The Carbon Cycle in Land and Water Systems

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The six chapters (Chapters 10-15) in Part III consider the current and future carbon balance of terrestrial and aquatic ecosystems in North America. Although the amount of carbon exchanged between these ecosystems and the atmosphere each year through photosynthesis and plant and microbial respiration is large, the net balance for all of the ecosystems combined is currently a net sink of 370-505 million tons of carbon (Mt C) per year[1]. This net sink offsets only about 20-30% of current fossil-fuel emissions from the region (1856 Mt C per year in 2003) (Chapter 3 this report). The cause of this terrestrial carbon sink is uncertain. Although management has the potential for removing carbon from the atmosphere and storing it in vegetation and soil, most of the current sink is not the result of current management practices. Instead, most of it may be attributed to a combination of past management and the response of terrestrial ecosystems to environmental changes.

The large sink in the forests of Canada and the United States, for example, is, in some measure, the consequence of continued forest growth following agricultural abandonment that occurred in the past. This is partly the result of past and current management practices (*e.g.*, fire suppression), and partly the result of forest responses to a changing environment (climatic change, carbon dioxide [CO_2] fertilization, and the increased mobilization of nutrients). The relative importance of these broad factors in accounting for the current sink is unknown. Estimates vary from attributing nearly 100% of the sink in United States forests to regrowth (Caspersen *et al.*, 2000; Hurtt *et al.*, 2002) to attributing nearly all of it to CO_2 fertilization (Schimel *et al.*, 2000). The attribution question is critical because the current sink may be expected to increase in the future if the important mechanism is CO_2 fertilization, for example, but may be expected to decline if the important mechanism is forest regrowth (forests accumulate carbon more slowly as they age). Understanding the history of land use, management, and disturbance is critical because disturbance and recovery are major determinants of the net terrestrial carbon flux.

Land-use change and management have been, and will be, important in the carbon balance of other ecosystems besides forests. The expansion of cultivated lands in Canada and the United States in the 1800s released large amounts of carbon to the atmosphere (Houghton *et al.*, 1999), leaving those lands with the potential for recovery (*i.e.*, a future carbon sink), if managed properly. For example, recent changes in farming practice may have begun to recover the carbon that was lost decades ago. Recovery of carbon in soil, however, generally takes longer than its loss through cultivation. Grazing lands, although not directly affected by cultivation, have, nevertheless, been managed in the United States through fire suppression. The combined effects of grazing and fire suppression are believed to have promoted the invasion of woody vegetation, possibly a carbon sink at present. Wetlands are also a net carbon sink, but the magnitude of the sink was larger in the past than it is today, again, as a result of land-use change (draining of wetlands for agriculture and forestry). The only lands that seem to have escaped management are those lands overlying permafrost (perennially frozen ground), and they are clearly subject to change in the future as a result of global warming. Settled lands, by definition, are managed, and are dominated by fossil-fuel emissions. Nevertheless, the accumulation of carbon in urban and

Understanding the history of land use, management, and disturbance is critical because disturbance and recovery are major determinants of the net terrestrial carbon flux.

[1] The lower estimate is from this overview, the larger estimate from Chapter 3, with most of the difference attributable to uncertainty in the sink from woody encroachment. See Table III.1, footnote h, for discussion of this range.

suburban trees suggests a net sequestration of carbon in the biotic component of long-standing settled lands. Residential lands recently cleared from forests, on the other hand, are sources of carbon (Wienert and Hamburg, 2006).

From the perspective of carbon and climate, ecosystems are important if (1) they are currently large sources or sinks of carbon or (2) they have the potential to become large sources or sinks of carbon in the future through either management or environmental change, where "large" sources or sinks, in this context, are determined by the product of area (hectares) times flux per unit area (or flux density) (megagrams of carbon [Mg C] per hectare per year).

The largest carbon sink in North America (270 Mt C per year) is associated with forests (Chapter 11 this report) (Table III-1). The sink includes the carbon accumulating in wood products (*e.g.*, in increasing numbers of houses and landfills) as well as in the forests themselves. A sink is believed to exist in wetlands (Chapter 13 this report), including the wetlands overlying permafrost (Chapter 12 this report), although the magnitude of this sink is uncertain. More certain is the fact that the current sink is considerably smaller than it was before wetlands were drained for agriculture and forestry. The other important aspect of wetlands is that they hold more than half of the carbon in North America. Thus, despite the current net sink in these systems, their potential for future emissions is large.

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Although management has the potential to increase the carbon sequestered in agricultural (cultivated) lands, these lands today are nearly in balance with respect to carbon (Chapter 10 this report). The carbon lost to the atmosphere from cultivation of organic soils (soils dominated by organic matter) is approximately balanced by the carbon accumulated in mineral soils (soils consisting of more inorganic material, such as sand or clay). In the past, before cultivation, these soils held considerably more carbon than they do today, but 25-30% of that carbon was lost soon after the lands were initially cultivated. In large areas of grazing lands, there is the possibility that the invasion and spread of woody vegetation (woody encroachment) is responsible for a significant net carbon sink at present (Chapter 10 this report). The magnitude (and even sign) of this flux is uncertain, however, in part because some ecosystems lose carbon below-ground (soils) as they accumulate it aboveground (woody vegetation), and in part because the invasion and spread of exotic grasses into semi-arid lands of the western United States are increasing the frequency of fires, reversing woody encroachment, and releasing carbon (Bradley *et al.*, 2006).

The emissions of carbon from settled lands are largely considered in the chapters in Part II and in Chapter 14 of this report. Non-fossil carbon seems to be accumulating in trees in these lands, but the net changes in soil carbon are uncertain.

The only ecosystems that appear to release carbon to the atmosphere at present are the coastal waters. The estimated flux of carbon is close to zero (and difficult to determine) because the gross fluxes (from river transport, photosynthesis, and respiration) are large and variable in both space and time.

The average net fluxes of carbon expressed as Mg C per hectare per year in Table III-1 are for comparative purposes. They show the relative flux density for different types of ecosystems. These annual fluxes of carbon are rarely determined with direct measurements of flux, however, because of the extreme variability of fluxes in time and space, even within a single ecosystem type. Extrapolating from a few isolated measurements to an estimate for the whole region's flux is difficult. Rather, the net changes are more often based on differences in measured stocks over intervals of 10 years, or longer (Chapter 3 this report), or are based on the large and rapid changes per hectare that are reasonably well documented for certain forms of management, such as the changes in carbon stocks that result from the conversion of forest to cultivated land. Thus, most of the flux estimates in Table III-1 are long-term and large-area estimates.

Nevertheless, average flux density is one factor important in determining an ecosystem's role as a net source or sink for carbon. The other important factor is area. Permafrost wetlands, for example, are currently a small net sink for carbon. They cover a large area, however, hold large stocks of carbon, and, thus, have the potential to become a significant net source of carbon if the permafrost thaws with global warming (Smith *et al.*, 2001; Smith *et al.*, 2005a; Osterkamp and Romanovsky, 1999; Osterkamp *et al.*, 2000). Forests clearly dominate the net uptake and storage of carbon in North America, although wetlands and settled lands have mean flux densities that are above average.

The two factors (flux density and area) demonstrate the level of management required to remove a significant amount of carbon from the atmosphere and keep it on land. Under current conditions, sequestration of 100 Mt C per year, for example (about 7% of fossil-fuel emissions from North America), requires nearly half the forest area (Table III-1). As discussed above, the cause of this sequestration is uncertain, but enhancing it through management over a few hundred million hectares would require considerable effort. Nevertheless, the cost (in \$/metric ton CO_2) may be low relative to other options for managing carbon. For example, forestry activities are estimated to have the potential to sequester 100-200 Mt C per year in the United States at prices ranging from less than \$10/ton of CO_2 for improved forest management, to \$15/ton for afforestation, to \$30-50/ton for production of biofuels (Chapter 11 this report). Somewhat smaller sinks of 10-70 Mt C per year might be stored in agricultural soils at low to moderate costs (\$3-30/ton CO_2) (Chapter 10 this report). The maximum amounts of carbon that might be accumulated in forests and agricultural soils are not known, thus, the number of years these rates of sequestration might be expected to continue is also unknown. It seems unlikely that the amount of carbon currently held in forests and agricultural lands could double. Changes in climate will also affect carbon storage, but the net effect of management and climate is uncertain.

Table III.1 Ecosystems in North America: their areas, net annual fluxes of carbon (negative values are sinks), and carbon stocks (including both vegetation and soils).

Type of ecosystem	Area (10^6 ha)	Current mean flux density (Mg C per ha per year)	Current flux (Mt C per year)	Carbon stocks (Mt C)	Mean carbon stocks (Mt C per ha)
Agriculture	231	0.0	0±15[a]	18,500	80
Grass, shrub and arid	558	–0.01	–6[b]	59,950	107
Forests	771	–0.35	–269[c]	171,500	222
Permafrost lands					
Peatlands	51	–0.13	–6.7	57,700	1130
Mineral soils[d]	517	–0.03	–14	98,780	191
Non-permafrost wetlands					
Peatlands	86	–0.12	–10	126,400	1470
Mineral soils	105	–0.21	–22.3	38,100	363
Estuarine	4.5	–2.3	–10.2	900	200
Settled lands[e]	104	–0.31	–32	~1,000	10
Coastal waters	384	0.05	19		
Sum	2427[f]	–0.15[g]	–370[h]	572,830[f]	
Total	2126[i]			480,000[j]	225[g]

a. Fossil-fuel inputs to crop management are not included. Some of the carbon sequestration is occurring on grasslands as well as croplands, but the inventories do not separate these fluxes. The near-zero flux is for Canada and the United States only. Including Mexican croplands would likely change the flux to a net source because croplands are expanding in Mexico, and the carbon in biomass and soil is released to the atmosphere as native ecosystems are cultivated.

b. Fossil-fuels are not included. The small net sink results from the Conservation Reserve Program in the United States. Including Mexico is likely to change the net sink to a source because forests are being converted to grazing lands. Neither woody encroachment nor woody elimination is included in this estimate of flux because the uncertainties are so large.

c. Includes an annual sink of 68 Mt C per year in wood products as well as a sink of 201 Mt C per year in forested ecosystems.

d. Includes zones with continuous, discontinuous, sporadic, and isolated permafrost; that is, not all of the lands are strictly over permafrost.

e. Urban trees only (does not include soil carbon). Note that this sink is accounted for as part of the forest sink in Chapter 3 (Table 3.1).

f. Sum does not include coastal waters. The summed area is larger than the total area (note i) because of double counting. For example, an estimated 75 × 10^6 hectares (ha) of permafrost lands in Canada are forested (and may be included in forest area as well as permafrost area), 26 x 10^6 ha of wetlands in the United States are forested, and 54 x 10^6 ha of wetlands are shrublands. In addition, an estimated 75 x 10^6 ha of other wooded lands are included as both forests and rangelands, and ~70 x 10^6 ha of grasslands and shrublands are counted also as non-permafrost lands within areas defined as sporadic or isolated permafrost (see note d).

g. Weighted average; does not include coastal waters.

h. Does not include coastal waters. The total annual sink of 370 Mt C is lower than the estimate of 505 Mt C presented in Chapter 3 (Table 3.1). The largest difference results from the flux of carbon attributed to woody encroachment. Chapter 3 includes a sink of 120 Mt C per year; Table III-1, above, presents a net flux of zero (see note b). Other differences between the two estimates include: (1) an additional sink in Table III-1 of 14 Mt C per year in permafrost mineral soils and (2) a sink of 25 Mt C per year in rivers and reservoirs that is included in Table 3.1 but not in Table III-1. In addition, there are small differences in the estimates for agricultural lands and grasslands.

i. Areas (10^6 ha) (*The Times Atlas of the World*, 1990)

Globe	North America	Canada	United States	Mexico
14,900	2,126	992	936	197

j. Total carbon stocks are reduced by the areas double counted (see note f).

Despite the limited nature of carbon uptake and storage in offsetting the global emissions of carbon from fossil fuels, local and regional activities may, nevertheless, offset local and regional emissions of fossil carbon. This offset, as well as other co-benefits, may be particularly successful in urban and suburban systems (Chapter 14 this report).

The effects and cost of managing aquatic systems are less clear. Increasing the area of wetlands, for example, would presumably increase the sequestration of carbon; but it would also increase emissions of methane (CH_4), countering the effect of carbon storage. Fertilization of coastal waters with iron has been proposed as a method for increasing oceanic uptake of CO_2, but neither the amount of carbon that might be sequestered nor the side effects are known (Chapter 15 this report).

A few studies have estimated the potential magnitudes of future carbon sinks as a result of management (Chapters 10, 11 this report). However, the contribution of management, as opposed to the environment, in today's sink is unclear (Chapter 3 this report), and for the future, the relative roles of management and environmental change are even less clear. The two drivers might work together to enhance terrestrial carbon sinks, as seems to have been the case during recent decades (Prentice *et al.*, 2001) (Chapter 2 this report). On the other hand, they might work in opposing directions. A worst-case scenario, quite possible, is one in which management will become ineffective in the face of large natural sources of carbon not previously experienced in the modern world. In other words, while management is likely to be essential for sequestering carbon, it may not be sufficient to preserve the current terrestrial carbon sink over North America, let alone to offset fossil-fuel emissions.

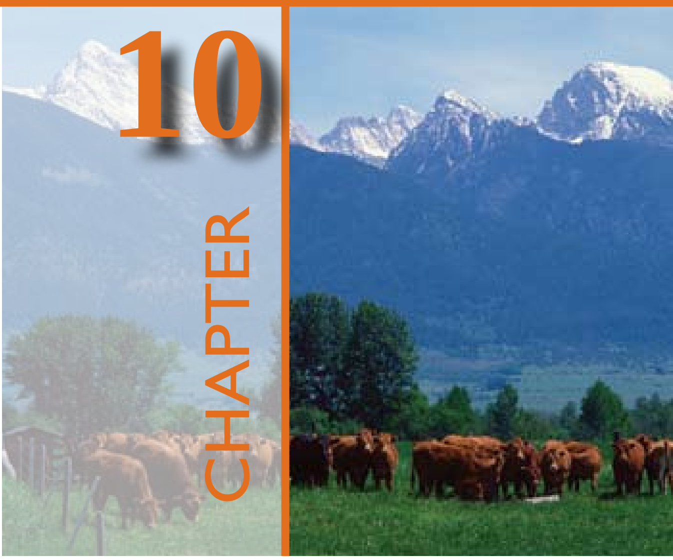
At least one other observation about storing carbon in terrestrial and aquatic ecosystems should be mentioned. In contrast to the hundreds of millions of hectares that must be managed to sequester 100 Mt C annually, a few million hectares of forest fires can release an equivalent amount of carbon in a single year. This disparity in flux densities underscores the fact that a few million hectares are disturbed each year, while hundreds of millions of hectares are recovering from past disturbances. The natural fluxes of carbon are large in comparison to net fluxes. The observation is relevant for carbon management, because the cumulative effects of managing small net sinks to mitigate fossil-fuel emissions will have to be understood, analyzed, monitored, and evaluated in the context of larger, highly variable, and uncertain sources and sinks in the natural cycle.

