes could take place.

The implications of these types of land-use changes for existing ecological models are interesting to consider. For the most part, current ecological models are built on potential forest areas or maps of current land uses (e.g., Bachelet et al. 2004, Cramer et al. 2001, Scholze et al. 2006). For tropical regions, the mitigation efforts described above largely imply reductions in deforestation, thus preservation of existing forest areas in tropical regions. Thus, if reductions in deforestation were included as an option for climate change action, the results of the ecological models for tropical regions would likely be robust because the ecological models already implicitly assume no future deforestation. The limiting factor, of course, is that climate change could alter the relative productivity of farmland in tropical regions, thus altering the relative costs of reducing deforestation. For temperate regions, the mitigation results imply an expansion of forest land. Ecological models that rely on potential vegetation likely already overestimate climate change impacts, and those that rely on current distributions of forests likely underestimate climate change impacts.

Third, results from ecological models examining climate change should influence estimates of the costs of mitigation. Economic modelers thus far have not accounted for climate change impacts when generating marginal abatement cost curves for sequestration. In the face of this limitation, several possibilities exist for whether accounting for climate change impacts would lead to higher or lower estimated costs of mitigation. In the short term, it was noted above that climate change would have its largest implications in boreal regions, and through growth effects on trees. In regions with positive tree growth effects, climate change would reduce sequestration costs as long as the relative value of agricultural land does not rise too much. In regions with negative tree growth effects, climate change would increase sequestration costs.

Ecological models that rely on potential vegetation likely already overestimate climate change impacts, and those that rely on current distributions of forests likely underestimate climate change impacts.

In the medium and long term, forested ecosystems are likely to be influenced by additional factors, including mortality from forest fires and other disturbances, and changes in the distribution of important tree species. Forest management activities to reduce fire frequency and intensity in forests, so as to conserve carbon in the landscape, could increase mitigation costs. Furthermore, an expansion of forest area owing to mitigation suggests more overall hectares burned, which potentially increases the costs of fighting fires. If the geographical distribution of specific tree species changes, or if the geographical distribution of optimal agricultural land changes, as is possible during the medium term, then costs of carbon sequestration could rise owing to rising opportunity cost of holding land in forests or cost of search processes associated with finding the right tree species to plant.

In summary, in the short term, climate change appears to improve the efficiency of mitigation efforts. In the medium to longer run, climate change impacts

Ralph Alig

Changes in agricultural productivity can affect a Nation's forest area.

appear to raise the risks associated with mitigation, and consequently raise the costs of avoiding climate change.

Aside from the interaction between mitigation and climate change, if mitigation efforts are undertaken, they are likely to have substantial impacts on timber markets by affecting prices. Over the long run, most economic studies show that mitigation expands timber supply and reduces timber prices (Murray et al. 2005, Sohngen and Mendelsohn 2003). However, in the short term, mitigation could actually increase timber prices if options such as increasing rotation ages are used. In fact, if large-scale mitigation efforts are undertaken inefficiently, they could have relatively rapid impacts in timber markets by altering the relative value of forestry and agriculture (Alig et al. 1997, Murray et al. 2004, Sohngen and Brown 2004).

Thus, although it is possible to increase carbon sequestration in forests through afforestation, the net effects on overall carbon sequestration from large-scale and quick startup programs may not be as large as anticipated because land markets respond by moving some unprotected forests back into agriculture (i.e., deforestation). Alig et al. (1997), for example, found an approximate 1-to-1 correspondence between hectares that are moved to forests from agriculture and hectares that move the opposite direction, suggesting that large-scale and quick startup afforestation in the U.S. situation may not be the most efficient method for carbon sequestration. Murray et al. (2004) found similarly large "leakage" effects for some regions of the United States, and Sohngen and Brown (2004) found smaller, although still potentially substantial, "leakage" effects for tropical regions. More recent efforts suggest that efficient policies with flux constraints or carbon pricing could provide net sequestration, and that these would increase timber supply both in the short run and long run (Adams et al. 1999; Hoen and Solberg 1994, 1997, 1999; Murray et al. 2005; Sohngen and Mendelsohn 2003).

The development of the so-called second-generation biofuel technology will be of special interest here, as it may cause a large shift in demand for forest fiber.

A related issue that has not been widely examined is the question of substitution between wood and other energy-intensive products like steel, concrete, and aluminum. Energy-intensive constraints on carbon (e.g., high carbon prices) would increase the production costs of energy-intensive products and thus increase the demand for wood products that substitute for them. Such substitution has a permanent effect on the concentration of atmospheric CO_2. The existing empirical studies indicate that the potential here is rather high (e.g., Buchanan and Bry Levine 1999; Burschel et al. 1993; Petersen and Solberg 2002, 2003, 2004, 2005; Raymer 2006). However, higher prices for traditional energy products would also spur the demand for bioenergy products, which as noted above, could have substantial impacts upon land use globally. In particular, large areas of land could be converted from existing forests to support growing needs for bioenergy products based on agricultural crops (Clarke et al. 2007). The development of the so-called second-generation biofuel technology (using hemi-cellulose for producing biofuels) will be of special interest here, as it may cause a large shift in demand for forest fiber. The fiber for this purpose does not have to be of high quality and could easily use salvage harvest biomass, thus counterbalancing the impacts to industrial forestry of damages caused by climate change. The development of technology is closely linked to policy instruments—for example, the new European Union regulations that 20 percent of the transport fuels in the European Union should, by 2020, be based on renewable resources is one main driver for developing the second-generation biofuel technology.

Conclusions

This paper provides a general overview of potential climate change impacts on the forest sector in the short, medium, and long run. The results of our review of the literature suggest that climate change is likely to have small impacts in the near term out to 2025. Short-term impacts could become large, however, if climate change involves significant changes in regional weather patterns or dieback effects that cause timber stock losses. Existing ecological evidence implies that the earliest signs of climate change will be observed in boreal regions. Changes in boreal regions, however, are likely to have limited effects on global markets, although they could have large consequences for communities located near the impacts.

Climate change impacts are likely to accelerate in the medium and long term if mitigation and abatement efforts are not undertaken. Ecological studies suggest that precipitation can offset the effects of warming to some extent, but that there are substantial risks to forests in virtually all regions with global average temperature changes of more than 2.5 °C. These risks include additional disturbances (e.g., fires, insect infestations), changes in distribution of species, and conversion of forests to grasslands or other nonforested vegetation types. Because climate models cannot precisely predict how temperature and precipitation will change geographically, it is difficult to know with certainty where the impacts will occur.

Climate change impacts are likely to accelerate in the medium and long term if mitigation and abatement efforts are not undertaken.

Economic studies have shown that if forest productivity increases (decreases), timber prices will likely fall (increase). Large disturbances caused by climate change can have large influences on timber prices. In particular, large salvage efforts following dieback from forest fires would reduce timber prices, with the lower prices benefiting consumers and reducing producers' economic welfare for landowners. The largest effects during climate change may actually result from market adaptation, with temperate and boreal regions losing market share to subtropical and tropical regions. These trends already appear to be occurring, so climate change would likely only enhance the movement of industrial timber production from developed temperate regions to developing tropical and subtropical regions.

One important influence on forests and timber markets that has not been widely examined is the potential effect of changes in agricultural productivity and agricultural policies. Agriculture and forestry compete for the same land globally, and if climate change alters the productivity of agricultural land or global (European Union, U.S., World Trade Organization) agricultural policies change, then one would expect a change in the demand for agricultural land. Few models have examined the combined effects of climate change on agriculture and forestry, although the studies that have been conducted do not suggest substantial changes in overall land uses relative to the baseline (Alcamo et al. 1997, Alig et al. 2002). Using four climate change scenarios from a national climate change assessment in the late 1990s, Alig et al. (2002) found that climate change leads to less projected forest area than no climate change. Less cropland is projected to be converted to forests owing to increases caused by climate change in overall agricultural crop production and exports. Projected changes for livestock production and prices depend on the specific climate change scenario (and climate model, e.g., Hadley model), with some variation over regions and time.

Evidence is emerging that forestry can play an important role in overall climate change abatement efforts, and if this role emerges, it will entail large changes in how society uses land.

Mitigation efforts could have substantial impacts on timber markets, timber prices, and land use during the entire century. Evidence is emerging that forestry can play an important role in overall climate change abatement efforts, and if this role emerges, it will entail large changes in how society uses land (Alig et al. 2010). These effects include not only changes in the margin between agricultural and forest land, but also increases in the intensity of forest management. Less attention has been given to how mitigation efforts would be affected by the impacts of climate change on forests; however, because climate change is likely to have small, but positive effects on forest productivity in the near term, climate change could reduce the costs of mitigation efforts. Over the longer term, mitigation efforts could be more costly to sustain owing to climate impacts, particularly if society is not successful in reducing greenhouse gas emissions.

A number of studies have used ecological model results in economic models, but there has been little use of economic models in ecological studies.

One of the most important implications of this synthesis of the literature is that there is little evidence that ecologists and economists have worked seriously together to assess climate impacts in ecosystems. A number of studies have used ecological model results in economic models, but there has been little use of economic models in ecological studies. This is problematic because the scale of human influence on ecosystems is large (e.g., see the Millennium Ecosystem Assessment http://ma.caudillweb.com//en/Products.Global.Overview.aspx). One would expect the ecological effects to somehow be moderated or influenced by adaptation in markets, with timber producers and consumers behaving in ways that act to limit economic effects. Further, economists and ecologists working together on feedbacks could help advance the analysis of resiliency associated with climate change, both from ecological and socioeconomic viewpoints. Given that climate change can potentially affect many parts of the global ecosystem and economy, indicators of resiliency would aid in ranking policy responses to climate change. As part of the resiliency analysis, feedback loops would need to be considered. An example is macroeconomic factors that affect forest products markets, such that changing timber values from forest land can affect land use and the other ecosystem goods and services on that forest land, and those production relationships can be affected by climate change and any adaptation or mitigation responses. Interdisciplinary research could advance resiliency rankings while recognizing that economists and ecologists often work at quite different scales. For example, many ecologists work at finer scales than economists and focus on functions and processes viewed by some economists as being at a scale that is data poor and very detailed. Economists can provide analyses that help set the context regarding relative importance of giving more attention to certain feedback loops and ecosystem function resiliency indicators in our global system. Thus, future research is needed to fully integrate ecological and economic models to better understand how forest ecosystems and markets may be affected by climate change.

Changing timber values from forest land can affect land use and the other ecosystem goods and services on that forest land, and those production relationships can be affected by climate change and any adaptation or mitigation responses.

Another area that needs additional attention from the research community is the impact of climate change on nonwood forest products and services, such as biodiversity, recreation, edible fruits, and other nonwood products. These are more difficult to assess because our understanding of the demand for these products is incomplete globally, although knowledge is growing, and also because the uncertainty is rather high regarding the ecological effects from climate change (Kauppi and Solberg 1999). Most likely, however, impacts on nonwood forest products will differ dramatically from place to place, depending on the nature of climate change (Irland et al. 2001, Loomis and Crespi 1999, Mendelsohn and Markowski 1999, Wall 1998). In particular, industrial wood products are less susceptible to climate change because global market systems allow wood trade from region to region. With fewer such established links for nonwood forest products and services, they are likely to exhibit more vulnerability to climate change, at least locally. Impacts on some nonwood products and services, however, would be global regardless of whether or not they are traded across regions (e.g., biodiversity is a global public good, with potentially high public value in all regions).

Acknowledgments

The authors acknowledge financial support through a joint venture agreement from the U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.

English Equivalents

When you know:	Multiply by:	To get:
Hectares (ha)	2.47	Acres
Cubic meters (m^3)	35.3	Cubic feet
Cubic meters	0.00811	Acre-feet
Cubic meters per hectare (m^3/ha)	14.29	Cubic feet per acre
Grams (g)	0.0352	Ounces
Degrees Celius (C)	1.8C + 32	Degrees Fahrenheit
Metric tonnes (t)	1.102	Tons

Literature Cited

Australian Bureau of Agricultural and Resource Economics [ABARE]; Jaako Poyry Consulting, Inc. 1999. Global outlook for plantations. Res. Rep. 99.9. Canberra, Australia: ABARE. 145 p.

Adams, D.M.; Alig, R.J.; McCarl, B.A.; Callaway, J.M.; Winnett, S.M. 1999. Minimum cost strategies for sequestering carbon in forests. Land Economics. 75(3): 360–374.

Alcamo, J.; Kreileman, G.J.J.; Bollen, J.C.; van den Born, G.J.; Gerlagh, R.; Krol, M.S.; Toet, A.M.C.; de Vries, H.J.M. 1997. Baseline scenarios of global environmental change. Global Environmental Change. 6(4): 261–303.

Alig, R.J.; Adams, D.M.; Joyce, L.A.; Sohngen, B. 2004. Climate change impacts and adaptation in forestry: responses by trees and markets. Choices: The Magazine of Food, Farm and Resource Issues. Fall: 7–11.

Alig, R.J.; Adams, D.M.; McCarl, B.A. 2002. Projecting impacts of global climate change on the U.S. forest and agriculture sectors and carbon budgets. Forest Ecology and Management. 169: 3–14.

Alig, R.J.; Adams, D.M.; McCarl, B.A.; Callaway, J.M.; Winnett, S.M. 1997. Assessing effects of mitigation strategies for global climate change with an intertemporal model of the U.S. forest and agriculture sectors. Environmental and Resource Economics. 9: 259–274.

Alig, R.J.; Butler, B.J. 2004. Projecting large-scale area changes in land use and land cover for terrestrial carbon analyses. Environmental Management. 33(4): 443–456.

Alig, R.J.; Latta, G.S.; Adams, D.M.; McCarl, B.A. 2010. Mitigating greenhouse gases: the importance of land base interactions between forests, agriculture, and residential development in the face of changes in bioenergy and carbon prices. Forest Policy and Economics. 12(1): 67–75.

Amacher, G.A.; Malik, A.; Haight, R. 2005. Not getting burned, the importance of fire prevention in forest management. Land Economics. 81: 284–302.

Bachelet, D.; Neilson, R.P.; Hickler, T.; Drapek, R.J.; Lenihan, J.M.; Sykes, M.T.; Smith, B.; Sitch, S.; Thonicke, K. 2003. Simulating past and future dynamics of natural ecosystems in the United States. Global Biogeochemical Cycles. 17(2)1045: 14-1–14-21.

Bachelet, D.; Neilson, R.P.; Lenihan, J.M.; Drapek, R.J. 2004. Regional differences in the carbon source-sink: potential of natural vegetation in the U.S.A. Environmental Management. 33: S23–S43.

Boisvenue, C.; Running, S.W. 2006. Impacts of climate change on natural forest productivity—evidence since the middle of the 20th century. Global Change Biology. 12: 862–882.

Buchanan, A.H.; Bry Levine, S. 1999. Wood-based building materials and atmospheric carbon emissions. Environmental Science and Policy. 2: 427–437.

Burschel P.; Kursten, E.; Larson, B.C.; Weber, M. 1993. Present role of German forests and forestry in the national carbon budget and options to its increase. Water, Air, and Soil Pollution. 70: 325–340.

Calkin, D.E.; Gebert, K.M.; Jones, J.G.; Neilson, R.P. 2005. Forest Service large fire area burned and suppression expenditure trends, 1970–2002. Journal of Forestry. 103(4): 179–183.

Clarke, L.E.; Edmonds, J.A.; Jacoby, H.D.; Pitcher, H.M.; Reilly, J.M.; Richels, R.G. 2007. Scenarios of greenhouse gas emissions and atmospheric concentrations; and review of integrated scenario development and application. Sub-report 2.1A of synthesis and assessment Product 2.1 by the U.S. Climate Change Science Program and the Subcommittee on Global Change Research. Washington, DC: Department of Energy, Office of Biological and Environmental Research. 77 p.

Cramer, W.; Bondeau, A.; Woodward, F.I.; Prentice, I.C.; Betts, R.; Brovkin, V.; Cox, P.M.; Fisher, V.; Foley, J.A.; Friend, A.D.; Kucharik, C.; Lomas, M.R.; Ramankutty, N.; Sitch, S.; Smith, B.; White, A.; Young-Molling, C. 2001. Global response of terrestrial ecosystem structure and function to CO_2 and climate change: results from six dynamic global vegetation models. Global Change Biology. 7: 357–373.

Daigneault, A.; Sohngen, B.; Sedjo, R. 2008. Exchange rates and the competitiveness of the United States timber sector in a global economy. Forest Policy and Economics. 10(3): 108–116.

Gitay, H.; Brown, S.; Easterling, W.; Jallow, B. 2001. Ecosystems and their services. In: Climate change: impacts, adaptation, and vulnerability. Contribution of working group II to the third assessment report of the Intergovernmental Panel on Climate Change. Geneva, Switzerland: 72–98. Chap. 5.

Haight, R.G.; Smith, W.D.; Straka, T.J. 1995. Hurricanes and the economics of loblolly pine plantations. Forest Science. 41(4): 675–688.

Hoen, H.F.; Solberg, B. 1994. Potential and economic efficiency of carbon sequestration in forest biomass through silvicultural management. Forest Science. 40: 429–451.

Hoen, H.F.; Solberg, B. 1997. CO_2-taxing, timber rotations, and market implications. In: Sedjo, R.A.; Sampson, R.N.; Wisniewski, J., eds. Economics of carbon sequestration in forestry. Critical Reviews in Environmental Science and Technology: 27: 151–162.

Hoen, H.F.; Solberg, B. 1999. Policy options in carbon sequestration via sustainable forest management—an example from the North. In: Palo, M., ed. Forest transition and carbon fluxes—global scenarios and policies. World Development Studies No. 15. Helsinki: World Institute for Development Economics Research (WIDER): 117–132.

Houghton, J.T.; Meira Filho, L.G.; Callender, B.A.; Harris, N.; Kattenberg, A.; Maskell, K. 1996. The science of climate change. Contribution of working group I to the second assessment report of the Intergovernmental Panel on Climate Change. Cambridge, United Kingdom: Cambridge University Press. 77 p.

Houghton, R.A. 2003. Revised estimates of the annual net flux of carbon to the atmosphere from changes in land use and land management 1850–2000. Tellus. 55b: 378–390.

Intergovernmental Panel on Climate Change [IPCC]. 2007a. Impacts, adaptation and vulnerability. Working group II report. http://www.ipcc.ch. (September 30, 2009).

Intergovernmental Panel on Climate Change [IPCC]. 2007b. The physical science basis. Working group 1 report. http://www.ipcc.ch. (September 30, 2009).

Irland, L.; Adams, D.M.; Alig, R.J.; Betz, C.J.; Chen, C.; Hutchins, M.; McCarl, B.A.; Skog, K.; Sohngen, B.L. 2001. Assessing socioeconomic impacts of climate change on U.S. forest, wood-product markets, and forest recreation. BioScience. 51(9): 753–764.

Joyce, L.A.; Aber, J.; McNulty, S.; Dale, V.; Hansen, A.; Irland, L.; Neilson, R.; Skog, K. 2001. Potential consequences of climate variability and change for the forests of the United States. In: National Assessment Synthesis Team, ed. Climate change impacts on the United States: the potential consequences of climate variability and change. U.S. Global Change Research Program. Cambridge: Cambridge University Press: 489–522. Chap. 17. http://www.usgcrp.gov/usgcrp/Library/nationalassessment/. (October 2007).

Joyce, L.A.; Mills, J.R.; Heath, L.S.; McGuire, A.D.; Haynes, R.W.; Birdsey, R.A. 1995. Forest sector impacts from changes in forest productivity under climate change. Journal of Biogeography. 22: 703–713.

Kallio, A.M.T.; Moiseyev, A.; Solberg, B. 2006. Economic impacts of increased forest conservation in Europe: a forest sector analysis. Environmental Science and Policy. 491: 9–18.

Kauppi, P.; Solberg, B. 1999. Forests and climate change—ecological and economic challenges. In: Palo, M.; Uusivuori, J., eds. World forest, society and environment. World Forests: 121–126. Vol. 1.

King, G.A.; Neilson, R.P. 1992. The transient response of vegetation to climate change: a potential source of CO_2 to the atmosphere. Water, Air, and Soil Pollution. 94: 365–383.

Kirschbaum, M.U.F.; Fischlin, A. 1996. Climate change impacts on forests. In: Watson, R.T.; Zinyowera, M.C.; Moss, R.H., eds. Scientific-technical analysis of impacts, adaptations, and mitigation of climate change. Contribution of working group II to the second assessment of the Intergovernmental Panel on Climate Change. Cambridge, United Kingdom: Cambridge University Press: 1–14. Chap. 1.

Kurz, W.A.; Apps, M.J. 1999. A 70-year retrospective analysis of carbon fluxes in the Canadian forest sector. Ecological Applications. 9(2): 526–547.

Latta, G.; Temesgen, H.; Adams, D.; Barrett, T. 2010. Analysis of potential impacts of climate change on forests of the United States. Forest Ecology and Management. 259(4/5): 720–729.

Lindner M. 1999. Forest management strategies in the context of potential climate change. Forstwissenschafliches Centralblatt. 118(1): 1–13.

Lindner M. 2000. Developing adaptive forest management strategies to cope with climate change. Tree Physiology. 20(5-6): 299–307.

Loomis, J.; Crespi, J. 1999. Estimated effect of climate-change on selected outdoor recreation activities in the United States. In: Mendelsohn, R.; Neumann, J.E., eds. The impact of climate change on the United States economy. Cambridge, United Kingdom: Cambridge University Press: 289–314. Chap. 11.

Manne, A.S.; Richels, R.G. 2006. The role of non-CO_2 greenhouse gases and carbon sinks in meeting climate objectives. The Energy Journal. 27: 393–404.

McCarl, B.A.; Burton, D.M.; Adams, D.M.; Alig, R.J.; Chen, J. 2000. Forestry effects of global climate change. Climate Research. 15(3): 195–205.

Melillo, J.M.; McGuire, A.D.; Kicklighter, D.W.; Moore, B., III; Vorosmarty, C.J.; Schloss, A.L. 1993. Global climate change and terrestrial net primary production. Nature. 363: 234–240.

Mendelsohn, R.; Markowski, M. 1999. The impact of climate change on outdoor recreation. In: Mendelsohn, R.; Neumann, J.E., eds. The impact of climate change on the United States economy. Cambridge: Cambridge University Press: 145–157. Chap. 10.

Metz, B.; Davidson, O.; Swart, R.; Pan, J. 2001. Climate change 2001: mitigation. Cambridge, United Kingdom: Cambridge University Press. 752 p.

Mills, J.R.; Haynes, R.W. 1995. Influence of climate change on supply and demand for timber. In: Joyce, L.A., ed. Productivity of America's forests and climate change. Gen. Tech. Rep. RM-GTR-271. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 46–55.

Murray B.C.; McCarl, B.A.; Lee, H. 2004. Estimating leakage from forest carbon sequestration programs. Land Economics. 80(1):109–124.

Murray, B.C.; Sohngen, B.L.; Sommer, A.J.; Depro, B.M.; Jones, K.M.; McCarl, B.A.; Gillig, D.; DeAngelo, B.; Andrasko, K. 2005. EPA-R-05-006. Greenhouse gas mitigation potential in U.S. forestry and agriculture.

Washington, DC: U.S. Environmental Protection Agency, Office of Atmospheric Programs. 144 p.

Nilsson, S.; Shvidenko, A. 2000. Dynamics of carbon budget of the Russian forest sector in 1961–1998 and scenarios for 2010 [Abstract]. In: Apps, M.; Marsden, J., eds. Proceedings, The role of boreal forests and forestry in the global carbon budget. Edmonton, Canada: International Science Conference: 77–89.

Norby, R.J.; DeLucia, E.H.; Gielend, B.; Calfapietra, C.; Giardina, C.P.; King, J.S.; Ledford, J.; McCarthy, H.R.; Moore, D.J.P.; Ceulemans, R.; De Angelis, P.; Finzi, A.C.; Karnosky, D.F.; Kubiske, M.E.; Lukac, M.; Pregitzer, K.S.; Scarascia-Mugnozzan, G.E.; Schlesinger, W.H.; Oren, R. 2005. Forest response to elevated CO_2 is conserved across a broad range of productivity. Proceedings of the National Academy of Sciences. 102(50): 18052–18056.

Palo, M.; Lehto, E.; Uusivuori, J. 2000. Underlying causes of deforestation: modelling causes of deforestation with 477 subnational units. In: Palo, M.; Vanhanen, H., eds. World forests from deforestation to transition. World Forest Series. Helsinki, Finland: Kluwer: 101–124. Vol II.

Perez-Garcia, J.; Joyce, L.A.; McGuire, A.D.; Binkley, C.S. 1997. Economic impact of climatic change on the global forest sector. In: Sedjo, R.A.; Sampson, R.N.; Wisniewski, J., eds. Economics of carbon sequestration in forestry. Boca Raton, FL: Lewis Publishers: 123–129.

Perez-Garcia, J.M.; Joyce, L.A.; McGuire, D.; Xiao, X. 2002. Impacts of climate change on the global forest sector. Climatic Change. 54: 439–461.

Petersen, A.K.; Solberg, B. 2002. Greenhouse gas emissions, life-cycle inventory and cost-efficiency of using laminated wood instead of steel construction. Case: beams at Gardermoen airport. Environmental Science and Policy. 5(2): 169–182.

Petersen, A.K.; Solberg, B. 2003. Substitution between floor constructions in wood and natural stone: comparison of energy consumption, GHG emissions, and costs over the life cycle. Canadian Journal of Forest Research. 33(6): 1061–1075.

Petersen, A.K.; Solberg, B. 2004. Comparison of GHG emissions and costs over the life cycle of flooring materials. Climatic Change. 64: 143–167.

Petersen, A.K.; Solberg, B. 2005. Environmental and economic impacts of substitution between wood products and alternative materials: a review of micro-level analyses from Norway and Sweden. Forest Policy and Economics. 7(3): 249–259.

Raymer, A.K.P. 2006. A comparison of avoided greenhouse gas emissions when using different kinds of wood energy. Biomass and Bioenergy. 30: 605–617.

Reed, W.J. 1984. The effects of the risk of fire on the optimal rotation of a forest. Journal of Environmental Economics and Management. 11(2): 180–190.

Reilly, J.; Tubiello, F.; McCarl, B.; Abler, D.; Darwin, R.; Fuglie, K.; Hollinger, S.; Izaurralde, C.; Jagta, S.; Jones, J.; Mearns, L.; Ojima, D.; Paul, E.; Paustian, K.; Riha, S.; Rosenberg, N.; Rosenzweig, C. 2003. U.S. agriculture and climate change: new results. Climatic Change. 57: 43–69.

Sathaye, J.; Makundi, W.; Dale, L.; Chan, P.; Andrasko, K. 2006. HG mitigation potential, costs and benefits in global forests. Energy Journal. 27: 127–162.

Schimel, D.; Melillo, J.; Tian, H.Q.; McGuire, A.D.; Kicklighter, D.; Kittel, T.; Rosenbloom, N.; Running, S.; Thornton, P.; Ojima, D.; Parton, W.; Kelly, R.; Sykes, M.; Neilson, R.; Rizzo, B. 2000. Contribution of increasing CO_2 and climate to carbon storage by ecosystems in the United States. Science. 287(5460): 2004–2006.

Scholze, M.; Knorr, W.; Arnell, N.W.; Prentice, I.C. 2006. A climate-change risk analysis for world ecosystems. Proceedings of the National Academy of Sciences. 103(35): 13116–13120.

Shugart, H.H.; Antonovsky, M.Y.; Jarvis, P.G.; Sandford, A.P. 1986. CO_2, climatic change, and forest ecosystems. In: Bolin, B.; Doos, B.R.; Jager, J.; Warrick, R.A., eds. The greenhouse effect, climatic change and ecosystems. New York, Wiley: 475–521.

Smith, T.M.; Shugart, H.H. 1993. The potential response of global terrestial carbon storage to a climate-change. Water Air and Soil Pollution. 70(1-4): 629–642.

Sohngen, B.; Alig, R. 2000. Mitigation, adaptation, and climate change: results from recent research on U.S. timber markets. Environmental Science and Policy. 3: 235–248.

Sohngen, B.; Brown, S. 2004. Measuring leakage from carbon projects in open economies: a stop timber harvesting project as a case study. Canadian Journal of Forest Research. 34: 829–839.

Sohngen, B.; Mendelsohn, R. 1998. Valuing the market impact of large scale ecological change: the effect of climate change on U.S. timber. American Economic Review. 88(4): 689–710.

Sohngen, B.; Mendelsohn, R. 2003. An optimal control model of forest carbon sequestration. American Journal of Agricultural Economics. 85(2): 448–457.

Sohngen, B.; Mendelsohn, R. 2007. A sensitivity analysis of carbon sequestration. In: Schlesinger, M.; Kheshgi, H.S.; Smith, J.; de la Chesnaye, F.C.; Reilly, J.M.; Wilson, T.; Kolstad, C., eds. Human-induced climate change: an interdisciplinary assessment. Cambridge, United Kingdom: Cambridge University Press: 217–226. Chap. 19.

Sohngen, B.; Mendelsohn, R.; Neilson, R. 1998. Predicting CO_2 emissions from forests during climate change: a comparison of human and natural response models. Ambio. 27(7): 509–513.

Sohngen, B.; Mendelsohn, R.; Sedjo, R. 1999. Forest management, conservation, and global timber markets. American Journal of Agricultural Economics. 81(1): 1–13.

Sohngen, B.; Mendelsohn, R.; Sedjo, R. 2000. Measuring climate change impacts with a global timber model. Working Paper. Department of Agricultural, Environmental, and Development Economics. Columbus, OH: Ohio State University. 67 p.

Sohngen, B.; Sedjo, R. 2000. Potential carbon flux from timber harvests and management in the context of a global timber market. Climatic Change. 44: 151–172.

Sohngen, B.; Sedjo, R. 2006. Carbon sequestration in global forests under different carbon price regimes. Energy Journal. 27: 109–126.

Solberg, B.; Brooks, D.; Pajuoja, H.; Peck, T.J.; Wardle, P.A. 1996. Long-term trends and prospects in world supply and demand for wood and implications for sustainable forest management—synthesis. In: Solberg, B., ed. Long-term trends and prospects in world supply and demand for wood and implications for sustainable forest management. Res. Rep. 6. Joensuu, Finland: European Forest Institute: 7–42.

Solberg, B.; Moiseyev, A.; Kallio, A.M.I. 2003. Economic impacts of accelerating forest growth in Europe. Forest Policy and Economics. 5: 157–171.

Solomon, A.M.; Kirilenko, A.P. 1997. Climate change and terrestrial biomass: What if trees do not migrate? Global Ecology and Biogeography Letters. 6: 139–148.

Tian, H.; Melillo, J.M.; Kicklighter, D.W.; McGuire, A.D.; Helfrich, J.; Moore, B., III; Vorosmarty, C.J. 1998. Effect of interannual climate variability on carbon storage in Amazonian ecosystems. Nature. 396: 664–667.

Trømborg, E.; Buongiorno, J.; Solberg, B. 2000. The global timber market: implications of changes in economic growth, timber supply and technological trends. Forest Policy and Economics. (1)1: 53–71.

United Nations Food and Agriculture Organization [UNFAO]. 1999. State of the world's forests, 1999. Rome, Italy. 135 p.

United Nations Food and Agriculture Organization [UNFAO]. 2005. Global forest resources assessment 2005: progress towards sustainable forest management. FAO Forestry Pap. 147. Rome, Italy. http:\\www.fao.org. (November 2009).

U.S. Department of Agriculture, Forest Service [USDA FS]. 1988. The South's fourth forest: alternatives for the future. Res. Rep. No. 24. Washington, DC. 512 p.

Vegetation/Ecosystem Modeling and Analysis Project [VEMAP]. 1995. Vegetation/Ecosystem Modeling and Analysis Project (VEMAP): comparing biogeography and biogeochemistry models in a continental-scale study of terrestrial ecosystem responses to climate change and CO_2 doubling. Global Biogeochemical Cycles. 9(4): 407–437.

Wall, G. 1998. Implications of global climate change for tourism and recreation in wetland areas. Climatic Change. 40: 371–389.

Watson, R.T.; Noble, I.R.; Bolin, B.; Ravindranath, N.H.; Verardo, D.J.; Dokken, D.J., eds. 2000. Land use, land-use change, and forestry, 2000. Special report of the Intergovernmental Panel on Climate Change. Cambridge, United Kingdom: Cambridge University Press. 377 p.

Watson, R.T.; Zinyowera, M.C.; Moss, R.H. 1996. Scientific-technical analysis of impacts, adaptations, and mitigation of climate change. Contribution of working group II to the second assessment of the Intergovernmental Panel on Climate Change. Cambridge, United Kingdom: Cambridge University Press. 245 p.

