Chapter 11. North American Forests

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KEY FINDINGS

- North American forests contain more than 170 Gt of carbon, of which 28% is in live biomass and 72% is in dead organic matter.
- North American forests were a sink of approximately 350 Mt C yr^{-1} for the decade of the 1990s. This number is highly uncertain.
- There is general understanding that forests of North America were a source of CO_2 to the atmosphere during the 19th and early 20th century as forests were converted to agricultural land; this process continues today in Mexico where forests are a source of 50-62 Mt C yr^{-1}. Only in more recent decades have forests of Canada and the United States become a sink as a consequence of the recovery of forests following the abandonment of agricultural land.
- Many factors that cause changes in carbon stocks of forests and wood products have been identified, including land-use change, timber harvesting, natural disturbance, increasing atmospheric CO_2, climate change, nitrogen deposition, and tropospheric ozone. Existing monitoring and modeling capability is still somewhat inadequate for a definitive assessment of the relative importance of these factors. Consequently, there is a lack of general consensus about how these different natural and anthropogenic factors contribute to the current sink, and the relative importance of factors probably varies by country.
- There have been several continental- to subcontinental-scale assessments of future changes in carbon and vegetation distribution in North America, but the resulting projections of future trends for North American forests are highly uncertain. Some of this is due to uncertainty in future climate, but there is also considerable uncertainty in forest response to climate change and in the interaction of climate with other natural and anthropogenic factors.

- Forest management strategies can be adapted to manipulate the carbon sink strength of forest systems. The net effect of these management strategies will depend on the area of forests under management, management objectives for resources other than carbon, and the type of disturbance regime being considered.
- Decisions concerning carbon storage in North American forests and their management as carbon sources and sinks will be significantly improved by (1) filling gaps in inventories of carbon pools and fluxes, (2) a better understanding of how management practices affect carbon in forests, and (3) the increased availability of decision support tools for carbon management in forests.

INTRODUCTION

The forest area of North America totals 771 million hectares, about 20% of the world's forest area (Food and Agriculture Organization 2001) (see Table 11-1). About 45% of this forest area is classified as boreal, mostly in Canada and some in Alaska. Temperate and tropical forests constitute the remainder of the forest area.

Table 11-1. Area of forest land by biome and country, 2000 (1000 ha).

North American forests are critical components of the global carbon cycle, exchanging large amounts of CO_2 and other gases with the atmosphere and oceans. Forests and wood products constitute more than 60% of the total annual carbon sink on land in North America (–557 Mt C yr^{-1}; see Chapter 3), including the –23 Mt C yr^{-1} stored in land defined by the census as urban and suburban trees in the United States. In this chapter we present the most recent estimates of the role of forests in the North American carbon balance, describe the main factors that affect forest carbon stocks and fluxes, and discuss management options and research needs.

CARBON STOCKS AND FLUXES

Ecosystem Carbon Stocks And Pools

North American forests contain more than 170 Gt of carbon, of which 28% is in live biomass and 72% is in dead organic matter (Table 11-2). Among the three countries, Canada's forests contain the most carbon and Mexico's forests the least.

Table 11-2. Carbon stocks in forests by ecosystem carbon pool and country (Mt C).

In Canada, mean carbon density values for forest biomass range from about 20 t C ha^{-1} in the eastern portion of the boreal forest to over 140 t C ha^{-1} in Pacific Cordilleran forests. Dead organic matter (DOM) values range from 138 t C ha^{-1} in the western boreal to nearly 250 t C ha^{-1} in the subarctic. DOM represents 60–90% of total C density, with a countrywide average of 83% (Kurz and Apps, 1999).

In the United States, the total carbon currently stored in forest ecosystems is 66,575 Mt C (Heath and Smith 2004), of which forest land in Alaska constitutes 14,000 Mt C (Birdsey and Heath, 1995). For the conterminous United States, about 40% of the total ecosystem carbon is in the aboveground carbon pool, which includes live trees, understory vegetation, standing and down deadwood, and the forest floor. About 8% is in roots of live trees, and the remainder, a little more than half, is in the soil (Heath and Smith, 2004). DOM represents roughly 63% of the total ecosystem carbon stocks in U.S. forests.

In Mexico, in unmanaged forested areas, temperate forests contain 4,500 Mt C, tropical forests contain 4,100 Mt C, and semiarid forests contain 5,000 Mt C. In forest plantations 800 Mt C are sequestered in long and short rotations, restoration, and bioenergy plantations. Managed temperate and tropical forests store 500 Mt C, and protected forests store 2,000 Mt C. Agroforestry systems harbor 100 MtC.

Net North American Forest Carbon Fluxes

According to nearly all published studies, North American lands are a net carbon sink (Pacala *et al.*, 2001); however, the magnitude of the Canadian and Mexican forest contribution to the land carbon sink is categorized as highly uncertain (meaning there is 95% certainty that the actual value is within ±100% of the reported estimate). The estimated carbon sink of the United States forests is categorized as uncertain (meaning that there is a 95% certainty that the actual value is within 50% of the reported estimate.) A summary of currently available data from greenhouse gas inventories and other sources suggests that the magnitude of the North American forest carbon sink was approximately –350 Mt C yr^{-1} for the decade of the 1990s (Table 11-3).

Table 11-3. Change in carbon stocks for forests and wood products by country (Mt C yr^{-1}).