The major challenge for future research is quantification of the mechanisms responsible for current (and future) fluxes of carbon. In particular, what are the relative effects of management (including land-use change), environmental change, and natural disturbance in determining sources and sinks of carbon for today and tomorrow? Will the current natural sinks continue, grow in magnitude, or reverse to become net sources? What is the role of soils in the current (and future) carbon balance (Davidson and Janssens, 2006)? What are the most cost-effective means of managing carbon?

Answering these questions will require two scales of measurement: (1) an expanded network of intensive research sites dedicated to understanding basic processes (*e.g.*, the effects of management and environmental effects on carbon stocks), and (2) extensive national-level networks of monitoring sites, through which uncertainties in carbon stocks (inventories) would be reduced and changes, directly measured. Elements of these measurements are underway, but the effort has not yet been adequate for resolving these questions.

KEY UNCERTAINTIES AND GAPS IN UNDERSTANDING THE CARBON CYCLE OF NORTH AMERICA

- As mentioned above, the net flux of carbon resulting from woody encroachment and its inverse, woody elimination, is highly uncertain. Even the sign of the flux is in question.
- Rivers, lakes, dams, and other inland waters are mentioned in Chapter 15 as being a source of carbon, but they are claimed elsewhere to be a sink (Chapter 3 this report). The sign of the net carbon flux attributable to erosion, transport, deposition, accumulation, and decomposition is uncertain (*e.g.*, Stallard, 1998; Lal, 2001; Smith *et al.*, 2005b).
- Several chapters cite studies that have attempted to quantify the potential for management to increase carbon sinks in the future, but no studies have yet attempted to estimate the potential future sources of carbon for North America as they have for the globe (*e.g.*, Friedlingstein *et al.*, 2006; Jones *et al.*, 2005). Global models that include the feedbacks between climatic change and the carbon cycle have all shown decreased carbon sinks over the next century. In North America, warming of wetlands and thawing of permafrost, in particular, are likely to increase emissions of carbon to the atmosphere, CH_4 as well as CO_2; and periods of unusually low rainfall, combined with warming trends, are likely to release carbon from the ecosystems of the Mountain West and the southwestern United States through increasing their vulnerability to wildfires and insect outbreaks (Potter *et al.*, 2003 and 2005).



CHAPTER 10

Agricultural and Grazing Lands

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KEY FINDINGS

- Agricultural and grazing lands (cropland, pasture, rangeland, shrublands, and arid lands) occupy 789 million hectares (1.95 billion acres), which is 47% of the land area of North America, and contain 78.5 ± 19.5[1] billion tons of organic carbon (17% of North American terrestrial carbon) in the soil alone.
- The emissions and uptake and storage of carbon on agricultural lands are mainly determined by two conditions: management and changes in the environment. The effects of converting forest and grassland to agricultural lands and of agricultural management (e.g., cultivation, conservation tillage) are reasonably well known and have been responsible for historic losses of carbon in Canada and the United States (and for current losses in Mexico); the effects of climate change or of elevated concentrations of atmospheric carbon dioxide are uncertain.
- Conservation-oriented management of agricultural lands (e.g., use of conservation tillage, improved cropping and grazing systems, reduced bare fallow, set-asides of fragile lands, and restoration of degraded soils) can significantly increase soil carbon stocks.
- Agricultural and grazing lands in the United States and Canada are currently near neutral with respect to their soil carbon balance, but agricultural and grazing lands in Mexico are likely losing carbon due to land-use change. Although agricultural soils are estimated to currently uptake about 19-20 million tons of carbon per year, the cultivation of organic soils releases approximately 6-12 million tons of carbon per year. On-farm fossil-fuel use (around 31 million tons of carbon per year), agricultural liming (1.2 million tons of carbon per year), and manufacture of agricultural inputs including fertilizer (approximately 6 million tons of carbon per year) yields a net source from the agricultural sector of about 25-30 million tons of carbon per year.
- As much as 120 million tons of carbon per year may be accumulating through woody encroachment of arid and semi-arid lands of North America; this value is highly uncertain. Woody encroachment is generally accompanied by decreased forage production, and ongoing efforts to reestablish forage species are likely to reverse carbon accumulation by vegetation.
- Projections of future trends in agricultural land area and soil carbon stocks are unavailable or highly uncertain because of uncertainty in future land-use change and agricultural management practice.
- Annualized prices of $15/metric ton carbon dioxide, could yield mitigation amounts of 46 million tons of carbon per year captured in agricultural soils and 14.5 million tons of carbon per year from reductions in fossil-fuel use. At lower prices of $5/metric ton carbon dioxide, the corresponding values would be 34 million tons of carbon per year and 9 million tons of carbon per year, respectively.
- Policies designed to suppress emissions of one greenhouse gas need to consider complex interactions to ensure that *net* emissions of total greenhouse gases are reduced. For example, increased use of fertilizer or irrigation may increase crop residues and carbon uptake and storage, but may stimulate emissions of methane or nitrous oxide.

[1] The uncertainty in this value is given as one standard error of the mean.

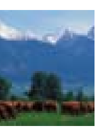
- Many of the practices that lead to carbon capture and storage or to reduced carbon dioxide and methane emissions from agricultural lands not only increase production efficiencies, but lead to environmental co-benefits, for example, improved soil fertility, reduced erosion, and pesticide immobilization.
- An expanded network of intensive research sites would allow us to better understand the effects of management on carbon cycling and storage in agricultural systems. An extensive national-level network of soil monitoring sites in which changes in carbon stocks are directly measured would allow us to reduce the uncertainty in the inventory of agricultural and grazing land carbon. Better information about the spatial extent of woody encroachment, the amount and growth of woody vegetation, and variation in impacts on soil carbon stocks would help reduce the large uncertainty of the carbon impacts of woody encroachment.

BOX 10.1: Nitrous Oxide Emissions From Agricultural and Grazing Lands

Nitrous oxide (N_2O) is the most potent greenhouse gas in terms of global warming potential, with a radiative forcing 296 times that of CO_2 (IPCC, 2001). Agricultural activities that add mineral or organic nitrogen (fertilization, plant N_2 fixation, manure additions, *etc.*) augment naturally occurring N_2O emissions from nitrification and denitrification by 0.0125 kg N_2O per kg nitrogen applied (Mosier *et al.*, 1998a). Agriculture contributes significantly to total global N_2O fluxes through soil emissions (35% of total global emissions), animal waste handling (12%), nitrate leaching (7%), synthetic fertilizer application (5%), grazing animals (4%), and crop residue management (2%). Agriculture is the largest source of N_2O in the United States (78% of total N_2O emissions), Canada (59%), and Mexico (76%).

10.1 INVENTORY

10.1.1 Background



Agricultural and grazing lands (cropland, pasture, rangeland, shrublands, and arid lands)[2] occupy 47% of the land area in North America (59% in the United States, 70% in Mexico, and 11% in Canada), and contain 17% of the terrestrial carbon. Most of the carbon in these ecosystems is held in soils. Live vegetation in cropland generally contains less than 5% of total carbon, whereas vegetation in grazing lands contains a greater proportion (5–30%), but still less than that in forested systems (30–65%). Agricultural and grazing lands in North America contain 78.5 ± 19.5 (±1 standard error) billion tons of organic carbon (Gt C) in the soil (Table 10.1). Significant increases in vegetation carbon stocks in some grazing lands have been observed and, together with soil carbon stocks from croplands and grazing lands, likely contribute significantly to the large North American terrestrial carbon sink (Houghton *et al.*, 1999; Pacala *et al.*, 2001; Eve *et al.*, 2002; Ogle *et al.*, 2003). These lands also emit greenhouse gases: fossil-fuel use for on-farm machinery and buildings, for manufacture of agricultural inputs, and for transportation account for 3–5% of total carbon dioxide (CO_2) emissions in developed countries (Enquete Commission, 1995); activities on agricultural and grazing lands, like livestock production, animal waste management, biomass burning, and rice cultivation emit 35% of global anthropogenic methane (CH_4) (27% of United States', 31% of Mexican, and 27% of Canadian CH_4 emissions) (Mosier *et al.*, 1998b; CISCC, 2001; Ministry of the Environment, 2006; EPA, 2006); and agricultural and grazing lands are the largest anthropogenic source of nitrous oxide (N_2O) emissions (CAST, 2004; see Box 10.1). However, agricultural and grazing lands are actively managed and have the capacity to take up and store carbon. Thus improving management could lead to substantial reductions in CO_2 and CH_4 emissions and could sequester carbon to offset emissions from other lands or sectors.

Agricultural and grazing lands are actively managed and have the capacity to take up and store carbon. Thus improving management could lead to substantial reductions in CO_2 and CH_4 emissions.

10.1.2 Carbon Dioxide Fluxes From Agricultural and Grazing Land

The main processes governing the carbon balance of agricultural and grazing lands are the same as for other ecosystems: the photosynthetic uptake and assimilation of CO_2 into organic compounds, the release of gaseous carbon through respiration (primarily CO_2 but also CH_4), and fire. Like other terrestrial ecosystems in general, for which CO_2 emissions are approximately two orders of magnitude greater than CH_4 emissions, carbon cycling in most agricultural and grazing lands is dominated by fluxes of CO_2 rather than CH_4. In agricultural lands, carbon assimilation is directed towards production of food, fiber, and forage by manipulating species composition and growing conditions (soil fertility, irrigation, *etc.*). Biomass, being predominantly herbaceous (*i.e.*, non-woody), is a small, transient carbon pool (compared to forests) and hence soils constitute the dominant carbon stock. Cropland systems can be among the most productive ecosystems, but in some cases restricted growing season length, fallow periods, and grazing-induced shifts in species

[2] We refer collectively to pasture, rangeland, shrublands, and arid lands as grazing lands since grazing is their primary use, even though not all of these lands are grazed.

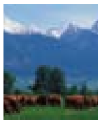
Table 10.1 Soil organic carbon pools in agricultural and grazing lands in Canada, Mexico, and the United States. The data values are given in Gt C. The area (in millions of hectares) for each climatic zone is in parentheses. Current soil carbon stocks are secondary quantities derived from an initial starting point of undisturbed native ecosystems carbon stocks, which were quantified using the intersection of (Moderate Resolution Imaging Spectroradiometer-International Geosphere-Biosphere Programme) MODIS-IGBP[a] land cover types (Friedl *et al.*, 2002) and mean soil carbon contents to 1-m depth from Sombroek *et al.* (1993), spatially arrayed using Food and Agriculture Organization soil classes (ISRIC, 2002), and summed by climate zone. These undisturbed native ecosystem carbon stock values were then multiplied by soil carbon loss factors for tillage- and overgrazing-induced losses (Nabuurs *et al.*, 2004; Ogle *et al.*, 2004) to estimate current soil carbon stocks (see Figure 10.2). Uncertainties (± one standard error) were derived from uncertainty associated with soil carbon stocks and soil carbon loss factors.

Practice	Temperate dry[b,c]	Temperate wet	Tropical dry	Tropical wet	Total
		Agricultural lands			
Canada	1.79±0.35 (17.3)	1.77±0.36 (22.1)	–	–	3.60±0.77 (39.4)
Mexico	–	–	0.24±0.06 (3.9)	0.53±0.14 (10.2)	0.81±0.22 (14.1)
United States	3.31±0.74 (34.8)	8.66±2.18 (108.4)	0.35±0.08 (5.6)	1.53±0.33 (28.4)	14.05±3.20 (177.1)
Total	**5.16±1.07 (52.1)**	**10.57±2.42 (130.5)**	**0.61±0.14 (9.5)**	**2.18±0.54 (38.6)**	**18.5±4.16 (230.6)**
		Grazing lands			
Canada	2.17±0.55 (18.4)	9.49±1.27 (40.8)	–	–	11.66±4.88 (59.2)
Mexico	–	–	7.20±1.62 (99.1)	2.19±0.58 (20.3)	9.99±2.60 (119.4)
United States	16.89±3.62 (209.9)	5.67±1.39 (55.0)	4.26±0.98 (68.1)	4.30±0.89 (46.7)	32.88±7.18 (379.7)
Total	**19.34±4.27 (228.3)**	**21.07±5.80 (95.8)**	**12.59±2.73 (167.1)**	**6.94±1.86 (67.0)**	**59.95±14.65 (558.2)**

[a] Cropland area was derived from the IGBP cropland land cover class plus the area in the cropland/natural vegetation IGBP class in Mexico and one-half of the area in the cropland/natural vegetation IGBP class in Canada and the United States. Grazing land area includes IGBP woody savannas, savannas, and grasslands in all three countries, plus open shrubland in Mexico and open shrublands (not in Alaska) in the United States.

[b] Temperate zones are those located above 30° latitude. Tropical zones (below 30° latitude) include subtropical regions.

[c] Dry climates were defined as those where the ratio of mean annual precipitation (MAP) to potential evapotranspiration (PET) is less than one; in wet areas, MAP/PET is greater than one.