Watson, R.T.; Zinyowera, M.C.; Moss, R.H. 1998. The regional impacts of climate change: an assessment of vulnerability. A special report of the Intergovernmental Panel on Climate Change working group II. Cambridge, United Kingdom: Cambridge University Press. 129 p.

Westerling, A.L.; Hidalgo, H.G.; Cayan, D.R.; Swetnam, T.W. 2006. Warming and earlier spring increase western U.S. forest wildfire activity. Science. 313: 940–943.

Weyant, J.P.; de la Chesnaye, F.C.; Blanford, G.J. 2006. Overview of EMF-21: multigas mitigation and climate policy. Energy Journal. 27: 1–32.

Chapter 3: Modeling Land-Use Changes as Mitigation Options Involving the Forestry and Agricultural Sectors

by Ralph J. Alig

United States Land Resources

The United States has a wealth of land, 931.5 million ha in total, with about one-third of that in forest. Of the 304 million ha of forest land, about two-thirds (208 million ha) is classified as timberland that meets productivity standards and is available for timber management and harvests (Smith et al. 2009) (also see glossary). Despite major historical transfers of land to agriculture, the United States still has a very large forested area, roughly two-thirds of the land that was forested in 1600. Over the 20th century, U.S. forest land area declined about 4 million ha in net, with the largest recent losses to developed uses (Alig et al. 2010b).

One source of information on forest resource and utilization trends are the periodic Resources Planning Act assessments, which document current resource conditions and trends, and project future changes (e.g., USDA FS 2001). This information helps to establish benchmarks and future milestones for long-term performance indicators, and the timber assessment (e.g., Adams and Haynes 2007), for example, utilizes 50 years of historical data, and makes projections 50 years into the future. The assessment draws upon more than 70,000 permanent data plots across the United States. The assessment considers the broad workings of the economy, such as continuing increase in recycling and efficiency in paper production. The most recent land base assessment projects a 6-percent reduction in forest-land cover by 2062, as projected increases in urban and developed uses will likely intensify competition for remaining land between the agricultural and forestry sectors (Alig et al. 2010b). These projections are based on assumptions of no significant carbon payments or availability of other significant climate change policies, as this base case is a reference point representing policies frozen in place from the past decade. In the next section, we will review studies using alternative projections under different policies (e.g., carbon payments to landowners).

The extent of the U.S. timberland base and its many forest types provide multiple options for responding to climate-induced changes.

The extent of the U.S. timberland base and its many forest types provide multiple options for responding to climate-induced changes. Many biological and economic opportunities exist to increase forest growth on the sizable U.S. timberland base, many of which would increase carbon stores (e.g., Birdsey et al. 2000, Hair et al. 1996, Vasievich and Alig 1996). Tree planting on marginal agricultural land has been suggested numerous times as one strategy for increasing terrestrial carbon stores (e.g., Adams et al. 1999, Moulton and Richards 1990). Currently, less than 10 percent of U.S. timberland is planted. The majority of the planting is in the South, predominantly on private lands and conifer species (e.g., loblolly pine [*Pinus taeda* L.]).

Although the planting of trees to create forest plantations has emerged as a major activity in recent decades, about 90 percent of U.S. forest land area has naturally regenerated stands. Major U.S. forest regions have widely different potentials to attract private investments in tree planting and in forest production more generally (Alig et al. 2001). Rapid tree growth generally translates into higher potential economic returns to investors; tree growth is fastest in the South and high-rainfall areas of the Pacific Northwest.

The South, comprising 13 states, has accounted for about 80 percent of U.S. tree planting. The region has large areas of marginal agricultural land that could be planted to trees, and is near major wood-processing facilities that are relatively close to the large concentration of population in the East. In 1998, 10 states in the South each planted more than 40 000 ha (100,000 acres), collectively more than 810 000 ha (2 million acres), 77 percent of the U.S. total (Moulton 1999). In the remainder of the United States in 1998, the West had 16 percent of the Nation's tree planting while the North had 4 percent. Spikes in tree planting on nonindustrial private lands are associated with major government programs, such as the Soil Bank Program in the late 1950s and the Conservation Reserve Program of the latter half of the 1980s (fig. 3-1).

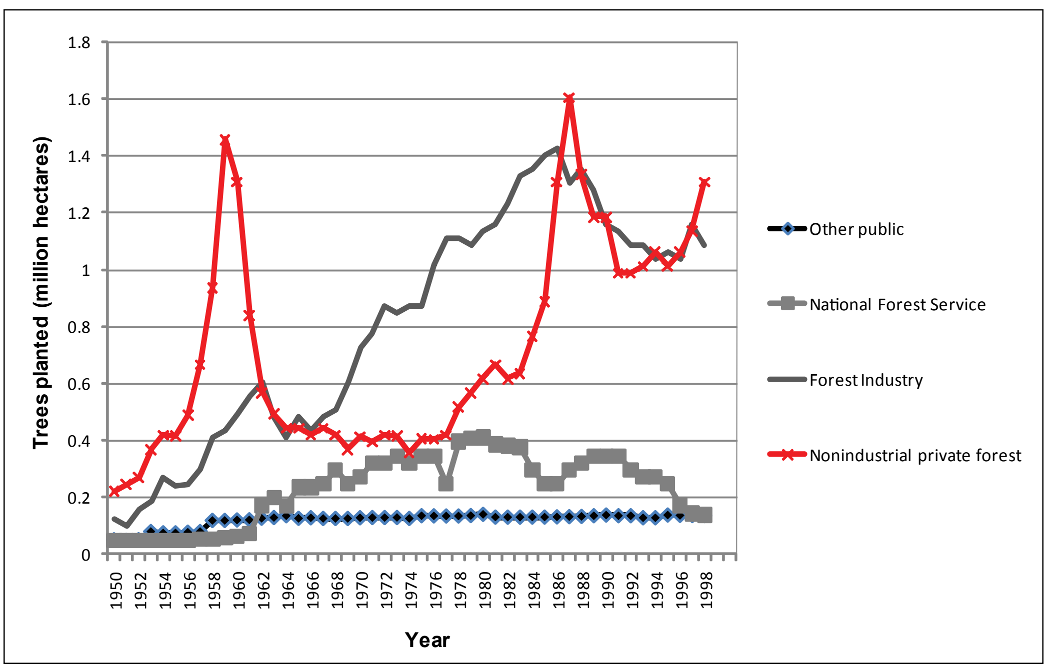


Figure 3-1—Tree planting by ownership in the United States, 1950–1998 (adapted from Moulton 1999).

The South has been the leading tree planting area in the United States for a number of reasons, including a favorable climate (long growing season and generally abundant precipitation), excellent markets for wood owing to the heavy concentration of forest industry in the region, and comparatively less competition for land from agriculture. The South does have an important and diversified agricultural sector, based largely on fruits and vegetables (citrus, onions, peaches, and other truck crops), rice, tobacco, cotton, poultry, hogs, and other meats. The South is not a significant producer of major field crops such as corn and wheat, which in some other regions require large areas.

The South enjoys a cost advantage in that southern pine seedlings (e.g., loblolly and slash pine [*Pinus elliottii* Engelm.]) need only be grown in nurseries for 1 year before they are ready for field planting. Currently, high-quality, genetically improved southern pines are available in the South at a lower cost than conifer seedlings in the North (white pine, red pine [*Pinus* L. sp.], and spruces [*Picea* A. Dietr. sp.]) and West (Douglas-fir [*Pseudotsuga menziesii* (Mirb.) Franco] and ponderosa pine [*Pinus ponderosa* C. Lawson]). Those outside the South are typically grown for 2 to 3 years, and may have to be transplanted within the nursery.

The U.S. South also typically has lower opportunity costs regarding potential net returns from competing land uses such as agriculture. The South is a key supplier of fiber for papermaking and contains about two-thirds of the fast-growing coniferous plantations in the world, equal in 1997 to about 12 million ha (30 million acres) of southern pine plantations (Adams and Haynes 2007, Alig and Butler 2004). Planted pine area in the South increased more than 10-fold since 1952, evidence of how quickly some changes in the forest resource can occur. The area of planted pine in the South is projected to increase by more than 40 percent over the next 50 years (Alig and Butler 2004), in a base case without any carbon-related payments to landowners.

Potential for further expansion of tree planting will depend in part on changes in incentives to landowners.

Potential for further expansion of tree planting will depend in part on changes in incentives to landowners, discussed in more detail in chapter 5 in this volume. If substantial carbon-related payments are made available, this could markedly affect the attractiveness of forest plantings relative to other land uses (Alig et al. 2010a). Moulton and Richards (1990) identified more than 107 million ha (265 million acres) of environmentally sensitive (e.g., erodible or wet soils) or lower productivity agricultural land that was suitable for tree planting in the United States. Their estimates include substantial amounts of afforestable land outside the South. The large majority is currently in cropland. Such afforestation opportunities are on private land and concentrated in the East. Afforestation of even a portion of such agricultural land could economically and substantially increase forest growth and

USDA Forest Service

The South and Midwest have relatively large amounts of land suitable for afforestation as a mitigation option.

carbon sequestration (e.g., Alig et al. 1997, 2010a; Birdsey et al. 2000; Vasievich and Alig 1996). Planting of short-rotation woody crops on marginal agricultural lands (e.g., Alig et al. 1999b) for use in generating biomass energy could also affect interactions between the forestry and agricultural sectors, including increased use of woody biomass in co-firing wood processing facilities (White 2010).

An earlier range of estimates of how much carbon could be stored is provided for a hypothetical program to afforest 9.3 to 18.2 million ha (23 to 45 million acres) of marginal cropland and pastureland (Birdsey et al. 2000: table 8.3). That program would be phased in over a 10-year period and could effect a change in carbon storage of 50 teragrams (Tg) carbon/yr, at an annual cost of $350 to 770 million, with 20 to 30 years to achieve the program's carbon sequestration target.

A key question regarding afforestation is how many financially attractive afforestation opportunities will be implemented by landowners.

A key question regarding afforestation is how many financially attractive afforestation opportunities will be implemented by landowners. For example, the Conservation Reserve Program was the Nation's largest 5-year tree planting program in the 1980s and 1990s. Original estimates were that about twice as many acres would be planted to trees if owners responded to financially attractive opportunities. However, ultimately only about half of the potential financially attractive acres were planted to trees (Alig 2003). Estimating the likelihood of tree planting by different owners can involve a complex of factors (Alig et al. 1990a), including financial feasibility, constraints, and environmental and esthetic considerations. Option values may be important to owners, and cultural tendencies also influence receptivity to tree planting by some owners (Fairweather 1996). Consideration of nonmarket benefits and the consequences of potential irreversibility of land conversion may significantly affect a landowner's willingness to agree to a major land-use change that will result in carbon being sequestered. When these factors are considered, the costs of carbon sequestration can be affected (Alig 2003, also see chapter 4 in this volume). In addition, many owners face capital constraints when faced with large upfront planting costs. Alig et al. (1999a) showed that afforestation amounts could be reduced markedly below economically optimal levels if such constraints are effective.

Several studies have noted that when owners did afforest former agricultural land under past programs, there was a relatively high retention rate of such plantations 10 to 15 years after establishment (Alig et al. 1980; Kurtz et al. 1980, 1994) and well beyond the program date. This ran counter to expectations by some that farmers would quickly convert back to agriculture use after the incentive program ended. Owners recognized potential timber value of young plantations after establishment. A number of stands were in need of silvicultural treatment according to timber prescriptions, often having overstocked conditions as owners applied relatively little management after establishment. Further, many such stands are often regenerated back to forest after harvest.

When owners did afforest former agricultural land under past programs, there was a relatively high retention rate of such plantations 10 to 15 years after establishment.

Ownership changes in the forest land base may result in different land management objectives or new private owners with different available resources to invest in forest management. Changes in forest ownership have been substantial in recent decades (Best and Wayburn 2001). Although a national set of associated data is not yet available, some broad outlines of the changes are becoming evident. The South, in particular, has seen tens of millions of hectares change hands in recent decades (Clutter et al. 2005). Traditional industrial ownership of forest land has shrunk considerably, as land divested by industry in the South is now owned by timber management investment organizations (TIMOs) and Real Estate Investment Trusts (REITs) and is classified as nonindustrial private forest (NIPF) land. Such owners do not have processing facilities (e.g., mills) that would require a steady supply of timber, and so they have more flexibility to move into or out of a specific forest type or region to meet financial goals. Most owners also do not have the same level of investment in forestry research or firefighting materials as traditional industrial owners.

Among forest ownerships, the NIPF ownership is generally the most affected by land-use conversions and changes in land-use policy affecting private land.

At the same time, on average, owners of NIPF forest land are getting older, and nonheirs may take ownership of some land during transitions in families. This can also lead to smaller average forest parcel sizes (Butler and Leatherberry 2004), as part of the parcelization process. Among forest ownerships, the NIPF ownership is generally the most affected by land-use conversions and changes in land-use policy affecting private land. Change in total forest area is the net result of the conversion of forest land to nonforest and the shifting of nonforest to forest land by natural reversion or afforestation.

Given prospects of land-use changes, avoided deforestation as a mitigation option has been examined in a number of studies (e.g., Malmsheimer et al. 2008). Land-use changes from forest to nonforest use releases forest-stored greenhouse gases (GHGs) into the atmosphere. In the United States between 1982 and 1997, more than 8 million ha were deforested. The destination of about half of the

USDA Forest Service

Afforestation on erodible or other environmentally sensitive agricultural land can have substantial other co-benefits, such as reduction of water pollution, in addition to contributing to climate change mitigation.

converted forest area was to urban and developed uses, with more than 4 million ha of U.S. nonfederal forests converted to developed uses according to National Resources Inventory (NRI) estimates. That is an area larger than the combined current forest area of five Northeastern States (Connecticut, Delaware, Maryland, New Jersey, and Rhode Island). Between 1992 and 1997, the rate of deforestation increased and the proportion of urban and developed uses as a destination for deforested acres increased to 55 percent of the total deforestation (USDA NRCS 2007), with more than 400 000 ha converted to developed uses per year.

The largest increases in U.S. developed area in recent decades have been in the Southeastern region of the country (13 states from Virginia to Texas). Aside from the United States as a whole, this region provides more timber harvest than any country in the world (Wear and Greis 2002). Between 1982 and 1997, the U.S. South had 7 of the 10 states with the largest average annual additions of developed area according to the USDA Natural Resource Conservation Service (2007). The top three—Texas, Florida, and North Carolina—each added more developed area than did the country's most populous state, California. A contributing factor to expansion of developed area there and in other regions is the decreasing number of people per household (Alig et al. 2004b), owing to decreasing family size, popularity of second homes, divorce rate, and growing number of older adults living alone.

Land-use changes can be prompted by actions in different sectors, with significant drivers of land-use change being changes in population and personal incomes.

Adding another 125 million people in the United States over the next half century will lead to more urban and developed area and affect costs of land conversion. Expanded developed area may lead to a net reduction in the area of private timberland available for carbon sequestration and increase forest carbon sequestration costs, with conversions to urban and developed uses outweighing timberland area additions from agriculture in the business-as-usual case. Projections are that loss of forest land to other uses will be substantial (e.g., Alig et al. 2010b), causing net release of net GHGs currently stored in those forests and also precluding future sequestration opportunities for such forests to take carbon dioxide (CO_2) out of the air and store it as they grow. Conversion of forests can also reduce open space and threaten the ability of diverse forest-land-based ecosystems to provide a variety of habitats for wildlife; help to cleanse the air and water; supply timber, fuelwood, and other harvested products; serve as places for recreation; and provide other goods and environmental services such as mitigation of global climate change (Alig 2007).

Estimates of costs for carbon programs differ notably across some studies (see chapter 4 this volume for more details), in part because of differences in scope and underlying assumptions built into the analyses. This includes whether opportunity costs of the land and market effects on land and resource prices are incorporated, causing estimates of carbon sequestration to rise. For example, Birdsey et al. (2000) indicated a broad range of U.S. cost estimates. Plantinga and Mauldin (2001) also pointed out that climatic change could have a potentially large effect on the costs of afforestation. Their analysis suggests that regions that are now cost-effective for afforestation may not be so in the future, and vice versa.

Mitigation Options Involving the Forestry and Agricultural Sectors

Forests currently play an important role among major land uses, offsetting approximately 13 percent of U.S. GHG emissions in 2007 (US EPA 2009). United States forestry can play a substantial role in climate change mitigation through carbon sequestration—tree, litter, soil, understory, and harvested wood products—and bioenergy feedstocks (see White 2010 for information on forest bioenergy aspects). Mitigation activities in the forest sector generally can be competitive compared to opportunities in agriculture (e.g., reduced soil tillage, manure management), including afforestation of agricultural land. Estimates are affected by regional differences, dynamics of forest growth and carbon sequestration, forest ownership differences, and interactions with other sectors of the economy.

Given the large portion of the U.S. land base in forest and agricultural uses, a linked model of the two sectors facilitates investigating impacts of climate change

A linked model of the forest and agriculture sectors facilitates investigating impacts of climate change on those uses.

on those uses (e.g., Alig et al. 2002), land-based strategies that could contribute to mitigation (Adams et al. 1999, Alig et al. 1997), and adaptation options resulting from market-based actions (e.g., Alig et al. 2002, 2004a). In the 1990s, the lines of modeling from the forest and agricultural sectors were merged in developing the FASOM model (Adams et al. 1996), which linked the two sectors for climate change analyses. Unique features of the FASOM-GHG modeling system include linkage of forestry and agricultural commodity markets, with a connection of those markets to private land-use decisions for forest, crop, and grassland. The model uses a 5-year time step and has full carbon accounting in the forest and agricultural sectors, including from forest through final products and disposal.

Research supporting the FASOM-GHG model started decades ago, with Adams and Haynes (2007) on the forestry side introducing the Timber Assessment Market Model (TAMM) in the USDA Forest Service's Forest and Rangeland Renewable Resources Planning Act (RPA) assessments. The basic structure of the forest sector modeling grew to include a family of models applied in decennial RPA Timber Assessments: TAMM (Adams and Haynes 1996), NAPAP (North American Pulp and Paper model; e.g., Ince 1994), ATLAS (Aggregate Timberland Assessment System; Mills and Kincaid 1992), and AREACHANGE (Alig et al. 1990b, 2003; Alig and Butler 2004). Timber inventory data and estimates of current and future timber yields were taken in large part from the ATLAS input used for the 2000 RPA Timber Assessment and 2005 Update (Adams and Haynes 2007, Alig and Bair 2006). The AREACHANGE models provide timberland area and forest type allocations to the ATLAS model. TAMM and NAPAP are market projection models of the solid wood and fiber products sectors in the United States and Canada. In ATLAS, harvested lands are regenerated (grown) according to exogenous assumptions regarding the intensity of management and associated yield volume changes. The timberland base is adjusted for gains and losses projected over time by the AREACHANGE models, including afforestation of the area moving from agriculture into forestry.

On the agricultural side, the Agricultural Sector Model (ASM) is a spatially disaggregated agricultural sector model representing the United States in terms of 63 production regions and 10 market regions depicting trade with a number of foreign countries. The ASM depicts production in an equilibrium year and is thus an intermediate-run model giving implications for policy after it has been fully worked into the sector. The ASM model has been in use for more than 20 years. The model has been used to study effects of climate change, greenhouse gas mitigation, El Niño forecasting, conservation tillage, new cropping technology, pesticide bans, and farm program revisions, among numerous other applications (McCarl et al. 1998). Because large areas of land can move between forestry and agricultural

Emily Jederlinich

Improved land-use data regarding recent trends would aid in climate change analyses.

uses, responses from (and options in) the agricultural sector must be considered for effective GHG policy analysis. Increased demands for land will affect forestry's potential contributions to addressing climate change—particularly demands related to biofuels production this decade.

The FASOM-GHG model is dynamic in that it solves jointly for the multi-market, multiperiod equilibrium in the linked agricultural and forestry sectors. The nonlinear programming model depicts the allocation of land over time for competing activities in the two sectors. A solution reflects price and quantity equilibria established in each sector in each period, where producers and consumers have perfect knowledge of market conditions in all periods. To our knowledge, the FASOM-GHG model is unique in its modeling of multiple forest-related markets, including both logs and mill-processed products. A key capability of the model involving the forest sector is to be able to examine deforestation, reforestation, and afforestation, based on maximizing net returns to different land uses (Alig et al. 2001), and linkage of land-use changes and land management (e.g., forest thinning). This platform aids in evaluating welfare and market impacts of alternative policies for sequestering carbon in forest ecosystems and agricultural practices. The model has 9 forestry regions and 11 agricultural regions (Adams et al. 1996). The FASOM model includes full carbon accounting of the U.S. forest sector from forest through final products and disposal.

A key capability of the model involving the forest sector is to be able to examine deforestation, reforestation, and afforestation, based on maximizing net returns to different land uses.

The FASOM model was expanded and enhanced in the 2000s on both the forestry and agricultural sides. Products (e.g., softwood lumber) were added on the forestry side as well as log markets, along with an increased number of forest types (e.g., for the South, seven planted-pine management intensity classes), limits on periodic shifts between land uses, updated exogenous projections of conversion of agricultural and forest land to urban and developed uses, and expanded forest bioenergy modeling. Extensive modifications were made on the agricultural side as well, including improvement of agricultural carbon sequestration dynamics, expansion of the scope of agricultural-sector GHG emission source and mitigation strategy coverage, and addition of carbon accounting related to use of fossil fuels. After exiting the forest or agricultural land base for conversion to urban and developed uses, carbon is tracked on the developed land, a capability added in recent years.

Preventing GHG emissions from deforestation is increasingly being promoted, both internationally and domestically (e.g., Malmsheimer et al. 2008). The capability of forests to remove GHGs from the atmosphere over time in the United States has been affected by land-use changes such as deforestation. In the earliest part of the 20th century, most deforestation was due to conversion to agriculture, but in recent decades the largest losses of forest area have been to developed uses (Alig et al. 2010b). It is important to be able to model jointly afforestation, reforestation, and deforestation to capture the different effects on GHG storage and release, as well as to account for the total area changes involving forests.

An early application of the FASOM model by Alig et al. (1997) demonstrated the leakage possibilities of afforesting substantial amounts of land to sequester GHG, especially increasing carbon sequestration. They estimated that the response of land markets to afforesting more than 4 million ha in the South in one decade would be to essentially have no net gain in forest area—due to countervailing transfers of other forest land to agricultural use as the price of forest land dropped and agricultural land value rose because of the afforestation amount.

Greenhouse gas emission mitigation actions in the forestry and agriculture sectors can be less costly than comparable actions in other sectors.

In general, leakage as an unintended consequence of policies can occur via unintended (and unregulated) adjustments in land use between forest and agricultural sectors in response to a sequestration policy. Policy design can affect the amount of leakage. Other issues involving policy design include consideration of permanence and whether current sequestration is followed by future GHG releases. Additionality is another issue (Maness 2009) and is discussed in chapter 5 in this volume.

Several analyses using the FASOM-GHG model indicate that GHG emission mitigation actions in the forestry and agriculture sectors can be less costly than comparable actions in other sectors (most notably transportation and power generation). One earlier national analysis using FASOM model runs estimated that between 10 and 25 percent of current U.S. GHG emissions could be offset through a combination of actions in forestry and agriculture, including reduced tillage, afforestation, improved forest management, improved nutrient management, manure management, and bioenergy production (Murray et al. 2005). The study estimated that costs of these actions would range from $16.50 to $27.50/tonne ($15 to $25/ton) of CO_2 mitigated. For the sectors involved, the costs for these actions multiplied by offset production could translate into an additional $9 to $42 billion per year in increased gross revenue for the agricultural and forestry sectors.

Mitigation strategies involving forest ecosystems will be affected by climate change. Potential effects of global climate change on the U.S. forest sector, including impacts on forest carbon inventories, may include modifications of growth and geographic distribution of forests. Alig et al. (2002) examined global change scenarios from the National Climate Change Assessment, based on a combination of global circulation (Canadian and Hadley) and ecological process (Century, Terrestrial Ecosystem Model) models. The analyses used an equilibrium climate scenario based on transient Canadian and transient Hadley scenarios, with a baseline scenario using average climate for the 1961–1990 period. The climate change scenario was the average of the projected climate for 2070 to 2100. Results at that time indicated the likelihood of an overall increase in forest productivity in the United States, leading to an increase in long-term timber inventory (Irland et al. 2001).

With more forest inventory, timber harvests in most scenarios rise over the next 100 years, lowering timber prices, and reducing costs of wood and paper products.

Eric White

Private landowners control the most forest area in the United States and have the most opportunities for forest-based mitigation opportunities.

Future forest area could differ markedly under different price assumptions for CO_2

Total economic welfare is higher than in the base case for all climate change scenarios, owing to overall higher forest productivity. Adjustments related to market-based incentives include interregional migration of timber production, substitution in timber consumption, altered forest stand management (e.g., change in timber rotation length), salvage of dead or dying trees, shifts in planting stock, and changes in fertilization and thinning regimes. Aggregate welfare effects of climate change for the forest sector are relatively small, consistent with the McCarl et al. (2000) findings that they are relatively limited even under extreme scenarios.

Recent scenario analysis involving possible markets for carbon under hypothetical cap-and-trade policies has shown that future forest area could differ markedly under different price assumptions for CO_2 (Alig et al. 2010a). In comparison to a baseline projection of a reduction in timberland area under business as usual, Alig et al. (2010a) indicated that a price of more than \$25 per metric tonne of CO_2 could be needed to eliminate the projected loss in timberland area (fig. 3-2). Higher CO_2 prices could induce enough afforestation to offset timberland area losses to other uses such as developed uses and agriculture. This national analysis projected that about 10 million ha could be afforested at prices of \$25 CO_2 per metric tonne over the next 50 years and more than double that area with \$50 CO_2 prices. The majority would be in the South Central, Corn Belt, and Southeast regions.

The analysis by Alig et al. (2010a) with an optimization model using mathematical programming involved a perfect foresight assumption, and the importance of such assumptions will be reviewed in chapter 4. An econometric approach (e.g., Lubowski et al. 2006, Plantinga and Mauldin 2001, Plantinga et al. 1999) produces

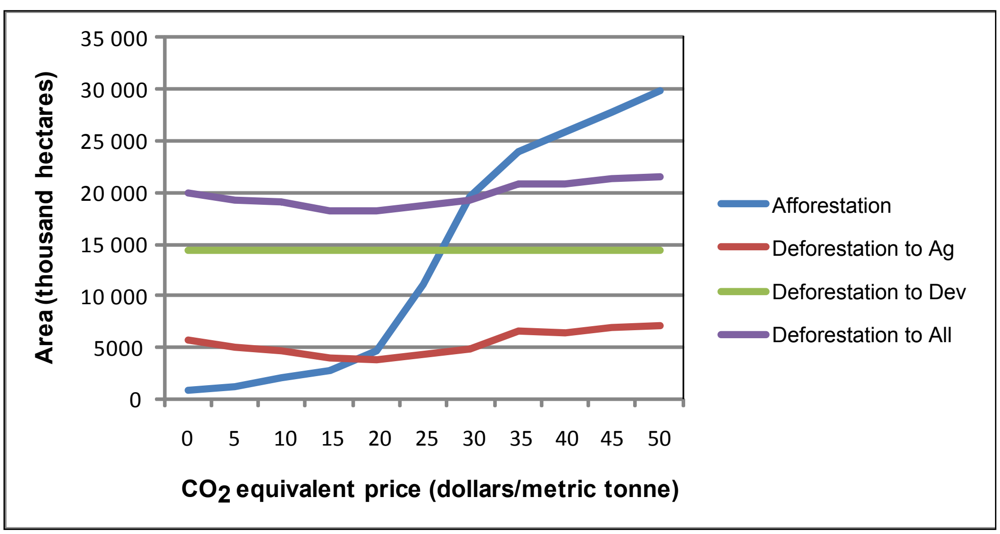


Figure 3-2—Projected changes in U.S. timberland area (contiguous 48 states) owing to baseline projection of deforestation to developed uses and hypothetical carbon dioxide (CO_2) prices per metric tonne (adapted from Alig et al. (2010a).

higher cost estimates than engineering and optimization methods (see chapter 4), reflecting a number of factors that affect land-use decisions in practice but that are difficult to measure explicitly and include in engineering and optimization models. These include landowner uncertainty in the face of irreversible investments, non-monetary returns to landowners from forest and agricultural uses of land, liquidity constraints, and other private or market costs or benefits.

Besides climate change mitigation, any policies to encourage the conversion of agricultural land to forest use may generate additional environmental benefits or co-benefits associated with afforestation. For example, Plantinga and Wu (2003) estimated the reductions in agricultural externalities, such as soil erosion and nitrogen and atrazine pollution, for a hypothetical afforestation program. They quantified the values of reduced soil erosion and benefits from enhanced wildlife habitat. Such values were the same order of magnitude as the costs of carbon sequestration policy, indicating that the co-benefits of forest carbon sinks are important factors to consider in designing a portfolio of climate mitigation strategies.

Besides climate change mitigation, any policies to encourage the conversion of agricultural land to forest use may generate additional environmental benefits or co-benefits associated with afforestation.

Summary

Potential actions for reducing net GHG emissions involve a wide variety of sinks and sources in the forestry and agricultural sectors. Afforestation is a key activity involving the two sectors. Afforestation opportunities are concentrated in the East on private lands. Availability of carbon-related payments could markedly influence the financial attractiveness of such opportunities to landowners. Co-benefits of forest sinks are an important factor to consider in designing a portfolio of climate mitigation strategies. Planting of short-rotation woody crops on marginal agricultural lands (e.g., Alig et al. 1999b) for use in generating biomass energy could also affect interactions between the forestry and agricultural sectors, including increased use of woody biomass in co-firing wood processing facilities (White 2010).

Co-benefits of forest sinks are an important factor to consider in designing a portfolio of climate mitigation strategies.

The studies reviewed contain a wide range of assumptions or approaches in the climate change effects, mitigation, and adaptation area pertaining to (a) simultaneous consideration of the agricultural (crops, livestock) and forestry sectors; (b) discount rate; (c) inclusion of carbon prices or values; (d) dynamic modeling of forest stands; (e) inclusion of non-CO_2 gases in GHG accounting; (f) GHG accounting overall; (g) modeling of forest and agricultural lands to developed uses (e.g., urban); (h) examination of bioelectricity as part of bioenergy analyses; and (i) consideration of indirect land-use changes, among other differences. This is not totally unexpected as the literature moves forward, especially in a topic area that is growing rapidly. An example of one research gap involving adaptation and mitigation activities is the potential interplay between the two over time and how that may affect costs of both adaptation and mitigation. Another is extended investigation of external benefits from increased afforestation, such as water quality improvements, reduced forest fragmentation, enhanced biodiversity, and other benefits. Public timberlands also contain many young stands that can sequester large amounts of forest carbon (Depro et al. 2008), and potential interactions with private timberland opportunities warrants more investigation.

Data needed to enhance land-use analyses in climate change studies include updated estimates of afforestation yields on former cropland and pastureland by region. Such data are needed that would allow examination of different forest management regimes (e.g., low, medium, and high levels of forest plantation management) following afforestation. These types of data would be useful in essentially all regions if carbon markets emerge to a degree where forest investments associated with forest carbon payments can compete with agricultural enterprises. For example, the Corn Belt may be a region with relatively good forest production potential but where forest use has traditionally been quite limited because of land-use competition from agriculture. Forward-looking studies in such regions would benefit from current estimates of afforestation costs and yields. In addition, data from landowner surveys (e.g., Butler and Leatherberry 2004, Kline et al. 2000) about willingness to undertake mitigation activities would usefully augment forest survey data.

The potential contributions of U.S. forestry and agriculture to climate change mitigation are framed within the need to help feed and house an additional 3 billion people globally, increased land demand for bioenergy production, and land needed to house another 125 million U.S. residents by 2050.

Additional work is needed to enhance design of efficient portfolios of land-related activities across sectors, such as afforestation, modified agricultural tillage, avoided deforestation, and other activities, including consideration of how they may vary over space and time. Such global questions involving forestry's and agriculture's potential contributions to climate change mitigation are framed within a national context of increased demands for cropland, forage, and wood products to help feed and house an additional 3 billion people globally by 2050, increased land demand for bioenergy production, and tens of millions of hectares of land needed to house another 125 million U.S. residents by midcentury.