Canadian forests and forest products may be a net sink of about –109 Mt C yr^{-1} (Table 11-3). These estimates pertain to the area of forest considered to be “managed” under international reporting guidelines, which is 53% of the total area of Canada’s forests. The estimates also include the carbon changes that result from land-use change. Changes in forest soil carbon are not included. High interannual variability is averaged into this estimate—the annual change varied from approximately –190 Mt C in 1990 to –70 Mt C in 2003 (Environment Canada, 2005).

In the United States, forest ecosystem carbon stocks are estimated to be a net sink of –236 Mt C yr^{-1}, and for wood products, the estimated sink is –57 Mt C yr^{-1} (Table 11-3). Most of the net sink is in aboveground carbon pools, which account for –146 Mt C yr^{-1} (Smith and Heath, 2005). The net sink for the belowground carbon pool is estimated at –90 Mt C (Pacala *et al*., 2001). The size of the carbon sink in U.S. forest ecosystems appears to have declined slightly over the last decade (Smith and Heath, 2005). In contrast, a steady or increasing supply of timber products now and in the foreseeable future (Haynes, 2003) means that the rate of increase in the wood products carbon pool is likely to remain steady.

For Mexico, the most comprehensive available estimate for the forest sector suggests a source of +52 Mt C per year (Masera *et al*., 1997). This estimate does not include changes in the wood products carbon pool. The main cause of the estimated source is deforestation, which is offset to a much lesser degree by restoration and recovery of degraded forestland.

Large-scale estimates of ecosystem carbon fluxes can only be explained by a more detailed examination of the dynamics of individual forest stands that have unique combinations of disturbance history, management intensity, vegetation, and site characteristics. How carbon fluxes change over time in response to disturbance helps explain the aggregated estimates at larger scales. Extensive land-based measurements of forest/atmosphere carbon exchange reveal patterns and causes of sink or source strength. Representative estimates for North America are summarized in Appendix 11.A.

TRENDS AND DRIVERS

Overview of Trends and Drivers of Change in Carbon Stocks

Many factors that cause changes in carbon stocks of forests and wood products have been identified, but there is some agreement on the relative magnitude of their influence (Barford *et al*., 2001; Caspersen *et al*., 2000; Goodale *et al*., 2002; Körner 2000; Schimel *et al*., 2000). The long-term effects of land-use change, timber harvesting, natural disturbance, increasing atmospheric CO_2, climate change, nitrogen deposition, and tropospheric ozone are all considered major factors affecting carbon stocks in forests and wood products. Furthermore, the relative impacts of these different drivers can vary in magnitude, depending on the type of forest and the kind of landscape involved. It is particularly difficult, yet very important for policy and management, to separate the effects of direct human actions from natural factors.

North American forest ecosystems are a net C sink of roughly –312 Mt C yr^{-1} (Table 11-3), but there is a lack of consensus about precisely how natural and anthropogenic factors have contributed to this overall estimate, and the relative importance of factors varies by country. In Canada, one study estimated that impacts of wildfire and insects caused emissions of about +40 Mt C yr^{-1} of carbon to the atmosphere over the last two decades (Kurz and Apps, 1999). Yet another study concluded that the positive effects of climate, CO_2, and nitrogen deposition outweighed the effects of increased natural disturbances, making

Canada's forests a net carbon sink in the same period (Chen *et al*., 2003). In the United States between 1953 and 1997, carbon stocks in forest ecosystems (excluding soils) increased by about 175 Mt C yr^{-1}, and for the approximate year 2000, the average annual increase in forest ecosystem carbon stocks is 146 Mt C yr^{-1} (Smith and Heath, 2005). This declining trend is based mainly on dynamics of vegetation change following a long history of land-use change and management (Birdsey *et al*., 2006). Mexico emits 52.3 Mt C yr^{-1} as a consequence of land use change, including deforestation, forest degradation, forest fires and forest regeneration (Masera *et al* 1997; de Jong *et al*., 2000). These driving factors are expected to continue influencing forests in the near future.

Effects of Land-Use Change

Since 1990, approximately 549,000 ha of former cropland or grassland in Canada have been abandoned and are reverting to forest, while 71,000 ha of forest have been converted to cropland, grassland, or settlements, for a net increase in forest area of 478,000 ha (Environment Canada 2005). Land-use change in Canada caused a net increase in total carbon storage of about –50 Mt C yr^{-1} in 1990, with the sink strength declining through 2003 to about –20 Mt C yr^{-1}.

In the last century more than 130 million hectares of land in the conterminous United States were either afforested (62 million ha) or deforested (70 million ha) (Birdsey and Lewis 2003). Even though the net change in the area of forest land was not significant during that time, the magnitude of the shifts in land use caused significant redistribution of carbon stocks among land categories. Over the longer term, Houghton *et al*. (1999) estimated that cumulative changes in forest carbon stocks for the period from 1700 to 1990 in the United States were about +25 Gt C, primarily from conversion of forestland to agricultural use and reduction of carbon stocks for wood products.

Mexican forests emit +50 to +62 Mt C yr^{-1} to the atmosphere as a consequence of land use change (Masera *et al*., 1997). In Mexico, deforestation and forest degradation were responsible for an annual forest loss of 720,000 ha in the late 1980s and early 1990s (Masera *et al*., 1997). The deforestation rate of unmanaged forests was about 619,000 ha per year in 1990; however, based on total forest cover change between 1993 and 2000, Palacio *et al*. (2000) estimated a deforestation rate of 880,000 ha yr^{-1}. Deforestation is primarily driven by conversion of tropical forest to pastures (73% of deforested tropical evergreen forest, and 61% of deforested tropical deciduous forest, Masera *et al*., 2001). About 13 to 15% of deforested land gets converted to agricultural land (Masera *et al*., 2001). The highest deforestation rates occur in the tropical deciduous forests (304,000 ha in 1990) and the lowest in temperate broadleaf forests (59,000 ha in 1990) (Masera *et al.*, 2001). Carbon fluxes in tropical rainforests in La Selva Lacandona resulting from a 31% reduction of closed forest cover between 1976 and 1996 correspond to total emissions of 41.7 ± 12.1 Mt C [95% confidence interval (CI)] with 31.9 ± 7.0 Mt C (95% CI) from vegetation and 9.5 ± 10.4 Mt C (95% CI) from soils (de Jong *et al*., 2000).