BOX 10.2: Inorganic Soil Carbon in Agricultural and Grazing Ecosystems

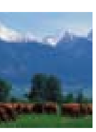
Inorganic carbon in the soil is comprised of primary carbonate minerals, such as calcite ($CaCO_3$) or dolomite ($CaMg[CO_3]_2$), or secondary minerals formed when carbonate (CO_3^{2-}), derived from soil CO_2, combines with base cations (*e.g.*, Ca^{2+}, Mg^{2+}) and precipitates within the soil profile in arid and semi-arid ecosystems. Weathering of primary carbonate minerals in humid regions can be a source of CO_2, whereas formation of secondary carbonates in drier areas is a sink for CO_2; however, the magnitude of either flux is highly uncertain. Agricultural liming involves addition of primary carbonate minerals to the acid soils to increase the pH. In Canada and the United States, about 0.1 and 1.1 Mt C per year is emitted from liming (Sobool and Kulshreshtha, 2005; EPA, 2006). Inorganic carbon stocks in North America have been estimated at 66.8 Gt C (Sombroek *et al.*, 1993).

composition or production can reduce carbon uptake relative to that in other ecosystems. These factors, along with tillage-induced soil disturbances and removal of plant carbon through harvest, have depleted soil carbon stocks by 20-40% (or more) from pre-cultivated conditions (Davidson and Ackerman, 1993; Houghton and Goodale, 2004). Soil organic carbon stocks in grazing lands (see Box 10.2 for information on inorganic soil carbon stocks) have been depleted to a lesser degree than for cropland (Ogle *et al*., 2004), and in some regions biomass has increased due to suppression of disturbance and subsequent woody encroachment (see Box 10.3). Woody encroachment is potentially a significant sink for atmospheric CO_2, but the magnitude of the sink is poorly constrained (Houghton *et al*., 1999; Pacala *et al*., 2001). Since woody encroachment leads to decreased forage production, management practices are aimed at reversing it, with consequent reductions in biomass carbon. Disturbance-induced increases in decomposition rates of above-ground litter and harvest removal of some (30–50% of forage in grazing systems, 40–50% in grain crops) or all (*e.g.*, corn for silage) of the above-ground biomass, have drastically altered carbon cycling within agricultural lands and thus the sources and sinks of CO_2 to the atmosphere.

BOX 10.3: Impacts of Woody Encroachment Into Grasslands on Ecosystem Carbon Stocks

Encroachment of woody species into grasslands—caused by overgrazing-induced reduction in grass biomass and subsequent reduction or elimination of grassland fires—is widespread in the United States and Mexico, decreases forage production, and is unlikely to be reversed without costly mechanical intervention (Van Auken, 2000). Encroachment of woody species into grassland tends to increase biomass carbon stocks by one million grams of carbon (1 Mg C) per hectare per year (Pacala *et al.*, 2001), with estimated net sequestration of 120–130 Mt C per year in encroaching woody biomass (Houghton *et al.*, 1999; Pacala *et al.*, 2001). In response to woody encroachment, soil organic carbon stocks can significantly increase or decrease, thus predicting impacts on soil carbon or ecosystem carbon stocks is very difficult (Jackson *et al.*, 2002). Invasion of grass species into native shrublands tends to lead to the release of soil organic carbon (Bradley *et al.*, 2006).

Much of the carbon lost from agricultural soil and biomass pools can be recovered with changes in management practices.



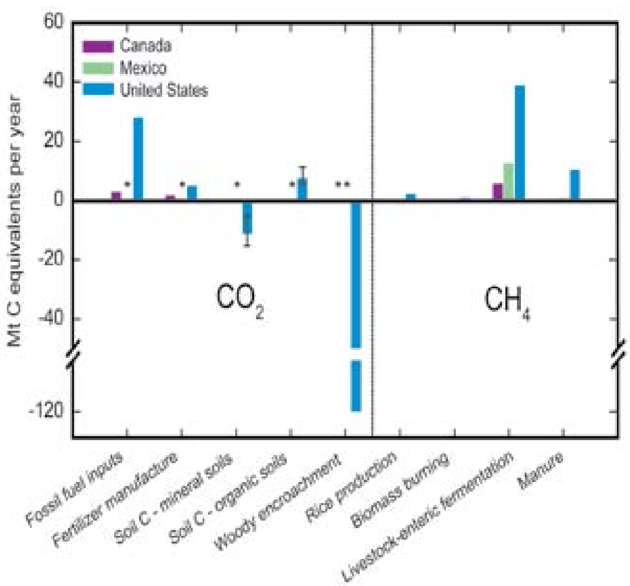


Figure 10.1 North American agricultural and grazing land CO_2 (left side) and CH_4 (right side), adjusted for global warming potential. All units are in Mt C-equivalent per year for years around 2000. Negative values indicate net flux from the atmosphere to soil and biomass carbon pools (*i.e.*, sequestration). All data are from Canadian (Matin *et al.,* 2004) and U.S. (EPA, 2006) National Inventories and from the second Mexican National Communication (CISCC, 2001), except for Canadian (from Kulshreshtha *et al.,* 2000) and U.S. fossil-fuel inputs (from Lal *et al.,* 1998) and woody encroachment (from Houghton *et al.,* 1999). Values are for 2003 for Canada, 1998 for Mexico, and 2004 for the United States. A global warming potential of 23 for methane was used to convert emissions of CH_4 to CO_2 equivalents (IPCC, 2001) and a factor of 12/44 to convert from CO_2 to carbon. Asterisks indicate unavailable data. Data ranges are indicated by error bars where available.

Much of the carbon lost from agricultural soil and biomass pools can be recovered with changes in management practices that increase carbon inputs, stabilize carbon within the system, or reduce carbon losses, while still maintaining outputs of food, fiber, and forage. Increased production, increased residue carbon inputs to the soil, and increased organic matter additions have reversed historic soil carbon losses in long-term experimental plots (*e.g.*, Buyanovsky and Wagner, 1998). However, the management practices that promote soil carbon sequestration would need to be maintained over time to avoid subsequent losses of sequestered carbon. Across Canada and the United States, mineral soils have been sequestering 2.5† and 17.0 ± 0.45 million metric tons of carbon (Mt C) per year[3] (Ministry of the Environment, 2006; Ogle *et al*., 2003; EPA, 2006), respectively, largely through increased production and improved management practices on annual cropland (Figure 10.1, Table 10.2). Conversion of agricultural land to grassland, like under the Conservation Reserve Program in the United States (7.6–11.5 Mt C per year on 31.5 million acres [12.5 million hectares] of land), and afforestation have also sequestered carbon in agricul-

[3] † A dagger symbol indicates that the magnitude and/or range of uncertainty for the given numerical value(s) is not provided in the references cited.

Table 10.2 North American agricultural and grazing land carbon fluxes for the years around 2000. All units are in Mt C per year. Negative numbers (in parentheses) indicate net flux from the atmosphere to soil and biomass carbon pools. Unless otherwise noted, data are from Canadian (Matin *et al.*, 2004) and United States' National Inventories (EPA, 2006), and from the Second Mexican National Communication (CISCC, 2001). Values are for 2003 for the United States and Canada, and 1998 for Mexico. A factor of 12/44 was used to convert from CO_2 to carbon and a factor of 12/16 to convert CH_4 to carbon

	Canada	Mexico	United States	Total
CO_2				
On-farm fossil-fuel use	2.9[a]	ND	28[b]	30.9
Fertilizer manufacture	1.7	ND	4.7	6.4
Mineral soil carbon sequestration	(2.5)	ND	(17±0.45)	(19.1) – (20.0)
Organic soil cultivation	0.1	ND	8.3±3.2	5.6 – 11.9
Agricultural liming	0.1	ND	1.1	1.2
Woody encroachment	ND	ND	(120)[c]	(120)
Total	2.3	ND	(114.7) – (120.1)	(117) – (122.4)
CH_4				
Rice production	0	0.011	0.25±0.28	0.26
Biomass burning	<0.01	<0.01	0.03±0.02	0.05
Livestock	0.62	1.48	3.67±0.53	5.93
Manure	0.18	0.05	1.28±0.24	1.60
Total	0.80	1.54	5.23	7.84

ND = no data reported.
[a] From Kulshreshtha *et al.* (2000).
[b] From Lal *et al.* (1998).
[c] From Houghton *et al.* (1999).

tural and grazing lands (Follett *et al.*, 2001a). In contrast, cultivation of organic soils (*e.g.*, peat-derived soils) is releasing an estimated 0.1 and 8.3 ± 3.2 Mt C per year† from soils in Canada and the United States (Matin *et al.*, 2004; Ministry of the Environment, 2006; Ogle *et al.*, 2003; EPA, 2006). Compared with other systems, the high productivity and management-induced disturbances of agricultural systems promote movement and redistribution (through erosion, runoff, and leaching) of organic and inorganic carbon, sequestering potentially large amounts of carbon in sediments and water (Raymond and Cole, 2003; Smith *et al.*, 2005; Yoo *et al.*, 2005). However, the net impact of soil erosion on carbon emissions to the atmosphere remains highly uncertain.

Production, delivery, and use of field equipment, fertilizer, seed, pesticides, irrigation water, and maintenance of animal production facilities contribute 3–5% of total fossil-fuel CO_2 emissions in developed countries (Enquete Commission, 1995). On-farm fossil-fuel emissions together with manufacture of fertilizers and pesticides contribute emissions of 32.7 Mt C per year† within the United States (Lal *et al.*, 1998) and 4.6 Mt C per year in Canada (Kulshreshtha *et al.*, 2000) (Table 10.2). Energy consumption for heating and cooling high intensity animal production facilities is among the largest CO_2 emitters within the agricultural sector (Enquete Commission, 1995).

Much of the ammonia production and urea application (United States: 4.3 Mt C per year; Mexico: 0.4 Mt C per year; Canada: 1.7 Mt C per year) and phosphoric acid manufacture (United States: 0.4 Mt C per year; Mexico: 0.2 Mt C per year; Canada: not reported) are devoted to agricultural uses.

10.1.3 Methane Fluxes From Agricultural and Grazing Lands

Cropland and grazing land soils act as both sources and sinks for atmospheric CH_4. Methane formation is an anaerobic process and is most significant in waterlogged soils, like those under paddy rice cultivation (United States: 0.25 ± 0.28 Mt CH_4-C per year; Mexico: 0.01 Mt CH_4-C per year†; Canada: negligible, not reported; Table 10.2). Methane is also formed by incomplete biomass combustion of crop residues (United States: 0.03 ± 0.02 Mt CH_4-C per year; Mexico: <0.01 Mt CH_4-C per year; Canada: negligible, not reported; Table 10.2). Methane oxidation in soils is a global sink for about 5% of CH_4 produced annually and is mainly limited by CH_4 diffusion into the soil. However, intensive cropland management tends to reduce soil CH_4 consumption relative to forests and extensively managed grazing lands (CAST, 2004). Management-induced changes in CH_4-C fluxes have a smaller impact on terrestrial carbon cycling than changes in CO_2-C fluxes (Table 10.2), but relatively greater radiative forcing for CH_4 amplifies the impact of increasing atmospheric CH_4 concentrations on net radiative forcing (Figure 10.1). Recent research has shown that live plant biomass and litter produce substantial amounts of CH_4, potentially making plants as large a source of CH_4 as livestock (Keppler *et al.*, 2006). If this is the case, activities that increase plant biomass (and sequester CO_2) may lead to increased CH_4 production (Keppler *et al.*, 2006).

10.1.4 Methane Fluxes From Livestock

Enteric fermentation (the process of organic matter breakdown by gut flora within the gastrointestinal tract of animals, particularly ruminants) allows for the digestion of fibrous

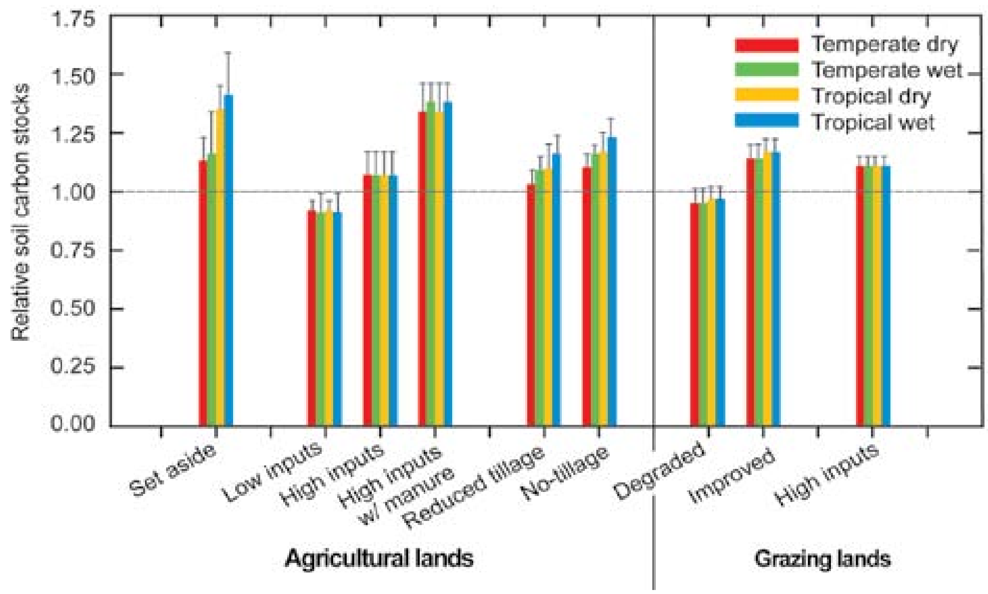
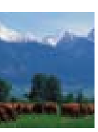


Figure 10.2 Relative soil carbon following implementation of new agricultural or grassland management practices. Conventionally tilled, medium-input cultivated land and moderately grazed grasslands with moderate inputs are defaults for agricultural and grazing lands, respectively. Default soil carbon stocks (like those in Table 10.1) can be multiplied by one or more stock change factors to estimate carbon sequestration rates (over a 20-year time period). The dashed horizontal line indicates default soil carbon stocks (*i.e.*, those under conventional-tillage cropland or undegraded grazingland, with medium inputs). Temperature/precipitation divisions are the same as those described in Table 10.1. Data are from Nabuurs *et al.* (2004) and Ogle *et al.* (2004).



materials by livestock, but the extensive fermentation of the ruminant diet requires 5–7% of the dietary gross energy to be belched out as CH_4 to sustain the anaerobic processes (Johnson and Johnson, 1995). Methane emissions from livestock contribute significantly to total CH_4 emissions in the United States (3.7 ± 0.53 Mt CH_4-C per year, 20% of total United States' CH_4 emissions), Canada (0.78 ± 0.14 Mt CH_4-C per year, 22% of total) (Ministry of the Environment, 2006; Sobool and Kulshreshtha, 2005), and Mexico (1.5 Mt CH_4-C per year, 27% of total)[†] with the vast majority of enteric CH_4 emissions from beef (72%) and dairy cattle (23%) (Table 10.2). Emissions from ruminants are tightly coupled to feed consumption, since CH_4 emission per unit of feed energy is relatively constant, except for feedlot cattle with diets high in cereal grain contents, for which the fractional loss falls to one-third to one-half of normal rates (Johnson and Johnson, 1995). Between 1990 and 2002, CH_4 emissions from enteric fermentation fell 2% in the United States but increased by 20% in Canada (EPA, 2000; Matin *et al.*, 2004).

Where the incentives (e.g., carbon offset market payments, government subsidies) to adopt new practices exceed the opportunity costs, farmers will adopt new practices.