English Equivalents

When you know:	Multiply by:	To get:
Hectares (ha)	2.47	Acres
Grams (g)	0.0352	Pounds
Tonnes or metric tons	1.102	Tons
Dollars per tonne ($/t)	0.907	Dollars per ton
Teragrams (Tg)	1,102,311	Tons

Literature Cited

Adams, D.M.; Alig, R.; McCarl, B.; Callaway, M.; Winnett, S. 1996. The Forest and Agricultural Sector Optimization Model (FASOM): model structure and policy applications. Res. Pap. PNW-RP-495. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 60 p.

Adams, D.M.; Alig, R.J.; McCarl, B.A.; Callaway, J.M.; Winnett, S.M. 1999. Minimum cost strategies for sequestering carbon in forests. Land Economics. 75(3): 360–374.

Adams, D.M.; Haynes, R.W. 1996. The 1993 Timber Assessment Market Model: structure, projections and policy simulations. Gen. Tech. Rep. PNW-GTR-358. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 34 p.

Adams, D.M.; Haynes, R.W., eds. 2007. Resource and market projections for forest policy development: twenty-five years of experience with the U.S. RPA Timber Assessment. Dordrecht, The Netherlands: Springer. 589 p.

Alig, R.J. 2003. U.S. landowner behavior, land use and land cover changes, and climate change mitigation. Silva Fennica. 37(4): 511–527.

Alig, R.J. 2007. A United States view on changes in land use and land values affecting sustainable forest management. Journal of Sustainable Forestry. 24(2/3): 209–227.

Alig, R.J.; Adams, D.M.; Chmelik, J.; Bettinger, P. 1999a. Private forest investment and long-run sustainable harvest volumes. New Forests. 17: 307–327.

Alig, R.J.; Adams, D.M.; Joyce, L.A.; Sohngen, B. 2004a. Climate change impacts and adaptation in forestry: responses by trees and markets. Choices: The Magazine of Food, Farm and Resource Issues. 2004(Fall): 7–11.

Alig, R.J.; Adams, D.M.; McCarl, B.A. 2002. Projecting impacts of global climate change on the U.S. forest and agriculture sectors and carbon budgets. Forest Ecology and Management. 169: 3–14.

Alig, R.J.; Adams, D.M.; McCarl, B.A; Callaway, J.M.; Winnett, S.M. 1997. Assessing effects of mitigation strategies for global climate change with an intertemporal model of the U.S. forest and agriculture sectors. Environmental and Resource Economics. 9: 259–274.

Alig, R.J.; Adams, D.M.; McCarl, B.A.; Ince, P. 1999b. Economic potential of short-rotation woody crops on agricultural land for pulp fiber production in the United States. Forest Products Journal. 50(5): 67–74.

Alig, R.J.; Adams, D.M.; Mills, J.R.; Haynes, R.W.; Ince, P.; Moulton, R. 2001. Alternative projections of the impacts of private investment on southern forests: a comparison of two large-scale forest sector models of the United States. Silva Fennica. 35(3): 265–276.

Alig, R.J.; Bair, L. 2006. Forest environmental investments and implications for climate change mitigation. Journal of Environmental Quality. 35(4): 1389–1395.

Alig, R.J.; Butler, B.J. 2004. Area changes for forest cover types in the United States, 1952 to 1997, with projections to 2050. Gen. Tech. Rep. PNW-GTR-613. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 106 p.

Alig, R.J.; Kline, J.; Lichtenstein, M. 2004b. Urbanization on the U.S. landscape: looking ahead in the 21st century. Landscape and Urban Planning. 69(2/3): 219–234.

Alig, R.J.; Latta, G.S.; Adams, D.M.; McCarl, B.A. 2010a. Mitigating greenhouse gases: the importance of land base interactions between forests, agriculture, and residential development in the face of changes in bioenergy and carbon prices. Forest Policy and Economics. 12(1): 67–75.

Alig, R.J.; Lee, K.J.; Moulton, R.J. 1990a. Likelihood of timber management on nonindustrial private forests: evidence from research studies. Gen. Tech. Rep. SE 60. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station. 17 p.

Alig, R.J.; Mills, T.J.; Shackelford, R.J. 1980. Most soil bank plantings in the South have been retained; some need follow-up treatment. Southern Journal of Applied Forestry. 4(1): 60–64.

Alig, R.J.; Murray, B.C.; Hohenstein, W.G.; Haight, R.G. 1990b. Changes in area of timberland in the United States, 1952–2040, by ownership, forest type, region, and state. Gen. Tech. Rep. SE 64. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station. 34 p.

Alig, R.J.; Plantinga, A.; Ahn, S.; Kline, J. 2003. Land use changes involving forestry for the United States: recent trends and projections. Gen. Tech. Rep. PNW-GTR-587. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 92 p.

Alig, R.J.; Plantinga, A.; Haim, D.; Todd, M. 2010b. Area changes in U.S. forests and other major land uses, 1982–2002, with projections to 2062. Gen. Tech. Rep. PNW-GTR-815. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 140 p.

Best, C.; Wayburn, L. 2001. America's private forests: status and stewardship. The Pacific Forest Trust. Washington, DC: Island Press. 268 p.

Birdsey, R.A.; Alig, R.J.; Adams, D.M. 2000. Mitigation options in the forest sector to reduce emissions or enhance sinks of greenhouse gases. In: Joyce, L.A.; Birdsey, R.A., eds. The impact of climate change on America's forests. Gen. Tech. Rep. RMRS-GTR-59. Portland, OR: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 112–127. Chap. 8.

Butler, B.; Leatherberry, E. 2004. America's family forest owners. Journal of Forestry. 102(7): 4–9.

Clutter, M.; Mendell, B.; Newman, D.; Wear, D.; Greis, J. 2005. Strategic factors driving timberland ownership changes in the U.S. South. Draft version presented to the Southern Group of State Foresters. http://www.srs.fs.usda.gov/econ/pubs/southernmarkets/strategic-factors-and-ownership-v1.pdf. (September 2008).

Depro, B.M.; Murray, B.C.; Alig, R.J.; Shanks, A. 2008. Public land, timber harvests, and climate mitigation: quantifying carbon sequestration potential on U.S. public timberlands. Forest Ecology and Management. 255(3–4): 1122–1134.

Fairweather, J. 1996. Understanding why farmers plant trees. New Zealand Forestry. 41(2): 34–38.

Hair, D.; Sampson, R.N.; Hamilton, T.E. 1996. Summary: forest management opportunities for increasing carbon storage. Forests and Global Change. 2: 237–254.

Ince, P. 1994. Recycling and long-range timber outlook. Res. Pap. FPL-RP-534. Madison, WI: U.S. Department of Agriculture, Forest Service, Forest Products Laboratory. 110 p.

Irland, L.; Adams, D.M.; Alig, R.J.; Betz, C.J.; Chen, C.; Hutchins, M.; McCarl, B.A.; Skog, K.; Sohngen, B.L. 2001. Assessing socioeconomic impacts of climate change on U.S. forest, wood-product markets, and forest recreation. BioScience. 51(9): 753–764.

Kline, J.D.; Alig, R.J.; Johnson, R.L. 2000. Owner incentives to protect riparian habitat. Ecological Economics. 33(1): 29–43.

Kurtz, W.B.; Alig, R.J.; Mills, T.J. 1980. Retention and condition of Agricultural Conservation Program conifer plantings. Journal of Forestry. 78(5): 273–276.

Kurtz, W.B.; Noweg, G.; Moulton, R.J.; Alig, R.J. 1994. An analysis of the retention, condition, and land-use implications of tree plantings established under the Soil Bank Program, the Forestry Incentives Program, and the Agricultural Conservation Program. Rep. 464. Columbia, MO: University of Missouri Agricultural Experiment Station. 67 p.

Lubowski, R.; Plantinga, A.; Stavins, R. 2006. Land-use change and carbon sinks: econometric estimation of the carbon sequestration supply function. Journal of Environmental Economics and Management. 51: 135–152.

Malmsheimer, R.W.; Heffernan, P.; Brink, S.; Crandall, D.; Deneke, F.; Galik, C.; Gee, E.; Helms, J.A.; McClure, N.; Mortimer, M.; Ruddell, S.; Smith, M.; Stewart, J. 2008. Forest management solutions for mitigating climate change in the United States. Journal of Forestry. 106(3): 115–173.

Maness, T.C. 2009. Forest management and climate change mitigation: good policy requires careful thought. Journal of Forestry. 107(3): 119–124.

McCarl, B.A.; Adams, D.M.; Alig, R.J.; Chen, J. 2000. Effects of global climate change on the U.S. forest sector: response functions derived from a dynamic resource and market simulator. Climate Research. 15(3): 195–205.

McCarl, B.A.; Chang, C.C.; Atwood, J.D.; Nayda, W.I. 1998. Documentation of ASM: the U.S. Agricultural Sector Model. Tech. Pap. 23. College Station, TX: Texas A&M University, Department of Agricultural Economics. 77 p.

Mills, J.R.; Kincaid, J.C. 1992. The aggregate timberland assessment system—ATLAS: a comprehensive timber projection model. Gen. Tech. Rep. PNW-GTR-281. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 160 p.

Moulton, R. 1999. Tree planting in the United States, 1998. Forest Service tree planting statistics (Unnumbered rep.). Washington, DC: U.S. Department of Agriculture. 33 p.

Moulton, R.; Richards, K. 1990. Costs of sequestering carbon through tree planting and forest management in the United States. Gen. Tech. Rep. WO-58. Washington, DC: U.S. Department of Agriculture. 56 p.

Murray, B.C.; Sohngen, B.L.; Sommer, A.J.; Depro, B.M.; Jones, K.M.; McCarl, B.A.; Gillig, D.; DeAngelo, B.; Andrasko, K. 2005. Greenhouse gas mitigation potential in U.S. forestry and agriculture. EPA-R-05-006. Washington, DC: U.S. Environmental Protection Agency, Office of Atmospheric Programs. 144 p.

Plantinga, A.J.; Mauldin, T. 2001. A method for estimating the cost of CO_2 mitigation through afforestation. Climatic Change. 49: 21–40.

Plantinga, A.J.; Mauldin, T.; Miller, D. 1999. An econometric analysis of the costs of sequestering carbon in forests. American Journal of Agricultural Economics. 81: 812–824.

Plantinga, A.J.; Wu, J. 2003. Co-benefits from carbon sequestration in forests: evaluating reductions in agricultural externalities from an afforestation policy in Wisconsin. Land Economics. 79(1): 74–85.

Smith, W.B.; Miles, P.D.; Perry, C.H.; Pugh, S.A. 2009. Forest resources of the United States, 2007. Gen. Tech. Rep. WO-78. Washington, DC: U.S. Department of Agriculture, Forest Service, Washington Office. 336 p.

U.S. Department of Agriculture, Forest Service [USDA FS]. 2001. 2000 RPA assessment of forest and range lands. FS-687. Washington, DC. http://fia.fs.fed.us/pl/rpa/. (November 2008).

U.S. Department of Agriculture, Natural Resources Conservation Service [USDA NRCS]. 2007. National Resources Inventory—2003 Annual NRI. Washington, DC. http://www.nrcs.usda.gov/technical/NRI/2003/nri03landuse-mrb.html. (January 2010).

U.S. Environmental Protection Agency [US EPA]. 2009. Inventory of U.S. greenhouse gas emissions and sinks: 1990–2007. EPA Report. Washington, DC. (April 2009).

Vasievich, M.; Alig, R. 1996. Opportunities to increase timber growth and carbon storage on timberlands in the contiguous United States. In: Sampson, R.N.; Hair, D., eds. Forests and global change, forest management opportunities. Washington, DC: American Forests: 91–104. Vol. 2.

Wear, D.; Greis, J. 2002. Southern forest resource assessment: summary of findings. Journal of Forestry. 100(7): 6–14.

White, E.M. 2010. Woody biomass for bioenergy in the United States—a briefing paper. Gen. Tech. Rep. PNW-GTR-825. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 45 p.

Chapter 4: What Explains Differences in the Costs of Carbon Sequestration in Forests? A Review of Alternative Cost Estimation Methodologies

by Judith Dempsey, Andrew J. Plantinga, and Ralph J. Alig

Introduction

For the past two decades, starting with Sedjo and Solomon (1989), there has been ongoing research focused on estimating the cost of forest-based carbon sequestration. On balance, the results of this research indicate that the costs of carbon sequestration in forests are similar or lower in comparison to energy-based mitigation approaches.[1] However, there are still questions raised by differences in carbon sequestration cost estimates across studies. Some of the difference may be due to regional variations in land rental rates and rates of carbon uptake. But part of the difference appears to arise from different methodologies used to compute costs. Three main approaches have been taken in the literature: “bottom-up” engineering models, sectoral optimization studies, and econometric models (Richards and Stokes 2004). A review of the literature reveals that these different cost estimation methods do not provide consistent cost estimates of forest-based carbon sequestration. It is important to understand the effects of the calculation method on the final results before accepting the cost estimates provided by any study. Thus, this literature review will focus on examining differences in cost estimates obtained by different cost estimation methodologies. Our review will focus on studies of afforestation—the conversion of nonforest land to forest—and will not consider forest management strategies to sequester carbon; earlier studies found that afforestation offers larger opportunities to sequester carbon in forests (e.g., Moulton and Richards 1990). We also restrict our attention to the United States.

Each of the three cost estimation approaches are concerned with measuring opportunity costs—what the landowner gives up when he or she converts land from a nonforest use such as agriculture to forest.`

Each of the three cost estimation approaches are concerned with measuring opportunity costs—what the landowner gives up when he or she converts land from a nonforest use such as agriculture to forest. Bottom-up and sectoral optimization studies measure these costs as the lost value from the land plus costs of establishing and maintaining trees. A key assumption in these studies is that all relevant costs can be estimated explicitly by, for example, the average cropland rent or sale price

[1] For example, Lubowski et al. (2006) found carbon sequestration costs in the same range as those found for energy-based studies below 500 million tons per year and lower costs above that point.

in an area plus estimates of tree planting costs based on historical patterns. Econometric studies, on the other hand, are based on how landowners actually respond to incentives they face in the marketplace. Historical data are used to model and predict future landowner behavior. This approach has the potential to measure many unobservable factors implicitly in the parameters of the model and thus lead to more realistic and nuanced predictions of how landowners will behave. These unobservable factors may include nonmonetary benefits that the landowner obtains from the land in a particular use. Because of its potential to measure unobservable factors, the econometric method tends to produce cost estimates that are higher than those from engineering and optimization studies.

Another important difference between the studies is the modeling of price effects. As a carbon sequestration incentive attracts more agricultural land into forest, one should expect prices for agricultural commodities to rise and prices for forest commodities to fall (assuming the new forests can be harvested). This endogenous price effect should raise the costs of converting more land into forest. Only some of the engineering cost and econometric studies account for price effects. In contrast, the chief advantage of the sectoral optimization studies is their ability to model forest and agricultural commodity markets and the linkages between them. There is evidence that optimization studies produce higher cost estimates than engineering studies (Adams et al. 1993), though these estimates tend to remain lower than estimates from econometric studies, as we will discuss below.

There are many assumptions that must be made in the calculation of carbon sequestration costs. Because of the range of assumptions, the results of different studies, even studies using similar calculation methods, cannot be directly compared. To isolate the effects of the calculation method, this review will consider several papers that have normalized groups of carbon sequestration studies. These papers each have slightly different criteria for normalization, but all provide comparable estimates of carbon sequestration costs for the group of studies evaluated. The main finding in these studies is that the econometric method produces higher cost estimates than the engineering and optimization approaches.

Overview of Methodologies for Estimating Carbon Sequestration Cost

Modeling Choices

Gorte (2009) discussed many factors that can influence carbon sequestration cost estimates. Ultimately, analysts are faced with a number of choices, and must be aware of how these choices affect their results. As emphasized by Gorte (2009), there is significant variability in the volume of carbon that is stored in a forest.

These differences can appear by region of the country, the variety of tree species, and forestry practices. Many of the available carbon sequestration studies focus on specific regions, which makes direct comparisons difficult. The history of land use at a site can affect the estimated volume of carbon. For example, agricultural land and previously forested lands will typically have different initial levels of soil carbon and this will affect the volume of carbon sequestered subsequently. Newell and Stavins (2000) examined differences in carbon sequestration costs between tree plantations and naturally regenerated stands. The costs per ton of carbon are found to be similar because the higher rates of carbon uptake by plantations are offset by their higher upfront establishment costs.

Gorte (2009) also discussed the effects that harvesting practices can have on total sequestered carbon. Trees may be continually harvested, or a stand may be allowed to grow indefinitely. Some portions of harvested trees may be assumed to remain onsite, or all may be removed for processing into wood products or for energy generation. Depending on the assumptions made in the study, carbon sequestration rates and related costs can differ significantly. Newell and Stavins (2000) discussed the possibility that periodic harvesting could actually increase carbon sequestration in the long run if harvested wood is put into long-lived wood products and new forests are grown in their place. However, these authors, in addition to Lubowski et al. (2006), found that periodic harvesting decreases total carbon sequestration, thereby increasing the price per ton of carbon sequestered. This result is likely to be sensitive to assumptions regarding the length of time carbon is stored in wood products and other factors.

Ralph Alig

The forest sector can contribute to atmospheric greenhouse gas reduction, while also providing other environmental, economic, and social benefits.

The discount rate adds an additional possible variant between studies. Discounting is used in carbon sequestration cost studies so that monetary and carbon flows occurring at different points in time are comparable. Boscolo et al. (1998) pointed out that the discount rate can be uncertain and even controversial. As the chosen discount rate rises, estimated future costs will have a smaller and smaller effect on the present value of costs. Newell and Pizer (2001) stated that the results of a study will be sensitive to the choice of discount rate. Newell and Stavins (2000) examined the effect of varying discount rates on sequestration costs and the amount of forest conversion induced by a specific tax or subsidy. They found that marginal sequestration costs (the cost of sequestering an additional ton of carbon) rise with the discount rate. This is because higher discount rates diminish the importance of future carbon flows relative to the upfront costs of establishing forests. On the other hand, more land is converted to forest because the upfront payments for carbon sequestration are large relative to the future losses in agricultural revenues, which decrease with a higher discount rate.

Three methods have been used to develop a summary measure of a ton of sequestered carbon: the flow summation approach, the average storage approach, and the levelization/ discounting approach.

Related to the choice of discount rate is the method used to summarize carbon flows that occur through time. Stavins and Richards (2005) pointed out that carbon sequestration does not occur at one discrete moment, but instead throughout the life cycle of a forest. Within the literature, three methods have been used to develop a summary measure of a ton of sequestered carbon: the flow summation approach, the average storage approach, and the levelization/discounting approach (we refer readers to Richards and Stokes (2004) for technical details of these approaches). Richards and Stokes (2004) noted that the method used in a particular study can affect the results by up to an order of magnitude. However, we found that the levelization/discounting method is increasingly accepted as the appropriate way to treat carbon flows through time. It involves discounting flows in the same manner (and at the same rate) as costs.

Finally, Richards (2004) discussed the importance of choosing an appropriate baseline scenario of original land use. Each study needs to be able to compare the baseline case of what would have happened without the sequestration program to what would be accomplished with the sequestration policy. Newell and Stavins (2000) reported large differences in carbon sequestration when they considered different baselines defined in terms of agricultural price growth.

Econometric, Engineering, and Optimization Studies

As described by Stavins and Richards (2005), econometric carbon sequestration cost studies analyze data from actual land-use changes with the goal of identifying the relationship between land-use choices and relative returns in the forestry, agricultural, and urban land sectors. To illustrate the econometric method, it is helpful to provide a quick overview of the approach taken by Lubowski et al. (2006). This study used National Resources Inventory (NRI) data that provide observations of plot-level land-use changes. Additionally, using a variety of sources, the authors constructed estimates of annual net returns for each county and major land types represented in the NRI data. An econometric model was developed to estimate the relationship between land-use changes over a specific period and annual net returns to each use. The methodology allows unobservable factors affecting actual land-use decisions to be measured implicitly.

As stated earlier, the econometric calculation method removes a level of uncertainty that is found in many engineering models, leading in principle to more realistic predictions of how landowners will behave. Because they are based on actual historical data, these models can implicitly capture such factors as landowner uncertainty in the face of irreversible investments, nonmonetary returns to landowners from forest and agricultural uses of land, liquidity constraints, and other private or market costs or benefits (Lubowski et al. 2006). Some examples of econometric models are those by Newell and Stavins (2000), Plantinga et al. (1999), and Stavins (1999).

The survey-based study by van Kooten et al. (2002) demonstrates why the unobservable incentives that are captured in the econometric method are so important. Van Kooten et al. evaluated survey results from 182 Canadian farmers

Eric White

Private timberland management costs and land conversion costs are important factors in decisions about forest production and use of forests to contribute to climate change mitigation.

and found that they were not solely motivated by potential profit from their lands. For example, the farmers indicated that they placed significant value on retaining control over their lands. Although these farmers placed some value on carbon sequestration, they preferred methods that allowed their land to still be used for agriculture.

A downside of the engineering method is the inability to account for unobservable factors affecting landowner decisions.

As discussed by Richards (2004), bottom-up engineering models (examples include Dudek and Le Blanc 1990, Moulton and Richards 1990, New York State 1991, Richards 1997, Richards et al. 1993, Sedjo and Solomon 1989, van Kooten et al. 1992) generally use regional average land prices or land rents to estimate the opportunity cost of converting land from one use or practice to another. These values measure the foregone profits from agricultural production. The calculated opportunity cost is combined with the conversion costs for moving land from agricultural to forest use, and this becomes the total opportunity cost of the land. In the simplest approach, total annualized costs for each region are divided by an annualized measure of carbon sequestration for each region to obtain the cost per unit of carbon sequestered (or the average cost of carbon sequestration). An alternative approach, more useful because it facilitates comparisons across studies, is to construct a schedule of marginal costs. This gives the cost of an additional ton of carbon at each level of annual carbon sequestration (see, for example, fig. 4-1). The engineering method has advantages of using observable information and transparency of the calculations. However, a downside of the engineering method, and a likely source of the differences in cost estimates provided by the engineering method compared to the econometric approach, is the inability to account for unobservable factors affecting landowner decisions.

The third approach makes use of sectoral optimization models (examples include Adams et al. 1993, 1996, 1999; Alig et al. 1997, 2002; Sohngen and Mendelsohn 2003). These studies typically combine market models of agriculture and forestry and account for the interaction between the two sectors. Similar to engineering models, optimization models are not able to account for unobservable cost and benefits to landowners; however, they do account for the increasing scarcity of agricultural land as more of it is converted to forestry.[2] This scarcity increases agricultural commodity prices and corresponding returns to agriculture, which feed back into the carbon sequestration policy in two ways. First, as agricultural land becomes more valuable, it increases the cost of converting additional agricultural land into forest. Second, the increase in agricultural returns causes some forest land to be converted back to agriculture, reducing carbon sequestration and driving up

[2] Although engineering and optimization models do not explicitly account for unobservables, parameters can be varied to reflect uncertainty about their true values.

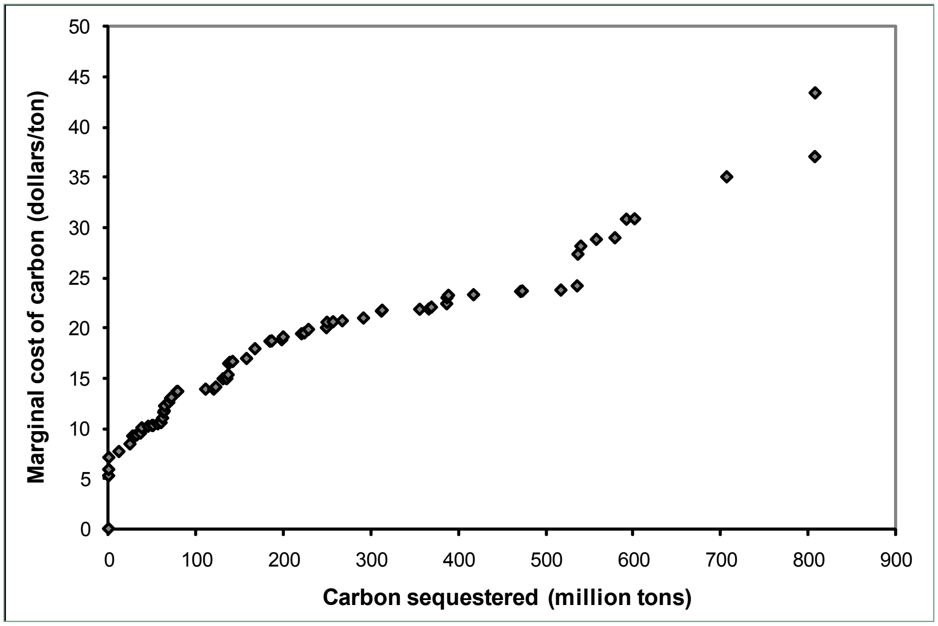


Figure 4-1—Example of marginal cost curve of forest carbon sequestration from a bottom-up engineering approach, adapted from figure 3 in Moulton and Richards (1990).

costs of sequestration. Therefore, cost estimates tend to be higher than bottom-up engineering studies that ignore such price effects. Adams et al. (1993) found that including these price effects raised marginal cost estimates at the highest carbon sequestration level by an amount between 57 and 470 percent.

Review of Representative Studies

Engineering Study

Moulton and Richards (1990) is a representative bottom-up engineering study. For this study, data were collected for each farm production region in the United States (e.g., Corn Belt, Lake States, Northeast). For each region, areas of land in crop, pasture, and forest were identified and classified in terms of characteristics (dry, wet, soil quality, etc.) and potential for sequestering additional carbon. For example, the areas of dry and wet crop and pasture land with potential for afforestation were estimated for each region. Each land type was then matched with different treatments (e.g., tree planting for agricultural lands, active management for forest lands) yielding seven land type/treatments for each farm production region.

Opportunity cost is calculated by determining the likely rental cost per acre (i.e., the amount a landowner would charge somebody else to use their land). Rental rates are typically highest for cropland, second highest for pasture, and lowest for forest land. These rates also differ by region of the country. As an example, the rental rates (in 1990 dollars) for the Cornbelt are \$200/ha (\$81/acre) for cropland,

$61.06/ha ($24.72/acre) for pasture, and $21.37/ha ($8.65/acre) for forest land. Contrast this with the Appalachian rental rates of $150.67/ha ($61/acre) for cropland, $68.69/ha ($27.81/acre) for pasture, and $24.03/ha ($9.73/acre) for forest land. Note that these rates were fixed in the study and so there was no mechanism for the opportunity cost to change as land is converted into forest and agricultural land becomes more scarce. The authors also included the annualized cost of a specified forestry treatment (e.g., tree planting). Finally, the average annual rate of carbon sequestration over a 40-year time horizon was estimated. The total annual costs divided by average annual sequestration gives the average cost per ton of carbon for each region and land type/treatment.

Moulton and Richards constructed a marginal cost schedule by arraying these average cost per ton estimates against the cumulative amount of carbon sequestered (fig. 4-1). Each point on the graph corresponds to a region/land type/treatment combination, and marginal costs are seen to range from about $5.50/tonne ($5/ton) to about $50/tonne ($45/ton) (in 1990 dollars) at 726 million tonnes (800 million tons) sequestered. The Moulton and Richards study was one of the first careful analyses of carbon sequestration costs. The authors pointed out several limitations of the study that have been addressed (to varying degrees) in later studies. These include the lack of attention to indirect costs or benefits to society (such as environmental effects); no consideration of the effects of timber harvesting on the carbon budget; and lack of precision in estimates of opportunity costs.

A number of factors can potentially affect land-use decisions that are difficult to account for explicitly: the irreversibility of conversion to forests, the cost of knowledge, and nonmarket benefits.

Econometrics Study

The Plantinga et al. (1999) study is an early example of an econometric study that was motivated by potential shortcomings of previous engineering cost studies. The authors discussed a number of factors that can potentially affect land-use decisions, but which are difficult to account for explicitly: the irreversibility of conversion to forests, which may discourage landowners from making an otherwise profitable decision; the cost of knowledge, which might keep farmers from switching their land from agriculture to forest; and nonmarket benefits that might affect land-use decisions. The authors reviewed the findings of the previous econometric analyses by Plantinga (1997) and Stavins (1999) and reported that higher costs are found in econometric anlayses than in previous engineering studies. These higher costs are attributed to the unobservable factors discussed above.

Plantinga et al. (1999) extended the study by Plantinga (1997). The goal of the study was to compare carbon sequestration costs among regions of the United States, to model the effects of population increases, and to examine alternative program designs. An econometric model of aggregate land-use shares was estimated using county-level data from Maine, South Carolina, and Wisconsin. As stated before, these data will potentially reflect costs and benefits of different land uses that could not be explicitly measured with the engineering calculation method. Once the relationship between land-use shares and net returns to alternative uses was estimated, a simulation of afforestation subsidies was performed to estimate the response to carbon sequestration incentives. Following a procedure similar to that described above, a marginal cost curve for carbon sequestration was derived.

This econometric analysis provided cost estimates that are higher than those found in most earlier engineering or optimization analyses including Moulton and Richards (1990), Adams et al. (1993), Parks and Hardie (1995), and Alig et al. (1997). Direct comparisons are possible with the results in Moulton and Richards (1990). It was found that the Plantinga et al. cost estimates were approximately four times the cost for carbon sequestration in Maine and Wisconsin, and 1.1 times that in South Carolina. The results indicate the importance of deriving cost estimates from observed landowner behavior.

Sectoral Optimization Study

Alig et al. (1997) stated that in past analyses of carbon sequestration in forests, land market interactions, timber harvest, and forest investment have received far less attention than the biophysical relationships between forest biomass and carbon sequestration. To address this issue, the authors applied a linked model of the U.S. forest and agricultural sectors to investigate the effect of different carbon sequestration programs. Specifically, they investigated whether the effects of forest carbon sequestration programs differ significantly from those found in previous studies, with particular attention paid to the costs and mixture of land-based adjustments that occur when a specific carbon sequestration target is met. The authors used the Forest and Agricultural Sector Optimization Model (FASOM). In this model, the objective function is defined in terms of the welfare of producers and consumers in the U.S. agriculture and forest sectors over a finite time horizon. Under certain conditions, maximization of this objective function yields the solution that would be achieved in an competitive market equilibrium.

Alig et al. focused on five scenarios: the base case, a simulated fixed-afforestation program assuming a specific land transfer, and three specific carbon-flux target scenarios assuming specific carbon sequestration targets. The base case was calculated, and all other projections were measured relative to this scenario. Using this method, it was found that the fixed-increment specific carbon sequestration target (carbon target 1) was more efficient than any other scenario. It was significantly more efficient than the scenario with the specific land-use target. According to the authors, this suggests that land-use changes do not have to be permanent, and that minimum social cost strategies may involve more complex land transitions than previously stated. Carbon sequestration costs are not listed in this study, which instead looked at overall social welfare. However, Richards and Stokes (2004) derived the average cost of carbon (dollars/metric ton) using data from this paper and employing the levelization discounting method. They found that the fixed-afforestation scenario had an average cost of $81/t, whereas the fixed increment scenario had an average cost of $24/t.

Comparison of Normalized Cost Estimates

As discussed in the "Overview of Methodologies for Estimating Carbon Sequestration Cost" section, analysts must make many choices when estimating carbon sequestration costs. Because of this, it is often impossible to directly compare cost estimates from different studies. The difference in the estimates may be due to the cost estimation method, but also to the many other assumptions that the analyst must make. Fortunately, there have been a number of reviews of the literature that develop comparable cost estimates from groups of studies. Each of these reviews normalized the data using different criteria, but because the normalization was consistently applied to each set of studies one can draw conclusions about how cost estimates differ among engineering, econometric, and optimization approaches.

Van Kooten et al. (2004) conducted a meta-analysis of a large set of carbon sequestration cost studies. Meta-analysis is a technique that attempts to determine how features of a study affect the results. Commonly applied in the medical field, it uses regression analysis to quantify how the design of studies affects the results obtained. Van Kooten et al. examined 981 estimates from 55 studies of the costs of creating carbon offsets using forestry, and then estimated an econometric model that analyzes how the different assumptions of each study affect the associated carbon sequestration cost estimates. This analysis controlled for many of the possible inconsistencies across studies, mentioned earlier, and found that many of them cause significant differences in cost estimates.