Effects of Forest Management

The direct human impact on North American forests ranges from very minimal for protected areas to very intense for plantations (Table 11-4). Between these extremes is the vast majority of forestland, which has a wide range of human impacts that seems to vary by country.

Table 11-4. Area of forestland by management class and country, 2000 (1000 ha).

Forests and other wooded land in Canada occupy about 404 Mha, of which 214 Mha (53%) are under active forest management (Environment Canada 2005). Managed forests are considered to be under the direct influence of human activity and not reserved. Less than 1% of the area under active management is harvested annually. Apps *et al*. (1999) used a carbon budget model to simulate carbon in harvested wood products (HWP) for Canada. Approximately 800 Mt C were stored in the Canadian HWP sector in 1989, of which 50 Mt C were in imported wood products, 550 Mt C in exported products, and 200 Mt C in wood products produced and consumed domestically.

Between 1990 and 2000, about 4 Mha yr^{-1} were harvested in the U.S., two-thirds by means of some form of partial-cut harvest and one-third by a clearcut method (Birdsey and Lewis 2003). Between 1987 and 1997, about 1 Mha yr^{-1} were planted with trees, and about 800,000 ha were treated to improve the quality and/or quantity of timber produced (Birdsey and Lewis 2003). Harvesting in U.S. forests accounts for substantially more tree mortality than natural causes such as wildfire and insect outbreaks (Smith *et al*., 2004). In 2002, about 170 Mt C of tree biomass were removed from forests by harvest, offset by 280 Mt C of net primary productivity (which includes growth and mortality from natural causes), making U.S. tree biomass a net sink of –110 Mt C yr^{-1} (Smith and Heath 2005). The harvested wood resulted in -57 Mt C added to landfills and products in use, and an additional 88 Mt C were emitted from harvested wood burned for energy (Skog and Nicholson 1998).

About 80% of the forested area in Mexico is socially owned by communal land grants (*ejidos*) and rural communities. About 95% of timber harvesting occurs in native temperate forests (SEMARNAP 1996). Extensive overexploitation (e.g., illegal deforestation and fuelwood extraction) of natural resources from forests have caused dramatic land degradation in forested land (21.4 Mha affected in 1990). It is estimated that illegal wood extraction reaches 13.3 million m^3 of wood every year (Torres 2004). Unlike U.S. and Canadian forests, Mexican forests have been affected since pre-Columbian times by the almost ubiquitous influence of a large proportion of the rural population, which controls the carbon fluxes and stocks through fire; wood extraction; legal and illegal logging; shifting agriculture practices; and conversion of land to plantations (e.g., coffee), fields for agricultural crops (e.g., sugar cane), and pastures. Also, the differences in property rights, land ownership, and associated management policies (and lack thereof), which are preeminently important in Mexico, where most of the land is communal, also contribute to different socioeconomic controls over the carbon cycle.

Effects of Climate and Atmospheric Chemistry

Some studies indicate that the combined effects of climate and atmospheric chemistry changes on carbon sequestration are likely to be significantly smaller than the effects of land management and land-use change (Caspersen *et al*., 2000; Schimel *et al*., 2000), but existing monitoring and modeling capability is still somewhat inadequate for a definitive assessment of the relative importance of these factors (U.S. Climate Change Science Program 2003). Environmental factors, including climate variability, nitrogen deposition, tropospheric ozone, and elevated CO_2, have been recognized as significant factors affecting the carbon cycle of forests (Aber *et al*., 2001; Ollinger *et al*., 2002). Experimental studies have clearly shown that rising atmospheric CO_2 increases photosynthesis in plants. Recent reviews of ecosystem-scale studies known as Free Air CO_2 Exchange (FACE) experiments suggest an increase in net primary productivity (NPP) of 12–23% over all species (Norby *et al*., 2005; Nowak *et al*., 2004). However, at the ecosystem scale, it is uncertain whether this effect results in a lasting increase in sequestered carbon or causes a more rapid cycling of carbon between the ecosystem and the atmosphere (Korner *et al*., 2005; Lichter 2005). Experiments have also shown that the effects of rising CO_2 are significantly moderated by increasing tropospheric ozone (Karnosky *et al*., 2003; Loya *et al*., 2003). When nitrogen is also considered, reduced soil fertility limits the response to rising CO_2, but nitrogen deposition can increase soil fertility to counteract that effect (Johnson *et al*., 1998; Oren *et al*., 2001).

Effects of Natural Disturbances

Wildfires were the largest disturbance in the twentieth century in Canada (Weber and Flannigan, 1997). In the 1980s and 1990s, the average total burned area was 2.6 Mha yr^{-1} in Canada's forests, with a maximum 7.6 Mha yr^{-1} in 1989. Carbon emissions from forest fires are substantial and arise mostly from northern forests (boreal, subarctic). Emissions range from less than +1 Mt C yr^{-1} in the interior of British Columbia to more than +10 Mt C yr^{-1} in the western boreal forest. Total emissions from forest land in Canada averaged approximately +27 Mt C yr^{-1} between 1959 and 1999 (Amiro *et al*., 2001). Much of the Canadian forest is expected to experience increases in fire severity (Parisien *et al*., 2005) and burn areas (Flannigan *et al*., 2005). Outbreaks of forest pests are also likely (Volney and Hirsch, 2005). While some of this disturbance may be reduced through enhanced suppression efforts, a long-term increase in impacts of disturbance is likely in the future, with associated losses of forest carbon stocks.