Methane emissions during manure storage (United States: 1.3 ± 0.24 Mt CH_4 per year; Mexico: 0.06 Mt CH_4 per year [†]; Canada: 0.3 ± 0.05 Mt CH_4 per year) are governed by the amount of degradable organic matter, degree of anoxia, storage temperature, and duration of storage. Unlike enteric CH_4, the major sources of manure CH_4 emissions in the United States are from swine (44%) and dairy cattle (39%). Manure CH_4 production is greater for production systems with anoxic lagoons, largely anoxic pits, or manure handled or stored as slurry. Between 1990 and 2002, CH_4 emissions from manure management increased 25% in the United States and 21% in Canada (EPA, 2000; Matin *et al.*, 2004).

10.2 DRIVERS AND TRENDS

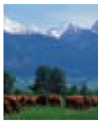
The extent to which agriculture will contribute to greenhouse gas mitigation will largely depend on government policy decisions, but mitigation opportunities will also be constrained by technological advances and changing environmental conditions (see discussion below). Estimates from national inventories suggest that United States' and Canadian agricultural soils are currently near neutral or small net sinks for CO_2, which has occurred as a consequence of changing management (*e.g.*, reduced tillage intensity) and government programs designed for purposes other than greenhouse gas mitigation (*e.g.*, soil conservation, commodity regulation). However, to realize the much larger potential for soil carbon sequestration (see section below) and for significant reductions in CH_4 (and N_2O) emissions, specific policies targeted at greenhouse gas reductions are required. It is generally recognized that farmers (and other economic actors) are, as a group, "profit-maximizers," which implies that to change from current practices to ones that reduce net emissions, farmers will incur additional costs (termed "opportunity costs"). Hence, where the incentives (*e.g.*, carbon offset market payments, government subsidies) to adopt new practices exceed the opportunity costs, farmers will adopt new practices. Crop productivity, production input expenses, marketing costs, *etc.* (which determine profitability) vary widely within (and between) countries. Thus, the payment needed to achieve a unit of emission reduction will vary, among and within regions. In general, each successive increment of carbon sequestration or emission reduction comes at a progressively higher cost (this relationship is often shown in the form of an upward bending marginal cost curve).

The interaction of changes in technological and environmental conditions, including crop growth improvements, impacts of CO_2 increase, nitrogen deposition, and climate change, will shape future trends in greenhouse gas emissions and mitigation from agricultural and grazing lands. A continuation of the yield increases seen in the past several decades for agricultural crops (Reilly and Fuglie, 1998) would tend to enhance the potential for soil carbon sequestration (CAST, 2004). Similarly, increased plant growth due to higher concentrations of CO_2 (and nitrogen deposition) has been projected to boost carbon uptake on agricultural (and other) lands, offsetting some or all of the climate-change induced reductions in productivity projected in some regions of North America (NAS, 2001). However, recent syntheses from field-scale FACE (Free-Air Carbon dioxide Enrichment) studies of croplands (Long *et al.*, 2006) and grasslands (Nowak *et al.*, 2004) suggest that the growth enhancement from CO_2 fertilization may be much less than previously thought. Feedbacks between temperature and soil carbon stocks could counteract efforts to reduce greenhouse gases via carbon sequestration within agricultural ecosystems. Increased temperatures tend to increase the rate of biological processes—including plant respiration and organic matter decay, and CO_2 release by soil organisms—particularly in temperate climates that prevail across most of North America. Because soil carbon stocks, including those in agricultural lands, contain such large amounts of carbon, small percentage increases in the rate of soil organic matter decomposition could lead to substantially increased emissions (Jenkinson *et al.*, 1991; Cox *et al.*, 2000). There is currently a scientific debate about the relative temperature sensitivity of the different constituents making up soil organic matter (*e.g.*, Kätterer *et al.*, 1998; Giardina and Ryan, 2000; Ågren and Bosatta, 2002; Knorr *et al.*, 2005), reflecting uncertainty in the possible degree and magnitude of climate change feedbacks. Despite this uncertainty, the potential for climate and other environmental feedbacks to influence the carbon balance of agricultural systems by perturbing productivity (and carbon input rates) and organic matter turnover, and potentially soil N_2O and CH_4 fluxes, cannot be overlooked.

10.3 OPTIONS FOR MANAGEMENT

10.3.1 Carbon Sequestration

Agricultural and grazing land management practices capable of increasing carbon inputs or decreasing carbon outputs, while still maintaining yields, can be divided into two classes: those that impact carbon inputs, and those that affect carbon release through decomposition and disturbance. Reversion to native vegetation or setting agricultural land aside as grassland, such as in the Canadian Prairie Cover Program and the U.S. Conservation Reserve Program, can increase the proportion of photosynthesized carbon retained in the system and sequester carbon in the soil[4] (Conant *et al.*, 2001; Post and Kwon, 2000; Follett *et al.*, 2001b) (Figure 10.2). In annual cropland, improved crop rotations, yield enhancement measures, organic amendments, cover crops, improved fertilization and irrigation practices, and reduced bare fallow tend to increase productivity and carbon inputs, and thus soil carbon stocks (Lal *et al.*, 1998; Paustian *et al.*, 1998; VandenBygaart *et al.*, 2003) (Figure 10.2). Tillage, traditionally used for soil preparation and weed control, disturbs the soil and stimulates decomposition and loss of soil carbon. Practices that substantially reduce (reduced-till) or eliminate (no-till) tillage-induced disturbances are being increasingly adopted and generally increase soil carbon stocks while maintaining or enhancing productivity levels (Paustian *et al.*, 1997; Ogle *et al.*, 2003) (Figure 10.2). Estimates of the technical potential for annual cropland soil carbon sequestration are on the order of 50–100 Mt C per year in the United States (Lal *et al.*, 2003; Sperow *et al.*, 2003) and 3.3–6.4 Mt C per year in Canada (Boehm *et al.*, 2004).



Within grazing lands, historical overgrazing has substantially reduced productive capacity in many areas, leading to loss of soil carbon stocks (Conant and Paustian, 2002) (Figure 10.2). Conversely, improved grazing management and production inputs (like fertilizer, adding (nitrogen-fixing) legumes, organic amendments, and irrigation) can increase productivity, carbon inputs, and soil carbon stocks (Conant *et al.*, 2001), potentially storing 0.44 Mt C per year[†] in Canada (Lynch *et al.*, 2005) and as much as 16–54 (mean = 33.2) Mt C per year in the United States (Follett *et al.*, 2001a). Such improvements will carry a carbon cost, par-

[4] The bulk of carbon sequestration potential in agricultural and grazing lands is restricted to soil carbon pools, though carbon can be sequestered in woody biomass in agroforestry systems (Sheinbaum and Masera, 2000). Woody encroachment on grasslands can also store substantial amounts of carbon in biomass, but the phenomenon is neither well-controlled nor desirable from the standpoint of livestock production, since it results in decreased forage productivity, and the impacts on soil carbon pools are highly variable and poorly understood.

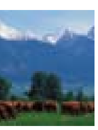
ticularly fertilization and irrigation, since their production and implementation require the use of fossil fuels.

10.3.2 Fossil-Fuel Derived Emission Reductions

Converting from conventional plowing to no-tillage can reduce on-farm fossil-fuel emissions by 25–80% and total fossil-fuel emissions by 14–25%.

The efficiency with which on-farm (from tractors and machinery) and off-farm (from production of agricultural input) energy inputs are converted to agricultural products varies several-fold (Lal, 2004). Where more energy-efficient practices can be substituted for less efficient ones, fossil-fuel CO_2 emissions can be reduced (Lal, 2004). For example, converting from conventional plowing to no-tillage can reduce on-farm fossil-fuel emissions by 25–80% (Frye, 1984; Robertson *et al.*, 2000) and total fossil-fuel emissions by 14–25% (West and Marland, 2003). Substitution of legumes for mineral nitrogen can reduce energy input by 15% in cropping systems incorporating legumes (Pimentel *et al.*, 2005). More efficient heating and cooling (*e.g.*, better building insulation) could reduce CO_2 emissions associated with housed animal facilities (*e.g.*, dairy). Substitution of crop-derived fuels for fossil fuels could decrease net emissions.

Energy intensity (energy per unit product) for the United States' agricultural sector has declined since the 1970s (Paustian *et al.*, 1998). Between 1990 and 2000, fossil-fuel emissions on Canadian farms increased by 35%[†] (Sobool and Kulshreshtha, 2005).



10.3.3 Methane Emission Reduction

Reducing flood duration and decreasing organic matter additions to paddy rice fields can reduce CH_4 emissions. Soil amendments such as ammonium sulfate and calcium carbide inhibit CH_4 formation. Coupled with adoption of new rice cultivars that favor lower CH_4 emissions, these management practices could reduce CH_4 emission from paddy rice systems by 16–70% (mean = 40%) of current emissions (Mosier *et al.*, 1998b).

Biomass burning is uncommon in most Canadian and United States' crop production systems; less than 3% of crop residues are burned annually in the United States (EPA, 2006). Biomass burning in conjunction with land clearing and with subsistence agriculture still occurs in Mexico, but these practices are declining. The primary path for emission reduction is reducing residue burning (CAST, 2004).

Practices that sequester carbon in agricultural and grazing land soils improve soil fertility, buffering capacity, and pesticide immobilization.

Refinement of feed quality, feed rationing, additives, and livestock production efficiency chains can all reduce CH_4 emissions from ruminant livestock with minimal impacts on productivity or profits (CAST, 2004). Boadi *et al.* (2004) review several examples of increases in energy intensity. Wider adoption of more efficient practices could reduce CH_4 production from 5–8% to 2–3% of gross feed energy (Agriculture and Agri-Food Canada, 1999), reducing CH_4 emissions by 20–30% (Mosier *et al.*, 1998b).

Methane emissions from manure storage are proportional to duration of storage under anoxic conditions. Handling solid rather than liquid manure, storing manure for shorter periods of time, and keeping storage tanks cool can reduce emissions from stored manure (CAST, 2004). More important, capture of CH_4 produced during anaerobic decomposition of manure (in covered lagoons or small- or large-scale digesters) can reduce emissions by 70–80% (Mosier *et al.*, 1998b). Use of digester systems is spreading in the United States, with 50 digesters currently in operation and 60 systems in construction or planned (NRCS, 2005). Energy production using CH_4 captured during manure storage will reduce energy demands and associated CO_2 emissions.

10.3.4 Environmental Co-benefits From Carbon Sequestration and Emission Reduction Activities

Many of the practices that lead to carbon sequestration and reduced CO_2 and CH_4 emissions not only increase production efficiencies but also lead to environmental co-benefits. Practices that sequester carbon in agricultural and grazing land soils improve soil fertility, buffering capacity, and pesticide immobilization (Lal, 2002; CAST, 2004). Increasing soil carbon content makes the soil more easily workable and reduces energy requirements for field operations (CAST, 2004). Decreasing soil disturbance and retaining more surface crop residues enhance water infiltration and prevent wind and water erosion, improving air quality. Increased water retention plus improved fertilizer management reduces nitrogen losses and subsequent nitrate (NO_3^-) leaching and downstream eutrophication.

10.3.5 Economics and Policy Assessment

Policies for agricultural mitigation activities can range from transfer payments (such as subsidies, tax credits, *etc.*) to encourage greenhouse gas mitigating practices or taxes or penalties to discourage practices with high emissions, to emission offset trading in a free market-based system with governmental sanction. Currently the policy context of the three North American countries differs greatly. Canada and the United States are both Annex 1 (developed countries) within the United Nations Framework Convention on Climate Change (UNFCCC), but Canada is obligated to mandatory emission reductions as a party to the Kyoto Protocol, while the United States currently maintains a national, voluntary

emission reduction policy outside of Kyoto. Mexico is a non-Annex 1 (developing) country and thus is not currently subject to mandatory emission reductions under Kyoto.

At present, there is relatively little practical experience upon which to judge the costs and effectiveness of agricultural mitigation activities. Governments are still in the process of developing policies and, moreover, the economics of various mitigation activities will only be known when there is a significant economic incentive for emission reductions, *e.g.*, through regulatory emission caps or government-sponsored bids and contracts. However, several economic analyses have been performed in the United States, using a variety of models (*e.g.*, McCarl and Schneider, 2001; Antle *et al.*, 2003; Lewandrowski *et al.*, 2004). Most studies have focused on carbon sequestration, and less work has been done on the economics of reducing CH_4 and N_2O emissions. While results differ between models and for different parts of the country, some preliminary conclusions have been drawn (see Boehm *et al.*, 2004; CAST, 2004).

- Additional carbon (10–70 Mt C per year), above current rates, could be sequestered in soils at low to moderate costs ($10–100 per metric ton of carbon).
- Mitigation practices that maintain the primary income source (*i.e.*, crop/livestock production), such as conservation tillage and pasture improvement, have a lower cost per ton sequestered carbon compared with practices where mitigation would be a primary income source (*i.e.*, foregoing income from crop and/or livestock production), such as land set-asides, even if the latter have a higher biological sequestration potential.
- With higher energy prices, major shifts in land use in favor of energy crops and afforestation may occur at the expense of annual cropland and pasture.
- Policies based on per-ton payments (for carbon actually sequestered) are more economically efficient than per-hectare payments (for adopting specific practices, see Antle *et al.*, 2003), although the former have a higher verification cost (*i.e.*, measuring actual carbon sequestered versus measuring adoption of specific farming practices on a given area of land).

A recent study commissioned by the U.S. Environmental Protection Agency (EPA, 2005), evaluated some agricultural mitigation options for different policy scenarios, including constant CO_2 price scenarios for 2010–2110, where the price represents the incentive required for the mitigation activity. Annualized prices of $15/ton of CO_2 would yield mitigation amounts of 46 Mt C per year through agricultural soil carbon sequestration and 14.5 Mt C per year from fossil-fuel use reduction (compared with the estimated United States' national ecosystem carbon sink of 480 Mt C per year). At lower prices of $5/ton CO_2, the corresponding values would be 34 Mt C per year (for soil sequestration) and 9 Mt C per year (for fossil-fuel reduction), respectively, reflecting the effect of price on the supply of mitigation activities[5].