Van Kooten and Sohngen (2007) provided an update to van Kooten et al. (2004) that is more useful for our purposes. The 2007 analysis includes information on costs of carbon uptake and storage in forest ecosystems from 68 studies. More than three-quarters of the studies used a bottom-up approach, 18 percent used an optimization approach, and only 6 percent used an econometric approach. There were also many other differences in the assumptions of the different studies. The authors found, not surprisingly, that studies accounting for the opportunity cost of land produce higher cost estimates. A dummy variable for the opportunity cost of land is statistically significant and adds about $27.55/t ($25/ton) of carbon dioxide (CO_2) to costs. A dummy variable for studies using an econometric approach was also

Feasibility of joint production in forest ecosystems is influenced by costs of forest management activities, including opportunity costs of extending timber rotations to sequester additional carbon in woody biomass.

included in one model specification. The coefficient had a positive and significant coefficient, which indicates that cost per ton estimates in econometric studies are about 16 percent higher than those from engineering studies. In another regression, the authors found that optimization studies produce estimates that are higher by \$46/tonne (\$42/ton) CO_2, which represents about a 50-percent increase over the mean value of cost estimates in the studies considered.

Bottom-up engineering studies do not generally account for market adjustments resulting from forest conversion, but they have the benefits of transparency and being easy to interpret.

Richards and Stokes (2004) compared results from 36 studies, the majority of which used engineering calculation methods. Their review focused on studies with significance for global, national, and regional policies and did not include analyses of individual projects. Among the conclusions of this paper was that "although full carbon sequestration studies all contain essentially the same components, they are not comparable on their face due to the inconsistent use of terms, geographic scope, assumptions, and methods." The paper contains a detailed discussion of the different calculation methods described above. The authors noted that bottom-up engineering studies do not generally account for market adjustments resulting from forest conversion, but they have the benefits of transparency and being easy to interpret. Several engineering studies that do account for the feedback effects on the marginal cost of land are those by Richards et al. (1993) and Richards (1997). Richards and Stokes (2004) mentioned several optimization models (Adams et al. 1999, Alig et al. 1997) that allow for the possibility that forest may be converted back to agriculture as agricultural land becomes scarcer. This is the so-called carbon "leakage" effect, which could potentially offset the accomplishments of a carbon sequestration policy. The discussion of econometric studies emphasizes that these studies analyze historical data to determine how landowners have behaved in the past and use this to predict how they would behave in the future in response to incentives created by a carbon sequestration policy. Therefore, econometric studies use a revealed-preference approach based on actual practices rather than assuming a specific model of landowner behavior.

Richards and Stokes (2004) compared the results of studies using several criteria, but the most important one for this literature review is a comparison of studies based in the United States, differentiated according to cost estimation method. Three studies were chosen for direct comparison based on the similarity of their basic assumptions: the econometric approaches applied by Stavins (1999) and Plantinga et al. (1999) and the bottom-up approach used by Richards (1997). All three studies developed marginal cost curves, which allows the comparison of costs across varying levels of carbon sequestration. At low levels of sequestration, the bottom-up method estimated higher sequestration costs (figs. 4-2 and 4-3). However, the marginal cost curves for the econometric studies are much steeper, and as total tons of carbon sequestered per year increases, the costs increase much more rapidly. At approximately 7.0 million tonnes per year, the cost estimates by Stavins are higher than those by Richards (we refer here to the modified Stavins curve, which was scaled up to include the same total area as in the study by Richards) (fig. 4-2).

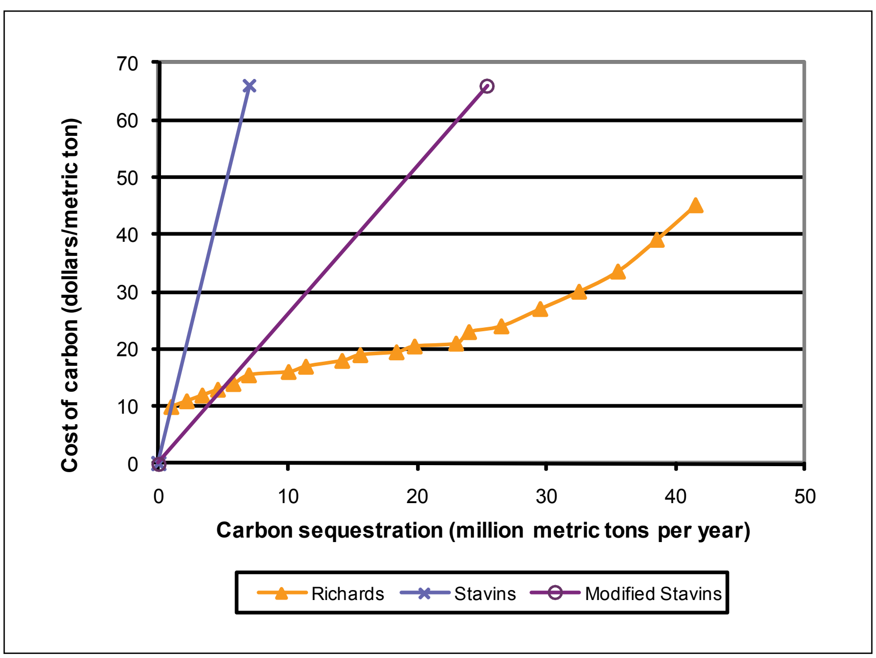


Figure 4-2—Comparison of marginal cost curves from two econometric (Richards and Stavins) and one bottom-up engineering study (modified Stavins), adapted from figure 3 in Richards and Stokes (2004).

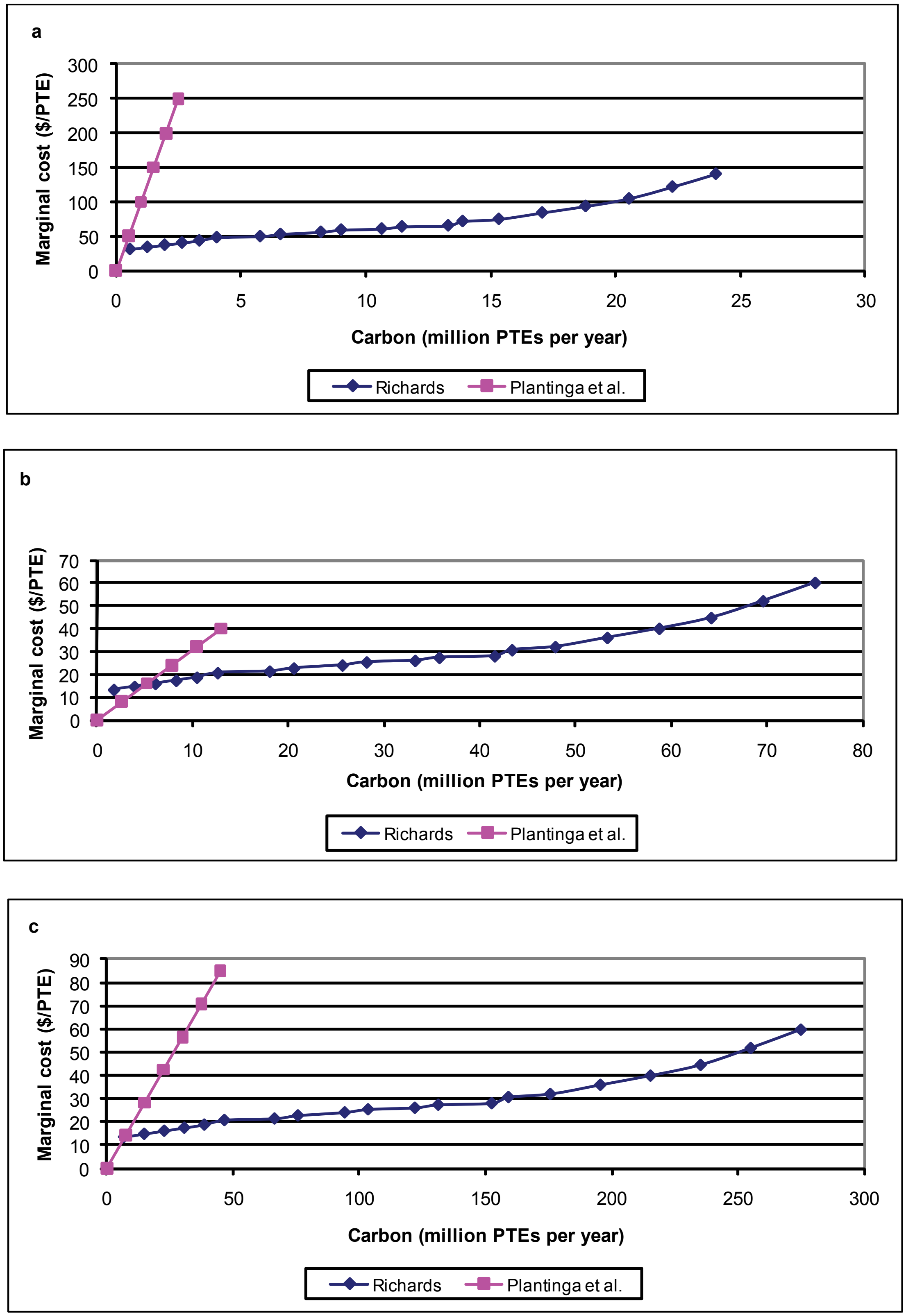


Figure 4-3—Comparison of marginal cost curves of forest carbon sequestration for (a) Maine, (b) South Carolina, and (c) Wisconsin, comparing Plantinga et. al (1999) and Richards (1997); adapted from figure 4 in Richards and Stokes (2004). PTE = present tons equivalent.

At 20 million t of carbon per year, marginal costs according to the modified Stavins curve are double those of Richards. For the comparison with Plantinga et al. (1999), data for Maine, South Carolina, and Wisconsin were drawn from Richards (fig. 4-3). The same pattern emerges, with the econometric marginal cost curve being much steeper than the marginal cost curve calculated by the bottom-up engineering approach. At low levels of carbon sequestration, marginal costs in Plantinga et al. are double those in Richards.

Lubowski et al. (2006) performed an econometric evaluation of carbon sequestration costs. The marginal cost estimates from this study are higher at most carbon sequestration levels than previous estimates. This study differentiates itself from previous studies by three principal features. First, six major land uses are modeled. Second, micro data are used that are comprehensive of the contiguous United States. Third, key commodity prices are treated as endogenous in the simulations of the carbon sequestration supply function. Because this is one of the few econometric studies that allows for price feedbacks from the policy, it allows us to identify the cost differences between the econometric and optimization approaches.

Lubowski et al. (2006) compared their results to those from other studies (fig. 4-4). They found that over the range of carbon prices considered in previous studies, their cost estimates were higher than those obtained using optimization models (Adams et al. 1993, Callaway and McCarl 1996) and bottom-up engineering cost methods (Richards et al. 1993). Compared to an earlier econometric analysis by Stavins (1999), they found similar costs at low carbon sequestration levels,

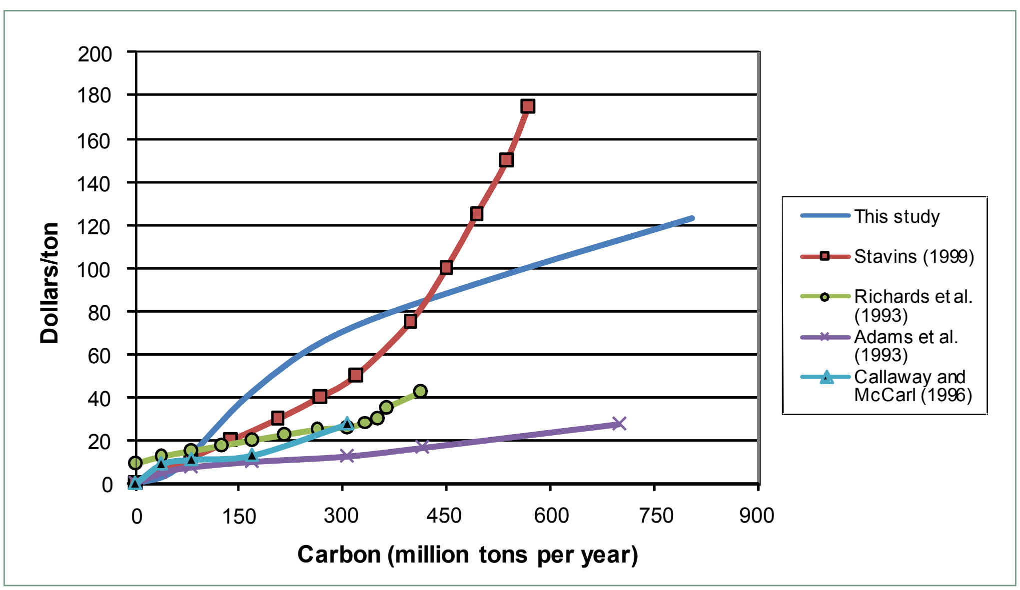


Figure 4-4—Comparison of marginal cost curves for forest carbon sequestration by Lubowski et al. (2006) with optimization models (Adams et al. 1993, Callaway and McCarl 1996) and bottom-up engineering cost methods (Richards et al. 1993); adapted from figure 5 in Lubowski et al. (2006).

but lower costs at higher carbon sequestration levels. The difference between this study and the study by Stavins was attributed to the different land use categories and scope (Stavins's national results were extrapolated from a model estimated for a group of counties). Consistent with the other papers presented in this review, Lubowski et al. (2006) found that econometric estimates are higher in the range of forested acres that would be needed to sequester significant amounts of carbon. Although the econometrically estimated marginal costs begin at zero dollars per ton, they surpass those from the engineering and optimization studies after 100 million tons of annual carbon sequestered.

Econometric estimates are higher in the range of forested acres that would be needed to sequester significant amounts of carbon.

Stavins and Richards (2005) identified 11 previous analyses that they considered to be good candidates for comparison and synthesis (table 4-1). Results from these studies were normalized by adjusting for constant-year dollars, discount rates, geographic scope, and reporting in equivalent annual costs. Compared to the other studies discussed above, the normalization in Stavins and Richards is more thorough and, thus, this study provides the best basis for comparing cost estimates across studies. After the normalization, there were still significant differences in carbon sequestration costs, which the authors attributed to different underlying biological and economic assumptions as well as different analytical methods employed. Among the national studies, the reported range, after normalization to 1997 dollars, was \$28 to \$83/tonne (\$25 to \$75/ton) for a program size of 272 million tonnes (300 million tons) of annual carbon sequestration, and \$33 to \$99/

Table 4-1—Carbon (C) sequestration cost studies included in normalization

Authors	Scope	Potential quantity reported	Range of costs reported	Potential normalized quantity	Range of normalized costs
		Short tons C/year	*\$/short tons C*	*Short tons C/year*	*\$/short tons C*
Moulton and Richards (1990)	National	809×10^6	5–43	809×10^6	5–57
Dudek and Leblanc (1990)	National	NA[a]	23.9–38.4	37.9×10^6	60
New York State (1991)	New York	1.2×10^6	14–54	75×10^6	8–53
Adams et al. (1993)	National	700×10^6	18–55	700×10^6	24–71
Richards et al. (1993)	National	NA[b]	8–60	448×10^6	11–81
Parks and Hardie (1995)	National	120×10^6	4–82	120×10^6	2–37
Alig et al. (1997)	National	44×10^6	22	44×10^6	27
Richards (1997)	National	495×10^6	9–125	495×10^6	10–143
Stavins (1999)	Delta States	13.9×10^6	0–664	722×10^6	0–816
Plantinga et al. (1999)	Maine, South Carolina, and Wisconsin	NA[c]	0–250	77×10^6	0–263
Lubowski et al. (2003)	National	1700×10^6	7–275	1700×10^6	7–275

[a] The total quantity was not reported.
[b] Potential yield was reported as the cumulative amount of carbon over 160 years.
[c] Quantities were reported in "present ton equivalents" rather than tons per year.
Table 8 in Stavins and Richards (2005).

tonne ($30 to $90/ton) for a program size of 454 million tonnes (500 million tons) of annual carbon sequestration. The authors noted that a sequestration program of 454 million tonnes (500 million tons) per year would offset approximately one-third of annual U.S. carbon emissions.

Stavins and Richards constructed a marginal cost curve (or supply function) for carbon sequestration using the normalized results for the national studies (fig. 4-5). Curves are shown for engineering (Dudek and LeBlanc 1990, Moulton and Richards 1990, Parks and Hardie 1995, Richards 1997, Richards et al. 1993), econometric (Lubowski et al. 2003[3]), and optimization studies (Adams et al. 1993). Stavins and Richards found that normalization narrows the spread of the different cost estimation methods. For example, compare the position of the Adams et al. (1993) curve in figure 4-4 to its position in figure 4-5. There is still evidence that econometric methods produce higher costs. The marginal cost curves from Lubowski et al. (2003) are the highest over the range of 227 to 454 million tonnes (250 to 500 million tons) per year, but lie below those of Dudek and LeBlanc (1990), Parks and

[3] Note that the curves from Lubowski et al. 2003 are the same as those from Lubowski et al. 2006.

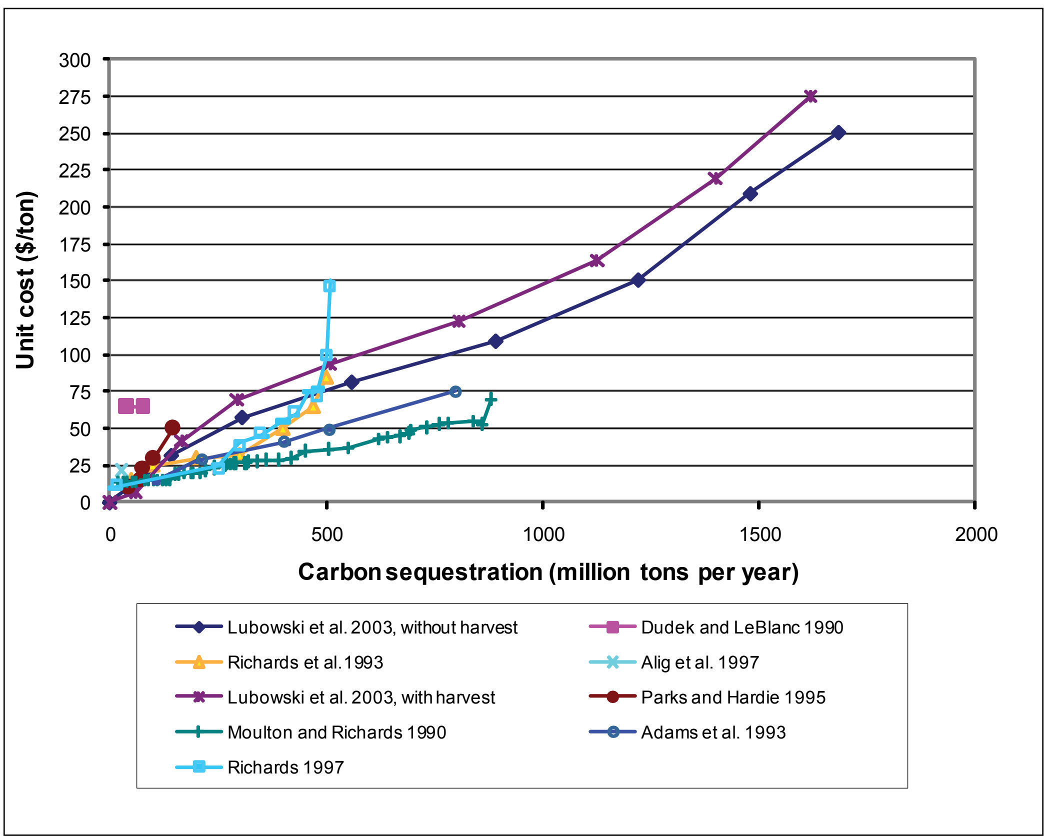


Figure 4-5—Comparison of normalized marginal cost curves (or supply function) for forest carbon sequestration for national studies based on engineering, econometric, and optimization studies; adapted from figure 5 in Stavins and Richards (2005).

Hardie (1995), and Alig et al. (1997) at lower levels of carbon sequestration. As well, the cost curve from Richards (1997)—an engineering study—is the highest after 454 million tonnes (500 million tons).

Finally, tables 4-1 and 4-2 report normalized cost estimates from Stavins and Richards (2005) and Stavins (1999) for groups of studies that include regional analyses. Again, there is evidence that the highest cost estimates are produced by econometric analyses.

Table 4-2—Comparison with results from other studies

	Total quantity		Average cost		Marginal cost	
Study	Land	Carbon	Land	Carbon	Land	Carbon
	Million acres	*Million tons/yr*	*$/acre per yr*	*$/ton*	*$/acre per yr*	*$/ton*
This study[a]						
United States normalization	342	518	106	70	≤ 200	≤ 136
Delta states	5	7	58	38	≤ 100	≤ 66
Moulton and Richards (1990)						
United States[b]	269	690	—	27	≤ 81	≤ 37
Delta states cropland	25	67	50	22	—	—
Richards et al. (1993)						
United States[c]	244	416	—	—	—	≤ 41
Delta states cropland[d]	11	29	42	18	≤ 52	≤ 22
Adams et al. (1993)[e]	274	700	—	—	—	≤ 27
Nordhaus (1991)[f]	248	44	81	64	—	—
Parks and Hardie (1995)[g]	9	22	49	21	—	≤ 24
Rubin et al. (1992)[h]	71	73	—	23	—	—
Dudek and LeBlanc (1990)[i]	14	—	—	38	—	—
Plantinga (1995)[j]	0.65	1.5	—	—	—	6-13
Callaway and McCarl (1996)[k]	187	280	—	—	—	≤ 25

[a] Pine plantation, periodically harvested, at a 5-percent discount rate.
[b] Permanent stands on cropland and pastureland only, i.e., not forest land.
[c] Figure for total U.S. carbon sequestration is an annuity calculated at 5 percent over 160 years.
[d] These figures were used, but not reported, in Richards et al. (1993). Reference is to a permanent pine stand, based on data provided in a personal communication from Richards (1994). Carbon costs and tonnages were annualized over 160 years at a 5-percent discount rate.
[e] Nationwide results for a scenario with harvesting and sale of timber (Stavins 1999: table 1 p. 79 and table 4 p. 83), recalculated at a 5-percent discount rate.
[f] Permanent forestation of "marginal U.S. land" (Stavins 1999: table 8 p. 60). For this and other studies, we have converted to acres at a rate of 1 hectare = 2.477 acres and to short tons at a rate of one metric ton = 1.102 short tons.
[g] Figures are for U.S. cropland-only scenario (Stavins 1999: table 1 p. 127). Marginal costs were computed from marginal cost formula for figure 4 (Stavins 1999: p. 131) using 22 million tons per year and annualized using a 4-percent discount rate over 10 years.
[h] Nationwide results converted from original study (Stavins 1999: table 3 p. 261) at a rate of 3.67 tons of carbon dioxide (CO_2) equals one ton of carbon, and into short tons from metric tons.
[i] An average permanent stand of U.S. tree species, from Stavins 1999: table 3 p. 36; CO_2 converted to carbon.
[j] Figures are for a 14-county region of Wisconsin for the scenario assuming a least-cost program at a 4-percent discount rate and a constant annual sequestration rate of 2.25 tons of carbon per acre (Stavins 1999: table II). Hectares converted to acres.
[k] Calculations use a 5-percent discount rate, employ carbon yield functions from Birdsey (1992), and do not allow for farm programs.
Source: Table 3 from Stavins (1999).

Results and Conclusions

The optimization approach produces higher cost estimates than the engineering approach because it accounts for price feedbacks from the policy that are typically ignored in engineering studies.

Two decades of research supports the conclusion that afforestation can be used to offset CO_2 emissions at costs that are comparable to or lower than those with energy-based approaches such as energy efficiency and fuel switching. Nevertheless, differences in carbon sequestration cost estimates among studies are deserving of further investigation. Part of the explanation is that researchers analyze different regions with different forest species and opportunity costs of land. In this review, we have tried to identify differences that arise from the methodology used to compute costs, focusing on the three dominant modeling approaches—bottom-up engineering, sectoral optimization, and econometric models. Our main finding is that the optimization approach produces higher cost estimates than the engineering approach because it accounts for price feedbacks from the policy that are typically ignored in engineering studies. As more land is converted from agriculture to forest, agricultural commodity prices should rise and forest commodity prices should fall, depressing incentives to move more land into forest. In addition, we found that the econometric approach produces higher cost estimates than both the engineering and optimization methods. The most likely reason is that the econometric approach can account for a number of factors that affect land use decisions in practice but that are difficult to measure explicitly for engineering and optimization models. These include landowner uncertainty in the face of irreversible investments, nonmonetary returns to landowners from forest and agricultural uses of land, liquidity constraints, and other private or market costs or benefits.

For policymakers interested in knowing how much U.S. forest carbon sequestration can contribute to an emissions reduction target, a conservative estimate based on the evidence from many studies is that 454 million tonnes (500 million tons) could be sequestered annually at a price of $110/tonne ($100/ton) of carbon. This is the estimate found by Stavins (1999) and Lubowski et al. (2006), two econometric studies, and Richards et al. (1993) and Richards (1997), two engineering studies that account for price feedbacks. It should be emphasized, again, that the cost estimates in this literature are the opportunity costs, or the social costs, of offsetting carbon emissions through afforestation. The costs to the government of achieving emissions reduction targets are likely to be considerably higher. Indeed, if one considers that raising public funds to pay for a carbon sequestration policy can have welfare implications, then the social costs of carbon sequestration may be underestimated. This discussion points to the need for further research on a number of implementation issues that still must be carefully addressed before a large-scale carbon sequestration program can be put into effect. For instance, to reduce the budgetary impacts of a carbon sequestration program, the government would need to address the issue that landowners will have more information about their opportunity costs than the government does. A number of contracting schemes can be employed in this context to reduce government costs.

Acknowledgments

Funding from the U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station is gratefully acknowledged. The authors thank Ken Richards and Ned Stone for helpful comments.

English Equivalents

When you know:	Multiply by:	To get:
Hectares (ha)	2.47	Acres
Grams (g)	0.352	Pounds
Metric tonnes	1.102	Tons

References

Adams, D.M.; Alig, R.J.; McCarl, B.A.; Callaway, J.M.; Winnett, S.M. 1996. The Forest and Agricultural Sector Optimization Model: model structure and applications. Res. Pap. PNW-RP-495. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 44 p.

Adams, D.M.; Alig, R.J.; McCarl, B.A.; Callaway, J.M.; Winnett, S.M. 1999. Minimum cost strategies for sequestering carbon in forests. Land Economics. 75(3): 360–374.

Adams, R.M.; Adams, D.M.; Callaway, J.M.; Chang, C.C.; McCarl, B.A. 1993. Sequestering carbon on agricultural land; social cost and impacts on timber markets. Contemporary Policy Issues. 11(1): 76–87.

Alig, R.J.; Adams, D.M.; McCarl, B.A. 2002. Projecting impacts of global climate change on the U.S. forest and agriculture sectors and carbon budgets. Forest Ecology and Management. 169: 3–14.

Alig, R.J.; Adams, D.M.; McCarl, B.A; Callaway, J.M.; Winnett, S.M. 1997. Assessing effects of mitigation strategies for global climate change with an intertemporal model of the U.S. forest and agriculture sectors. Environmental and Resource Economics. 9: 259–274.

Birdsey, R.A. 1992. Carbon storage and accumulation in United States forest ecosystems. Gen. Tech. Rep. WO-59. U.S. Department of Agriculture, Forest Service. Washington, DC. 51 p.

Boscolo, M.; Vincent, J.R.; Panayotou, T. 1998. Discounting costs and benefits in carbon sequestration projects. Environment Discussion Pap. No. 41. Cambridge, MA: Harvard Institute for International Development. 43 p.

Callaway, J.M.; McCarl, B. 1996. The economic consequences of substituting carbon payments for crop subsidies in U.S. agriculture. Environmental and Resource Economics. 7(196): 15–43.

Dudek, D.; LeBlanc, A. 1990. Offsetting new CO_2 emissions: a rational first greenhouse policy step. Contemporary Policy Issues. 8: 29–42.

Gorte, R.W. 2009. Carbon sequestration in forests. Congressional Research Service. 7-5700. www.crs.gov; RL31432.http://www.fas.org/sgp/crs/misc/RL31432.pdf. (November 2009).

Lubowski, R.; Plantinga, A.; Stavins, R. 2003. Land-use change and carbon sinks: econometric estimation of the carbon sequestration supply function. Working Paper 23. Cambridge, MA: John. F. Kennedy School of Government, Harvard University. 78 p.

Lubowski, R.; Plantinga, A.; Stavins, R. 2006. Land-use change and carbon sinks: econometric estimation of the carbon sequestration supply function. Journal of Environmental Economics and Management. 51: 135–152.

Moulton, R.; Richards, K. 1990. Costs of sequestering carbon through tree planting and forest management in the United States. Gen. Tech. Rep. WO-58. Washington, DC: U.S. Department of Agriculture. 56 p.

New York State. 1991. Analysis of carbon reduction in New York State. Report of the New York State Energy Office, in consultation with NYS Department of Environmental Conservation and NYS Department of Public Service. New York: NYS Energy Office. 46 p.

Newell, R.; Pizer, W. 2001. Discounting the benefits of climate change mitigation: How much do uncertain rates increase valuations? On file with: Pew Center for Global Climate Change, Washington, DC. 39 p.

Newell, R.; Stavins, R. 2000. Climate change and forest sinks: factors affecting the costs of carbon sequestration. Journal of Environmental Economics and Management. 40(3): 211–235.

Nordhaus, W.D. 1991. The cost of slowing climate change: a survey. Energy Journal. 12: 37–65.

Parks, P.; Hardie, I. 1995. Least-cost forest carbon reserves: cost-effective subsidies to convert marginal agricultural land to forests. Land Economics. 71(1): 122–136.

Plantinga, A.J. 1995. The costs of carbon sequestration in forests: a positive analysis. Terrestrial carbon sequestration: an economic synthesis workshop. Bergendal, Sweden, May 15–19. Washington, DC: American Forests.

Plantinga, A.J. 1997. The costs of carbon sequestration in forests: a positive analysis. Critical Reviews in Environmental Science and Technology. 27(Nov): S268–S278.

Plantinga, A.J.; Mauldin, T.; Miller, D. 1999. An econometric analysis of the costs of sequestering carbon in forests. American Journal of Agricultural Economics. 81: 812–824.

Richards, K. 1997. Estimating costs of carbon sequestration for a United States greenhouse gas policy. Report on file with: Ken Richards, Indiana Universtiy, Department of Public and Environmental Affairs, 1315 E Tenth St., Bloomington, IN 47405.

Richards, K. 2004. A brief overview of carbon sequestration economics and policy. Environmental Management. 33(4): 545–558.

Richards, K.R.; Moulton, R.; Birdsey, R. 1993. Costs of creating carbon sinks in the U.S. Energy Conservation and Management. 34(9-11): 905–912.

Richards, K.R.; Stokes, C. 2004. A review of forest carbon sequestration cost studies: a dozen years of research. Climatic Change. 63(1-2): 1–48.

Rubin, E.S.; Cooper, R.N.; Frosch, R.A.; Lee, T.H.; Marland, G.; Rosenfeld, A.H.; Stine, D.D. 1992. Realistic mitigation options for global warming. Science. 257: 148–149.

Sedjo, R.; Solomon, A. 1989. Greenhouse warming: abatement and adaptation. In: Crosson, P.; Darmstadter, J.; Easterling, W.; Rosenberg, N., eds. July 1989 RFF Proceedings. Washington, DC: Resources for the Future: 110–119.

Sohngen, B.; Mendelsohn, R. 2003. An optimal control model of forest carbon sequestration. American Journal of Agricultural Economics. 85(2): 448–457.

Stavins, R.N. 1999. The costs of carbon sequestration: a revealed preference approach. American Economic Review. 89: 994–1009.

Stavins, R.N.; Richards, K. 2005. The cost of U.S. forest based carbon sequestration. Arlington, VA: Pew Center on Global Climate Change. 40 p.

van Kooten, G.C.; Arthur, L.; Wilson, W. 1992. Potential to sequester carbon in Canadian forests: some economic considerations. Canadian Public Policy. XVIII(2): 127–138.

van Kooten, G.C.; Eagle, A.; Manley, J.; Smolak, T. 2004. How costly are carbon offsets? A meta-analysis of forest carbon sinks. Working Pap. 2004-01. Victoria, BC: University of Victoria, Department of Economics, Resource Economics and Policy Analysis (REPA) Group. 37 p.

van Kooten, G.C.; Shaikh, S.L.; Suchanek, P. 2002. Mitigating climate change by planting trees: the transaction costs trap. Land Economics. 78(4): 559–572.

van Kooten, G.C.; Sohngen, B. 2007. Economics of forest ecosystem carbon sinks: a review. Victoria, BC: University of Victoria, Department of Economics, Resource Economics and Policy Analysis (REPA) Group. 82 p.