Estimated carbon emissions from four major insect pests in Canadian forests (spruce budworm, jack pine budworm, hemlock looper, and mountain pine beetle) varied from +5 to 10 Mt C yr^{-1} in the 1970s to less than +2 Mt C yr^{-1} in the mid-1990s[1]. Large emissions occurred in the 1970s and early 1980s as a result of extremely large spruce budworm outbreaks in Ontario and Quebec (18 to 30 Mha in each province). The area of outbreaks and associated carbon emissions has recently increased as a result of the mountain pine beetle outbreak in British Columbia, which affected 3.7 Mha in 2003, when emissions were approximately +4 Mt C yr^{-1}.

Natural disturbance is commonplace in U.S. forests, where insects, diseases, and wildfire combined affect more than 30 Mha per decade (Birdsey and Lewis 2003). Damage from weather events (hurricanes, tornados, ice storms) may exceed 20 Mha per decade (Dale *et al.*, 2001). There are few estimates of the impact of selected natural disturbances on carbon pools of temperate forests. McNulty (2002) estimated that large hurricanes in the United States could convert 20 Mt C of live biomass into detrital carbon pools. The impacts of fire are clearly significant. According to one estimate, the average annual carbon emissions from biomass burning in the contemporary United States ranges from 9 to 59 Mt C (Leenhouts 1998).

Pests and diseases are important natural disturbance agents in temperate forests of Mexico; however, no statistics exist on the extent of the affected land area. The number and area of sites affected by forest fires in Mexico have fluctuated considerably between 1970 and 2002 with a clear tendency of an increasing number of fire events (4,000–7,000 in the 1970s and 1,800–15,000 in the 1990s), and overall, larger areas are being affected (0.08–0.25 Mha in 1970s and 0.05–0.85 Mha in 1990s). During El Nino years, increasing drought increases fire frequencies (Torres 2004). Between 1995 and 2000, an average 8,900 fire events occurred per year and affected about 327,000 ha of the forested area. Currently, no estimates are available on the contribution of these fires to CO_2 emissions.

Projections of Future Trends

There have been several continental- to subcontinental-scale assessments of future changes in carbon and vegetation distribution in North America (VEMAP Members, 1995; Pan *et al.*, 1998; Neilson *et al.*, 1998; Joyce *et al.*, 2001). For the conterminous United States, the VEMAP study suggested that under most future climate conditions, NPP would respond positively to changing climate (20.8% ± 2.4%) but

[1]These estimates are the product of regional carbon density values, the proportion of mortality in defoliated stands given in Kurz and Apps (1999), data on area affected taken from NFDP (2005), and the proportion of C in insect-killed stands that is emitted directly to the atmosphere (0.1) from the disturbance matrix for insects used in the CBM-CFS (Kurz *et al*., 1992).

that total carbon storage would remain relatively constant (2.0% ± 3.5%). Under most climate scenarios the West gets wetter; when coupled with higher CO_2 and longer growing seasons, simulations show woody expansion and increased sequestration of carbon as well as increases in fire (Bachelet *et al*., 2001). However, recent scenarios from the Hadley model show some drying in the Northwest, which produces some forest decline (Price *et al*. 2004). Many simulations show continued growth in eastern forests through the end of the twenty-first century while others show the opposite, especially in the Southeast. Eastern forests could experience a period of enhanced growth in the early stages of warming, due to elevated CO_2, increased precipitation, and a longer growing season. However, further warming could bring on increasing drought stress, reducing the carrying capacity of the ecosystem and causing carbon losses through drought-induced dieback and increased fire and insect disturbances.

Large portions of the Canadian and Alaskan forest are expected to be particularly sensitive to climate change due to its high latitude and interior continental location (Hogg and Bernier, 2005). Climate change effects on forest growth could be positive (e.g., increased rates of photosynthesis and increased water use efficiency) or negative (decreased water availability, higher rates of respiration) (Baldocchi and Amthor, 2001). It is difficult to predict the direction of these changes and they will likely vary by species and local conditions of soils and topography (Johnston and Williamson, 2005). Because of the large area of boreal forests and expected high degree of warming, Canada and Alaska require close monitoring over the next few decades as these areas will likely be critical to determining the carbon balance of North America.

Future trends for Mexican forests are less certain. Deforestation will continue to cause large carbon emissions in the years to come. However, government programs (since 2001) are trying to reduce deforestation rates and forest degradation, implement sustainable forestry in native forests, promote commercial plantations and diverse agroforestry systems, and promote afforestation and protection of natural areas (Masera *et al*., 1997).

OPTIONS FOR MANAGEMENT

Forest management strategies can be adapted to manipulate the carbon sink strength of forest systems. The net effect of these management strategies on carbon stocks will depend on the area of forests under management, management objectives for resources other than carbon, and the type of disturbance regime being considered. The following sections describe current management strategies and provide some general information about how ecological principles might be applied to actively manipulate forest and atmosphere carbon stocks.