10.3.6 Other Policy Considerations

Agricultural mitigation of CO_2 through carbon sequestration and emission reductions for CH_4 (and N_2O), differ in ways that impact policy design and implementation. Direct emission reductions of CH_4 and CO_2 from fossil-fuel use are considered "permanent" reductions, while carbon sequestration is a "non-permanent" reduction, in that carbon stored through conservation practices could potentially be re-emitted if management practices revert back to the previous state or otherwise change so that the stored carbon is lost. This *permanence* issue applies to all forms of carbon sinks. In addition, soil carbon storage, with a given change in management (*e.g.*, tillage reduction, pasture improvement, afforestation), will tend to level off at a new steady state level after 15–30 years, after which there is no further accumulation of carbon (West *et al.*, 2004). Enhanced management practices must be sustained to maintain these higher carbon stocks. Key implications for policy are that the value of sequestered carbon could be discounted compared to direct emission reductions to compensate for the possibility of future emissions. Alternatively, long-term contracts will be needed to build and maintain carbon stocks, which will tend to increase the price per unit of sequestered carbon. However, even temporary storage of carbon has economic value (CAST, 2004), and various proposed concepts of leasing carbon storage or applying discount rates could accommodate carbon sequestration as part of a carbon offset trading system (CAST, 2004). In addition, switching to practices that increase soil carbon (and hence, improve soil fertility) could be more profitable to farmers in the long-run, so that additional incentives to maintain the practices once they become well established may not be necessary (Paustian *et al.*, 2006).

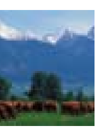
Another policy issue relating to carbon sequestration is *leakage* (also termed "slippage" in economics), whereby mitigation actions in one area (*e.g.*, geographic region, pro-

[5] These estimates were produced using a national-scale economic sector model which estimates the linkage between CO_2 prices and the supply of mitigation activities, for specified price scenarios. Hence, the model can produce a range of CO_2 mitigation amounts as a function of price, but the model was not used to estimate the uncertainty of mitigation amounts at a given price level.

BOX 10.4: Agricultural and Grazing Land N_2O Emission Reductions

When mineral soil nitrogen content is increased by nitrogen additions (*i.e.*, fertilizer), a portion of that nitrogen can be transformed to N_2O as a byproduct of two microbiological processes (nitrification and denitrification) and lost to the atmosphere. Coincidental introduction of large amounts of easily decomposable organic matter and NO_3^- from either a plow down of cover crop or manure addition greatly stimulates denitrification under wet conditions (Peoples *et al.*, 2004). Some practices intended to sequester atmospheric carbon in soil could prompt increases in N_2O fluxes. For example, reducing tillage intensity tends to increase soil moisture, leading to increased N_2O fluxes, particularly in wetter environments (Six *et al.*, 2004). Synchronizing organic amendment applications with plant nitrogen uptake and minimizing manure storage under anoxic conditions can reduce N_2O emissions by 10–25% and will increase nitrogen use efficiency which can decrease indirect emissions (in waterways) by 5–20% (CAST, 2004).

duction system) stimulate additional emissions elsewhere. For forest carbon sequestration, leakage is a major concern. For example, reducing harvest rates in one area (thereby maintaining higher biomass carbon stocks) can stimulate increased cutting and reduction in stored carbon in other areas, as was seen with the reduction in harvesting in the Pacific Northwest during the 1990s (Murray *et al.*, 2004). Preliminary studies suggest that leakage is of minor concern for agricultural carbon sequestration, since most practices would have little or no effect on the supply and demand of agricultural commodities. However, there are uncertain and conflicting views on whether land-set asides in which land is taken out of agricultural production, such as the Conservation Reserve Program in the United States, might be subject to significant leakage.



A further question, relevant to policies for carbon sequestration, is how practices for conserving carbon affect emissions of other greenhouse gases. Of particular importance is the interaction of carbon sequestration with N_2O emissions, because N_2O is such a potent greenhouse gas (Robertson and Grace, 2004; Six *et al.*, 2004; Gregorich *et al.*, 2005). (See Box 10.4). In some environs, carbon-sequestration practices, such as reduced tillage, can stimulate N_2O emissions, thereby offsetting part of the benefit; elsewhere, carbon-conserving practices may suppress N_2O emissions, amplifying the net benefit (Smith *et al.*, 2001; Smith and Conen, 2004; Conant *et al.*, 2005; Helgason *et al.*, 2005).

Similarly, carbon-sequestration practices might affect emissions of CH_4, if the practice, such as increased use of forages in rotations, leads to higher livestock numbers. These examples demonstrate that policies designed to suppress emission of one greenhouse gas, need to also consider complex interactions to ensure that *net* emissions of total greenhouse gases are reduced.

A variety of other factors will affect the willingness of farmers to adopt greenhouse gas reducing practices and the efficacy of agricultural policies, including perceptions of risk, information and extension efforts, technological developments, and social and ethical values (Paustian *et al.*, 2006). Many of these factors are difficult to incorporate into traditional economic analyses. Pilot mitigation projects, along with additional research using integrated ecosystem and economic assessment approaches (*e.g.*, Antle *et al.*, 2001), will allow us to get a clearer picture of the actual potential of agriculture to contribute to greenhouse gas mitigation efforts.

10.4 RESEARCH AND DEVELOPMENT NEEDS

Expanding the network of intensive research sites dedicated to understanding basic processes, coupled with national-level networks of soil monitoring/validation sites, could reduce inventory uncertainty and contribute to attributing changes in ecosystem carbon stocks to changes in land management (see Bellamy *et al.*, 2005). Expansion of both networks should be informed about how different geographic areas and ecosystems contribute to uncertainty and the likelihood that reducing uncertainty could inform policy decisions. For example, changes in ecosystem carbon stocks due to woody encroachment on grasslands constitute one of the largest, but least certain, aspects of terrestrial carbon cycling in North America (Houghton *et al.*, 1999; Pacala *et al.*, 2001). Better information about the spatial extent of woody encroachment, the amount and growth of woody biomass, and variation in the impacts on soil carbon stocks would help reduce that uncertainty. Identifying location, cause, and size of this sink could help identify practices that may promote continued sequestration of carbon and would constrain estimates of carbon storage in other lands, possibly helping to identify other policy options. Uncertainty in land use, land-use change, soil carbon responses to management (*e.g.*, tillage) on particular soils, and impacts of cultivation on soil carbon stocks (*e.g.*, impacts of erosion) are the largest contributors to uncertainty in the Canadian and United States' national agricultural greenhouse gas inventories (Ogle *et al.*, 2003; VandenBygaart *et al.*, 2003). Finally, if the goal of a policy instrument is to reduce greenhouse gas emissions, net impacts on CO_2, CH_4, and N_2O emissions, which are not as well understood, should be considered.

North American Forests

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KEY FINDINGS

- North American forests contain roughly 170 ± 40 billion tons of carbon, of which approximately 28% is in live vegetation and 72% is in dead organic matter.
- North American forests were a net carbon sink of -270 ± 130 million tons of carbon per year over the last 10 to 15 years.
- Deforestation continues in Mexico where forests are a source of carbon dioxide to the atmosphere. Forests of the United States and parts of Canada have become a carbon sink as a consequence of the recovery of forests following the abandonment of agricultural land.
- Carbon dioxide emissions from Canada's forests are highly variable because of interannual changes in area burned by wildfire.
- The size of the carbon sink in United States' forests appears to be declining based on inventory data from 1952 to the present.
- Many factors that cause changes in carbon stocks of forests have been identified, including land-use change, timber harvesting, natural disturbance, increasing atmospheric carbon dioxide, climate change, nitrogen deposition, and ozone in the lower atmosphere. There is a lack of consensus about how these different natural and human-caused factors contribute to the current sink, and the relative importance of factors varies geographically.
- There have been several continental- to sub continental-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 human-caused 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, (3) a better estimate of potential changes in forest carbon under climate change and other factors, and (4) the increased availability of decision support tools for carbon management in forests.

11.1 INTRODUCTION

BOX 11.1: CCSP SAP 2.2 Uncertainty Conventions

***** = 95% certain that the actual value is within 10% of the estimate reported,
**** = 95% certain that the estimate is within 25%,
*** = 95% certain that the estimate is within 50%,
** = 95% certain that the estimate is within 100%, and
* = uncertainty greater than 100%.
† = The magnitude and/or range of uncertainty for the given numerical value(s) is not provided in the references cited.

The forest area of North America totals 771 million hectares (ha), 36% of the land area of North America and about 20% of the world's forest area (Food and Agriculture Organization, 2001)† (see Table 11.1 and Box 11.1 for estimates and uncertainty conventions, respectively). 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.

North American forests are critical components of the global carbon cycle, exchanging large amounts of carbon dioxide (CO_2) and other gases with the atmosphere and oceans. 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, describe how forests affect the carbon cycle through CO_2 sequestration and emissions, and discuss management options and research needs.

11.2 CARBON STOCKS AND FLUXES

11.2.1 Ecosystem Carbon Stocks and Pools

North American forests contain more than 170 billion tons of carbon (Gt C), 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.

Carbon density (the amount of carbon stored per unit of land area) is highly variable. In Canada, the majority of carbon storage occurs in boreal and cordilleran forests (Kurz and Apps, 1999). In the United States, forests of the Northeast, Upper Midwest, Pacific Coast, and Alaska (with 14 Gt C) store the most carbon. In Mexico, temperate forests contain 4.5 Gt C, tropical forests contain 4.1 Gt C, and semiarid forests contain 5.0 Gt C.

Table 11.1 Area of forest land by biome and country, 2000 (1000 ha)[a]. See Box 11.1 for uncertainty conventions.

Ecological zone:	Canada[b]	U.S.[c]	Mexico[d]	Total
Tropical/subtropical	0*****	115,200*****	30,700*****	145,900*****
Temperate	101,100*****	142,400*****	32,900*****	276,400*****
Boreal	303,000*****	45,500*****	0*****	348,500*****
Total	404,100*****	303,100*****	63,600*****	770,800*****

[a] The certainty for estimates in this table are listed in Box 11.1. See sources for estimates (*e.g.*, see Bechtold and Patterson, 2005 for the United States).
[b] Canadian Forest Service (2005)
[c] Smith *et al.* (2004)
[d] Palacio-Prieto *et al.* (2000)

11.2.2 Net North American Forest Carbon Fluxes

According to nearly all published studies, North American lands are a net carbon sink (Pacala *et al.*, 2001). 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 -269 million metric tons of carbon (Mt C) per year over the last decade or so, with United States' forests accounting for most of the sink (Table 11.3). This estimate is likely to be within 50% of the true value.

Canadian forests were estimated to be a net sink of -17 Mt C per year from 1990-2004 (Environment Canada, 2006) (Table 11.3). These estimates pertain to the area of forest considered to be "managed" under international reporting guidelines, which is 83% 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 included; urban forests are excluded (Chapter 14 this report). High interannual variability is averaged into this estimate—the annual change varied from approximately -50 to +40 between 1990 and 2004. Years with net emissions were generally years with high forest fire activity (Environment Canada, 2005) (Figure 11.1).

Most of the net sink in United States' forests is in aboveground carbon pools, which account for -146 Mt C per year (Smith and Heath, 2005). The net sink for the below-ground carbon pool is estimated at -90 Mt C (Pacala *et al.*, 2001) (Table 11.3). The size of the carbon sink in United States' forest ecosystems appears to have declined slightly over the last decade (Smith and Heath, 2005). In

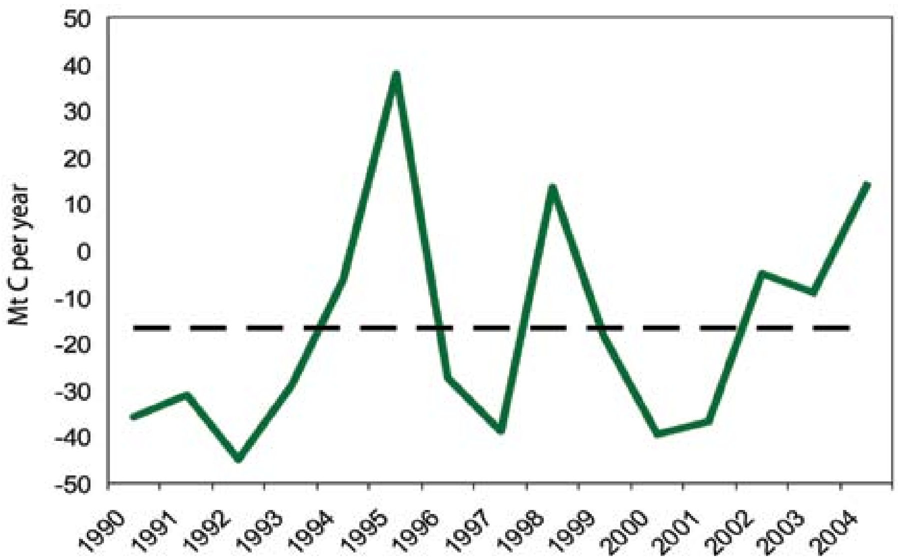


Figure 11.1 Average and annual estimates of change in carbon stocks for forest ecosystems of Canada, 1990-2004. Interannual variability is high because of changes in rates and impacts of disturbances such as fire and insects (from Environment Canada, 2006).

Table 11.2 Carbon stocks in forests by ecosystem carbon pool and country (Mt C)[a]. See Box 11.1 for uncertainty conventions.

Ecosystem carbon pool:	Canada[b]	U.S.[c]	Mexico[d]	Total
Biomass	14,500****	24,900****	7,700****	47,100****
Dead organic matter[e]	71,300****	41,700****	11,400****	124,400****
Total	85,800****	66,600****	19,100****	171,500****

[a] The certainty for estimates in this table are listed in Box 11.1. See sources for estimates (Heath and Smith, 2000; Smith and Heath, 2000). The estimated carbon stock in North American forests is thus 171,500 ± 43,000 Mt C.
[b] Kurz and Apps (1999)
[c] Heath and Smith (2004), Birdsey and Heath (1995)
[d] Masera *et al.* (2001)
[e] Includes litter, coarse woody debris, and soil carbon.

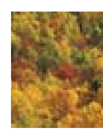
contrast, a steady or increasing supply of timber products now and in the foreseeable future (Haynes, 2003) means that the rate of increase in the wood products carbon pool is likely to remain steady.

For Mexico, the most comprehensive available estimate for the forest sector suggests a source of +52 Mt C per year in the 1990s (Masera *et al.*, 1997) (Table 11.3). 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.