Chapter 5: Literature Review: An Economic Analysis of Incentives for Carbon Sequestration on Nonindustrial Private Forests (NIPFs)

by Christian Langpap and Taeyoung Kim

Introduction

There is widespread recognition of the potential role forests can play in contributing to greenhouse gas (GHG) reductions through carbon sequestration (Brand 1998, Lubowski et al. 2006, Metz et al. 2001). Nonindustrial private forests (NIPF) make up a significant portion of U.S. forests (approximately 40 percent). Thus, it is crucial to assess the role that NIPF landowners can play in broader carbon sequestration efforts.

Because the resulting carbon sequestration benefits are to a large extent external to individual landowners, incentives may be necessary to induce them to adopt these management options.

Management actions by NIPF owners that could increase carbon sequestration on their lands include afforestation of land used for agriculture, increasing rotation length, intensive management (e.g., juvenile spacing, fire control, fertilization), changing stocking density, or choosing alternative tree species (Shaikh et al. 2007, Sohngen and Mendelsohn 2003, Stainback and Alavalapati 2002). Because the resulting carbon sequestration benefits are to a large extent external to individual landowners, incentives may be necessary to induce them to adopt these management options (Stainback and Alavalapati 2002). Specific incentives could include carbon sequestration subsidies and carbon release taxes, carbon rental fees, cost-sharing agreements, and agglomeration bonuses.[1] A key question is how effective these different policies can be in eliciting the desired management choices by NIPF owners. This includes assessing and understanding (1) baseline behavior, i.e., NIPF landowner management choices and implications for carbon sequestration in the absence of any incentives and (2) whether and how management choices could be modified by incentive-based policies, i.e., how landowners would respond to different types of incentives and the implications for carbon sequestration and incentive policy design.

Other factors to consider in assessing the potential of NIPFs for carbon sequestration include forest fragmentation and other spatial considerations, including the possibility of economies of scale for carbon sequestration (e.g., aggregators) and

[1] The emergence of markets for ecosystem services in general, and carbon markets in particular, could also create an important incentive for carbon sequestration. However, reviewing the existing research on carbon markets lies outside the scope of this literature review.

the potential role of agglomeration incentives; the impact of carbon markets on the value of forest land and the implications for transitions between forest and agriculture through increased afforestation or reduced deforestation; and other forest management objectives that may prevent landowners from undertaking carbon sequestration activities or reduce the extent to which they do so. In this report we lay the foundation for research that will begin to address these questions.

Carbon Sequestration in Forests

A large and growing literature addresses the general topic of carbon sequestration in forests. Most of the attention to date has focused on issues related to the afforestation of agricultural land. A smaller body of work addresses carbon sequestration in existing forests, but does not necessarily focus on NIPFs. Only a relatively small number of studies deal specifically with the issue of carbon sequestration in NIPFs.

Afforestation of Agricultural Land

Land heterogeneity leads to sharply increasing marginal costs of sequestration as higher quality agricultural lands are converted to forest.

Much of the literature that examines afforestation of agricultural land has focused on estimating the costs of carbon sequestration. Parks and Hardie (1995) simulated the impacts of subsidies for sequestering carbon in new forests established on agricultural land. They derived a supply schedule for carbon sequestered in marginal agricultural land converted to forest and used this supply schedule to develop criteria for enrolling lands in a national carbon sequestration program. Plantinga et al. (1999), Stavins (1999), Newell and Stavins (2000), and Lubowski et al. (2006) used econometric models of land use to simulate the effects of a payment (subsidy) for forested agricultural land and a tax on deforested land. They found that sequestration increases with a subsidy or a combined subsidy/tax policy, but their main emphasis was on estimating marginal costs of sequestration.

Plantinga et al. (1999) suggested that afforestation is a relatively low-cost way of reducing carbon concentrations. Stavins (1999) stressed that marginal costs of sequestration are not trivial, and that land heterogeneity leads to sharply increasing marginal costs as higher quality agricultural lands are converted to forest. Newell and Stavins (2000) found that costs of sequestration can be higher if trees are harvested periodically rather than permanently established, that marginal costs increase with discount rates, that higher agricultural prices lead to higher costs or less sequestration, and that delayed deforestation can sequester carbon at lower cost than increased forestation. Lubowski et al. (2006) estimated marginal costs of carbon sequestration greater than those from previous engineering cost analyses and sectoral optimization models. They found that the estimated sequestration supply function is similar to the carbon abatement supply function from energy-based analyses, suggesting that forest-based carbon sequestration merits inclusion in a cost-effective portfolio of domestic U.S. climate change strategies.

Plantinga and Wu (2003) simulated the response by private landowners to subsidies for converting agricultural land (cropland and pasture) to forest in Wisconsin. They examined the environmental impacts of afforestation that go beyond carbon sequestration, such as modification of wildlife habitat and reductions in agricultural pollution, and found that the additional environmental benefits would be substantial, on the same order of magnitude as the costs of the subsidy program. Gillig et al. (2004) examined the effects of carbon payments in the form of a sequestration subsidy or an emission tax. They estimated response functions that depict the effects of carbon prices, energy prices, domestic agricultural demand, and foreign agricultural demand on GHG emission reductions and sequestration. Their results suggest that restricting carbon payments only to afforestation or deforestation or only to agricultural sequestration substantially reduces potential mitigation. Policies that include both sectors consistently yield the largest quantity of GHG offsets in their simulations.

Other papers have explicitly modeled the links between agricultural land, forest land, and timber markets, and examined the potential for offsetting changes in land use (from forest to agriculture) resulting from price feedbacks. Adams et al. (1993) addressed the link between forest and agricultural sectors to capture price feedbacks between forest, agricultural, and land markets. They found empirical evidence of a rise in agricultural prices, a fall in timber prices, and changes in stakeholder welfare that could result from large-scale afforestation programs. They

Emily Jederlinich

Landowner behavior is a major determinant of land use and land cover changes, an important consideration for policy analyses concerned with climate change adaptation and mitigation.

argued that private landowners might need to be compensated to keep their land in forests. Otherwise, the price changes resulting from afforestation could provide an incentive for offsetting land-use changes from forest to agriculture, creating carbon leakage. Alig et al. (1997) used a model of the U.S. forest and agriculture sectors in which land-use choices and forest management decisions are endogenous. They examined the costs and land base adjustments from meeting a carbon sequestration target to minimize net social welfare costs. Their results suggest that policy-induced land-use changes to forestry may induce compensating land-use changes through their impact on markets. They argued that land-use shifts to meet policy targets may not be permanent, and found that the main forms of adjustment to meet policy targets are to shift land use from agriculture to forest and to manage forests more intensively.

Transaction costs of getting landowners to convert their land from agriculture to plantation forests appear to be a significant obstacle, possibly increasing the costs of afforestation projects beyond what conventional economic analysis suggests.

Finally, some papers have reported similar studies in other countries. For example, De Jong et al. (2000) examined the response of small (subsistence) farmers and communities in southern Mexico to incentives to switch from their current land use to forestry or adopt management measures to increase sequestration, such as agroforestry or improved forest management. They calculated the expected response to financial incentives in the form of carbon sequestration rents ranging from \$0 to \$40 per megagram of carbon (MgC) sequestered. They found that the amount of carbon sequestered would rise rapidly from 1 million MgC to 38 million MgC when the incentive level increases from \$5 to \$15 per MgC, owing mainly to natural forest management and fallow improvement.

Van Kooten et al. (2002) examined the institutions and incentives needed to encourage landowners in Canada to adopt tree planting on a large scale. They used data from a survey of farmers in western Canada's grain belt region to provide insights concerning transaction costs and the design of appropriate institutions and economic incentives for creating additional terrestrial carbon sinks at least cost. Their results suggest that transaction costs of getting landowners to convert their land from agriculture to plantation forests appear to be a significant obstacle, possibly increasing the costs of afforestation projects beyond what conventional economic analysis suggests. Over one-quarter of their survey respondents indicated that they would be unwilling to enter into an afforestation program voluntarily, even if they are fully compensated for lost agricultural revenues and tree planting costs. The authors argued that a possible reason is that later improving land for any subsequent agricultural use by removing trees is considered costly, both financially and in terms of utility.

Zelek and Shively (2003) measured the costs of carbon sequestration on tropical farms in the Philippines. They empirically estimated the value of agricultural land and the opportunity cost of converting fallow and agricultural land to forest and agroforest. They also derived the rates of carbon sequestration for timber and agroforestry systems and computed incentive-compatible compensating payment schedules for farmers who sequester carbon. To compute agricultural opportunity costs, they used a combination of data from household surveys conducted in the watershed and results from farm-level simulations. They found that agroforestry systems are a lower cost alternative to pure forest conversion, providing carbon storage at a marginal cost that is up to 23 percent lower than the marginal cost of carbon storage through conversion to a pure tree stand.

Shaikh et al. (2007) examined the costs of planting trees on marginal agricultural land in western Canada and the compensation landowners would require for converting pasture and cropland to forestry. They conducted a contingent valuation survey of landowners to incorporate nonmarket values, risk attitudes, and unobservable transaction costs. They found that farmers are unwilling to plant trees on agricultural land without financial incentives, but that the necessary incentives could be less than net returns to agricultural activities on marginal agricultural land due to nonmarket benefits from trees perceived by farmers. Nevertheless, they found that average costs of carbon sequestration generated this way would exceed the projected value of the corresponding carbon credits under a carbon emissions trading program.

USDA Forest Service

Landowners have retained subsidized afforested stands at high rates and well beyond government program life.

Ahn (2008) used an econometric land-use share model with province-level data to calculate the costs of carbon programs through afforestation in Korea. Estimation results show that an increase in forest returns causes landowners to convert agricultural land to forest, and suggested that carbon sequestration can be a cost-effective mitigation policy in Korea, although it is not necessarily the least-cost option. Ahn argued that enhancing the management of timberland to increase carbon stored per unit area is just as important as expanding afforestation. Rodríguez-Vicente and Marey-Pérez (2009) analyzed NIPF owners' management choices to transition between farming and forestry in northern Spain. They used a survey of NIPF owners and focused on past conversion of forest to meadow and marginal meadow to woodland. They examined the structural attributes of the forest holding and past changes and future land-use intentions of NIPFs in the region for the 1999–2003 period. They found that the greatest forest management activity was associated with larger (productive) forest holdings, less divided forest land, and more time available to dedicate to forestry activities. Conversion of forest land into meadow responded to increasing demand for agricultural land and landowner's occupation as a farmer. Past transformation of marginal meadow to forest and future intentions to increase forest area depended on past experience with forestry and whether it had been profitable, and on the occupation of the landowner (retired farmers and nonagricultural professionals). Intention to change forest species also depended on forest profitability.

This literature does not focus specifically on NIPFs or on studying the effectiveness of providing incentives for carbon sequestration. However, these studies rely on financial incentives, mostly tax/subsidy combinations, to measure the costs of afforestation programs. They strongly suggest that financial incentives and changes in relative returns to land use affect landowner behavior and can be used to increase carbon sequestration in private forests.

Carbon Sequestration in Existing Forests

A number of papers focused more specifically on increasing carbon sequestration in existing forests. Van Kooten and Sohngen (2007) provided a useful summary of this literature.

Plantinga and Birdsey (1993) developed a carbon budget model to examine the effects of forest management practices on carbon storage in private U.S. forests. The U.S. Forest Sector Model was used to project changes in forest resources under the assumption of market equilibrium for wood products. Changes in the forest carbon inventory result from tree growth and management activities, in particular harvesting. A base-run scenario projects increases in carbon storage in private timberlands by 2040; however, this increase is offset by carbon emissions resulting from harvesting. The study concluded that, if current trends in private timberland management continue, the effectiveness of these lands as a carbon sink may be limited. Although carbon budget surpluses are expected in near decades, increasing deficits are projected in the future as harvests increase to meet higher demand for wood products.

Englin and Callaway (1993) investigated the impact of carbon payments on the optimal rotation age of Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco). Using a range of carbon values from \$10 to \$200 per metric ton, they found that with carbon payments, the rotation age is longer than the traditional Faustmann rotation age and is positively correlated with the price of carbon. Plantinga and Birdsey (1994) conducted a theoretical study to incorporate carbon payments into the forest rotation problem. They concluded that, in most cases, the optimal rotation is infinite when only carbon benefits are included in the analysis, whereas when both carbon and timber are included, the optimal rotation would be between the optimal carbon-only rotation and the optimal timber-only rotation. Huang and Krongrad (2001) estimated the annual financial compensation that utility companies would have to pay private forest landowners to encourage sequestering additional carbon and calculated the average cost to sequester a ton of carbon. They calculated the amounts needed to compensate forest landowners who apply economically sub-optimal rotations to sequester maximum carbon or to motivate private landowners to convert unstocked lands into productive forest lands to sequester carbon. They found that the annual compensation values tend to increase as real interest rates increase: the minimum annual compensation is \$0.84 per ha using an interest rate of 2.5 percent, whereas the maximum annual compensation is \$72.79 per ha using an interest rate of 12.5 percent.

Murray (2003) used an analytical model of timber and carbon rotation and data from different forest settings to examine the effects of carbon sequestration incentives on the optimal management of an individual forest stand. He found that the incentive modifies the optimal timing of harvest and the return to forest land use. Sohngen and Mendelsohn (2003) added carbon sequestration to an optimal control model of GHGs, and modeled an incentive for sequestration that takes the form of a carbon rental fee for each additional ton of carbon stored. They found that carbon rental payments increase the value of land used for forests, causing more land conversion to forest and increasing rotation lengths and management intensity (increasing stock density). However, they argued that changing management intensity is more costly and less effective at carbon storage than is afforestation. Sohngen and Brown (2006) used a land-use share model to examine the mix of upland hardwoods and softwoods in a three-state region of the South Central Untied States (Arkansas, Louisiana, and Mississippi). The land-use share model was combined with a simulation model to examine the types of subsidies that could be used to maintain the stock of hardwoods in this region. The results suggest that subsidies of approximately $12 to $27 per ha per year would maintain the area of hardwood forests and reduce carbon emissions over the next 30 years.

Guthrie and Kumareswaran (2009) used a theoretical model to examine the effectiveness of carbon credit payments in providing incentives for private forest owners to increase forest land and lengthen rotations. They focused on alternative payment systems that could be used to allocate carbon credits to forest owners, including allocation of credits depending on the amount of carbon actually sequestered at a point in time, and allocation according to the long-run carbon sequestration potential of land, where landowners receive payment as long as the land is planted with trees. They found that allocating carbon credits can significantly alter harvesting decisions by making forestry more profitable, lowering the timber price at which the landowner abandons forestry (i.e., switches land use) as well as the price at which it is optimal to harvest and replant. Their results suggest that payments based on actual carbon stock lead to longer rotation periods, whereas payments for long-run potential sequestration induce shorter rotation periods. Payments based on actual sequestration leads to greater benefits from sequestration at a lower cost.

Emily Jederlinich

Unintended consequences of policies can lead to significantly different outcomes than envisioned, including leakage associated with market responses to changes in supply and demand.

In research focused on other countries, van Kooten et al. (1995) examined the implications of carbon subsidies and taxes for economically optimal harvest decisions and for carbon sequestration in forests in western Canada. Subsidies are intended to encourage planting and management activities that promote tree growth, and taxes discourage harvest and the subsequent release of carbon. They found that (for the most likely range of parameters) rotation ages would increase by roughly 20 percent over the level where no carbon costs or benefits are considered.

These papers examine carbon sequestration in forests, but do not focus specifically on NIPFs. Nevertheless, their results broadly agree with those from the literature that examines afforestation. In particular, they suggest that incentive programs including taxes or carbon payments or other types of subsidies can impact the management decisions of forest owners in ways that can lead to increased carbon sequestration.

For private forests, incentives may be necessary to induce landowners to consider carbon sequestration benefits in their production decisions.

Carbon Sequestration in NIPFs

There is a small literature that discusses carbon sequestration in the specific context of NIPFs. Stainback and Alavalapati (2002) suggested that forests being managed for commodity production could sequester additional carbon by lengthening the rotation and producing more products with a long product life, such as sawtimber, and fewer products with a shorter product life, such as pulpwood. They argued that, for private forests, incentives may be necessary to induce landowners to consider carbon sequestration benefits in their production decisions. They examined how internalizing carbon benefits onto private pine plantations in the Southeast United States would impact forest management, specifically the optimal rotation age, the product mix produced, and the amount of carbon supplied in slash pine forests. They set up a model in which landowners are compensated for carbon sequestered as trees grow and carbon is stored in timber products (sawtimber and pulpwood) and are taxed for carbon emissions at harvest. They found that a carbon subsidy and tax policy would increase the amount of carbon sequestered in a forest stand in two ways: by lengthening the rotation age and increasing the amount of biomass produced in the stand, and by increasing the proportion of the biomass put into long-lived end products such as sawtimber. This incentive system would have a significant impact on the management decisions of private forest-land owners. Positive values for carbon significantly increase forest-land rents. This increase in private forest-land rents induces landowners to use a larger portion of their land for timber production, thereby increasing timber supply at the extensive margin. The increase in forest-land rents could also reduce forest conversion to other uses such as urban development. They also argued that the increase in land values has

Nonindustrial private forest owners' responses to incentives that try to affect harvest decisions are complicated by the fact that owners have multiple land management objectives.

implications for the implementation of a carbon subsidy and tax policy, as the fact that landowners would gain substantially from such a policy indicates that it could be implemented on a voluntary basis similar to the way the Conservation Reserve Program is implemented.

Alig (2003) summarized land-use and land cover changes; identified the drivers of deforestation, afforestation, reforestation, and timber harvest; and highlighted the implications for carbon sequestration in forests, including NIPFs. He argued that U.S. landowners have responded significantly to past government programs for tree planting (afforestation), such as subsidized tree planting for environmental goals, and that projections indicate that U.S. private timberlands have considerable potential for additional wood production and more carbon sequestration under intensified management. However, he noted that NIPF owners' responses to incentives that try to affect harvest decisions, such as delaying harvest, are complicated by the fact that many owners do not have timber production as a primary goal and that some owners have multiple land management objectives.

Im et al. (2007) examined the welfare impacts and costs of a carbon tax and subsidy program for enhancing the sequestration of carbon on the existing private forest land base. Forest owners are subsidized for the carbon they accumulate and taxed for the carbon released by harvesting. They developed a theoretical model of a forest owner's response to the carbon tax and demonstrated how the forest owner would adjust harvest in various circumstances. They employed a model of the log market in Oregon to develop specific estimates of the impacts of the carbon tax on harvest and management actions and to examine the cost-effectiveness of the carbon tax as a mitigation option in the forestry sector. The simulated carbon tax leads to reduced harvest and increased carbon stock in standing trees and understory biomass. Changes in the level of silvicultural investments differ by owner, depending on the nature of their initial inventory. Average rotation age increases, varying in extent across ownerships and site qualities. Their estimates of the marginal cost of sequestering carbon in Oregon private forests are shown to be within the range of costs for projects considering afforestation alone in some eastern regions of the United States.

Fletcher et al. (2009) examined the willingness of NIPFs in Massachusetts to sell carbon credits in several hypothetical carbon sequestration programs. They used a pilot survey of 17 NIPFs in western Massachusetts that contained a choice experiment for six alternative carbon credit programs with different eligibility requirements, time commitment, expected payment, and penalty for early withdrawal. They used regression models to examine the relationship between program ratings and attributes and socioeconomic characteristics of NIPFs, as well as

Ralph Alig

Future developments in other sectors are likely to continue to impact the forest sector and its relative advantage in sequestering greenhouse gases.

their willingness to sell credits at various prices. They found that program rating increases with the expected payment and the length of commitment and decreases with the early withdrawal penalty. They also found that at current carbon credit prices, very few participants (less than 7 percent) would be willing to sell. However, the small sample size of this study suggests that these results should be interpreted with caution.

Our review of the literature on carbon sequestration in forests reveals that little attention has been paid to the potential effects of incentives for carbon sequestration in NIPFs. The broader literature on afforestation of agricultural land and carbon sequestration in forests suggests that incentives can be effective in changing land-use and forest management decisions in ways that could increase the amount of carbon sequestered in forests. The extent to which these general results apply specifically to NIPFs, however, remains an open question given that, as Alig (2003) pointed out, their motivations for owning and managing forest land may differ from those of other forest owners and landowners. To gain additional insight into the behavior of NIPFs, next we review the broader literature on how they respond to incentive programs with alternative environmental goals, such as sustainable forestry practices or biodiversity conservation.

Incentives and NIPF Landowner Behavior

A large literature explores NIPF landowner characteristics, objectives, and forest management decisions. This literature identifies a variety of policy tools that may influence the management decisions of NIPF owners, including education, technical assistance, regulation, and financial incentives. The most common financial incentives include cost sharing or grants for developing forest management plans or implementing forest management practices, including planting and stand improvement, and tax incentives through the federal income tax and state property tax systems (Jacobson et al. 2009a, 2009b). Summaries of this literature include Alig et al. (1990), Amacher et al. (2003), Greene et al. (2005, 2006), Kilgore et al. (2007), and Joshi and Arano (2009).

Many of these papers focus on how landowner attributes and incentives shape forest management decisions or sustainable forestry practices. Several authors have reviewed the literature and concluded that NIPF owners are largely unaware of the existence of incentives or do not understand how they apply to them and that often they would carry out the supported practice even in the absence of incentives. They found that NIPF owners are largely unresponsive to property tax and capital gains provisions, and that forest property tax programs are only modestly successful in achieving their goals (Greene et al. 2005, 2006; Jacobson et al. 2009a, 2009b; Kilgore et al. 2007). They also identified approaches that have consistently been found to provide adequate incentives for NIPFs to practice sustainable forestry: technical and management planning assistance, cost sharing, and direct contact with a forester or natural resource specialist. When tree planting on agricultural land is induced, a relatively large percentage is retained over a long period (e.g., Alig et al. 1980).

When tree planting on agricultural land is induced, a relatively large percentage is retained over a long period.

Jacobson et al. (2009a, 2009b) additionally used a survey of forestry agency officials who administer public incentive programs to examine whether, given changing forest ownership patterns and program emphases, financial incentives in the northern and southern regions are effective in promoting sustainable forestry practices, whether some programs are more effective than others, and the characteristics of effective programs. They evaluated eight federal incentive programs and three nonfederal programs. They found that forestry officials rated federal incentive programs as only adequate for NIPF owner awareness and appeal. They argued that one possible reason for the low appeal is a general wariness of participating in government programs for fear of loss of independence and fear of government control over management choices. Overall, forestry officials think that financial incentives are effective in promoting sustainable management practices. The results suggest that, in general, programs targeted specifically to forest owners are rated higher than programs targeted to ranchers and farmers in addition to forest owners.

Joshi and Arano (2009) also found that NIPF owners are largely unaware of incentive programs available to them, and thus argued that much remains to be done to encourage NIPF landowner investment in forestry activities. They suggested that existing programs have had limited success because they emphasize timber production, whereas landowners usually own forests for a variety of reasons, including recreation and wildlife or as a site for their home, and timber production may not be their main priority. They used data from a mail survey of 2,100 NIPF owners in West Virginia to evaluate the factors affecting their decisions to engage in timber harvest, silvicultural activities (e.g., planting, fertilization), property management (e.g., road maintenance, access control), and wildlife habitat management and recreation improvement. They found that age, education, profession, income, ownership size, length of ownership, distance to residence, objective of ownership, and development of a management plan were significant determinants of at least one of the categories of management activities.

Hardie and Parks (1996) examined how the level of cost-sharing might have affected the number and size of forest tracts enrolled in the reforestation cost component of the Forestry Incentive Program (FIP) in the southern pine region of the United States. They reported acreage enrollment predictions for government cost shares ranging from 0 to 100 percent. They also developed predictions that offer some insight into the potential effects of an education program aimed at informing NIPF landowners about the reforestation component of the FIP. The data used in the analysis are from an area-frame survey conducted by the National Agricultural Statistics Service. The results suggest that few acres would be replanted after harvest if the cost-sharing programs did not exist. The results also show that total acreage predictions decrease much more rapidly with decreases in the rate of cost-sharing when size of tracts installed by landowners is predicted to respond to the level of

Ralph Alig

The NIPF ownership is the one most impacted in the forest sector by land-use changes, affecting the ownership's potential for greenhouse gas sequestration.

program support. The results also reveal that informing more NIPF owners about existing cost-sharing programs is as effective as increasing the amount offered in the cost-share.

Nagubadi et al. (1996) examined factors influencing Indiana NIPF landowner participation in forestry assistance programs based on actual participation decisions. They used data collected by mail survey from NIPF landowners (789 respondents) in Indiana during the winter of 1994. They found that information and management factors are the most influential in predicting the probability of participation in forestry assistance programs. Landowners who are motivated by commercial interests and are involved in commercial forestry activities have a higher probability of participation in forestry programs. The results also suggest that the size of landholding is an important determinant of participation in forestry assistance programs in general, and classified forestry programs in particular.

Conway et al. (2003) argued that bequest motives, debt (or the propensity to save), and nontimber activities such as hunting, hiking, or wildlife observation could be important in determining NIPF landowners' harvest decisions. They used a survey of NIPF owners in the Southeastern United States to obtain data for a model addressing these motives. Their results suggest that there are significant differences between landowners holding large versus small forest properties, between absentee and resident owners, high versus low debt loads, and those that have and do not have bequest intentions. They found that absentee owners are less likely to harvest and engage in nontimber uses of their forest, hold higher debt relative to income, and are more likely to bequeath standing timber. They showed that larger tracts are more likely to be harvested and used for nontimber activities and less likely to be bequeathed as standing timber. Finally, they concluded that landowners with high debt are more likely to engage in nonmarket activities on their land but also more likely to harvest and less likely to bequeath standing timber.

Nagubadi and Zhang (2005) developed a modified multinomial logit framework to model and predict land-use changes by forest ownerships and forest types and apply it to forests in Alabama and Georgia. They found that land quality, federal incentive programs that promote tree planting, and better returns for forestry than for agriculture are the main factors driving the increase in timberland in the two states. Higher income levels and a higher proportion of good-quality land, on the other hand, decrease forestry land use. They found that higher forestry returns increase the amount of industrial private forests, and nontimber values increase acreage of NIPF ownership. Higher population density increases NIPF ownership, whereas a higher proportion of better quality land decreases NIPF ownership. They argued that pooling all types of timberland use into a single category can hide differences among heterogeneous ownerships and forest types and lead to incorrect predictions of land-use change.

Ross-Davis et al. (2005) examined ownership characteristics and values of landowners in Indiana who had planted trees between 1997 and 2001 and their motivations for planting trees, and related these ownership characteristics, values, and motivations to seedling survival during the critical establishment phase. They gathered data through interviews of 151 individuals and field data collection from 87 sites. They found that landowners value their land for the privacy it provides, as a place of residence, and as a legacy for future generations. Landowners afforested primarily to provide for future generations, to supply food and habitat for wildlife, and to conserve the natural environment. Seedling survival was lowest on sites owned by individuals who did not value their land as a legacy for future generations. Raunikar and Buongiorno (2006) analyzed the revealed willingness to pay of NIPF owners in the south-central United States for the amenities of mixed-age and mixed-species forests (i.e., uneven-aged loblolly pine [*Pinus taeda* L.] plantations). They estimated nontimber value (NTV) by the income that owners are willing to forego to maintain natural stands instead of converting them to more profitable plantations. The results show that there are significant differences in the NTV of natural plots by ownership. The average NTV of natural plots was highest on public lands, second on NIPF lands, and least on industry lands. Specifically, they estimated that the average NIPF owner is willing to forgo 60 percent of the timber profit for the NTV of their more natural stands compared to a less diverse industrially managed even-aged plantation.

Other papers focused on the effects of incentive programs to promote conservation of biodiversity and preservation of endangered species habitat on NIPFs. Kline et al. (2000) examined the reasons that NIPF owners own forest land and their willingness to adopt management actions, such as restricting harvest to restore or protect riparian salmon habitat, in exchange for a financial incentive in the form of a federal income tax reduction. They use data from a survey of NIPF owners in western Oregon and Washington. They classified forest owners according to their timber and nontimber objectives and developed an empirical model of landowners' willingness to forgo harvest in riparian areas for 10 years as a function of the tax reduction, their socioeconomic characteristics, and forest ownership objectives. They found that a significant proportion of forest owners are motivated by objectives other than timber production and that, for many, habitat protection is consistent with their forest ownership objectives. They suggested that participation of some NIPFs could be enlisted through low-cost programs that include technical assistance and education regarding management practices that benefit riparian

A significant proportion of forest owners are motivated by objectives other than timber production . . . Participation of some NIPFs could be enlisted through low-cost programs that include technical assistance and education regarding management practices that benefit riparian species.

species. They found that NIPF owners who have mainly timber objectives tend to own larger forests and a larger proportion of NIPF land, making their participation desirable. Financial incentives such as tax relief or cost sharing may be necessary to elicit their participation. They found that mean incentive payments necessary to induce participation differ by ownership objectives. Required payments to forgo harvest are higher for owners who have primarily timber objectives than for owners with both timber and nontimber objectives or mainly recreation objectives.

Zhang and Flick (2001) examined the impact of financial incentives (cost-sharing and a tax incentive) and of potential regulatory land-use restrictions imposed by the Endangered Species Act on NIPF landowner investment behavior. They conducted an econometric analysis of recorded reforestation activities under the incentives program and the regulatory threat, using data obtained through a survey of NIPF owners in North Carolina and South Carolina. The results show that incentives and regulatory threat influence NIPF reforestation behavior in opposite directions: incentives increase reforestation investment, whereas the threat of land-use restrictions decreases it. The results imply that government financial assistance programs can be used to alleviate the disincentive provided by the Endangered Species Act in reforestation investment.

Langpap (2004, 2006) used a survey to examine participation in incentive programs for endangered species conservation by NIPF owners in western Oregon and Washington and analyzed the likely effects of assurances, cost sharing, and compensation incentives on their forest management decisions. He identified landowner and property characteristics that affect participation decisions for an incentive program designed to provide habitat for endangered species, and examined how these characteristics differ from those that determine participation in more general incentive programs. His results suggest that landowners who are younger, have acquired property more recently, own more woodland, and are interested in conservation and providing habitat are more likely to participate. He found that compensation and assurances could have a significant effect on landowners' management decisions, but cost sharing may not. His results also suggest that more effective incentive programs would combine financial incentives with assurances about future regulation. Mayer and Tikka (2006) evaluated six voluntary economic incentive programs for biodiversity conservation in Europe and North America. They found that important factors contributing to program success include an allowance for some economic productivity in enrolled forests, a long period since the inception of the program, and little interference from other incentive programs.

Nelson et al. (2008) developed an integrated model that predicts private land use decisions as a function of existing market conditions and incentive-based conservation payments and predicted the impact of land-use changes on ecosystem services and biodiversity conservation. They used data from Oregon to compare the provision of carbon sequestration and species conservation under five policies that offer payments for conservation. They compared the performance of targeted and untargeted land-use conservation payment schemes relative to baseline land-use patterns with no land-use conservation policy as well as relative to the efficiency frontier for various levels of land-use conservation program budgets. Their results suggest that incentives to restore land to natural cover increase both carbon sequestration and biodiversity conservation, but that there are tradeoffs between these two policy objectives. They showed that policies aimed at increasing the provision of carbon sequestration do not necessarily increase species conservation and that highly targeted policies do not necessarily do as well as more general policies. Furthermore, they show that none of the conservation payment policies considered produce increases in carbon sequestration and species conservation that approach the maximum potential gains on the landscape.

Incentives to restore land to natural cover increase both carbon sequestration and biodiversity conservation, but there are tradeoffs between these two policy objectives.

Matta et al. (2009) examined the willingness of forest owners to adopt management practices designed to enhance biodiversity. They used data from a 2005 survey of NIPF owners in Florida to analyze how land, landowner, and program characteristics influence NIPF landowner participation in incentive programs designed to provide habitat beyond existing Best Management Practices. Their results indicate that mean willingness to accept for adoption of practices at their highest level of restriction would range between \$37 and \$151/ha per year, suggesting that financial incentives would promote habitat conservation on NIPFs. The results also suggest that younger forest owners, and those with higher income, education, and more years of ownership are more willing to adopt the suggested practices. Distance from urban centers, residence on the property, and membership in forest or conservation organizations also increase the probability of participation.