Although the science of managing forests specifically for carbon sequestration is not well developed, some management principles are emerging to guide management decisions (Appendix 11.B). The prospective role of forestry in helping to stabilize atmospheric CO_2 depends on harvesting and disturbance rates, expectations of future forest productivity, the fate and longevity of forest products, and the ability to deploy technology and forest practices to increase the retention of sequestered CO_2. Market factors are also important in guiding the behavior of the private sector. The forest sector includes a variety of activities that can contribute to increasing carbon sequestration, including: afforestation, mine land reclamation, forest restoration, agroforestry, forest management, biomass energy, forest preservation, wood products management, and urban forestry (Birdsey *et al*., 2000).

In the United States, forestry activities could increase carbon sequestration by significant amounts, in the range of –100 to –200 Mt C yr^{-1} for the United States alone according to several studies (Birdsey *et al*., 2000; Lewandrowski 2004; Environmental Protection Agency, 2005; Stavins and Richards, 2005). The studies also suggest that the rate of annual mitigation would likely decline over time as low-cost forestry opportunities become scarcer, forestry sinks become saturated, and timber harvesting takes place.

For Canada, Price *et al*. (1997) used the Carbon Budget Model of the Canadian Forest Sector (CBM-CFS) to examine the effects of reducing natural disturbance, manipulating stand density, and changing rotation lengths for a forested landscape in northwest Alberta. By replacing natural disturbance (fire) with a simulated harvesting regime, they found that long-term equilibrium carbon storage increased from 105 to 130 Mt C in a boreal-cordilleran forest management unit. Controlling stand density following harvest had minimal impacts in the short term but increased landscape-level carbon storage by 13% after 150 years, as the older, low-productivity stands were replaced by younger, higher-productivity stands. The main reason for the increased carbon storage was that the natural disturbance return interval (50 yr) was considerably shorter than the harvest rotation (up to 100 yr).

In a separate modeling study using the CBM-CFS model, Kurz *et al*. (1998) investigated the impacts on landscape-level carbon storage of the transition from natural to managed disturbance regimes. For a boreal landscape in northern Quebec, a simulated fire disturbance interval of 120 yr was replaced by a harvest cycle of 120 yr. The net impact was that the average age of forests in the landscape declined from 110 yr to 70 yr, and total carbon storage in forests declined from 16.3 to 14.8 Mt C (including both ecosystem and forest products pools). In this case the disturbance frequencies were the same, so the decline in carbon storage occurred because the harvesting regime preferentially selected older, high-biomass-density stands.

Market approaches and incentive programs to manage greenhouse gases, particularly CO_2, are under development in the United States, the European Union, and elsewhere (Totten, 1999). Since forestry activities have highly variable costs because of site productivity and operational variability, most recent studies of forestry potential develop "cost curves," i.e., estimates of how much carbon will be sequestered by a given activity for various carbon prices (value in a market system) or payments (in an incentive system). There is also a temporal dimension to the analyses because the rate of change in forest carbon stocks is variable over time, with forestry activities tending to have a high initial rate of net carbon sequestration followed by a lower or even a negative rate as forests reach advanced age.

Here we address costs of three broad categories of forestry activities: afforestation (conversion of agricultural land to forest), improved management of existing forests, and use of woody biomass for fuel. In general, analyses suggest that improved management of existing forestlands may be attractive to landowners at a carbon prices below \$10 per ton of CO_2, that afforestation requires a moderate price of \$15 per ton of CO_2 or more to induce landowners to participate, and that biofuels become dominant at prices of \$30 to \$50 per ton of CO_2 (Lewandrowski, 2004; Stavins and Richards, 2005; Environmental Protection Agency, 2005). Table 11-5 shows a simple scenario of emissions reduction below baseline, annualized over the time period from 2010 to 2110, for forestry activities as part of a bundle of reduction options for the land base.

Table 11-5. Illustrative emissions reduction potential of various forestry activities in the United States under a range of prices and sequestration rates.

Co-benefits are vitally important for inducing good forest carbon management. For example, conversion of agricultural land to forest will generally have positive effects on water, air, and soil quality and on biodiversity. In practice, some forest carbon sequestration projects have already been initiated even though sequestered carbon has little current value (Winrock International, 2005). In many of the current projects, carbon is a secondary objective that supports other landowner interests, such as restoration of degraded habitat. But co-effects may not all be beneficial. Water quantity may decline because of increased transpiration by trees relative to other vegetation. And taking land out of crop production may affect food prices—at higher carbon prices, nearly 40 million ha may be converted from cropland to forest (Environmental Protection Agency, 2005). Implementation of a forest carbon management policy will need to carefully consider co-effects, both positive and negative.

DATA GAPS AND INFORMATION NEEDS FOR DECISION SUPPORT

Decisions concerning carbon storage in North American forests and their management as carbon sources and sinks will be significantly improved by (1) filling gaps in inventories of carbon pools and fluxes, (2) a better understanding of how management practices affect carbon in forests, and (3) the increased availability of decision support tools for carbon management in forests.

Major Data Gaps in Estimates of Carbon Pools and Fluxes

Effective carbon management options to increase the retention time of sequestered carbon require a thorough understanding of current carbon stock sizes and flux rates in boreal, temperate, and tropical forest ecosystems in North America. However, major gaps exist in the data used to estimate the pools of carbon and carbon fluxes for the forests of Canada, the United States, and Mexico. These gaps complicate the prediction of how natural, social, and economic drivers will change carbon stocks and fluxes. Forests in an area as large as North America are quite diverse, and comprehensive data sets that better represent this diversity are needed.