Landscape-scale estimates of ecosystem carbon fluxes reflect the dynamics of individual forest stands that respond to unique combinations of disturbance history, management intensity, vegetation, and site characteristics. Extensive land-based measurements of forest/atmosphere carbon exchange for forest stands at various stages of recovery after disturbance reveal patterns and causes of sink or source strength, which is highly dependent on time since disturbance. Representative estimates for North America are summarized in Appendix D. As forests are planted or regrow on abandoned farmland, or as they recover from fire, harvest, or other disturbance, there is an initial period of slow (or negative) carbon sequestration followed by a period of rapid carbon sequestration. Many forests continue sequestering significant amounts of carbon for 125 years or more after establishment (Smith *et al.*, 2006). Eventually, the rate of sequestration slows as forests reach a new balance of carbon uptake and release, and in old growth forests processes of carbon uptake are very nearly balanced by processes of release (Chapter 3, this report).

11.3 TRENDS AND DRIVERS

11.3.1 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 the relative importance of each remains difficult to quantify (Barford *et al.*, 2001; Caspersen *et al.*, 2000; Goodale *et al.*, 2002; Körner, 2000; Schimel *et al.*, 2000). Land-use

Table 11.3 Change in carbon stocks for forests and wood products by country (Mt C per year). See Box 11.1 for uncertainty conventions.

Carbon pool:	Canada[a]	U.S.[b]	Mexico[c]	Total
Forest ecosystem	–17**	–236***	+52**	–201
Wood products	–11**	–57***	ND[d]	–68
Total	–28**	–293***	+52**	–269

[a] Data for 1990-2004, taken from Environment Canada (2006), Goodale *et al.* (2002).
[b] From Smith and Heath (2005) (excluding soils), and Pacala *et al.* (2001) (soils). Estimates do not include urban forests.
[c] From Masera (1997)
[d] Estimates are not available.

change, timber harvesting, natural disturbance, increasing atmospheric CO_2, climate change, nitrogen deposition, and tropospheric ozone all have effects on carbon stocks in forests, with their relative influence depending on geographic location, the type of forest, and specific site factors. It is important for policy implementation and management of forest carbon to separate the effects of direct human actions from natural factors.

The natural and human-caused (anthropogenic) factors that significantly influence forest carbon stocks are different for each country, and still debated in the scientific literature. Natural disturbances are significant in Canada, but estimates of the relative effects of different kinds of disturbance are uncertain. One study estimated that impacts of wildfire and insects caused emissions of about +40 Mt C per year† of carbon to the atmosphere over the two decades (Kurz and Apps, 1999). Another study concluded that the positive effects of climate, CO_2, and nitrogen deposition outweighed the effects of wildfire and insects, making Canada's forests a net carbon sink in the same period (Chen *et al.*, 2003). In the United States, land-use change and timber harvesting seem to be dominant factors according to repeated forest inventories from 1952 to 1997 that show forest carbon stocks (excluding soils) increasing by about 175 Mt C per year. The most recent inventories show a decline in the rate of carbon uptake by forests, which appears to be mainly the result of changing growth and harvest rates following a long history of land-use change and management (Birdsey *et al.*, 2006; Smith and Heath, 2005). The factors behind net emissions from Mexico's forests are deforestation, forest degradation, and forest fires that are not fully offset by forest regeneration (Masera *et al.*, 1997; De Jong *et al.*, 2000).

The most recent inventories for the U.S. show a decline in the rate of carbon uptake by forests.

11.3.2 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)†. In 2004, approximately 25,000 ha were converted from forest to cropland, 19,000 ha from forest to settlements, and approximately 3,000 ha converted to wetlands. These land-use changes resulted in emissions of about 4 Mt C (Environment Canada, 2005)†.

In the last century more than 130 million ha of land in the conterminous United States were either afforested (62 million ha)† or deforested (70 million ha)† (Birdsey and Lewis, 2003). 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.

Emissions from Mexican forests to the atmosphere are primarily due to the impacts of deforestation to pasture and degradation of 720,000 to 880,000 ha per year† (Masera *et al.*, 1997; Palacio-Prieto *et al.*, 2000). 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)†.

Table 11.4 Area of forestland by management class and country, 2000 (1000 ha)[a]. See Box 11.1 for uncertainty conventions.

Management class:	Canada	U.S.	Mexico	Total
Protected	19,300*****	66,700*****	6,000*****	92,000*****
Plantation	4,500*****	16,200*****	200*****	20,900*****
Other	380,300*****	220,200*****	57,400*****	657,900*****
Total	404,100*****	303,100*****	63,600*****	770,800*****

[a]From Food and Agriculture Organization (2001), Natural Resources Canada (2005). The certainty for estimates in this table are listed in Box 11.1. See sources for estimates (*e.g.,* for the United States, see Bechtold and Patterson, 2005).

11.3.3 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 is impacted by a wide range of human activities and government policies that influence harvesting, wood products, and regeneration.

Forests and other wooded land in Canada occupy about 402 million ha. Approximately 310 million ha is considered forest of which 255 million ha (83%) 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 million ha per year were harvested in the United States, two-thirds by partial-cut harvest and one-third by clear-cut (Birdsey and Lewis, 2003). Between 1987 and 1997, about 1 million ha per year 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 United States' forests accounts for substantially more tree mortality than natural causes such as wildfire and insect outbreaks (Smith *et al*., 2004). 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). Illegal harvesting involves 13.3 million cubic meters of wood every year (Torres, 2004). The rural population is the controlling factor for changes in carbon stocks from wildfire, wood extraction, shifting agriculture practices, and conversion of land to crop and pasture use.

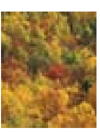
11.3.4 Effects of Climate and Atmospheric Chemistry

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). Some studies indicate that these effects are significantly smaller than the effects of land management and land-use change (Caspersen *et al*., 2000; Schimel *et al*., 2000). Recent reviews of ecosystem-scale studies known as Free Air CO_2 Exchange (FACE) experiments suggest that rising CO_2 increases net primary productivity by 12-23% over all species studied (Norby *et al*., 2005; Nowak *et al*., 2004). However, 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 (Körner *et al*., 2005; Lichter *et al*., 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 availability is also considered, reduced soil fertility limits the response to rising CO_2, but nitrogen deposition can increase soil fertility to counteract that effect (Finzi *et al*., 2006; Johnson *et al*., 1998; Oren *et al*., 2001). Observations of photosynthetic activity from satellites suggest that productivity changes due to lengthening of the growing season depend on whether areas were disturbed by fire (Goetz *et al*., 2005). Based on these conflicting and complicated results from different studies and approaches, a definitive assessment of the relative importance, and interactions, of natural and anthropogenic factors is a high priority for research (U.S. Climate Change Science Program, 2003).

11.3.5 Effects of Natural Disturbances

Wildfire, insects, diseases, and weather events are common natural disturbances in North America. These factors impact all forests but differ in magnitude by geographic region. 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 million ha per year in Canada's forests, with a maximum 7.6 million ha per year in 1989[†]. Carbon emissions from forest fires range from less than +1 Mt C per year in the interior of British Columbia to more than +10 Mt C per year in the western boreal forest. Total emissions from forest fires in Canada averaged approximately +27 Mt C per year between 1959 and 1999 (Amiro *et al.*, 2001)[†]. 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 per year in the 1970s to less than +2 Mt C per year in the mid-1990s[1]. Much of the Canadian forest is expected to experience increases in fire severity (Parisien *et al.*, 2005) and burn areas (Flannigan *et al.*, 2005), and continued outbreaks of forest pests are also likely (Volney and Hirsch, 2005).

In United States' forests, insects, diseases, and wildfire combined, affect more than 30 million ha per decade (Birdsey and Lewis, 2003). Damage from weather events (hurricanes, tornadoes, and ice storms) may exceed 20 million ha per decade (Dale *et al.*, 2001). Although forest inventory data reveal the extent of tree mortality attributed to all causes combined, estimates of the impacts of individual categories of natural disturbance on carbon pools of temperate forests are scarce. 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). McNulty (2002) estimated that large hurricanes in the United States could convert 20 Mt C of live biomass into detrital carbon pools.



Large portions of the Canadian and Alaskan forest are expected to be particularly sensitive to climate change.

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 million ha in the 1970s and 0.05-0.85 million ha in the 1990s). During El Nino years, increasing drought increases fire frequencies (Torres, 2004). Between 1995 and 2000, an average of 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. 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.

[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 carbon 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).

11.3.6 Projections of Future Trends

11.3.6.1 Canada

Large portions of the Canadian and Alaskan forest are expected to be particularly sensitive to climate change (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 in northern latitudes, 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.

11.3.6.2 United States

Assessments of future changes in carbon and vegetation distribution in the United States suggest that under most future climate conditions, net primary production (NPP) would respond positively to changing climate but total carbon storage would remain relatively constant (VEMAP Members, 1995; Pan *et al.*, 1998; Neilson *et al.*, 1998; Joyce *et al.*, 2001). Some climate scenarios indicate that much of the Northwest U.S. will receive more annual precipitation. 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 climate model show 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, but some 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. North American boreal forests are of particular concern due to substantial increases in fire activity projected under most future climate scenarios (Flannigan *et al.*, 2005).

11.3.6.3 Mexico

For Mexican forests, 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).

11.4 OPTIONS FOR MANAGEMENT

Forest management strategies can be adapted to increase the amount of carbon uptake by forest systems. Alternative strategies for wood products are also important in several ways: how long carbon is retained in use, how much wood is used for biofuel, and substitution of wood for other materials that use more energy to produce. The net effect of these management and production strategies on carbon stocks and emissions will depend on emerging government policies for greenhouse gas management, the area of forests under management, management objectives for resources other than carbon, and the type of management and production regime being considered.

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). Although the science of managing forests specifically for carbon sequestration is not well developed, some ecological principles are emerging to guide management decisions (Appendix E). The prospective role of forestry in helping to stabilize atmospheric CO_2 depends on government policy, 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.

For Canada, Price *et al.* (1997) examined 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. Controlling stand density following harvest had minimal impacts in the short term but increased landscape-level carbon storage by 13% after 150 years. 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).

Substantial increases in fire activity for North American boreal forests are projected under most future climate scenarios.

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.

In the United States, a bundle of forestry activities could potentially increase carbon sequestration from -100 to -200 Mt C per year according to several studies (Birdsey *et al.*, 2000; Lewandrowski *et al.*, 2004; Environmental Protection Agency, 2005; Stavins and Richards, 2005). 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.

Table 11.5 Illustrative emissions reduction potential of various forestry activities in the United States under a range of prices and sequestration rates[a].

Forestry activity	Carbon sequestration rate (t CO_2 per ha per year)	Price range (\$/t CO_2)	Emissions reduction potential (Mt CO_2 per year)
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

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

Economic analyses of the U.S. forestry potential have focused on three broad categories of activities: afforestation (conversion of agricultural land to forest), improved management of existing forests, and use of woody biomass for fuel. Improved management of existing forest lands may be attractive to landowners at carbon prices below \$10 per ton of CO_2; afforestation requires a moderate price of \$15 per ton of CO_2 or more to induce landowners to participate; and biofuels become dominant at prices of \$30-50 per ton of CO_2 (Lewandrowski *et al.*, 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 2010-2110, for forestry activities as part of a bundle of reduction options for the land base.

Production of renewable materials that have lower life-cycle emissions of greenhouse gases than non-renewable alternatives is a promising strategy for reducing emissions. Lippke *et al.* (2004) found that wood components used in residential construction had lower emissions of CO_2 from energy inputs than either concrete or steel.

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.

11.5 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.

11.5.1 Major Data Gaps in Estimates of Carbon Pools and Fluxes

Effective carbon policy and management to increase carbon sequestration and/or reduce emissions requires thorough understanding of current carbon stock sizes and flux rates, and responses to disturbance. Data gaps complicate analyses of the potential for policies to influence natural, social, and economic drivers that can change carbon stocks and fluxes. Forests in an area as large as North America are quite diverse, and comprehensive data sets that can be used to analyze forestry opportunities, such as spatially explicit historical management and disturbance rates and effects on the carbon cycle, would enable managers to change forest carbon stocks and fluxes. Although this report provides aggregate statistics on forest carbon by biome and country, users could benefit from spatially explicit estimates of forest carbon. Such an analysis might involve matching estimates based on forest inventories as presented by political unit and general forest type (Birdsey and Lewis, 2003) with data developed using remote sensing techniques (Running *et al.*, 2004). Research at the level of individual sites has proven the feasibility of this combination (*e.g.*, Van Tuyl *et al.*, 2005; Turner *et al.*, 2006). This kind of analysis could facilitate development of a forest carbon map for North America.

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 per year (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, although research is ongoing to combine various approaches and data sets: large-scale observations, process-based modeling, ecosystem experiments, and laboratory investigations (Foley and Ramankutty, 2004).

Data gaps exist for Canada, particularly regarding changes in forest soil carbon and forest lands that are considered "unmanaged" (17% 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 that 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 Carbon Budget Model of the Canadian Forest Sector (CBM-CFS3)(Kurz *et al.*, 2002); however, the CBM-CFS3 was not designed to incorporate 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.

11.5.2 Major Data Gaps in Knowledge of Forest Management Effects

There is insufficient information available to guide land managers in specific situations to change forest management practices to increase carbon sequestration, and there is some uncertainty about the longevity of effects (Caldeira *et al.*, 2004). This reflects a gap in the availability of inexpensive techniques for measuring, monitoring, and predicting changes in ecosystem carbon pools at the smaller scales appropriate for managers. There is more information available about management effects on live biomass and woody debris, and less about effects on soils and wood products. This imbalance in data has the potential to produce unintended consequences if predicted results are based on incomplete carbon accounting.

In the tropics, 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. Agroforestry also has potential in temperate agricultural landscapes, but as with forest management, there

is a lack of data about how specific systems affect carbon storage (Nair and Nair, 2003).

Refining management of forests to realize significant carbon sequestration, while at the same time continuing to satisfy the needs of forests and the services they provide (*e.g.*, timber harvest, recreational value, 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. For example, methods should be developed for enhancing the efficiency of forest management, increasing the carbon storage per acre from existing forests, or even increasing the acreage devoted to forest systems that provide carbon sequestration. Currently there is little information about how appropriate incentives might be applied to accomplish these goals 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.

Given the importance of forests in the global carbon cycle, success in enhancing the efficiency of forests as a renewable energy source could have important long-term and large-scale effects on global atmospheric carbon stocks.

11.5.3 Availability of Decision Support Tools

Few decision support tools for land managers that include complete carbon accounting are available; one example is the CBM-CFS3 carbon accounting model (Kurz *et al.*, 2002). Some are in development 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 in those markets, then the demand for decision support tools will encourage their development.