Some papers have examined incentives for biodiversity conservation in other countries. For example, Siikamaki and Layton (2007) programs compared the cost and conservation outcomes of an incentive payment program and a top-down program applied to the protection of small-scale biodiversity hotspots in Finland. They estimated the opportunity cost of conservation from a survey of forest owners in Finland that asked them about enrolling land for species protection in return for a payment. They combined landowners' assessments of their forests with data on species habitat to derive estimates of the biological benefits and costs of enrolling a site through a payment program or through a top-down mechanism. They found that a fairly simple program that allows owners to enroll land can achieve conservation targets in a cost-effective manner. They also found that the incentive payment

USDA Forest Service

The potential for the NIPF ownership to expand carbon storage in forests is large relative to some other ownerships.

program performed better than a species-only site selection approach, and was nearly as cost-effective as the hypothetical solution to the conservation program they used as a benchmark.

To summarize, a review of the literature on the effects of incentives on NIPF landowner behavior leads to sometimes conflicting conclusions. Most of the literature agrees that the response of NIPF owners to various incentives depends critically on the ownership motives of the landowner, as well as other landowner and property characteristics. However, there is much less agreement on the effectiveness of different types of incentives. If the goal is to promote forest management or sustainable forestry, then the consensus seems to be that financial incentives alone are largely ineffective in promoting the desired behavior. The noteworthy exception seems to be cost-sharing incentives. In addition, given landowners' lack of knowledge about incentives, information and technical assistance may be effective as well. An additional noteworthy aspect of this literature is that surveys reveal that landowners may often carry out the desired management activities even in the absence of incentives. This highlights the issue of whether there is additionality in incentives programs, which will be discussed later.

If the goal is to promote biodiversity and habitat conservation, the existing evidence seems to suggest that financial incentives can be effective in promoting desirable management choices by NIPF owners. Furthermore, there is evidence that in this case, cost sharing may again be the exception, with at least one study (Langpap 2006) finding that it may not be effective. There is also some evidence in this literature that landowners whose main ownership objective is not timber production may be willing to carry out some of the desired management activities at lower cost, which again raises the issue of additionality.

This divergence in results suggests that it is difficult to make generalizations about the impacts of incentives on NIPF owner behavior. Incentive effectiveness may depend on the specific policy goal and may differ across regions and over time, as well as with the characteristics of landowners. Thus, it might not be adequate to extrapolate from the existing literature to infer how NIPF owners may respond to incentives in the specific context of carbon sequestration.

It is difficult to make generalizations about the impacts of incentives on NIPF owner behavior. Incentive effectiveness may depend on the specific policy goal and may differ across regions and over time, as well as with the characteristics of landowners.

Incentives for Carbon Sequestration in Agriculture

To complement our review of the literature on carbon sequestration in forests, we provide a review of the recent economic literature on incentives for carbon sequestration in agriculture. Many of these papers examine how incentives affect agricultural landowners' decisions to adopt agricultural practices that increase carbon sequestration in agricultural soils.

Pautsch et al. (2001) examined different government and market-based instruments to increase soil carbon sequestration through increased adoption of conservation tillage. They used a model of the farmer's adoption decision and discussed the design of subsidy and market-based instruments, focusing on the institutions and practices surrounding agricultural policy. Then they used Natural Resources Inventory data (USDA Natural Resources Conservation Service) to estimate the probability that farmers adopt conservation tillage and combined the estimates with physical models of carbon sequestration to estimate and compare the costs of implementing a variety of subsidy and market-based schemes. They found that the lowest payment cost can be achieved using a price-discriminating subsidy (which varies for different private landowners as determined by market processes), although such an approach would not be politically viable. They found that a single subsidy is less efficient, but would have lower political and administrative costs. Their results indicate that payments associated with a price-discriminating subsidy would be as little as one-fourth the cost of a single-price subsidy. They also found that costs would be much higher when payments have to be made to all farmers employing conservation tillage rather than just those adopting in response to the subsidy.

Feng et al. (2000) used a dynamic model that includes both emission reductions and sequestration as sources of GHG reductions to investigate the value of carbon sequestration in agriculture, and demonstrated that this value is only a fraction of the value of emission abatement unless the sequestration is permanent. They also showed that to optimally reduce carbon emissions, sinks should be used as early as possible. They proposed three instruments to efficiently implement sinks: a pay-as-you-go system, a variable-length-contract system, and a carbon annuity account. In a pay-as-you-go system, landowners sell and purchase emissions based on the permanent reduction of carbon. In a variable-length-contract system, independent brokers would buy permits from sequestration sources and sell them to carbon emitters, thus providing the service of generating “permanent” carbon reductions from a series of independent temporary reductions. In a carbon annuity account, a landowner who sequesters carbon is paid the full value of the permanent reduction in GHGs achieved, but the payment is put directly into an annuity account. The earnings, but not the principal, of the account can be accessed as long as the carbon remains sequestered. The principal is removed if the carbon is released. If the carbon is sequestered permanently, the landowner eventually earns all of the interest payments.

The relative inefficiency of per-hectare contracts varies spatially and increases with the degree of spatial heterogeneity.

Antle et al. (2003) developed a method to investigate the efficiency of alternative types of policies or contracts for carbon sequestration in cropland soils, taking into account the spatial heterogeneity of agricultural production systems and the costs of implementing efficient contracts. They described contracts being proposed for implementation in the United States and other countries that would pay farmers for adoption of specified practices (per-hectare contracts), as well as more efficient contracts that would pay farmers per tonne of soil carbon sequestered, and show how to estimate the costs of implementing the more efficient contracts. They conducted a case study of a major agricultural region in the United States (the Northern Plains) that confirms that the relative inefficiency of per-hectare contracts varies spatially and increases with the degree of spatial heterogeneity. The results also show that per-hectare contracts are as much as five times as costly as per-tonne contracts. Measurement costs to implement the per-tonne contracts are found to be positively related to spatial heterogeneity but are estimated to be at least an order of magnitude smaller than the efficiency losses of the per-hectare contract for reasonable error levels. This finding implies that contracting parties could afford to bear a significant cost to implement per-tonne contracts and still achieve a lower total cost than would be possible with the less efficient per-hectare contracts.

Hartell (2004) asked what payments must be received by agricultural producers to induce them to supply certain quantities of beneficial nonfood outputs. In the empirical application, this study valued carbon sequestration in agricultural soil through the adoption of no-till cultivation using mathematical programming to derive representative price schedules. The shadow price (or marginal cost) schedule for various levels of yearly carbon sequestration is derived by iteratively imposing minimum quantity constraints. The results and derived supply schedules show that the level of monetary incentive and total budgetary outlays required to induce multi-functional carbon sequestration might be lower than anticipated, but this depends importantly on assumptions about the level of producer risk aversion.

Pendell et al. (2007) studied the carbon credit incentives needed to motivate adoption of no-tillage and manure applications to enhance soil carbon sequestration in corn production in Kansas. They examined the net returns from continuous corn production using conventional and no-tillage practices with nitrogen fertilization from either ammonium nitrate or beef cattle manure for sequestering carbon with and without incentives. The results indicate that no-tillage and manure fertilization increase carbon sequestration. Carbon credits or government program incentives are not required to entice risk-averse managers to use no-tillage, as no-till systems have the highest net returns and greatest sequestration rates, but are required to encourage manure use as a means of sequestering additional carbon even at historically high nitrogen prices.

Graff-Zivin and Lipper (2008) developed a farm-level model of the decision to adopt a technology that generates soil carbon sequestration co-benefits. They explored incentives of poor farmers to adopt production systems that increase soil carbon sequestration, focusing on the increased agricultural yield risk associated with the transition to a new farming system. They used a household dynamic optimization model of the decision to adopt conservation agriculture and supply soil carbon sequestration, where farmers optimize expected utility of profits

Emily Jederlinich

Forest carbon sequestration is a relatively new objective for some NIPF owners, and carbon markets are just emerging.

from agriculture and carbon sequestration. They considered two distinct impacts on agricultural productivity: the technological effects of the new system and the productivity effects of changes in soil carbon on agricultural output. Comparative static results indicate that increases in the soil carbon sequestration price and the discount rate have unambiguous impacts on equilibrium soil carbon levels, the former leading to higher and the latter to lower carbon sequestration levels. Increases in the price of agricultural output and risk aversion have ambiguous impacts, depending on the relative strength of the productivity and technology effects. The paper concludes with a discussion of designing soil carbon payment mechanisms to benefit low-income farmers. The results suggest that pooling soil carbon sequestration payments and devising other group schemes to help farmers share risk offer the potential of providing an effective way of stimulating agricultural development and poverty reduction through climate change mitigation initiatives.

Finally, some papers report similar research in other countries. Antle and Diagana (2003) argued that incentive mechanisms for carbon sequestration in agricultural soils could contribute to alleviating rural poverty, enhancing agricultural sustainability, and mitigating GHG emissions. They assessed the role that soil carbon sequestration could play in addressing soil degradation problems in developing countries. They concluded that emerging policies to mitigate GHG emissions, such as global carbon markets or the Kyoto Protocol's Clean Development Mechanism, and other international and national policies, could be used to create incentives for farmers in developing countries to adopt practices that benefit them and simultaneously help reduce GHG emissions. However, they cautioned that several significant challenges, such as lack of well-functioning legal and financial institutions or poorly defined property rights, would have to be overcome before poor farmers in developing countries would be able to take advantage of these opportunities. They cited a carbon loan that provides financing for conservation investments to be paid back by generating carbon credits as an example of an institutional innovation that could help farmers overcome adoption barriers caused by imperfect capital markets.

Weersink et al. (2005) assessed the extent to which agriculture can be part of meeting Canada's Kyoto commitments through direct means induced by a carbon market or indirectly through the voluntary adoption of GHG-reducing practices, such as reduced tillage. They considered three policy mechanisms that could affect the extent to which net GHG emissions are reduced: moral suasion, an offset market, and an inventory accounting system. They presented a conceptual model of a supply curve for carbon credits and reviewed the empirical evidence on the factors influencing this supply curve, including the potential price of carbon, the costs of a contract, and the opportunity cost of sequestration and emission reduction activities relative to their emission reduction potential. Based on the data, they argued that involvement by farmers in the offset market will be limited, but that there will be net GHG emission reductions from agriculture through voluntary adoption partially prompted by government extension efforts. They found that voluntary adoption of beneficial management practices will be the main way by which Canadian farmers will cut GHG emissions. Participation in the carbon-offset market will be limited owing to relatively low prices offered by large emitters for carbon credits, discounts applied to those prices for temporary sequestration, transaction costs and risk premiums associated with carbon contracts, and the low elasticity of supply of carbon dioxide abatement. Nevertheless, they argued that Canadian agriculture is likely to contribute significantly to reducing emissions through adoption of zero tillage and reduced fertilizer use, but that this contribution will respond mainly to personal economic objectives of farmers rather than to direct incentives through the offset program.

Canadian agriculture is likely to contribute significantly to reducing emissions through adoption of zero tillage and reduced fertilizer use, but this contribution will respond mainly to personal economic objectives of farmers rather than to direct incentives through the offset program.

Antle and Stoorvogel (2008) explored the impacts of payments for agricultural soil carbon sequestration on poverty of farm households and the sustainability of agricultural systems. They used a theoretical model combined with case studies in Kenya, Peru, and Senegal. They found that carbon contracts are likely to increase rural incomes and reduce the rate of soil carbon loss. This suggests that carbon payments could have a positive impact on sustainability while also reducing poverty.

To summarize, these papers suggest that, as in the case of forest owners, incentives can be effective in eliciting management decisions that increase carbon sequestration from agricultural landowners. Some of these papers suggest that the issue of additionality may be relevant when implementing incentive programs in agriculture as well. We turn to a discussion of this issue in the next section.

Additionality

Within the framework of incentive programs for provision of ecosystem services, additionality generally refers to whether agents supplying ecosystem services in exchange for an incentive are being compensated for services they would not have provided in the absence of the incentive. The concern is whether the incentive actually elicits additional provision of the ecosystem service.

For instance, Murray et al. (2007) addressed how credits generated by agricultural soil carbon sequestration (ASCS) activities can be adjusted to account for the phenomena of permanence, leakage, and additionality (PLA). The underlying objective is to understand and quantify what the net carbon benefits of an ASCS project are once we account for the fact that (1) the sequestered carbon may be stored impermanently, (2) the project may displace emissions outside the project boundaries, and (3) the project's carbon sequestration may not be entirely additional to what would have occurred anyway under business-as-usual conditions. Murray et al. examined the conceptual and policy rationale for adjusting ASCS credits for PLA, described methods for making these adjustments, and presented some evidence of the magnitude these effects could have on the economic returns. For the additionality of carbon sequestration, the authors introduced baseline estimation approaches and discussed data that could be used to develop baselines for the two main ASCS activities of interest and presented an example of a calculation of additionality.

Ferraro (2008) discussed contract design issues in the context of payments for environmental services and argued that reducing informational rents (costs generated by private information) is important in this context in order to maximize the amount of services obtained from limited budgets. He added that reducing informational rents also mitigates concerns about additionality. Wünscher et al. (2008) suggested a strategy to increase the additionality achieved by payments for environmental services (PES) by targeting benefit-cost ratios, incorporating multiple objectives, and explicitly considering the risk of environmental service loss as a spatial variable. They used data from Costa Rica to test the potential of the suggested targeting tool to boost the efficiency of that country's PES program in terms of additional environmental services per dollar spent. Their results suggest that the efficiency (benefit-to-cost ratio) of Costa Rica's PES program could be increased by using a targeting process that integrates spatial data rather than a targeting system based solely on priority areas.

The issue of additionality can be framed in the context of asymmetric information and contract design.

More generally, the issue of additionality can be framed in the context of asymmetric information and contract design (see Salanié 2005 or Bolton and Dewatripont 2005 for comprehensive modern treatments). There are a number of recent applications in the context of provision of ecosystem services. Gren (2004) examined efficient contracts for converting arable land into pollution sinks. Crépin (2005) examined the use of incentives to create wetlands in an asymmetric information context. She used a theoretical principal-agent model to show that contract choice can create welfare gains, and that the choice of contract depends on the distribution of the unobserved landowner type, on the elasticity of costs and benefits to wetland size changes, and on the costs of acquiring information. Sheriff (2009) developed an empirical methodology to use available data to develop beliefs regarding the technology and distribution of types in a regulated sector characterized by hidden information. He used the results to calibrate a second-best land conservation mechanism and evaluated its cost relative to simpler alternatives.

In the specific context of carbon sequestration in forests, the issue of additionality is mostly mentioned when discussing carbon offsets and the Clean Development

Mechanism (CDM) of the Kyoto Protocol in which developing countries can trade certified emission reductions to noncompliant countries in exchange for clean technologies and finance. Asuka and Takeuchi (2004) argued that poorly defined or relaxed additionality criteria may allow the over-generation of GHG reduction credits in excess of actual reductions. Beyond a certain threshold, these non-additional (business-as-usual) certified emissions can lead to economic losses for developing countries. Trexler et al. (2006) reviewed a variety of suggested additionality tests based, for instance, on whether GHG emissions are reduced below regulatory requirements, on whether a project uses technology it otherwise would not, or on whether the rate of return on a project would be too low without sequestration incentives, among other criteria. They then discussed the potential for false positives and negatives in these tests, and how they can be implemented as part of an additionality policy that seeks to reduce type I and type II errors.

Van Kooten and Sohngen (2007) defined additionality as getting credit only for carbon uptake above and beyond what occurs in the absence of carbon-uptake incentives. The additionality condition is satisfied if it can be demonstrated that a forest would be harvested and converted to another use in the absence of incentives. Carbon sequestered by incremental forest management actions, such as juvenile spacing, thinning, fire control, or fertilization would be eligible for carbon credits only if these activities would not otherwise have been undertaken. Afforestation projects satisfy additionality if they provide environmental benefits not captured

Ralph Alig

A key challenge in providing carbon offset credits will be determining a clear method for establishing a baseline of forest carbon sequestration.

Any climate agreement will require clear and mutually acceptable methods for determining baseline levels of carbon sequestration.

by the landowner (e.g., water quality, wildlife habitat) and that would not be undertaken in the absence of economic incentives.

Schneider (2009) examined how the additionality of CDM projects has been assessed in practice by systematically evaluating 93 projects. He concluded that current tools used for evaluating additionality need improvement, as they can be subjective and difficult to validate, based on undocumented assumptions, and lacking credibility. Maness (2009) argued that any climate agreement will require clear and mutually acceptable methods for determining baseline levels of carbon sequestration, so that carbon offset credits for projects are given only for the additional carbon sequestered beyond what would have been sequestered in the absence of the project.

Data Availability

In this section we briefly outline the data available on carbon sequestration in forests and NIPFs to assess the viability of conducting empirical studies on the effectiveness of incentives for carbon sequestration in NIPFs.

The USDA Forest Service, as directed by the Resources Planning Act, conducts a comprehensive national assessment of renewable resources, including forests. The *Forest Resources of the United States 2007* report includes, for example, data on forest-land area, net volume of timber, annual growth, mortality, and removals, and average area planted and harvested by region, subregion, state, productivity class, ownership group, and various other categories (Smith et al. 2009). It also includes estimated regional carbon storage and gross annual sequestration for the year 2000 and mean carbon per hectare by size-class and Environmental Protection Agency Level II Ecological Region for the years 2001–2006. These data, however, are highly aggregated, and do not provide information at the individual forest owner level.

The Forest Inventory and Analysis National Woodland Owner Survey, conducted by the USDA Forest Service (2009), asks forest owners about characteristics of their woodland, ownership motives, woodland use and management, intended future uses, and concerns, and obtains demographic information as well. This database provides considerable information at the individual landowner level, including age, gender, education level, and income of the forest owner; year, method, source of acquisition, and size of their holdings; harvesting and management activities conducted; conservation easements on the property; and participation in cost-sharing programs. However, it currently does not include any information on carbon or carbon management and sequestration.

The National Resources Inventory is a statistical survey of natural resource conditions and trends on nonfederal land in the United States, including privately owned lands, tribal and trust lands, and lands controlled by state and local governments, conducted by the USDA Natural Resources Conservation Service (2007). It provides nationally consistent statistical data on how these lands are used and on changes in land-use patterns for the period 1982–2003. Land-use categories analyzed include agriculture and forests.

In terms of data on incentives, Greene et al. (2009) collected a database of all federal, state, and private incentives available to NIPF owners. However, this database only provides a listing of available programs, not actual data on their implementation or effectiveness.

Summary and Concluding Comments

In this report, we have summarized the economics literature on incentives for carbon sequestration in forests, including afforestation of agricultural land and sequestration in existing forests and NIPFs, as well as the broader literature on effectiveness of incentives for forest management and biodiversity conservation on NIPFs and on carbon sequestration on agricultural land. We have also touched on the issue of additionality in the context of incentives for provision of ecosystem services and briefly reviewed relevant data sources.

Various types of incentives can effectively be used to promote afforestation and forest management practices to increase carbon sequestration.

There are large literatures on afforestation, carbon sequestration in forests, and incentives for NIPFs in general, but a very small literature on incentives for carbon sequestration on NIPFs. The broader literature on carbon sequestration and incentives suggest that various types of incentives can effectively be used to promote afforestation and forest management practices to increase carbon sequestration. However, the literature on NIPFs suggests that the effectiveness of incentives programs depends on a variety of factors, including the objective of the policy and landowner and property characteristics. Therefore, results from existing studies cannot necessarily be extrapolated to draw conclusions on the potential effectiveness of incentives for carbon sequestration on NIPFs. A separate study that focuses specifically on this topic would provide more reliable insights. Finally, our review suggests that the issue of additionality is relevant in this context, and that the design of any incentive scheme to elicit increased carbon sequestration needs to carefully consider how to minimize the costs caused by asymmetric information about landowner’s baseline behavior.

Acknowledgments

Prepared as part of a Joint Venture Agreement with the USDA Forest Service Pacific Northwest Research Station.

English Equivalents

When you know:	Multiply by:	To get:
Hectares (ha)	2.47	Acres
Metric tonnes	1.102	Tons
Megagrams (Mg)	1.102	Tons
Teragrams (Tg)	1,102,311	Tons

Literature Cited

Adams, R.M.; Adams D.M.; Callaway, J.M.; Chang, C.C.; McCarl, B.A. 1993. Sequestering carbon on agricultural land; social cost and impacts on timber markets. Contemporary Policy Issues. 11(1): 76–87.

Ahn, S. 2008. How feasible is carbon sequestration in Korea? A study on the costs of sequestering carbon in forest. Environmental and Resource Economics. 41(1): 89–109.

Alig, R.J. 2003. U.S. landowner behavior, land use and land cover changes, and climate change mitigation. Silva Fennica. 37(4): 511–527.

Alig, R.J.; Adams, D.M.; McCarl, B.A.; Callaway, J.M.; Winnett, S.M. 1997. Assessing effects of mitigation strategies for global climate change with an intertemporal model of the U.S. forest and agriculture sectors. Environmental and Resource Economics. 9: 259–274.

Alig, R.J.; Lee, K.J.; Moulton, R.J. 1990. Likelihood of timber management on nonindustrial private forests: evidence from research studies. Gen. Tech. Rep. SE-60. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station. 17 p.

Alig, R.J.; Mills, T.J.; Shackelford, R.J. 1980. Most soil bank plantings in the South have been retained; some need follow-up treatment. Southern Journal of Applied Forestry. 4(1): 60–64.

Amacher, G.S.; Conway, M.C.; Sullivan, J. 2003. Econometric analyses of nonindustrial forest landowners: Is there anything left to study? Journal of Forest Economics. 9(2): 37–164.

Antle, J.M.; Capalbo, S.; Mooney, S.; Elliott, E.; Paustiane, K. 2003. Spatial heterogeneity, contract design, and the efficiency of carbon sequestration policies for agriculture. Journal of Environmental Economics and Management. 46(2): 231–250.

Antle, J.M.; Diagana, B. 2003. Creating incentives for the adoption of sustainable agricultural practices in developing countries: the role of soil carbon sequestration. American Journal of Agricultural Economics. 85(5): 1178–1184.

Antle, J.M.; Stoorvogel, J.J. 2008. Agricultural carbon sequestration, poverty, and sustainability. Environment and Development Economics. 13: 327–352.

Asuka, J.; Takeuchi, K. 2004. Additionality reconsidered: lax criteria may not benefit developing countries. Climate Policy. 4(2): 177–192.

Bolton, P.; Dewatripont, M. 2005. Contract theory. Cambridge, United Kingdom: MIT Press. 740 p.

Brand, D. 1998. Opportunities generated by the Kyoto protocol in the forest sector. Commonwealth Forestry Review. 77: 164–169.

Conway, M.C.; Amacher, G.S.; Sullivan, J. 2003. Decisions nonindustrial forest landowners make: an empirical examination. Journal of Forest Economics. 9: 181–203.

Crépin, A. 2005. Incentives for wetland creation. Journal of Environmental Economics and Management. 50(3): 598–616.

De Jong, B.H.J.; Tipper, R.; Montoya-Gomez, G. 2000. An economic analysis of the potential for carbon sequestration by forests: evidence from southern Mexico. Ecological Economics. 33(2): 313–327.

Englin, J.; Callaway, J.M. 1993. Global climate change and optimal forest management. Natural Resource Modeling. 7: 191–202.

Feng, H.; Zhao, J.; Kling, C.L. 2000. Carbon sequestration in agriculture: value and implementation. Working paper 00-WP 256. Ames, IA: Iowa State University, Department of Economics. 31 p.

Ferraro, P.J. 2008. Asymmetric information and contract design for payments for environmental services. Ecological Economics. 65: 810–821.

Fletcher, L.S.; Kittredge, D.; Stevens, T. 2009. Forest landowners' willingness to sell carbon credits: a pilot study. Northern Journal of Applied Forestry. 26(1): 35–37.

Gillig, D.; McCarl, B.; Sands, R. 2004. Integrating agricultural and forestry GHG mitigation response into general economy frameworks: developing a family of response functions. Mitigation and Adaptation Strategies for Global Change. 9(3): 241–259.

Graff-Zivin, J.; Lipper, L. 2008. Poverty, risk, and the supply of soil carbon sequestration. Environment and Development Economics. 13(3): 353–373.

Greene, J.L.; Daniels, S.E.; Jacobson, M.G.; Kilgore, M.A.; Straka, T.J. 2009. Financial incentive programs for non-industrial private forest owners. http://www.srs.fs.usda.gov/econ/data/forestincentives/index.htm. (October 29, 2009).

Greene, J.L.; Daniels, S.E.; Kilgore, M.A.; Jacobson, M.G.; Straka, T.J. 2005. Research project C2 final report to the national commission on science for sustainable forestry. Existing and potential incentives for practicing sustainable forestry on non-industrial private forest lands. Washington, DC: National Council for Science and the Environment. 54 p. http://www.srs.fs.usda.gov/econ/data/forestincentives/ncssf-c2-final-report.pdf. (September 21, 2009).

Greene, J.L.; Kilgore, M.A.; Jacobson, M.G.; Daniels, S.E.; Straka, T.J. 2006. Existing and potential incentives for practicing sustainable forestry on non-industrial private forest lands. Proceedings, Southern Forest Economics Workshop. Raleigh, NC: Mississippi State University. 112 p.

Gren, I.M. 2004. Uniform or discriminating payments for environmental production on arable land under asymmetric information. European Review of Agricultural Economics. 31(1): 61–76.

Guthrie, G.; Kumareswaran, D. 2009. Carbon subsidies, taxes and optimal forest management. Environmental and Resource Economics. 43(2): 275–293.

Hardie, I.W.; Parks, P.J. 1996. Program enrollment and acreage response to reforestation cost-sharing programs. Land Economics. 72: 248–260.

Hartell, J.G. 2004. Pricing benefit externalities of soil carbon sequestration in multifunctional agriculture. Journal of Agricultural and Applied Economics. 36(2): 491–505.

Huang, C.H.; Kronrad, G.D. 2001. The cost of sequestering carbon on private forest lands. Forest Policy and Economics. 2(2): 133–142.

Im, E.; Adams, D.M.; Latta, G.S. 2007. Potential impacts of carbon taxes on carbon flux in western Oregon private forests. Forest Policy and Economics. 9(8): 1006–1017.

Jacobson, M.G.; Greene, J.L.; Straka, T.J.; Daniels, S.E.; Kilgore, M.A. 2009a. Influence and effectiveness of financial incentive programs in promoting sustainable forestry in the South. Northern Journal of Applied Forestry. 33(1): 35–41.

Jacobson, M.G.; Kilgore, M.A.; Straka, T.J.; Greene, J.L.; Daniels, S.E. 2009b. Financial incentive programs' influence in promoting sustainable forestry in the northern region. Northern Journal of Applied Forestry. 26(2): 61–67.

Joshi, S.; Arano, K.G. 2009. Determinants of private forest management decisions: a study on West Virginia NIPF landowners. Forest Policy and Economics. 11(2): 118–125.

Kilgore, M.A.; Greene, J.L.; Jacobson, M.G.; Straka, T.J.; Daniels, S.E. 2007. The influence of financial incentive programs in promoting sustainable forestry on the Nation's family forests. Journal of Forestry. June: 184–191.

Kline, J.D.; Alig, R.J.; Johnson, R.L. 2000. Owner incentives to protect riparian habitat. Ecological Economics. 33(1): 29–43.

Langpap, C. 2004. Conservation incentives programs for endangered species: an analysis of landowner participation. Land Economics. 80(3): 375–388.

Langpap, C. 2006. Conservation of endangered species: Can incentives work for private landowners? Ecological Economics. 57(4): 558–572.

Lubowski, R.; Plantinga, A.; Stavins, R. 2006. Land-use change and carbon sinks: econometric estimation of the carbon sequestration supply function. Journal of Environmental Economics and Management. 51: 135–152.

Maness, T.C. 2009. Forest management and climate change mitigation: good policy requires careful thought. Journal of Forestry. 107(3): 119–124.

Matta, J.R.; Alavalapati, J.R.R.; Mercer, D.E. 2009. Incentives for biodiversity conservation beyond the best management practices: Are forestland owners interested? Land Economics. 85(1): 132–143.

Mayer, A.L.; Tikka, P.M. 2006. Biodiversity conservation incentive programs for privately owned forests. Environmental Science and Policy. 9(7-8): 614–625.

Metz, B.; Davidson, O.; Swart, R.; Pan, J. 2001. Climate change 2001: mitigation. Cambridge, United Kingdom: Cambridge University Press. 752 p.

Murray, B.C. 2003. Economics of forest carbon sequestration. In: Sills, E.O.; Abt, K.L., eds. Forests in a market economy. Forestry Sciences. Dordrecht; Boston and London: Kluwer Academic. 72: 221–241.

Murray, B.C.; Sohngen, B.L.; Ross, M. 2007. Economic consequences of consideration of permanence, leakage and additionality for soil carbon sequestration projects. Climatic Change. 80(1-2): 127–143.

Nagubadi, R.V.; McNamara, K.T.; Hoover, W.L.; Mills, W.L., Jr. 1996. Program participation behavior of nonindustrial forest land owners: a probit analysis. Journal of Agricultural and Applied Economics. 28(2): 323–336.

Nagubadi, R.V.; Zhang, D. 2005. Determinants of timberland use by ownership and forest type in Alabama and Georgia. Journal of Agricultural and Applied Economics. 37(1): 173–186.

Nelson, E.; Polasky, S.; Lewis, D.J.; Plantinga, A.J.; Lonsdorf, E.; White, D.; Bael, D.; Lawler, J. 2008. Efficiency of incentives to jointly increase carbon sequestration and species conservation on a landscape. Proceedings of the National Academy of Sciences. 105(28): 9471–9476.

Newell, R.; Stavins, R. 2000. Climate change and forest sinks: factors affecting the costs of carbon sequestration. Journal of Environmental Economics and Management. 40(3): 211–235.

Parks, P.; Hardie, I. 1995. Least-cost forest carbon reserves: cost-effective subsidies to convert marginal agricultural land to forests. Land Economics. 71(1): 122–136.

Pautsch, G.R.; Kurkalova, L.A.; Babcock, B.A.; Kling, C.L. 2001. The efficiency of sequestering carbon in agricultural soils. Contemporary Economic Policy. 19: 123–134.

Pendell, D.L.; Williams, J.R.; Boyles, S.B.; Rice, C.W.; Nelson, R.G. 2007. Soil carbon sequestration strategies with alternative tillage and nitrogen sources under risk. Review of Agricultural Economics. 29(2): 247–268.

Plantinga, A.J.; Birdsey, R.A. 1993. Carbon fluxes resulting from U.S. private timberland management. Climatic Change. 23: 37–53.

Plantinga, A.J.; Birdsey, R.A. 1994. Optimal forest stand management when benefits are derived from carbon. Natural Resource Modeling. 8: 373–387.

Plantinga, A.J.; Mauldin, T.; Miller, D. 1999. An econometric analysis of the costs of sequestering carbon in forests. American Journal of Agricultural Economics. 81: 812–824.

Plantinga, A.J.; Wu, J. 2003. Co-benefits from carbon sequestration in forests: evaluating reductions in agricultural externalities from an afforestation policy in Wisconsin. Land Economics. 79(1): 74–85.

Raunikar, R.; Buongiorno, J. 2006. Willingness to pay for forest amenities: the case of non-industrial owners in the South Central United States. Ecological Economics. 56(1): 132–143.

Rodríguez-Vicente, V.; Marey-Pérez, M.F. 2009. Land-use and land-base patterns in non-industrial private forests: factors affecting forest management in northern Spain. Forest Policy and Economics. 11(7): 475–490.

Ross-Davis, A.L.; Broussard, S.R.; Jacobs, D.F.; Davis, A.S. 2005. Afforestation motivations of private landowners: an examination of hardwood tree plantings in Indiana. Northern Journal of Applied Forestry. 22(3): 149–153.

Salanié, B. 2005. The economics of contracts–a primer. 2nd ed. Cambridge, United Kingdom: MIT Press. 224 p.

Schneider, L. 2009. Assessing the additionality of CDM projects: practical experiences and lessons learned. Climate Policy. 9(3): 242–254.

Shaikh, S.L.; Sun, L.; van Kooten, G.C. 2007. Are agricultural values a reliable guide in determining landowners' decisions to create forest carbon sinks? Canadian Journal of Agricultural Economics. 55(1): 97–114.

Sheriff, G. 2009. Implementing second-best environmental policy under adverse selection. Journal of Environmental Economics and Management. 57(2): 253–268.

Siikamaki, J.; Layton, D.F. 2007. Potential cost-effectiveness of incentive payment programs for the protection of non-industrial private forests. Land Economics. 83(4): 539–560.

Smith, W.B.; Miles, P.D.; Perry, C.H.; Pugh, S.A. 2009. Forest resources of the United States, 2007. WO-GTR-78. Washington, DC: U.S. Department of Agriculture, Forest Service. 336 p. http://nrs.fs.fed.us/pubs/gtr/gtr_wo78.pdf. (June 2010).