In the United States, the range of estimates of the size of the land carbon sink is between 0.30 and 0.58 Mt C yr^{-1} (Pacala *et al*., 2001). Significant data gaps among carbon pools include carbon in wood products, soils, woody debris, and water transport (Birdsey 2004; Pacala *et al*., 2001). Geographic areas that are poorly represented in the available data sets include much of the Intermountain Western United States and Alaska, where forests of low productivity have not been inventoried as intensively as more productive timberlands (Birdsey 2004). Accurate quantification of the relative magnitude of various causal mechanisms at large spatial scales is not yet possible, given the limitations of our ability to combine various approaches and data sets: large-scale observations, process-based modeling, ecosystem experiments, and laboratory investigations (Foley and Ramankutty, 2004).

Large data gaps exist for Canada, particularly regarding changes in forest soil carbon and forestlands that are considered "unmanaged" (47% of forest lands). Aboveground biomass is better represented in forest inventories; however, the information needs to be updated and made more consistent among provinces. The new Canadian National Forest Inventory, currently under way, will provide a uniform coverage at a 20 × 20 km grid; it will be the basis for future forest carbon inventories. Data are also lacking on carbon fluxes, particularly those due to insect outbreaks and forest stand senescence. The ability to model forest carbon stock changes has considerably improved with the release of the CBM (Kurz *et al*., 2002); however the CBM does not consider climate change impacts (Price *et al*., 1999; Hogg and Bernier, 2005).

For Mexico, there is very little data about measured carbon stocks for all forest types. Information on forest ecosystem carbon fluxes is primarily based on deforestation rates, while fundamental knowledge of carbon exchange processes in almost all forest ecosystems is missing. That information is essential for understanding the effects of both natural and human-induced drivers (hurricanes, fires, insect outbreaks, climate change, migration, and forest management strategies), which all strongly impact the forest carbon cycle. Current carbon estimates are derived from studies in preferred sites in natural reserves with species-rich tropical forests. Therefore, inferences made from the studies on regional and national carbon stocks and fluxes probably give biased estimates on the carbon cycle.

Major Data Gaps in Knowledge of Forest Management Effects

With the exception of land use change (afforestation and deforestation), there is very little information available about how forest management affects various carbon pools, and there is some uncertainty about the longevity of effects (Caldeira *et al*., 2004). As with more general estimates of forest carbon pools and fluxes, there is more information available about effects on live biomass and woody debris than about soils and wood products. Agroforestry systems offer a promising economic alternative to slash-and-burn agriculture, including highly effective soil conservation practices and mid-term and long-term carbon mitigation options (Soto-Pinto *et al*., 2001; Nelson and de Jong, 2003; Albrecht and Kandji, 2003). However, a detailed assessment of current implementations of agroforestry systems in different regions of Mexico is missing. Refining management of forests to realize significant carbon sequestration while continuing to satisfy the other needs provided for by forests (e.g., timber, watershed management) will require a multi-criteria decision support framework for a holistic and adaptive management program of the carbon cycle in North American forests. This framework would necessarily influence considerations of policy and practice. Little is known about how this might be accomplished effectively, but given the importance of forests in the global carbon cycle, success in this endeavor could have important long-term and large-scale effects on global atmospheric carbon stocks.

Availability Of Decision-Support Tools

Few decision-support tools for managers are available, and they are either in early development modes or have been used primarily in research studies (Proctor *et al*., 2005; Potter *et al*., 2003). As markets emerge for trading carbon credits, and if credits for forest management activities have value, then the demand for decision-support tools will encourage their development.

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Table 11-1. Area of forest land by biome and country, 2000 (1000 ha)[1]

Ecological zone:	Canada[2]	U.S.[3]	Mexico[4]	Total
Tropical/subtropical	0	115,168	30,735	145,903
Temperate	101,100	142,445	32,851	276,396
Boreal/polar	303,000	45,461	0	348,461
Total	404,100	303,074	63,586	770,760

[1]There is 95% certainty that the actual values are within 10% of those reported in this table (e.g., for the United States see Bechtold and Patterson, 2005).
[2]Canadian Forest Service, 2005
[3]Smith *et al*., 2004
[4]Palacio *et al*., 2000

Table 11-2. Carbon stocks in forests by ecosystem carbon pool and country (Mt C)[1]

Ecosystem carbon pool:	Canada[2]	U.S.[3]	Mexico[4]	Total
Biomass	14,500	24,901	7,700	47,101
Dead organic matter[5]	71,300	41,674	11,400	124,374
Total	85,800	66,575	19,100	171,475

[1]There is 95% certainty that the actual values are within 25% of those reported in this table (Heath and Smith, 2000; Smith and Heath, 2000).
[2]Kurz and Apps, 1999
[3]Heath and Smith, 2004; Birdsey and Heath, 1995
[4]Masera *et al*., 2001
[5]Includes litter, coarse woody debris, and soil carbon

Table 11-3. Change in carbon stocks for forests and wood products by country (Mt C yr^{-1})

Carbon pool:	Canada[1]	U.S.[2]	Mexico[3]	Total
Forest Ecosystem	–99	–236	+52	–283
Wood Products	–10	–57	ND[4]	–67
Total	–109	–293	+52	–350

[1]Environment Canada (2005), Goodale *et al*. (2002). There is 95% certainty that the actual values are within 100% of those reported for Canada.
[2]From Smith and Heath, 2005 (excluding soils), and Pacala *et al*., 2001 (soils). Estimates do not include urban forests. There is 95% certainty that the actual values are within 50% of those reported for the United States.
[3]From Masera, 1997. There is 95% certainty that the actual values are within 100% of those reported for Mexico.
[4]Estimates are not available.