CHAPTER 12

Carbon Cycles in the Permafrost Region of North America

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KEY FINDINGS

- Much of northern North America (more than 6 million square kilometers) is characterized by the presence of permafrost (soils or rocks that remain frozen for at least two consecutive years). This permafrost region contains approximately 25% of the world's total soil organic carbon, a massive pool of carbon that is vulnerable to release to the atmosphere as carbon dioxide in response to an already detectable polar warming.
- The soils of the permafrost region of North America contain 213 billion tons of organic carbon, approximately 61% of the carbon in all soils of North America.
- The soils of the permafrost region of North America are currently a net sink of approximately 11 million tons of carbon per year.
- The soils of the permafrost region of North America have been slowly accumulating carbon for the last 5000–8000 years. More recently, increased human activity in the region has resulted in permafrost degradation and at least localized loss of soil carbon.
- Patterns of climate, especially the region's cool and cold temperatures and their interaction with soil hydrology to produce wet and frozen soils, are primarily responsible for the historical accumulation of carbon in the region. Non-climatic drivers of carbon change include human activities, including flooding associated with hydroelectric development, that degrade permafrost and lead to carbon loss. Fires, increasingly common in the region, also lead to carbon loss.
- Projections of future warming of the polar regions of North America lead to projections of carbon loss from the soils of the permafrost region, with upwards of 78% (34 billion tons) and 41% (40 billion tons) of carbon stored in soils of the Subarctic and northern-most coniferous (Boreal) regions, respectively, being severely or extremely severely affected by future climate change.
- Options for management of carbon in the permafrost region of North America, including construction methods that cause as little disturbance of the permafrost and surface as possible, are primarily those which avoid permafrost degradation and subsequent carbon losses.
- Most research needs for the permafrost region are focused on reducing uncertainties in knowing how much carbon is vulnerable to a warming climate and how sensitive that carbon loss is to climate change. Development and adoption of measures that reduce or avoid the negative impact of human activities on permafrost are also needed.

12.1 INTRODUCTION

It is especially important to understand the carbon cycle in the permafrost region of North America because the soils in this area contain large amounts of organic carbon that is vulnerable to release to the atmosphere as carbon dioxide (CO_2) and methane (CH_4) in response to climate warming. It is predicted that the average annual air temperature in the permafrost region will increase 3–4°C by 2020 and 5–10°C by 2050 (Hengeveld, 2000, see Box 12.1)[†]. The soils in this region contain approximately 61%[***] of the organic carbon occurring in all soils in North America (Lacelle *et al.*, 2000) even though the permafrost area covers only about 21%[***] of the soil area of the continent. Release of even a fraction of this carbon in greenhouse gases could have global consequences.

Permafrost is defined, on the basis of temperature, as soils or rocks that remain below 0°C for at least two consecutive years (van Everdingen, 1998 revised May 2005). Permafrost terrain often contains large quantities of ground ice in the upper section of the permafrost. If this terrain is well protected by forests or peat, this ground ice is generally in equilibrium with the current climate. If this insulating layer is not sufficient, however, even small temperature changes, especially in the southern part of the permafrost region, could cause degradation and result in severe thermal erosion (thawing). For example, some of the permafrost that formed in central Alaska during the Little Ice Age is now degrading in response to warming during the last 150 years (Jorgenson *et al.*, 2001).

Some of the permafrost that formed in central Alaska during the Little Ice Age is now degrading in response to warming during the last 150 years.

The permafrost region in North America is divided into four zones on the basis of the percentage of the land area underlain by permafrost (Figure 12.1). These zones are the Continuous Permafrost Zone (≥90 to 100%), the Discontinuous Permafrost Zone (≥50 to <90%), the Sporadic Permafrost Zone (≥10 to <50%), and the Isolated Patches Permafrost Zone (0 to <10%) (Brown *et al.*, 1997).

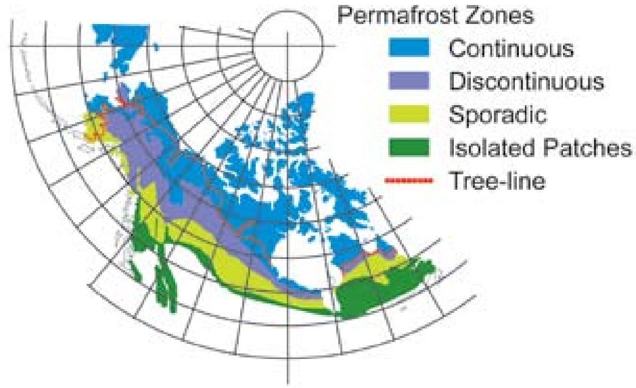


Figure 12.1 Permafrost zones in North America (Brown *et al.*, 1997).

These permafrost zones encompass three major ecoclimatic provinces (ecological regions) (Figure 12.2): the Arctic (north of the arctic tree line), the Subarctic (open canopy coniferous forest), and the Boreal (closed canopy forest, either

Figure 12.2 Arctic, Subarctic, and Boreal ecoclimatic provinces (ecological regions) in North America (Ecoregions Working Group, 1989; Baily and Cushwa, 1981).

BOX 12.1: CCSP SAP 2.2 Uncertainty Conventions

*****	=	95% certain that the actual value is within 10% of the estimate reported,
****	=	95% certain that the estimate is within 25%,
***	=	95% certain that the estimate is within 50%,
**	=	95% certain that the estimate is within 100%, and
*	=	uncertainty greater than 100%.
†	=	The magnitude and/or range of uncertainty for the given numerical value(s) is not provided in the references cited.

coniferous or mixed coniferous and deciduous). Peatlands (organic wetlands characterized by more than 40 cm of peat accumulation) cover large areas in the Boreal, Subarctic, and southern part of the Arctic ecoclimatic provinces.

Although northern ecosystems (Arctic, Subarctic, and Boreal) in North America cover approximately 14% of the global land area, they contain approximately 25% of the world's total soil organic carbon (Oechel and Vourlitis, 1994)[†]. In addition, Oechel and Vourlitis (1994) indicate that the tundra (Arctic) ecosystems alone contain approximately 12% of the global soil carbon pool, even though they account for only 6% of the total global land area[†]. Based on direct measure of the carbon density to one meter depth, the soil carbon pool should be doubled (Michaelson *et al*., 1996). The soils of the permafrost region of North America are currently a carbon sink and are unique because they are able to actively sequester carbon and store it for thousands of years.

The objectives of this chapter are to give the below-ground carbon stocks and to explain the mechanisms associated with the carbon cycle (sources and sinks) in the soils of the permafrost region of North America.

12.2 PROCESSES AFFECTING THE CARBON CYCLE IN A PERMAFROST ENVIRONMENT

12.2.1 Soils of the Permafrost Region

Soils cover approximately 6,211,340 square kilometers (km^2)[***] of the area of the North American permafrost region (Tables 12.1 and 12.2), with approximately 58%[***] of the land area being occupied by permafrost-affected (perennially frozen) soils (Cryosols/Gelisols) and the remainder by non-permafrost soils (Soil Carbon Database Working Group, 1993). Approximately 17%[***] of this area is associated with organic soils (peatlands), the remainder with mineral soils (Soil Carbon Database Working Group, 1993). It is important to distinguish between mineral soils and organic soils in the region because different processes are responsible for the carbon cycle in these two types of soils.

12.2.2 Mineral Soils

Table 12.1 Areas of mineral soils in the various permafrost zones.

Permafrost zones	Area (10^3 km^2)		
	Canada[a]	Alaska[b]	Total
Continuous	2001.80	353.46	2355.26
Discontinuous	636.63	479.15	1115.78
Sporadic	717.63	110.98	828.61
Isolated Patches	868.08	0.73	868.81
Total	4224.14	944.32	5168.46

[a] Calculated using the Soil Carbon of Canada Database (Soil Carbon Database Working Group, 1993).
[b] Calculated using the Northern and Mid Latitudes Soil Database (Cryosol Working Group, 2001).

Table 12.2 Areas of peatlands (organic soils) in the various permafrost zones.

Permafrost zones	Area (10^3 km^2)		
	Canada[a]	Alaska[b]	Total
Continuous	176.70	51.31	228.01
Discontinuous	243.51	28.74	272.25
Sporadic	307.72	0.62	308.34
Isolated Patches	221.23	13.05	234.28
Total	949.16	93.72	1042.88

[a] Calculated using the Peatlands of Canada Database (Tarnocai *et al.*, 2005).
[b] Calculated using the Northern and Mid Latitudes Soil Database (Cryosol Working Group, 2001).

The schematic diagram in Figure 12.3 provides general information about the carbon sinks and sources in mineral soils. Most of the permafrost-affected mineral soils are carbon sinks because of the slow decomposition rate due to cold and wet conditions and the process of cryoturbation, which moves surface organic matter into the deeper soil layers. Other processes, such as decomposition, wildfires, and thermal degradation, release carbon into the atmosphere and, thus, act as carbon sources.

For unfrozen soils and noncryoturbated frozen soils in the permafrost region, the carbon cycle is similar to that in soils occurring in temperate regions. In these soils, organic matter is deposited on the soil surface. Some soluble organic matter may move downward, but because these soils are not affected by cryoturbation, they have no mechanism for moving organic matter from the surface into the deeper soil layers and preserving it from decomposition and wildfires. Most of their below-ground carbon originates from roots and its residence time is relatively short.

The role of cryoturbation: Although permafrost-affected ecosystems pro-

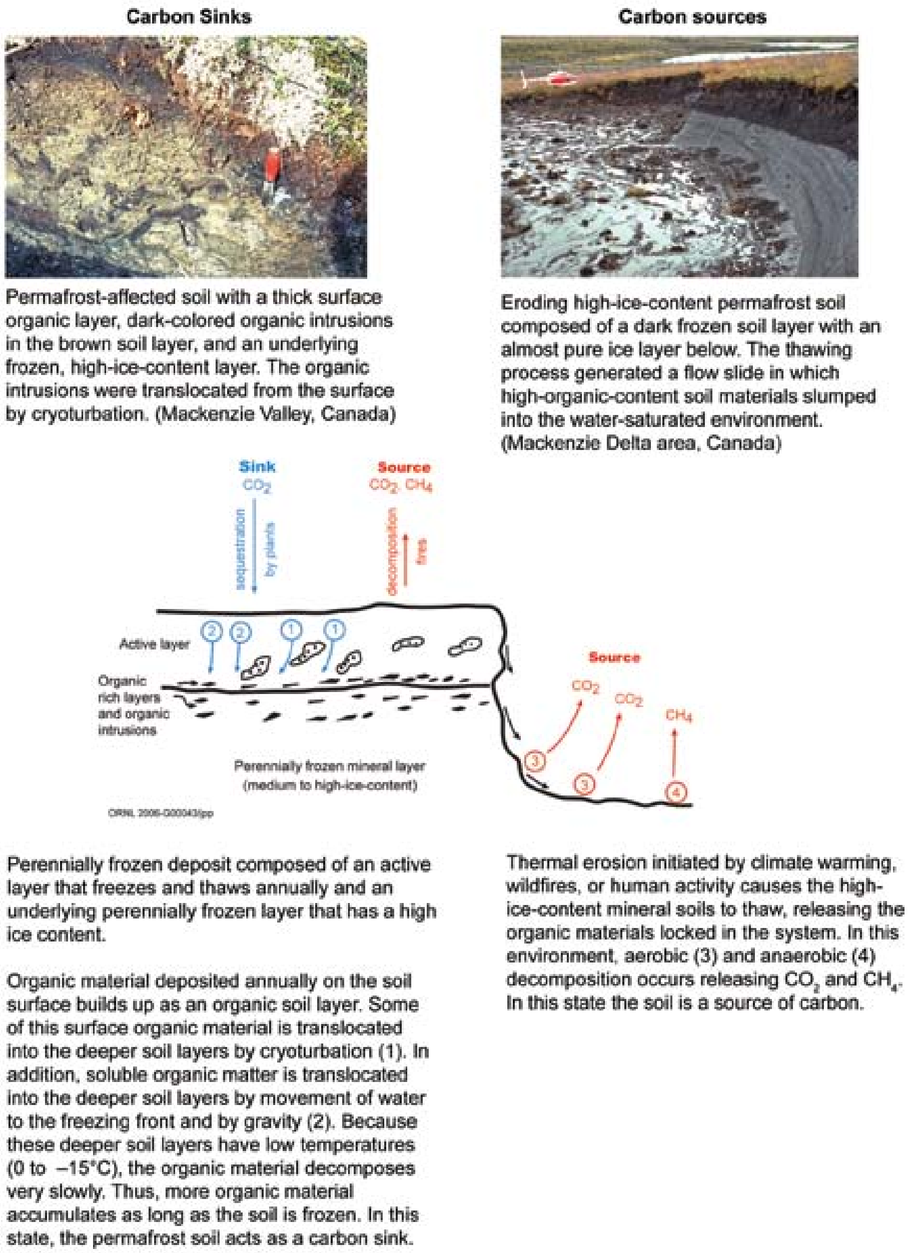


Figure 12.3 Carbon cycle in permafrost-affected upland (mineral) soils, showing belowground organic carbon sinks and sources.

decomposing as a result of biological activity. A large portion of this litter, however, builds up on the soil surface, forming an organic soil horizon. Cryoturbation causes some of this organic material to move down into the deeper soil layers (Bockheim and Tarnocai, 1998). Soluble organic materials move downward because of the effect of gravity and the movement of water along the thermal gradient toward the freezing front (Kokelj and Burn, 2005). Once the organic material has moved down to the cold, deeper soil layers where very little or no biological decomposition takes place, it may be preserved for many thousands of years. Radiocarbon dates from cryoturbated soil materials ranged between 490 and 11,200 yr BP (Zoltai *et al.*, 1978). These dates were randomly distributed within the soil and did not appear in chronological sequence by depth (the deepest material was not necessarily the oldest), indicating that cryoturbation is an ongoing process.

The permafrost table (top of the permafrost) is very dynamic and is subject to deepening due to factors such as removal of vegetation and/or the insulating surface organic layer, wildfires, global climate change, and other natural or human activities. When this occurs, the seasonally thawed

duce much less biomass than do temperate ecosystems, permafrost-affected soils that are subject to cryoturbation (frost-churning), a cryogenic process, have a unique ability to sequester a portion of this organic matter and store it for thousands of years. A number of models have been developed to explain the mechanisms involved in cryoturbation (Mackay, 1980; Van Vliet-Lanoë, 1991; Vandenberghe, 1992). The most recent model involves the process of differential frost heave (heave-subsidence), which produces downward and lateral movement of materials (Walker *et al.*, 2002; Peterson and Krantz, 2003).