Sohngen, B.; Brown, S. 2006. The influence of conversion of forest types on carbon sequestration and other ecosystem services in the South Central United States. Ecological Economics. 57(4): 698–708.

Sohngen, B.; Mendelsohn, R. 2003. An optimal control model of forest carbon sequestration. American Journal of Agricultural Economics. 85(2): 448–457.

Stainback, G.A.; Alavalapati, J.R. 2002. Economic analysis of slash pine forest carbon sequestration in the Southern U.S. Journal of Forest Economics. 8(2): 105–117.

Stavins, R.N. 1999. The costs of carbon sequestration: a revealed preference approach. American Economic Review. 89: 994–1009.

Trexler, M.C.; Broekhoff, D.J.; Kosloff, L.H. 2006. A statistically-driven approach to offset based GHG additionality determinations: What can we learn? Sustainable Development Law and Policy. 4(2): 30–40.

U.S. Department of Agriculture, Forest Service [USDA FS]. 2009. FIA national woodland owner survey. Washington, DC. http://fiatools.fs.fed.us/NWOS/tablemaker.jsp. (June 2010).

U.S. Department of Agriculture, Natural Resources Conservation Service [USDA NRCS]. 2007. National Resources Inventory—2007 Annual National Resources Inventory. Washington, DC. http://www.nrcs.usda.gov/technical/NRI/. (July 2010).

van Kooten, G.C.; Binkley, C.S.; Delcourt, G. 1995. Effect of carbon taxes and subsidies on optimal forest rotation age and supply of carbon services. American Journal of Agricultural Economics. 77(2): 365–374.

van Kooten, G.C.; Shaikh, S.L.; Suchanek, P. 2002. Mitigating climate change by planting trees: the transaction costs trap. Land Economics. 78(4): 559–572.

van Kooten, G.C.; Sohngen, B. 2007. Economics of forest ecosystem carbon sinks: a review. International Review of Environmental and Resource Economics. 1(3): 237–269.

Weersink, A.; Pannell, D.; Fulton, M.; Meyer-Aurich, A. 2005. Agriculture's likely role in meeting Canada's Kyoto commitments. Canadian Journal of Agricultural Economics. 53(4): 425–441.

Wünscher, T.; Engel, S.; Wunder, S. 2008. Spatial targeting for environmental services: a tool for boosting conservation benefits. Ecological Economics. 65: 822–833.

Zelek, C.A.; Shively, G.E. 2003. Measuring the opportunity cost of carbon sequestration in tropical agriculture. Land Economics. 79(3): 342–354.

Zhang, D.; Flick, W.A. 2001. Sticks, carrots, and reforestation investment. Land Economics. 77(3): 443–456.

Chapter 6: Public and Private Forest Ownership in the Context of Carbon Sequestration and Bioenergy Feedstock Production—Existing Research and Research Needs

by Eric M. White and Ralph J. Alig

Introduction

Forests continue to be recognized as having important roles in comprehensive policies and legislation to address climate change and increase renewable energy use. Much of the discussion around forests in the context of those topics has related to better understanding key physiological processes, developing projections of future markets for carbon or bioenergy, and examining the physical and economic outcomes of alternate policies. An important layer of complexity in considering the use of forests to address climate change and renewable energy development is forest-land ownership. In the United States, at the highest aggregation, forest ownership is divided into (1) those lands owned by private individuals and entities and (2) public lands. This briefing paper examines some of the existing literature to highlight considerations on how public and private forest ownership may influence the use of forests to address climate change and renewable energy development.

An important layer of complexity in considering the use of forests to address climate change and renewable energy development is forest-land ownership.

Forests constitute about 33 percent of the U.S. landscape and are common on both public and private land (Smith et al. 2009). In many places, forested landscapes are a mix of both public and private ownerships, and forests in both ownerships provide society with a variety of goods and services, from recreation opportunities to timber for the production of wood products. Public and private forests in the United States are also connected through a shared history. The federal National Forest System of today, which comprises the greatest extent of public forest land, was initially developed from forest reserves removed from settlement lands. Later, many national forests in the East were formed from lands that reverted to public ownership because of private land tax delinquency. In other cases, such as the "Oregon and California" lands, publicly owned forest lands were given to private companies in efforts to spur infrastructure development.

Just as forest landscapes can be made up of a diverse mix of forest owners, the conditions of forests in different ownership groups can differ widely. As shown later, public ownership forests tend to have a greater number of trees in older age classes relative to private forests. In general, public forests, relative to private

Few studies have examined the distinctions, linkages, and interactions between public and private forest land in the context of climate change and climate change policy.

forests, are more frequently managed for nonconsumptive uses such as recreation. However, public forests are managed by a variety of agencies at different levels of government with differing objectives, and the variety of conditions on public forests reflect those differences. Private forests are owned by individuals and families, traditional vertically integrated timber companies, and investment groups. Similar to public lands, private forests range from those intensively managed for timber production to forests where little if any active management occurs and objectives are primarily nonconsumptive uses of forest resources.

Today, the interactions and linkages between public and private forests range from the administrative (e.g., land parcel trades) to ecological (e.g., management of invasive species) to economic (e.g., impacts to the forest products sector) to legal (e.g., reciprocal fire suppression agreements). With the consideration of climate change policies, there is interest in considering how public and private forest lands may differ and complement one another in response to policy. Unfortunately, few studies have examined the distinctions, linkages, and interactions between public and private forest land in the context of climate change and climate change policy. The objectives of this research are to briefly describe the current areas and spatial patterns of public and private forest land ownership, to use existing knowledge to describe the differences and commonalities of public and private forest lands within the context of climate change, and to identify future research needs. Differences and commonalities of public and private forest lands are considered for four areas: carbon sequestration, bioenergy feedstock provisioning, timber harvest relationships, and decisionmaking.

Public and Private Forest-Land Ownership

The distribution and patterns of public and private forest land serve as a backdrop in considering the interaction of those ownerships in responding to climate change and energy policies. Forest land in the United States is 56 percent private and 44 percent public ownership (Butler 2009). Just less than half of publicly owned forests are managed by the USDA Forest Service. Reflecting the settlement patterns of the United States, forest lands in the East are most commonly private, and public forest lands are most common in the West (fig. 6-1). The general perception, particularly for the West, is that high-elevation forests are more commonly publicly owned, whereas lowland forests are often privately owned. Similar elevation patterns can also occur in the East, particularly along the Appalachian Mountains. Relative to historical levels, forest lands in the East have suffered the greatest reductions in area, largely because of conversion to agriculture and development (Smith et al.

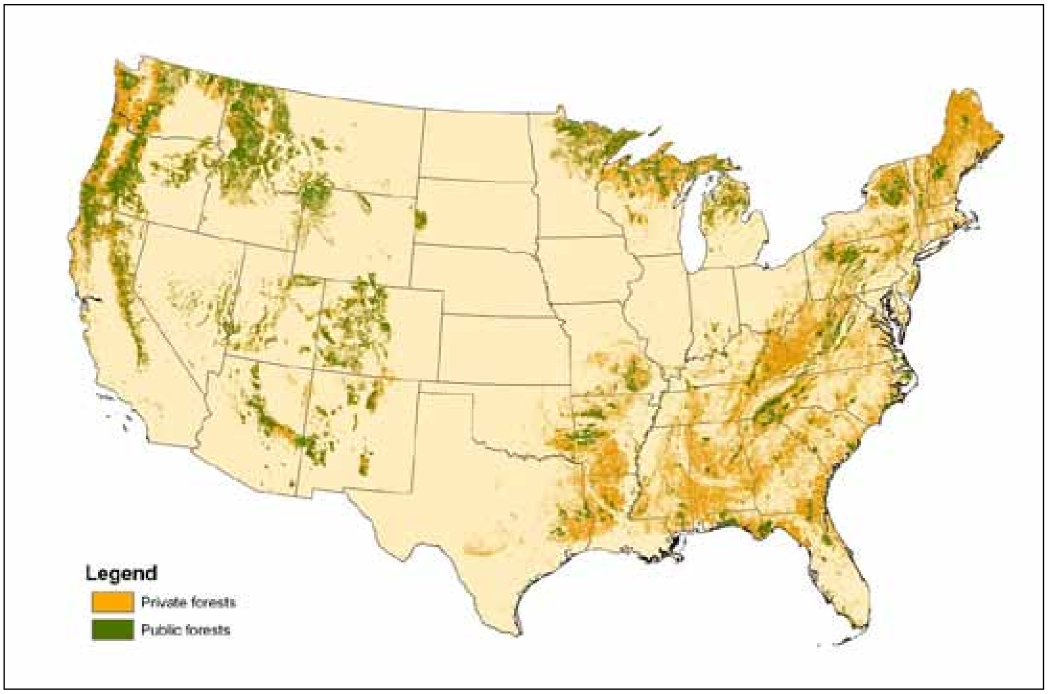


Figure 6-1—Public and private forest land in the United States.

2009). Eastern forests are also projected to experience the greatest amounts of future development (Stein et al. 2005, White et al. 2009).

The physical connections, via shared boundaries, between public and private forests are highlighted by considering two common local spatial arrangements of public and private forests. In many locales, public and private forests are held in fairly large blocks with a fairly clear delineation between the ownerships (fig. 6-2a). Conversely, in other areas, public and private forests are more intermingled, with forested ownerships contained in smaller contiguous blocks (fig. 6-2b). Because of east to west settlement of the United States, the former pattern is likely more common in the Western United States, whereas the latter is likely more common in the East—although both patterns occur in each region. The large-block arrangement suggests how public and private lands might be influenced differently by climate-related conditions that vary over space (e.g., temperature impacts that differ by elevation). Such a pattern of climate change could lead to disparate impacts by ownership group. The latter spatial arrangement is not as susceptible to disparate impacts to public and private owners from conditions that vary over space because the ownerships are intermingled. However, this intermingled arrangement highlights the degree to which forest management actions on one ownership may impact the other ownership and the extent to which the landscape provision of goods and services may be dependent on joint provisioning.

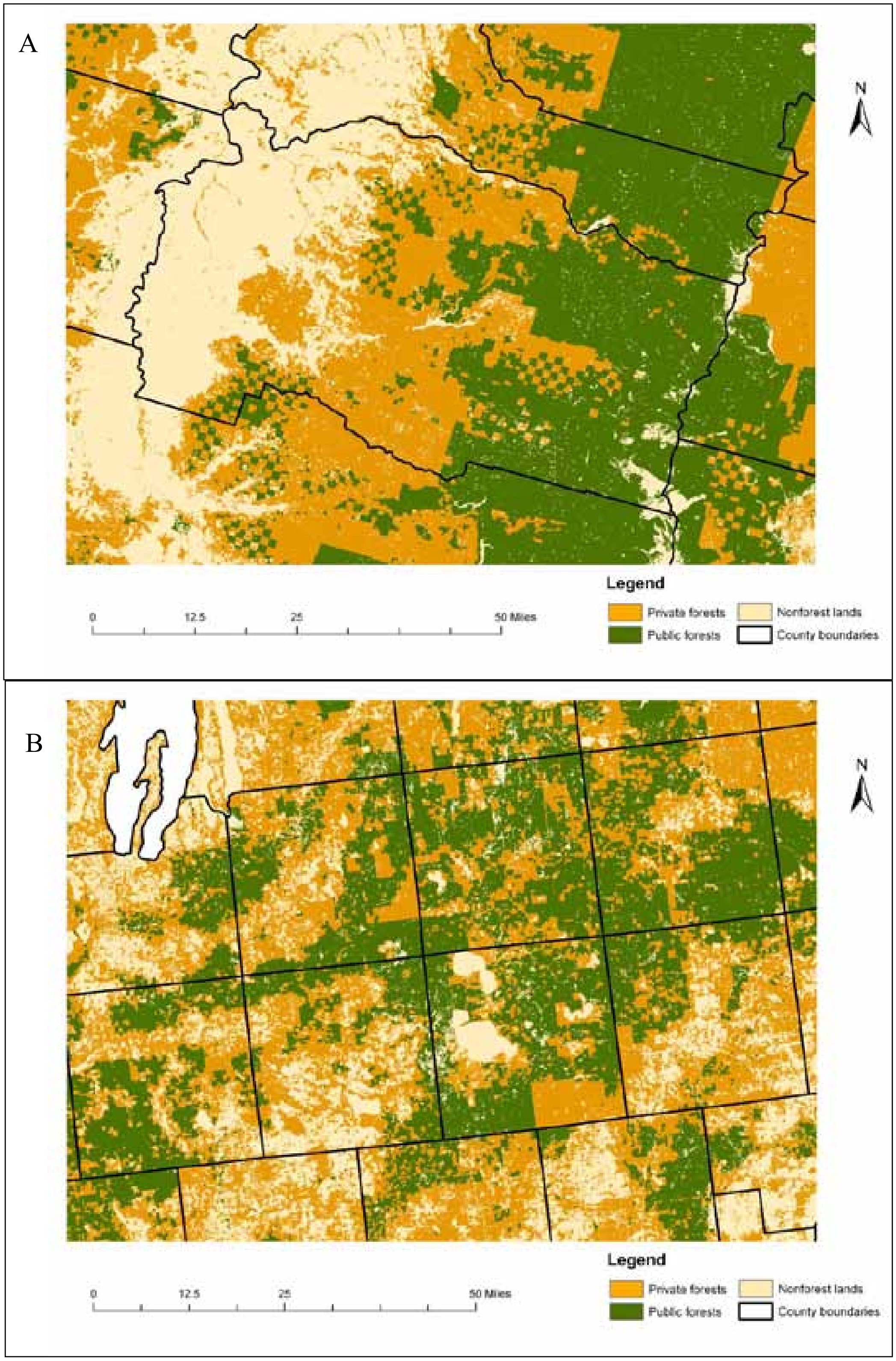


Figure 6-2—Two spatial patterns of U.S. public and private forest-land ownership, (a) Oregon and (b) Michigan. Data source: Theobald (2007).

Carbon Sequestration

The sequestration of emitted carbon by forests has been discussed in the context of both reducing existing atmospheric carbon and, within a program (cap and trade), to offset future carbon emissions produced by industry. In 2007, trees, agriculture soils, and other carbon sinks offset about 15 percent of U.S. greenhouse gas emissions (US EPA 2009). In addition to the ability to use forests to sequester emitted carbon, there has also been interest in avoiding the release of carbon stored in forests (in plant matter and in forest soils) that may occur because of land-use change, timber harvest, or disturbances such as fire. The release of carbon from individual forest stands reduces the flux of carbon sequestered to forests collectively (the net amount of carbon sequestered in a forest over a period) and if a release is severe enough, it can turn forests from carbon sinks to carbon sources. Because of the interest in carbon sequestration, the carbon fluxes and amount of carbon currently sequestered (i.e., carbon stocks) in public and private forests have been of interest.

Carbon Flux

Based on recent data, U.S. forests sequester an estimated 595 teragrams (Tg) carbon dioxide (CO_2) equivalent of carbon (1 Tg = 2.2 billion pounds) annually. Carbon sequestration in forests offsets about 8 percent of total U.S. greenhouse gas emissions (7150 Tg CO_2 equivalents) in 2007 (US EPA 2009). Since 1990, annual greenhouse gas emissions in the United States have been increasing. Over the same period, annual net carbon flux to forests has also increased, being about 17 percent greater than net carbon flux estimated for 1990. Most of this increase in flux can be attributed to increases in carbon sequestered in aboveground biomass, as a result of increased area of forest land and faster growing forests (USDA 2008). Currently, public forest lands have an aggregate carbon flux that is about 50 percent greater than the aggregate carbon flux on private forests (USDA 2008). Lower rates of flux on private forests likely result from greater land-use conversions and disturbance (including timber harvest) on private forests relative to public forests.

Currently, public forest lands have an aggregate carbon flux that is about 50 percent greater than the aggregate carbon flux on private forests.

Carbon Stocks

The forests of the United States account for an estimated 150 000 Tg CO_2 equivalent of carbon stocks (USDA 2008). Slightly less than 40 percent of these stocks are associated with forest soils and the remainder is in live and dead plant material (USDA 2008). Forest-land carbon stocks in the East (94 500 Tg CO_2 eq.) are about 70 percent greater than those in the West (55 300 Tg CO_2 eq.). The Northeastern States have slightly greater carbon stocks than the Southeastern States (USDA

2008). At the state level, carbon stocks in the live and dead biomass are greatest in California, Oregon, and Washington (fig. 6-3).

Currently, more forest-land carbon is stored in private forests (87 710 Tg CO_2 equivalent) than in public forests (62 132 Tg CO_2) (USDA 2008). Following the spatial pattern of forest ownership, private forest-land carbon stocks are greatest in the North and South regions (about 36 000 Tg CO_2 equivalents in each region) (fig. 6-4). Public forest-land carbon stocks are greatest in the Rocky Mountain and Pacific Coast regions. In the West, carbon stocks are greatest in public forests, mostly distributed in the older age classes. In the East, private forests account for the majority of sequestered carbon with stocks primarily, especially in the Southeast, in the younger age classes.

From some of the most recent data available, carbon stocks in forest industry and non-Forest Service public ownerships have been declining slightly; stocks in federal ownership have increased slightly; and stocks in nonindustrial private ownership have increased the most between 1987 and 1997 (table 6-1). These changes likely reflect both forest condition changes and changes in forest ownership (Birdsey and Lewis 2003). For forest industry lands, the greatest reductions were in the North. This likely reflects at least some divestiture of forest industry lands in that region in the early 1990s. The greatest gains for national forest land were in

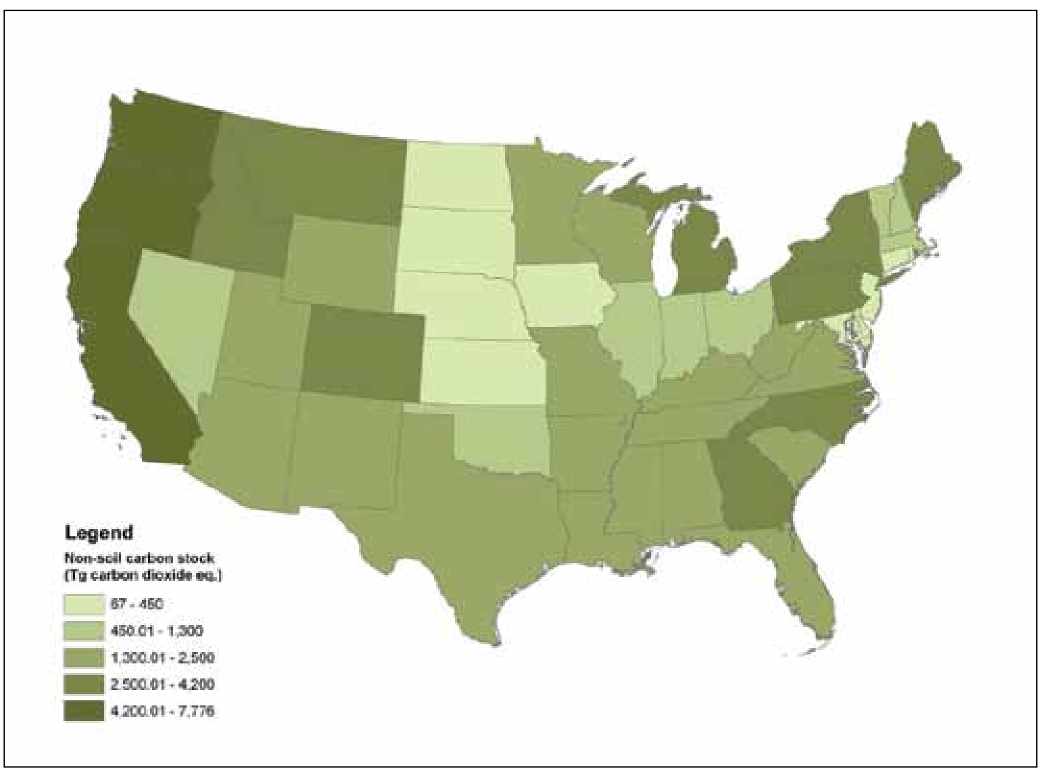


Figure 6-3—Estimated non-soil carbon stock on public and private forest land. Data source: USDA 2008.

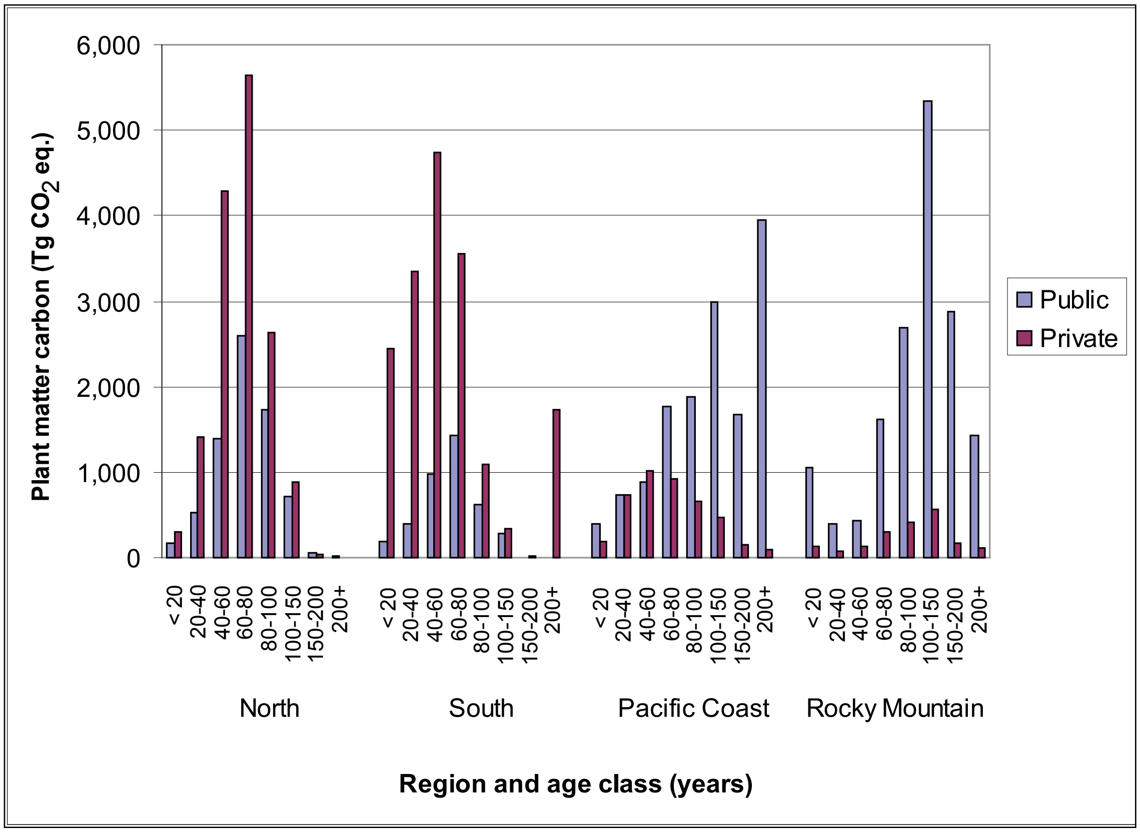


Figure 6-4—Distribution of forest carbon by region, ownership, and forest age class (Tg CO_2 is teragrams, equal to one million metric tons of carbon dioxide). Data source: USDA 2008.

the North. Nonindustrial private forest lands in the North and South experienced changes similar to each other.

Carbon is stored in the plant material of forests. When that plant material is harvested and converted to wood products (e.g., construction lumber), that carbon is then contained within the wood product (e.g., in newly constructed houses). This “fixed” carbon will remain in place until the wood products degrade, which often begins once they are discarded and subjected to decay or are burned. Carbon

Table 6-1—Carbon stock changes by ownership category and region, 1987–1997

Ownership	North	South	West	Total
		Percent		
National forest	5.9	4.7	4.5	4.6
Other public	7.4	19.2	-4.3	-1.0
Forest industry	-11.6	3.6	-1.9	-2.4
Nonindustrial private	7.1	6.5	5.2	6.4

Data source: Birdsey and Lewis 2003.

in wood products is not newly sequestered carbon but should be included when accounting for forest carbon stocks and carbon flux. In 2005, carbon stocks in existing harvested wood products accounted for an estimated 8700 Tg CO_2 equivalent of sequestered carbon. The net carbon pool in wood products increases by approximately 103 Tg CO_2 equivalent of carbon per year, counting carbon stored in new wood products and carbon released from discarded wood products (USDA 2008). Little work has been completed to differentiate wood product carbon by public or private sector forest source. However, it can be assumed that most of the hardwood products created in recent years are from timber harvested from private forests.

Declines in the amount of sequestered carbon over time result primarily from projected deforestation.

Projections of Private Carbon Stock

Carbon stocks in forested ecosystems on private lands are projected to decline over the next several decades under a business-as-usual case (fig. 6-5). Declines in the amount of sequestered carbon over time result primarily from projected deforestation. Forest lands in all private ownership types are subject to deforestation, but those lands owned by nonindustrial private forest owners have historically experienced the greatest amounts of deforestation for agriculture and urban development (Alig et al. 2003, 2010b). Because residential development typically includes some trees and perennial grasses in landscaping, deforestation for residential development

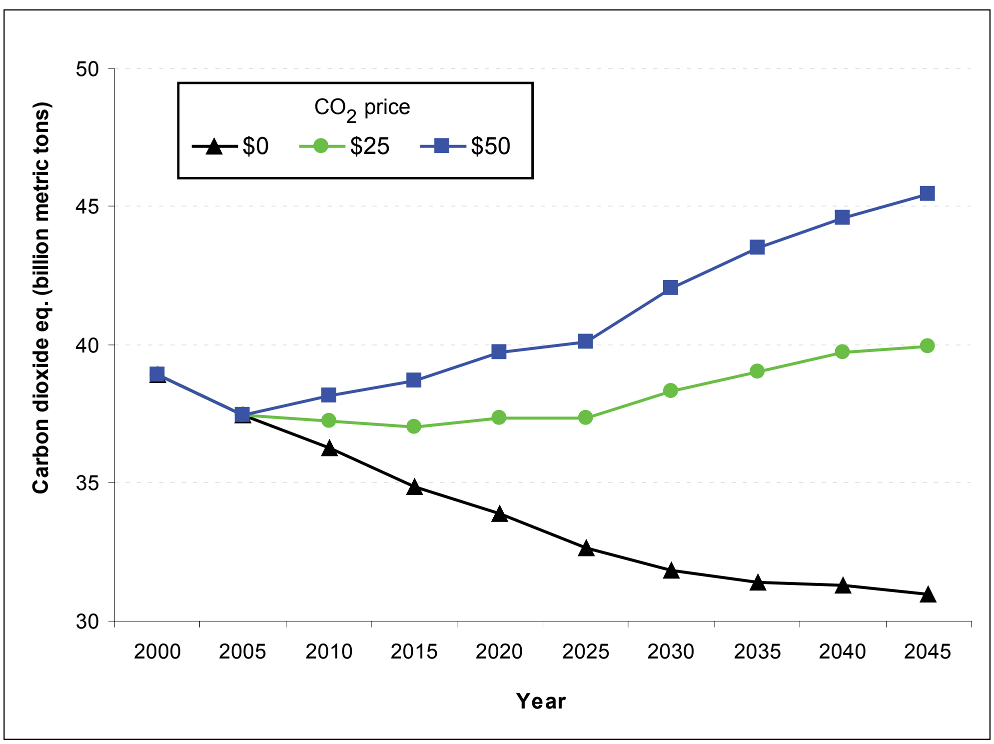


Figure 6-5—Projected carbon stocks in private forests under different carbon price scenarios. Adapted from Alig et al. (2010b).

has been assumed to have less negative consequences for carbon sequestration than agriculture land use (e.g., Cathcart et al. 2007). Under policies where carbon sequestered in trees is valued, forest ecosystem carbon stocks are projected to increase between the present and 2050. Carbon stocks are projected to increase through a combination of increased forest area (because of afforestation of agriculture land) and changes in forest management (e.g., lengthening harvest rotations and changing management intensities).

Projections of Public Carbon Stock

Non-soil carbon stocks on public lands are projected to increase between 2000 and 2050 (Smith and Heath 2004). Carbon stocks on national forest land are projected to be greatest in the Rocky Mountain and Pacific Northwest Regions. Aggregate carbon stocks on public lands not managed by the USDA Forest Service are projected to be highest in the North Central and Northeast regions. The amount of carbon sequestered annually on public lands is projected to slow slightly in the coming decades as public forests age and growth rates slow.

Depro et al. (2008) examined how public timber carbon sequestration might respond to changing timber harvest rates. Under business as usual, public forest lands in the Depro et al. study were projected to sequester, on average, 50 Tg of carbon annually between 2010 and 2050. National forests accounted for more than 60 percent of the projected sequestration. Because of projected aging of public forests, rates of carbon sequestration were projected to decrease over time under all harvest scenarios considered. Reducing public land timber harvest hypothetically from approximately 425 million m^3 (15 billion ft^3) per decade to near zero increased carbon stored in public forests by 40 to 50 percent. Conversely, increasing annual public forest harvest levels by approximately 566 million m^3 (20 billion ft^3) per decade, to the harvest levels projected in the 1989 national timber assessment (Haynes 1990), decreased the amount of carbon sequestered in public forests by 50 to 80 percent per decade. Even accounting for carbon sequestered in wood products, under a high timber harvest scenario, public forests were projected to become carbon sources rather than carbon sinks. It should be noted that the Depro et al. (2008) analysis included baseline levels of public forest disturbance (e.g., wildfire, insects, and disease) in the growth and yield estimates. As a first-generation study, the analysis of Depro et al. (2008) did not incorporate a private sector response to changes in public land timber harvest. Increased private harvesting could offset some of the additional carbon sequestered on public land under a no-public-harvest scenario. However, the vast majority of timber production in the United States is already associated with private lands (Adams et al. 2006).

Bioenergy Feedstock Provisioning

Increased use of biomass for the production of renewable and low-carbon electricity and liquid fuels may be an outcome of comprehensive climate legislation and may be an outcome of carbon emissions regulation. Woody biomass for bioenergy production can be obtained from a number of feedstocks from public and private forest lands (see White 2010 for a description). The current Renewable Fuels Standard in the Energy Independence and Security Act of 2007 and draft language in some proposed legislation is typically interpreted as not recognizing, for renewable electricity credit or carbon offsets, biomass from public lands and biomass from forests of certain characteristics. That topic is outside the scope of this paper and interested readers can refer to WFLC (2009).

Woody biomass already composes a significant component of current U.S. renewable energy consumption. Residues from timber mills are currently responsible for much of the bioenergy produced from biomass (see White 2010 for a discussion). In the future, under increased demand for woody biomass, material from timber harvest residues and hazardous fuel reduction on both public and private forests may be important feedstocks. These two feedstock sources are discussed in the next sections. In addition to harvest residues and hazard fuel reduction material, use of biomass feedstocks from short-rotation woody crops and other residues and wastes (e.g., construction debris) will also likely increase; however, feedstocks from these sources are associated almost exclusively with private forests or private companies.

Private forest lands would likely supply a greater volume of harvest residues than public lands under increased bioenergy feedstock demand.

Timber Harvest Residues

In 2006, approximately 130 million m^3 (4.6 billion ft^3) of residues were generated from timber harvesting activities (Smith et al. 2009). This woody material, left onsite, translates into approximately 58 million dry tonnes (64 million dry tons) of biomass material. Not all of this material would be technically or economically available for bioenergy production. In one study of harvest residue biomass, Gan and Smith (2006) estimated that about 32 million dry tonnes (36 million dry tons) of residues on public and private lands would actually be available under likely market conditions—enough material to generate 67.5 terawatt hours (TWh) of electricity—about 1.7 percent of the electricity available to the grid in 2007 (US DOE 2010).

Based on current timber harvest patterns, private forest lands would likely supply a greater volume of harvest residues than public lands under increased bioenergy feedstock demand. In 2002, less than 10 percent of the timber harvested in the United States came from public lands (Adams et al. 2006). Aggregate public land timber harvests were greatest in the North Central region and areas in the states of

Ralph Alig

Public timberlands represent about 20 percent of the U.S. timberland area and about 30 percent of the timber volume, with substantial opportunities to sequester additional carbon.

Oregon and Washington west of the crest of the Cascade Mountains. Private timberland harvest was greatest in the South Central region (although private harvest was also high in the Southeast, Northeast, and Pacific Northwest). The use of timber harvest residues for bioenergy feedstock would offer private landowners another revenue stream from harvests. However, the costs of handling and transporting biomass is high and feedstock values low, so revenues from this additional product stream are likely to be modest and in many cases not a driving factor in private forest management activity. For example, in Minnesota, the additional value from using timber harvest residues for bioenergy in hybrid poplar stands harvested for pulp was estimated to be positive but minor (Schmidt 2006).