Table 11-4. Area of forestland by management class and country, 2000 (1000 ha)[1]

Management class:	Canada	U.S.	Mexico	Total
Protected	19,321	66,668	6,010	91,999
Plantation	4,486	16,238	150	20,874
Other	380,293	220,168	57,426	657,887
Total	404,100	303,074	63,586	770,760

[1]From Food and Agriculture Organization 2001; Natural Resources Canada 2005. Estimates in this table are within 10% of the true value at the 95% confidence level (e.g. for the U.S. see Bechtold and Patterson 2005).

Table 11-5. Illustrative emissions reduction potential of various forestry activities in the United States under a range of prices and sequestration rates[1]

Forestry activity	Carbon sequestration rate (t CO_2 ha^{-1} yr^{-1})	Price range ($/t CO_2)	Emissions reduction potential (Mt CO_2 yr^{-1})
Afforestation	5.4–23.5	15–30	137–823
Forest management	5.2–7.7	1–30	25–314
Biofuels	11.8–13.6	30–50	375–561

[1]Adapted from Environmental Protection Agency (2005). Maximum price analyzed was $50/t CO_2.

APPENDIX 11A
ECOSYSTEM CARBON FLUXES

The recent history of disturbance largely determines whether a forest system will be a net source or sink of C. For example, net ecosystem productivity (NEP, gains due to biomass growth minus losses due to respiration in vegetation and soil) is being measured across a range of forest types in Canada using the eddy covariance technique. In mature forests, values range from –19.6 t C ha^{-1} yr^{-1} in a white pine plantation in southern Ontario (Arain and Restrepo-Coupe, 2005) to –3.2 t C ha^{-1} yr^{-1} in a jack pine forest in (Amiro *et al*., 2005; Griffis *et al*., 2003). In recently disturbed forests, NEP ranges from +58.0 t C ha^{-1} yr^{-1} in a harvested Douglas-fir forest (Humphreys *et al*., 2005) to +5.7 t C ha^{-1} yr^{-1} in a 7 year old harvested jack pine forest (Amiro *et al*., 2005). In general, forest stands recovering from disturbance are sources of carbon until uptake from growth becomes greater than losses due to respiration, usually within 10 years (Amiro *et al*., 2005).

In the United States, extensive land-based measurements of forest/atmosphere carbon exchange reveal patterns and causes of sink or source strength (Table 11A-1). Results show that net ecosystem exchange (NEE) of carbon in temperate forests ranges from a source of +12.7 t C ha^{-1} yr^{-1} to a sink of –5.9 t C ha^{-1} yr^{-1}. Forests identified as sources are primarily forests in the earliest stages of regeneration (up to about 8 years) following stand-replacing disturbances such as wildfire and logging (Law *et al*., 2002). Mature temperate deciduous broadleaf forests and mature evergreen coniferous forests were an average sink of –2.7 and –2.5 t C ha^{-1} yr^{-1}, respectively (12 sites, 54 site-years of data). Values ranged from a source of +0.3 for a mixed deciduous and evergreen forest to a sink of –5.8 for an aggrading deciduous forest, averaged over multiple years. Young temperate evergreen coniferous forests (8 to 20 years) ranged from a sink of –0.6 to –5.9 t C ha^{-1} yr^{-1} (mean 3.1). These forests are still rapidly growing and have not reached the capacity for carbon uptake.

Mature forests can be substantial sinks for atmospheric carbon. Disturbances that replace or remove forests can result in the land being a net source of carbon dioxide for a few years in mild climates to 10–20 years in harsh climates while the forests are recovering (Law *et al*., 2004; Clark *et al*., 2004). Thus, the range of observed annual NEE of carbon dioxide ranges from a source of about +13 t C ha^{-1} yr^{-1} in a clearcut forest to a net sink of –6 t C ha^{-1} in mature temperate forests.

For Mexican forests, estimates of net ecosystem carbon exchange are unavailable, but estimates from other tropical forests may indicate rates for similar systems in Mexico. In Puerto Rico, aboveground NPP in tropical forests range from –9.2 to –11.0 t C ha^{-1} yr^{-1} (Lugo *et al*., 1999). Belowground NPP measurements exist for only one site with –19.5 t C ha^{-1} yr^{-1} (Lugo *et al*., 1999). In Hawaii, aboveground and belowground NPP of native forests dominated by *Metreosideros polymorpha* vary depending on substrate age and precipitation regime. Aboveground NPP ranges between –4.0 to –14.0 t C ha^{-1} yr^{-1}, while belowground NPP ranges between –5.2 and –9.0 t C ha^{-1} yr^{-1} (Giardina *et al*., 2004). Soil carbon emissions along the substrate age gradient range from +2.2 to +3.3 t C ha^{-1} yr^{-1}, and along the precipitation gradient from +4.0 to +9.7 t C ha^{-1} yr^{-1} (Osher *et al*., 2003). NEP estimates are not available for these tropical forests, so their net impact on atmospheric carbon stocks cannot be calculated.

Table 11A-1. Comparison of net ecosystem exchange (NEE) for different types and ages of temperate forests. Positive NEE means the forest is a sink for atmospheric CO_2. Eighty-one site years of data are from multiple published papers from each of the AmeriFlux network sites, and a network synthesis paper (Law *et al*. 2002). NEE was averaged by site, then the mean was determined by forest type and age class. SD is standard deviation among sites in the forest type and age class.