Slightly more residues are generated from hardwood harvest than softwood harvest (Smith et al. 2009), but much of the private timber harvested currently is from southern softwoods. The disparity in residue production between forest types could push residue usage into northern regions where hardwoods are more common. In northern regions, public land owned by states and other public entities is common and is often used for timber production (e.g., in the northern Great Lakes Region). Thus, public forest lands in the North not managed by the USDA Forest Service may be well positioned for the provision of timber harvest residues. Hardwood harvest in the North has declined relative to the early 1990s but is still greater than that of the 1970s (Adams et al. 2006).

There are some concerns about reduction in available site nutrients because of timber harvest residue removal, but the literature is not currently definitive (Carter et al. 2006, Walmsley et al. 2009). However, guidelines that are being developed in some states for removing harvesting residue from logging sites (e.g., Minnesota FRC 2007) may help to mitigate any potential site productivity declines. If removal of harvest residues resulted in widespread private forest productivity declines, private land managers would need to increase management intensity (e.g., fertilizing, use of improved planting stock) to maintain the same levels of productivity. Decreased productivity and increased management intensity on private forest lands could result in greater pressure on public forest lands and nonindustrial private forests for provision of forest goods (e.g., timber) and services (e.g., wildlife habitat, clean water). This greater reliance on public lands could be complicated by the patchwork of public and private ownership, with the potential for difficulty in providing some services because of fragmented ownership (e.g., fig. 6-1).

Hazard-Fuel Reduction

Because of increased attention to large and costly wildland fires that cause damage to private property, there have been calls to implement widespread activities to reduce hazardous wildfire fuel loads on public and private lands. It is often suggested that small-diameter hazard-fuel material could be a key biomass feedstock for renewable energy production. Skog et al. (2006) quantified acres and volumes of material that could be removed from timberland in the Western United States under several hazard-fuel treatment scenarios. The amount of hazardous fuel volume on public lands far exceeds that on private lands. Under a representative scenario, the volume of biomass that could be removed from private lands was slightly less than half the volume that could be removed from public timberlands (table 6-2). In general, the Western States with the greatest forest areas have the greatest potential volumes of hazard material.

A significant challenge to widespread implementation of hazard-fuel reduction is the cost of treatment. Skog et al. (2006) found that no treatment scenario was profitable if all of the removed material were sold only for bioenergy. If larger stems that were removed could be sold for pulpwood and sawtimber, some hazard-fuel treatment scenarios resulted in positive net revenues. Subsidies of $22/green tonnes ($20/green ton) of chips in addition to the ability to sell larger stems for pulpwood and sawtimber allowed more treatments to become economically feasible. In addition to affecting economic feasibility, Skog et al. (2006) also found that allowing harvesting of larger stems as part of hazard-fuel reduction resulted in programs better able to meet targets for reducing the susceptibility of forest stands to wildfire crowning and torching. It is unclear whether hazardous-fuel removal programs on public lands would be required to cover costs. Where hazardous-fuel reduction could be completed on private forests (many lands that likely do not have management

Table 6-2—Volume of material removed under a simulated uneven-age hazard fuel thinning regime by timberland ownership

State	Private	National forest	Other federal	State and local	Total
		Million oven dry tons			
Arizona	2.0	6.9	0.0	0.0	8.9
California	50.0	65.1	0.6	1.7	117.4
Colorado	5.9	8.9	2.4	0.2	17.4
Idaho	13.2	35.7	3.5	5.3	57.7
Montana	14.8	38.2	3.2	2.6	58.9
Nevada	0.2	0.0	0.0	0.0	0.2
New Mexico	3.3	10.7	0.0	1.1	15.0
Oregon	16.3	28.3	8.4	2.1	55.1
South Dakota	0.0	1.1	0.0	0.0	1.1
Utah	1.6	3.9	0.3	1.1	6.9
Washington	12.8	18.4	1.1	6.4	38.8
Wyoming	2.3	3.1	1.8	0.1	7.3
Total	122.3	220.2	21.3	20.8	384.6

Adapted from Skog et al. 2006.

plans), it seems unlikely such treatments would be implemented if costs could not be covered either by revenue from selling the material or via subsidy. As such, it is likely that public forests would be the focus of a widespread hazardous-fuel reduction program and associated biomass for bioenergy. However, the ownership(s) on which hazard-fuel reduction occurs will likely be most influenced by the specific focus (public or private lands) of any potential future hazardous-fuel reduction program.

In general, changes in timber harvest levels on public lands lead to countervailing changes in harvest on private lands.

Public and Private Timber Harvest Linkages

In general, changes in timber harvest levels on public lands lead to countervailing changes in harvest on private lands (Adams et al. 1996). However, because only some of the material harvested under a hazard-fuel reduction program would have value for traditional commercial timber production, the relationship between public land hazard-fuel reduction programs and private forest harvest is not entirely clear. Building on the work reported in Skog et al. (2006), Ince et al. (2008) projected that widespread hazardous-fuel treatment on western public lands would result in "significant displacement" of timber that would have been harvested from private and state-owned forest land (up to 30 percent less than baseline projections). Increased production of timber from a hazardous-fuel reduction program was also projected to reduce stumpage prices for western softwood timber by up to approximately 40 percent in 2015 relative to the baseline. Hazard reduction programs that removed stems of a variety of sizes yielded greater reductions in timber harvest and stumpage price than treatments that thinned only the smallest stems. Because public timber output increases and prices fall, a hazardous-fuel reduction program was projected to decrease the welfare of timber producers but increase the welfare of wood product consumers. Research completed by others (Abt and Prestemon 2006, Keegan et al. 2004) is consistent with the findings of Ince et al. (2008).

Adams and Latta (2005) examined how a hypothetical federal forest-land restoration program would influence the private forest sector in a rural community. In their study, small material harvested during the restoration was left onsite and larger material was sold for timber production. Logging contractors completed the treatments and were provided a variety of subsidies. The setting for the Adams and Latta (2005) study was eastern Oregon, where timber milling capacity and timber harvest levels have been in decline. In this setting, a hazard-fuel reduction program increased the amount of timber harvested in the region with small reductions in timber prices, except in the most generous subsidy program. Implementation of a hazard-fuel reduction program slowed the projected reduction in timber mill capacity within the region, although over the long term, capacity was projected to be nearly as low, or lower, than in the base case with no restoration program. Adams and Latta (2005) found both reductions and increases in the values of various types of private timberland owing to changes in the output of the region's industry. Timber producers (primarily private forest owners) suffered welfare losses under the subsidy program but consumers (timber mills) gained welfare, as expected. Based on model output, timber producers were also found to change their management

Eric White

Public timberlands have been managed for multiple uses and ecosystem services including timber, range, wildlife habitat, watershed protection, recreation, and visual amenities, with increasing consideration given to carbon sequestration as another ecosystem service.

regimes when the fuel reduction program was in place—slightly reducing the share of acres in uneven-age management and increasing the share of acres in even-age management. Impacts on local mills and the forest sector from the fuel reduction program were sensitive to the type of subsidy program (including no subsidy) offered to logging contractors.

Goals and Decisionmaking

Differences in management goals and the process of decisionmaking and implementation is a factor in how public and private forest owners interact with one another and respond to climate change. Within the confines of regulations, private landowners can manage their lands for goals they identify. These goals may be narrow or broad and focused on production of goods (e.g., timber) or nonconsumptive uses (e.g., aesthetics). In theory, a sufficient number of private landowners are able to influence the activities on neighboring private land via the marketplace. In general, private landowners can influence public land policies only through the policy process. Looking forward, the general expectation is that private forest-land owners have the ability to respond rapidly and optimally to changes in environmental conditions and new climate change policies and opportunities. For example, in response to changing forest-growing conditions or changing policies, private landowners may adapt and choose to plant tree species better suited to the new conditions or to change management activities on existing forests (see chapter 1 for a discussion).

Private lands would appear to be well positioned to respond quickly to climate change and climate change policies. However, private forest landowners are currently experiencing widespread changes in ownership away from traditional timber industry and toward investment group ownership and small parcels owned by numerous private individuals. Pressure on forests for urbanization and residential development has been high and is projected to continue in the coming decades (Alig et al. 2003, 2010b; Stein et al. 2005). The timber industry is also currently faced with decreased demand for wood and paper products. These factors may make it challenging for private forest owners to optimally respond to climate change and such things as carbon markets. For example, it may be very difficult to get private individual forest owners to increase carbon sequestration rates given that these owners have a diverse array of management goals and often have no management plan (Birch 1997).

Public forests, particularly federal forests, are typically managed for a broad suite of goals that involve a number of consumptive and nonconsumptive uses. However, some public forests, such as those managed by local utilities, are managed primarily for a specific set of goals, such as the provision of water (e.g., Seattle Public Utilities 2008). Regardless, the goals for public forest lands and the management actions to achieve those goals are generally developed through processes involving numerous stakeholders. Reflecting the diversity of goals as well as the policymaking process, public forest lands are perceived to be slow to respond to changing environmental and market conditions. Generally, public land agencies influence management actions on private lands by providing information and technical assistance (e.g., the State and Private Forestry branch of the USDA Forest Service).

Federal public forests are less constrained than private forests by market conditions in adopting new management directions, although management decisions for many state-owned forest lands do explicitly incorporate market conditions. Some agencies, including many in the West, are responsible for providing revenue from forest resource uses such as timber harvest or grazing allotments to support state services. In addition to timber production, state and local forest agencies typically also provide a number of nonconsumptive resource opportunities. Federal forest agencies have recently exhibited, through significant increases in fire and fuels management capacity, that the government is able to make fairly rapid changes in response to perceived threats in at least some cases. However, in many other cases, changes to federal land policy continue to be slow to occur.

Needed Research

Research is needed to better understand how public and private forest lands are likely to interact in the provision of feedstocks for bioenergy.

The interactions between public and private forest lands in the context of climate change and climate change policies, as well as the provision of other ecosystem services, have not been well studied. However, these interactions are important in considering likely future conditions and the potential impacts of new policies. Several opportunities for additional lines of research are presented below. We focus on the United States here, but there are also opportunities to gain insights by comparing forest resource conditions, carbon sequestration, institutional arrangements, forest ownership, and social issues across regions of the world (e.g., Alig et al. 2006).

Research is needed to better understand how public and private forest lands are likely to interact in the provision of feedstocks for bioenergy. This might help to identify opportunities to increase the joint provision of feedstocks from landscapes that have a mixture of public and private forests. Currently, much effort is being expended discussing what feedstocks should be eligible for renewable energy credit. Even with limited eligibility standards in place, there is a need to better understand how increased demand for bioenergy feedstocks from private lands may influence the demands for goods and services obtained from public forest lands. Additionally, because the agriculture and forest sectors are linked and many of the feedstocks are substitutable, it is useful to consider both of those sectors in any analysis of bioenergy feedstocks (e.g., Alig et al. 2010a). It is possible that increased demand for biomass from private forest harvest residues would have little impact on the management of private forests. In one existing national-level study of the forest sector (McCarl et al. 2000), logging residues from harvest of traditional forest products were never utilized for bioenergy; however, the oil prices at the time were much lower than current ones. Research that quantifies how handling and transport costs differ between public and private forests would also be useful in identifying bioenergy feedstocks accessible at the lowest transportation cost.

How the forest sector responds to changes in public land timber harvest volume has been studied. However, relatively little is known about the threshold relationships that might exist between public land harvest volumes and the maintenance of adequate logging and milling infrastructure in local communities to support continued forest sector commercial activity (e.g., in eastern Oregon). In addition to traditional timber production, threshold relationships between local infrastructure and the provision of resources from public forest lands may also exist for items such as bioenergy feedstocks or other forest products. Additional research addressing these thresholds would be useful to identify potential unintended consequences from significant changes in public land management (e.g., a hypothetical halt of harvesting on public lands as modeled in Depro et al. 2008).

Private landowners, particularly individual forest-land owners, are diverse. Numerous research projects have been undertaken to quantify the motivations and willingness of private individual landowners to participate in conservation programs. Research efforts to summarize this existing work as well as efforts to quantify revealed behavior in responding to conservation programs (e.g., afforestation of erodible agricultural land) would be useful to better gauge the expected response by private forest-land owners to new climate change programs and associated new markets. This would help public forest agencies identify tools and information that would be useful to the private sector. Afforestation is one of the most productive approaches to generating carbon offsets as part of any potential cap and trade program. One recent study estimates that forest area could increase by up to 25 percent, mostly owing to afforestation, when carbon is valued (Alig et al. 2010a). Research that examines the willingness of agriculture landowners (including those who lease agriculture land to agriculture operators) to plant trees for climate change programs will help to place these forest and agriculture sector modeling results into perspective and identify opportunities for technological transfer from public land agencies.

Additionality, leakage, and permanence are three concerns commonly mentioned when considering climate change policies. Leakage is likely the most germane when considering the interaction between public and private forests in the context of climate change and climate change policy. The concept of leakage relates to how offset activities in one location or market may result in countervailing emissions in another location or market (see Kline et al. 2009 for a discussion of leakage). Some existing forest sector research is pertinent. For example, Adams et al. (1996) have shown that reductions in public harvest rates are followed by increases in private harvest. Within the context of climate change and associated policies, leakage is important when considering the overall efficacy of comprehensive climate legislation. Leakage between public and private forests is probably of greatest importance when considering policies implemented on publicly owned forests. However, leakage across regions is likely also of interest within the context of regional patterns of public and private land ownership.

Impacts on land values from climate change policies warrant further investigation. The form and extent of land value changes can depend on the type and size of the policy as well as whether taxes, subsidies, or other types of incentives are employed. For example, subsidies to promote delays in timing of timber harvest can affect land values for some forest stands even without timber price reductions (Adams and Latta 2005). Given the relatively long-term nature of forest production,

Eric White

Researchers are increasingly investigating linkages among carbon management, fire management, and bioenergy production on public timberlands, each of which can have a profound impact on the carbon balance, ecological integrity, and economic value of the forest.

it would be useful to consider changes in value over time. These types of investigations would be facilitated by expanded and consistent data coverage for land values, including for forest land, across the United States.

Finally, partnerships in management of public and private forest lands could continue to increase in popularity in the coming decades (see National Association of State and Private Foresters 2009). Research that examines ways to maximize the joint provision of goods and services from public and private forests, in the context of climate change, will help inform ways to achieve more effective policy. Lessons learned from past efforts at joint public and private timberland management, such as sustained timber yield units in the West, could help inform current policy deliberations. Additionally, policy implementation can be improved by research that identifies effective public agency programs for private landowners and the most effective approaches to public-private partnerships.

Conclusions

The public and private forests of the United States have a long history of connection and interaction. As climate change progresses and comprehensive policies are developed, consideration of public and private forest ownership will be important. Under baseline projections, carbon stocks are projected to decline on private forests but increase on public forests in the coming decades. When carbon is valued, private forest carbon is projected to increase because of afforestation and changes in forest management. Increases or reductions in public harvesting rates have been projected to lead to countervailing changes in public carbon stocks and flux relative to the baseline. Although research has confirmed that public and private forests are linked through the market by changes in timber harvest activity, additional research is needed to quantify projected responses in public and private forest management to alternate carbon market or policy formulations. Similar to the provision of bioenergy feedstocks, public lands could potentially participate in carbon markets, although public land participation is not certain. Whether carbon sequestered on public lands under a carbon market would be "additional" to carbon sequestered under "business as usual" needs to be considered, particularly on federal lands where current harvest rates are low.

The public and private forests of the United States have a long history of connection and interaction.

Both private and public forest ownership groups have advantages in the provision of some bioenergy feedstocks. Because private lands account for most U.S. timber harvest, those forests have the greatest capacity to provide harvest residues for biomass. Current expectations are that revenues from the sale of timber harvest residues would be minor and would probably not change private forest management. There is some concern about site productivity declines because of harvest residue removal. If harvest residue usage on private industry lands lead to widespread reduction in productivity, additional pressure could be placed on public lands to increase the provision of some forest goods and services. Public forests have the greatest volumes of material that could be treated as part of a hazard-fuel reduction program. Research has projected that hazard-fuel thinning programs would lead to a reduction in private timber harvest and stumpage values for softwoods. Research has consistently projected that logging contractors and timber mills benefit from hazard-fuel reduction programs, although this benefit is projected to be short lived in at least one study. There is currently much discussion over whether feedstocks from public forests would qualify for renewable energy credit under existing and proposed legislation.

The general perception is that privately owned forests are better positioned than public forests to respond rapidly to climate change and new climate change policies. However, private forests have gone through a change in traditional industry forest ownership, and much of private forest land is owned by a diverse group of individual owners, many without management plans. Public forest agencies can help private forest owners respond optimally to climate change and new policies by providing information and technical assistance. Recent responses by the USDA Forest Service to increase wildland fire and fuels management capacity may indicate that public forests do in fact have the capacity to make rapid changes in management in response to climate change and new policies in some cases. Even if public forests are not the focus of new climate policies, public forest agencies will be integral in helping private forests respond by providing information and support as well as participating in public-private partnerships.

The literature examining linkages and interactions between public and private forests within the context of climate change and climate change policies is limited. There are a number of research opportunities to quantify the connections between public and private forests. It is probable that both private and public ownerships will play important roles in the mitigation of and adaptation to climate change by the forest sector. Improved knowledge regarding the linkages between the two ownership groups should improve the effectiveness of climate change policies and help resource planners identify likely future conditions.

English Equivalents

When you know:	Multiply by:	To get:
Cubic meters (m^3)	35.3	Cubic feet (ft^3)
Grams (g)	0.0352	Pounds
Metric tonnes	1.102	Tons
Megagrams (Mg)	1.102	Tons
Teragrams (Tg)	1,102,311	Tons
Terawatt hours (TWH)	3.6×10^9	Megajoules

References

Abt, K.L.; Prestemon, J.P. 2006. Timber markets and fuel treatments in the Western U.S. Natural Resources Modeling. 19(1): 15–43.

Adams, D.M.; Alig, R.J.; McCarl, B.; Callaway, J.; Winnett, S. 1996. An analysis of the impacts of public timber harvest policies on private forest management in the United States. Forest Science. 42(3): 343–358.

Adams, D.M.; Haynes, R.W.; Daigneault, A.J. 2006. Estimated timber harvest by U.S. region and ownership, 1950–2002. Gen. Tech. Rep. PNW-GTR-659. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 64 p.

Adams, D.M.; Latta, G.S. 2005. Costs and regional impacts of restoration thinning programs on the national forests in eastern Oregon. Canadian Journal of Forest Resources. 35: 1329–1330.

Alig, R.J.; Krankina, O.; Yost, A.; Kuzminykh, J. 2006. Forest carbon dynamics in the Pacific Northwest (USA) and the St. Petersburg region of Russia: comparisons and policy implications. Climatic Change. 79: 335–360.

Alig, R.J.; Latta, G.; Adams, D.M.; McCarl, B. 2010a. Mitigating greenhouse gases: the importance of land base interactions between forests, agriculture, and residential development in the face of changes in bioenergy and carbon prices. Forest Policy and Economics. 12: 67–75.

Alig, R.J.; Plantinga, A.; Ahn, S.; Kline, J.D. 2003. Land use changes involving forestry in the United States: 1952 to 1997, with projections to 2050. Gen. Tech. Rep. PNW-GTR-587. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 92 p.

Alig, R.J.; Plantinga, A.; Haim, D.; Todd, M. 2010b. Area changes in U.S. forests and other major land uses, 1982-2002, with projections to 2062. Gen. Tech. Rep. PNW-GTR-815. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 140 p.

Birch, T.W. 1997. Private forest-land owners of the United States, 1994. Resour. Bull. NE-RB-134. Radnor, PA: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. 183 p.

Birdsey, R.A.; Lewis, G.M. 2003. Carbon in U.S. forests and wood products, 1987–1997: state-by-state estimates. Gen. Tech. Rep. NE-310. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northeastern Research Station. 42 p.

Butler, B. 2009. Forest ownership. In: Smith, W.B.; Miles, P.D.; Perry, C.H.; Pugh, S.A., eds. Forest resources of the United States, 2007. Gen. Tech. Rep. WO-78. Washington, DC: U.S. Department of Agriculture, Forest Service: 19–21.

Carter, M.C.; Dean, T.J.; Wang, Z.; Newbold, R.A. 2006. Impacts of harvesting and postharvest treatments on soil bulk density, soil strength, and early growth of *Pinus taeda* in the Gulf Coastal Plain: a long-term soil productivity affiliated study. Canadian Journal of Forestry Research. 36: 601–614.

Cathcart, J.; Kline, J.D.; Delaney, M.; Tilton, M. 2007. Carbon sequestration and Oregon's land use planning program. Journal of Forestry. 105(4): 167–172.

Depro, B.M.; Murray, B.C.; Alig, R.J.; Shanks, A. 2008. Public land, timber harvests, and climate mitigation: quantifying carbon sequestration potential on U.S. public timberlands. Forest Ecology and Management. 255: 1122–1134.

Gan, J.; Smith C.T. 2006. Availability of logging residues and potential for electricity production and carbon displacement in the USA. Biomass and Bioenergy. 30: 1011–1020.

Haynes, R. 1990. An analysis of the timber situation in the United States: 1989–2040. Gen. Tech. Rep. RM-GTR-199. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 268 p.

Ince, P.J.; Spelter, H.; Skog, K.E.; Kramp, A.; Dykstra, D.P. 2008. Market impacts of hypothetical fuel treatment thinning programs on federal lands in the Western United States. Forest Policy and Economics. 10: 363–372.

Keegan C.E., III; Fielder, C.E.; Morgan, T.A. 2004. Wildfire in Montana: potential hazard reduction and economic effects of a strategic treatment program. Forest Products Journal. 54(7/8): 21–25.

Kline, J.D.; Mazzotta, M.J.; Patterson, T.M. 2009. Toward a rational exuberance for ecosystem services markets. Journal of Forestry. 107(4): 204–212.

McCarl, B.A.; Adams, D.M.; Alig, R.J.; Chmelik, J.T. 2000. Competitiveness of biomass-fueled electrical power plants. Annals of Operations Research. 94: 37–55.

Minnesota Forest Resources Council [Minnesota FRC]. 2007. Biomass harvesting on forest management sites in Minnesota—draft report. http://www.forestrycenter.org/library.cfm?refid=98930. (March 3, 2010).

National Association of State and Private Foresters. 2009. All-lands policy platform: a seven-point plan for America's forests. http://www.stateforesters.org/files/110309-NASF-All-Lands-Policy-Platform.pdf. (December 2009).

Schmidt, D. 2006. Biomass energy opportunities from hybrid poplars in Minnesota. http://www.extension.umn.edu/Agroforestry/biomass/schmidt.pdf. (October 9, 2009).

Seattle Public Utilities. 2008. South Tolt Watershed management plan. http://www.cityofseattle.net/UTIL/stellent/groups/public/@spu/@ssw/documents/webcontent/spu01_004082.pdf. (March 4, 2010).

Skog, K.E.; Barbour, R.J.; Abt, K.L.; Bilek, E.M.; Burch, F.; Fight, R.D.; Hugget, R.J.; Miles, P.D.; Reinhardt, E.D.; Shepperd, W.D. 2006. Evaluation of silviculture treatments and biomass use for reducing fire hazard in Western States. Res. Pap. FPL-634. Madison, WI: U.S. Department of Agriculture, Forest Service, Forest Products Laboratory. 29 p.

Smith, J.E.; Heath, L.S. 2004. Carbon stocks and projections on public forest land in the United States, 1952–2040. Environmental Management. 33(4): 433–442.

Smith, W.B.; Miles, P.D.; Perry, C.H.; Pugh, S.A. 2009. Forest resources of the United States, 2007. Gen. Tech. Rep. WO-78. Washington, DC: U.S. Department of Agriculture, Forest Service. 336 p.

Stein, S.M.; McRoberts, R.E.; Alig, R.J.; Nelson, M.D.; Theobald, D.M.; Eley, M.; Dechter, M.; Carr, M. 2005. Forests on the edge: housing development on America's private forests. Gen. Tech. Rep. PNW-GTR-636. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 16 p.

Theobald, D.M. 2007. Protected lands of the continental US (CUS_UPPT_100). Unpublished data set. On file with: Dave Theobald, Human Dimensions of Natural Resources and the Natural Resource Ecology Lab, Forestry 245, Colorado State University, Fort Collins, CO 80523.

U.S. Department of Agriculture, Office of the Chief Economist [USDA]. 2008. U.S. agriculture and forestry greenhouse gas inventory: 1990–2005. Tech. Bull. 1921. Washington, DC: Global Change Program. 161 p.

U.S. Department of Energy [US DOE]. 2010. Table 8. Electricity supply, disposition, prices, and emissions. http://www.eia.doe.gov/oiaf/aeo/excel/aeotab_8.xls. (March 3, 2010).

U.S. Environmental Protection Agency [US EPA]. 2009. Inventory of U.S. greenhouse gas emissions and sinks: 1990–2007. http://www.epa.gov/climatechange/emissions/usinventoryreport.html. (March 4, 2010).

Walmsley, J.D.; Jones, D.L.; Reynolds, B.; Price, M.H.; Healey, J.R. 2009. Whole tree harvesting can reduce second rotation forest productivity. Forest Ecology and Management. 257: 1104–1111.

Western Forestry Leadership Coalition [WFLC]. 2009. Bioenergy—a western update. http://www.wflccenter.org/news_pdf/339_pdf.pdf. (November 2009).

White, E.M. 2010. Woody biomass for bioenergy and biofuels in the United States—a briefing paper. Gen. Tech. Rep. PNW-GTR-825. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 45 p.

White, E.M.; Morzillo, A.T.; Alig, R.J. 2009. Past and projected rural land conversion in the U.S. at state, regional, and national levels. Landscape and Urban Planning. 89: 37–48.

Glossary

afforestation—The forestation, either by human or natural forces, of nonforest land.

Conservation Reserve Program (CRP) land—A land cover/use category that includes land under a CRP contract. The CRP is a federal program established under the Food Security Act of 1985 to assist private landowners to convert highly erodible cropland to vegetative cover for 10 years.

cropland—A land cover/use category that includes areas used for the production of adapted crops for harvest. Two subcategories of cropland are recognized: cultivated and noncultivated. Cultivated cropland comprises land in row crops or close-grown crops and also other cultivated cropland, for example, hay land or pastureland that is in a rotation with row or close-grown crops. Noncultivated cropland includes permanent hay land and horticultural cropland.

developed land—In the National Resources Inventory (NRI), developed land consists of urban and built-up areas, as well as land devoted to rural transportation. This is a broader category than the "urban" land use considered in this study. This study has not attempted to model net returns to rural transportation use, so this report focuses only on the urban component of developed land.

forest land—Land at least 10-percent stocked by forest trees of any size, including land that formerly had such tree cover and that will be naturally or artificially regenerated. Forest land includes transition zones, such as areas between heavily forested and nonforested lands that are at least 10-percent stocked with forest trees and forest areas adjacent to urban and built-up areas. The minimum area for classification of forest land is 1 acre. Roadside, streamside, and shelterbelt strips of timber must have a crown width of at least 120 feet (36.6 m) to qualify as forest land. Unimproved roads and trails, streams, and clearings in forest areas are classified as forest if less than 120 feet wide.

Intergovernmental Panel on Climate Change (IPCC)—The IPCC was established to provide decisionmakers and others interested in climate change with an objective source of information about climate change. The IPCC does not conduct any research nor does it monitor climate-related data or parameters. Its role is to assess on a comprehensive, objective, open, and transparent basis the latest scientific, technical, and socioeconomic literature produced worldwide relevant to the understanding of the risk of human-induced climate change, its observed and projected impacts, and options for adaptation and mitigation. For more information, see http://www.ipcc.ch/.

land area—The area of dry land and land temporarily or partly covered by water, such as marshes, swamps, and river flood plains; streams, sloughs, estuaries, and canals less than 200 feet (61 m) wide; and lakes, reservoirs, and ponds less than 4.5 acres (1.8 ha).

land cover/use—A term that includes categories of land cover and categories of land use. Land cover is the vegetation or other kind of material that covers the land surface. Land use is the purpose of human activity on the land; it is usually, but not always, related to land cover. The NRI uses the term land cover/use to identify categories that account for all the surface area of the United States. The six major land-use categories considered in this study are (1) cropland, (2) pasture, (3) range, (4) Conservation Reserve Program (CRP), (5) forest, and (6) urban. These uses are described in this glossary.

large urban and built-up areas—These areas include developed tracts of 10 acres (4 ha) and more.

nonindustrial private forest (NIPF)—An ownership class of private lands where the owner does not operate commercial wood-using plants.

National Resources Inventory (NRI)—A statistical survey of land use and natural resource conditions and trends on U.S. nonfederal lands. The NRI is led by Natural Resources Conservation Service (NRCS), the Department of Agriculture's lead conservation agency. For more information, see http://www.nrcs.usda.gov/technical/NRI/.

other rural land—A land cover/use category that includes farmsteads and other farm structures, field windbreaks, barren land, and marshland. Some reports refer to this as NRI minor land cover/uses.

pastureland—A land cover/use category of land managed primarily for the production of introduced forage plants for livestock grazing. Pastureland cover may consist of a single species in a pure stand, a grass mixture, or a grass-legume mixture. Management usually consists of cultural treatments: fertilization, weed control, reseeding or renovation, and control of grazing. For the NRI, includes land that has a vegetative cover of grasses, legumes, or forbs, regardless of whether or not it is being grazed by livestock.

public—An ownership class composed of land owned by federal, state, county, or municipal governments.

rangeland—A land cover/use category on which the climax or potential plant cover is composed principally of native grasses, grasslike plants, forbs or shrubs suitable for grazing and browsing, and introduced forage species that are managed like rangeland. This would include areas where introduced hardy and persistent grasses, such as crested wheatgrass (*Agropyron cristatum* (L.) Gaertn.), are planted and such practices as deferred grazing, burning, chaining, and rotational grazing are used, with little or no chemicals or fertilizer being applied. Grasslands, savannas, many wetlands, some deserts, and tundra are considered to be rangeland. Certain communities of low forbs and shrubs, such as mesquite, chaparral, mountain shrub, and pinyon-juniper, are also included as rangeland.

residential area—The sum of area in lots used for housing units. Estimates of residential area, urban and rural, are based on data from the American Housing Surveys.

timberland—Forest land that is producing or is capable of producing crops of industrial wood and not withdrawn from timber utilization by statute or administrative regulation. (Note: Areas qualifying as timberland are capable of producing in excess of 20 ft^3/acre [14 m^3/ha] per year of industrial wood in natural stands. Currently inaccessible and inoperable areas are included.)

urban area—Nationally, there are two main sources of data on urban area. First, the U.S. Department of Commerce Bureau of the Census compiles urban area every 10 years, coincident with the census of population. Second, the Natural Resources Conservation Service, U.S. Department of Agriculture, publishes developed land, including urban components, at 5-year intervals as part of the NRI. Although the U.S. Geological Survey, National Aeronautics and Space Agency, Housing and Urban Development Department, and several local, state, and federal agencies also collect data or conduct special-purpose studies on urban area, the census and the NRI provide the only nationally consistent historical series. Because of differences in data-collection techniques and definitions, the NRI estimates of "large urban and built-up areas" is usually higher than the census "urban area" estimates for nearly all states. The census urban area series runs from 1950, whereas the NRI started providing a consistent series in 1982. Historically, the Economic Research Service (ERS) major land-use time series (MLUS) has used census urban area numbers. Prior to 1982, census urban area was the only reliable national source of urban area data available. Since 1945, census urban area has been used in the MLUS time series to maintain a consistent series. For comparison purposes, census urban area is checked against the NRI to help project and interpolate census trends between decennial census years.

urban and built-up areas—These areas consist of residential, industrial, commercial, and institutional land; construction and public administrative sites; railroad yards; cemeteries; airports; golf courses; sanitary landfills; sewage plants; water control structures; small parks; and transportation facilities within urban areas.

Pacific Northwest Research Station

Web site	http://www.fs.fed.us/pnw/
Telephone	(503) 808-2592
Publication requests	(503) 808-2138
FAX	(503) 808-2130
E-mail	pnw_pnwpubs@fs.fed.us
Mailing address	Publications Distribution Pacific Northwest Research Station P.O. Box 3890 Portland, OR 97208-3890

Printed on
Recycled Paper

U.S. Department of Agriculture
Pacific Northwest Research Station
333 SW First Avenue
P.O. Box 3890
Portland, OR 97208-3890