	NEE (t C ha^{-1} y^{-1})		
	Regenerating Clearcut (–1 ~ 3 years after disturbance) (1 site, 5 site-years)	Young forest (8 ~ 20 years old) (4 sites, 16 site-years)	Mature forest (>20 years old) (13 sites, 60 site-years)
Evergreen Coniferous Forests	–12.7 ~ 1.7, mean –7.1 (SD 4.7) (1 site, 5 site-years)	0.6 ~ 5.9, mean 3.1 (SD 2.6) (4 sites, 16 site-years)	0.6 ~ 4.5, mean 2.5 (SD 1.4) (6 sites, 20 site-years)
Mixed Evergreen and Deciduous Forests	NA	NA	0.3 ~ 2.1, mean –1.0 (SD 0.6) (1 site, 6 site-years)
Deciduous Broadleaf Forests	NA	NA	0.6 ~ 5.8, mean 2.7 (SD 1.8) (6 sites, 34 site-years)

APPENDIX 11B

PRINCIPLES OF FOREST MANAGEMENT FOR ENHANCING CARBON SEQUESTRATION

The net rate of carbon accumulation has been generally understood (Woodwell and Whittaker 1968) as the difference between gross primary production (gains) and respiration (losses), although this neglects important processes such as as leaching of DOC, emission of methane (CH4), fire, harvests or erosion that may contribute substantially to carbon loss and gain in forest ecosystems (Schulze *et al*., 1999; Harmon, 2001; Chapin *et al*., in review). The net ecosystem carbon balance (NECB) in forests is therefore defined as net ecosystem production, or NEP, plus the non-physiological horizontal and vertical transfers into and out of the forest stand.

With respect to the impacts of forest management on the overall carbon balance, some general principles apply (Harmon, 2001; Harmon and Marks, 2002; Pregitzer *et al*., 2004). First, forest management can impact carbon pool sizes via:

- changing production rates (since NEP = NPP—heterotrophic respiration Rh);
- changing decomposition flows (Rh) (e.g., Fitzsimmons *et al*. 2004);
- changing the amount of material transferred between pools; or
- changing the period between disturbances/ management activities.

The instantaneous balance between production, decomposition, and horizontal or vertical transfers into and out of a forest stand determines whether the forest is a net source or a net sink. Given that these terms all change as forests age, the disturbance return interval is a key driver of stand- and landscape-level carbon dynamics. Rh tends to be enhanced directly after disturbance, so as residue and other organic carbon pools decompose, a forest is often a net source immediately after disturbances such as management activity. NPP tends to increase as forests age, although in older forests it may decline (Ryan, 1997). Eventually, as stands age, NPP and Rh become similar in magnitude, although few managed stands are allowed to reach this age. The longer the average time interval between disturbances, the more carbon is stored. The nature of the disturbance is also important; the less severe the disturbance (e.g., less fire removal), the more carbon is stored.

Several less general principles can be applied to specific carbon pools, fluxes, or situations:

- Management activities that move live carbon to dead pools (such as CWD or soil C) over short periods of time will often dramatically enhance decomposition (Rh), although considerable carbon can be stored in decomposing pools (Harmon and Marks 2002). Regimes seeking to reduce the decomposition-related flows from residue following harvest may enhance overall sink capacity of these forests if these materials are used for energy generation or placed into forest products that last longer than the residue.
- Despite the importance of decomposition rates to the overall stand-level forest carbon balance, management of CWD pools is mostly impacted by recruitment of new CWD rather than by changing decomposition rates (Janisch and Harmon, 2002; Pregitzer and Euskirchen, 2004). Decreasing the interval between harvests can significantly decrease the store in this pool.
- Live coarse root biomass accounts for approximately 20–25% of aboveground forest biomass (Jenkins *et al*. 2003), and there is additional biomass in fine roots. Following harvest, this pool of live root biomass is transferred to the dead biomass pool, which can form a significant carbon store. Note that roots of various size classes and existing under varying environmental conditions decompose at different rates.
- Some carbon can be sequestered in wood products from harvested wood, though due to manufacturing losses only about 60% of the carbon harvested is stored in products (Harmon, 1996). Clearly, longer-lived products will sequester carbon for longer periods of time. The replacement of fossil fuel by biomass fuel can be counted as an emissions offset, if residue or manufacturing "waste" would otherwise be lost via decomposition or other processes. Faster-growing, larger trees (achieved via thinning, fertilization, or genetic improvement, for example) may also become products with longer lifespans, providing a positive feedback to carbon sequestration.

Little published research has been aimed at quantifying the impacts of specific forest management activities on carbon storage, but examples of specific management activities can be given.

Practices aimed at increasing NPP*:* fertilization; genetically improved trees that grow faster (Peterson *et al*., 1999); any management activity that enhances growth rate without causing a concomitant increase in decomposition (Stanturf *et al*., 2003; Stainback and Alavalapati, 2005).

Practices aimed at reducing Rh (i.e., minimizing the time forests are a source to the atmosphere following disturbance): low impact harvesting (that does not promote soil respiration); utilization of logging residues (biomass energy and fuels); incorporation of logging residue into soil during site prep (but note that this could also speed up decomposition); thinning to capture mortality; fertilization.

Since NECB changes with time as forests age, if a landscape is composed of stands with different ages then carbon gains in one stand can be offset by losses from another stand. The net result of these stand-level changes determines overall landscape-level carbon stores. Note that disturbance-induced Rh losses are typically larger than annual gains, such that a landscape where forest area is increasing might still be neutral with respect to carbon stocks overall. Thus, at the landscape level practices designed to enhance carbon sequestration must, on balance, replace lower-C-density systems with higher-C-density systems. Examples of these practices include: reducing fire losses; emphasizing very long-lived forest products; increasing the interval between disturbances; or reducing decomposability of dead material.

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