Part of the organic matter produced annually by the vegetation is deposited as litter on the soil surface, with some

layer (active layer) becomes deeper and the organic material is able to move even deeper into the soil (translocation). However, if such factors cause thawing of the soil and melting of the ground ice, some or all of the organic materials locked in the system could be exposed to the atmosphere. This change in soil environment gives rise to both aerobic and anaerobic decomposition, releasing carbon into the atmosphere as CO_2 and CH_4, respectively (Figure 12.3). At this stage, the soil can become a major carbon source.

If, however, the permafrost table rises (and the active layer becomes shallower) because of reestablishment of the vegetation or buildup of the surface organic layer, this deep organic material becomes part of the permafrost and is, thus, more securely preserved. This is the main reason that permafrost-affected soils contain high amounts of organic carbon not only in the upper (0–100 cm) layer, but also in the deeper layers. These cryoturbated, permafrost-affected soils are effective carbon sinks.

12.2.3 Peatlands (Organic Soils)

The schematic diagram in Figure 12.4 provides general information about the processes driving the carbon sinks and sources in peatland soils. The water-saturated conditions, low soil temperatures, and acidic conditions of northern peatlands provide an environment in which very little decomposition occurs; hence, the litter is converted to peat and preserved. This gradual buildup process has been ongoing in peatlands during the last 5000–8000 years, resulting in peat deposits that are an average of 2–3 meters (m) thick and, in some cases, up to 10 m thick. At this stage, peatlands can act as very effective carbon sinks for many thousands of years (Figure 12.4).

***Carbon dynamics*:** Data for carbon accumulation in various peatland types in the permafrost regions are given in Table 12.3. Although some values for the rate of peat accumulation are higher (associated with unfrozen peatlands), the value for frozen peatlands, which are more widespread, is 13.31 ± 2.20 grams of carbon per square meter (g C per m^2) per year (Robinson and Moore, 1999). Peat accumulations in the various ecological regions were calculated on the basis of the thickness of the deposit and the date of the basal peat. The rate of peat accumulation is generally highest in the Boreal region and decreases northward (Table 12.3). Note, however, that if the surface of the peat deposit has eroded, the calculated rate of accumulation (based on the age of the basal peat and a decreased deposit thickness) will appear to be higher than it should be. This is probably the reason for some of the high rates of peat accumulation found for the Arctic region, which likely experienced a rapid rate of accumulation during the Hypsithermal Maximum with subsequent erosion of the surface of some of the deposits reducing their thicknesses. Wildfires, decomposition, and leaching of soluble organic compounds release approximately one-third of the carbon input, causing most of the carbon loss in these peatlands.

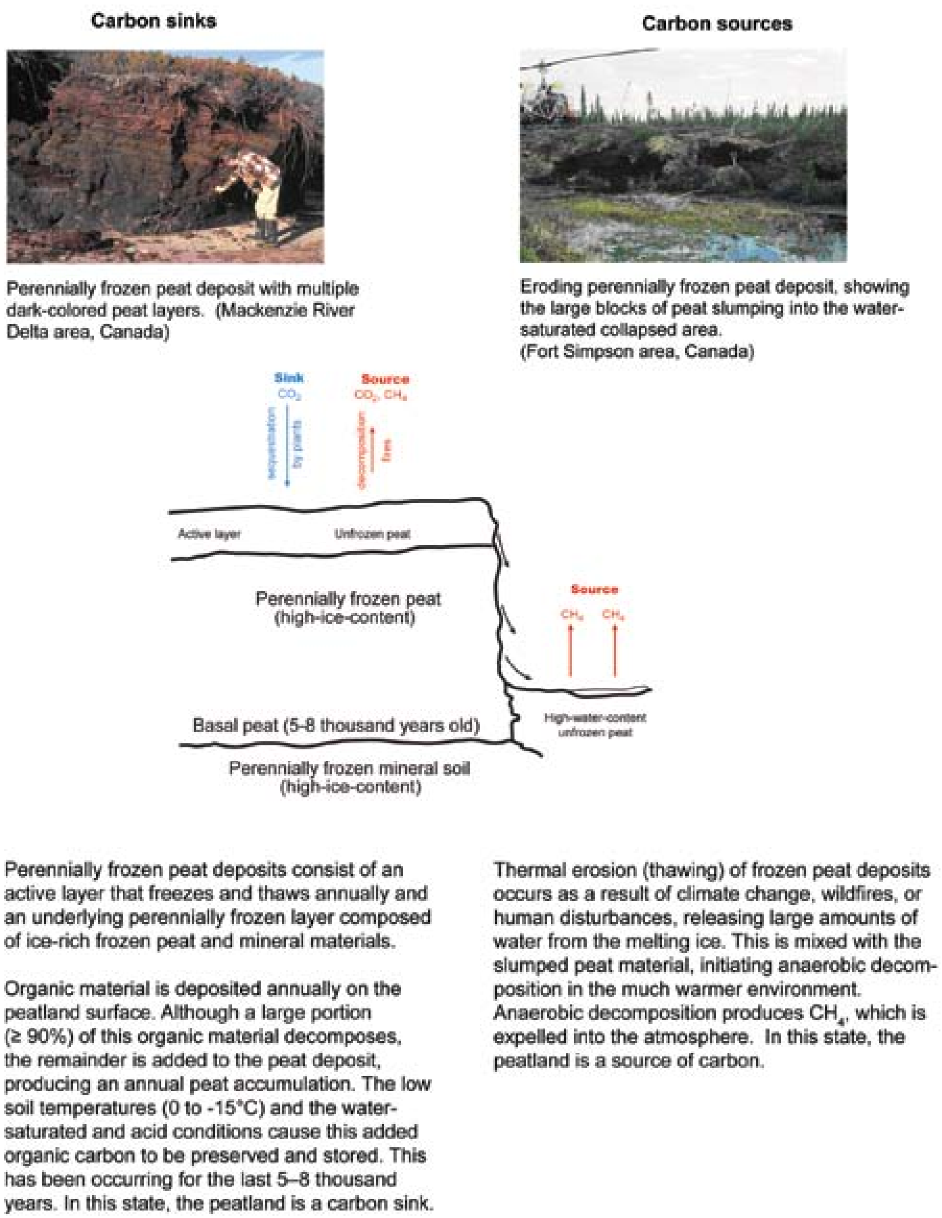


Figure 12.4 Carbon cycle in permafrost peatlands, showing below-ground organic carbon sinks and sources.

12.3 BELOW-GROUND CARBON STOCKS

The carbon content of mineral soils to a 1-m depth is 49-61 kilograms (kg) per m^2 for permafrost-affected soils and 12-17 kg per m^2 for unfrozen soils (Tables 12.4 and 12.5). The carbon content of organic soils (peatlands) for the total depth of the deposit is 81-129 kg per m^2 for permafrost-affected soils and 43-144 kg per

Table 12.3 Organic carbon accumulation and loss in various Canadian peatlands. Positive values indicate net flux into the atmosphere (source); negative values indicate carbon sequestration (land sinks).

Peatlands	Amount of carbon
Boreal peatlands	–9.8 Mt per year[a]
All Canadian peatlands	–30 Mt per year[b]
All mineral and organic soils	–18 mg per m^2 per year[c]
Rich fens	–13.58 ± 1.07 g per m^2 per year[d]
Poor fens (unfrozen, Discontinuous Permafrost Zone)	–20.34 ± 2.86 g per m^2 per year[d]
Peat plateaus (frozen, Discontinuous Permafrost Zone)	–13.31 ± 2.20 g per m^2 per year[d]
Collapse fens	–13.54 ± 1.50 g per m^2 per year[d]
Bogs (unfrozen, Discontinuous Permafrost Zone)	–21.81 ± 3.25 g per m^2 per year[d]
Dissolved organic carbon (DOC)	+2 g per m^2 per year[e]
Arctic peatlands	–0 to –16 cm/100 yr[f]
Subarctic peatlands	–2 to –5 cm/100 yr[f]
Boreal peatlands	–2 to –11 cm/100 yr[f]
Carbon release by each fire in northern boreal peatlands	+1.46 kg C per m^2[g]
Carbon release by fires in all terrain	+27 Mt per year[h]
Carbon release by fires in Western Canadian peatlands	+5.9 Mt per year[h]

[a] Zoltai *et al.* (1988).
[b] Gorham (1988).
[c] Liblik *et al.* (1997).
[d] Robinson and Moore (1999).
[e] Moore (1997).
[f] Calculated based on the thickness of the deposit and the date of the basal peat (National Wetlands Working Group, 1988).
[g] Robinson and Moore (2000).
[h] Turetsky *et al.* (2004).
Note: Except as explicitly indicated otherwise, no estimates of the confidence, certainty, or uncertainty of the numerical values in this table are available.

m^2 for unfrozen soils (Tables 12.4 and 12.5) (Tarnocai, 1998 and 2000).

Soils in the permafrost region of North America contain 213 billion tons (Gt) of organic carbon (Tables 12.6 and 12.7), which is approximately 61% of the organic carbon in all soils on this continent (Lacelle *et al.*, 2000). Mineral soils contain approximately 99 Gt of organic carbon in the 0- to 100-cm depth (Table 12.6). Although peatlands (organic soils) cover a smaller area than mineral soils (17% vs. 83%), they contain approximately 114 Gt of organic carbon in the total depth of the deposit, or more than half (54%) of the soil organic carbon of the region (Table 12.7).

12.4 CARBON FLUXES

12.4.1 Mineral Soils

Very little information is available about carbon fluxes in both unfrozen and perennially frozen mineral soils in the permafrost regions. For unfrozen upland mineral soils, Trumbore and Harden (1997) report a carbon accumulation of 60-100 g C per m^2 per year (Table 12.4). They further indicate that the slow decomposition results in rapid organic matter accumulation, but the turnover time due to wildfires (every 500–1000 years) eliminates the accumulated carbon except for the deep carbon derived from roots in the subsoil. The turnover time for this deep carbon is 100-1600 years. Therefore, the carbon stocks in these unfrozen soils are low, and the turnover time of this carbon is 100 to 1600 years.

As with unfrozen mineral soils, very little information has been published on the carbon cycle in perennially frozen mineral soils. The carbon cycle in these soils differs from that in unfrozen soils in that, because of cryogenic activities, these soils are able to move the organic matter deposited on the soil surface into the deeper soil layers. Assuming that cryoturbation was active in these soils during the last six thousand years (Zoltai *et al.*, 1978), an average of 9 million tons of carbon (Mt C)** have been added annually to these soils. Most of this carbon has been cryoturbated into the deeper soil layers, but some of the carbon in the surface organic layer is released by decomposition and, periodically, by wildfires. The schematic diagram in Figure 12.5 shows the carbon cycle in these soils.

12.4.2 Peatlands (Organic Soils)

Peatland vegetation deposits various amounts of organic material (litter) annually on the peatland surface. Reader and Stewart (1972) found that the amount of litter (dry biomass) deposited annually on the bog surface in boreal peatlands in Manitoba, Canada was 489-1750 g per m^2. Approximately 25% of the original litter fall was found to have decomposed during the following year. In the course of the study, they found that the average annual accumulation rate was 10% of the annual net primary production. Robinson and Moore (1999) and Robinson *et al.* (2003) found that, in the Sporadic Permafrost Zone, mean carbon accumulation rates over the past 100 years for unfrozen bogs and frost mounds were 88.6 ± 4.4 and 78.5 ± 8.8 g per m^2 per year, respectively. They also found that, in the Discontinuous Permafrost Zone, the mean carbon accumulation rate during the past 1200 years in frozen peat plateaus was 13.31 ± 2.20 g per m^2 per year, while in unfrozen fens

Table 12.4 Soil carbon pools and fluxes for the permafrost areas of Canada. Positive flux numbers indicate net flux into the atmosphere (source); negative values indicate carbon sequestration (land sinks).

Type	Peatlands		Mineral soils		Total
	Perennially frozen	Unfrozen	Perennially frozen	Unfrozen	
Current area (× 10^3 km^2)	422[a]	527[a]	2088[b]	2136[b]	5173
Current pool (Gt)	47[c]	65[a]	56[c]	28[b]	196
Current atm. flux (g per m^2 per year)	−5.7[d]	−15.2[e]			
Carbon accumulation (g per m^2 per year)	−13.3 ± 2.20[f]	−20.34 ± 2.86 to −21.81 ± 3.25[f]		−60 to −100[g]	
Carbon release by fires (g per m^2 per year)[h]	+7.57[i]				
Methane flux (g per m^2 per year)		+2.0[j]			

[a] Calculated using the Peatlands of Canada Database (Tarnocai *et al.*, 2005).
[b] Calculated using the Soil Carbon of Canada Database (Soil Carbon Database Working Group, 1993).
[c] Tarnocai (1998).
[d] Using C accumulation rate of 0.13 mg per ha per year (this report).
[e] Using C accumulation rate of 0.194 mg per ha per year (Vitt *et al.*, 2000).
[f] Robinson and Moore (1999).
[g] Trumbore and Harden (1997).
[h] Fires recur every 150–190 years (Kuhry, 1994; Robinson and Moore, 2000).
[i] Robinson and Moore (2000).
[j] Moore and Roulet (1995).
Note: Except as explicitly indicated otherwise, no estimates of the confidence, certainty, or uncertainty of the numerical values in this table are available.

and bogs the comparable rates were 20.34 ± 2.86 and 21.81 ± 3.25 g per m^2 per year, respectively.

Because peatlands cover large areas in the permafrost region of North America, their contribution to the carbon stocks is significant (Table 12.5). Zoltai *et al.* (1988) estimated that the annual carbon accumulation capacity of boreal peatlands is approximately 9.8 Mt[†]. Gorham (1988), in contrast, estimated that Canadian peatlands accumulate approximately 30 Mt C[†] annually.

Currently, wildfires are probably the greatest natural force in converting peatlands to a carbon source. Ritchie (1987) found that the western Canadian boreal forests have a fire return interval of 50-100 years, while Kuhry (1994) indicated that, for wetter Sphagnum bogs, the interval is 400-1700 years. For peat plateau bogs, each fire resulted in an average decrease in carbon mass of 1.46 kg per m^2 and an average decrease in height of 2.74 cm, which represents about 150 years of peat accumulation (Robinson and Moore, 2000). In recent years, the number of these wildfires has increased, as has the area burned, releasing increasing amounts of carbon into the atmosphere.

The schematic diagram presented in Figure 12.6 summarizes the carbon cycle in peatlands in the permafrost region. Based on average values for the rate of peat accumulation, approximately 17 g C per m^2 per year, or 18 Mt C, is added annually to peatlands in this region of North America. Approximately 1.46 kg C per m^2 is released to the atmosphere every 600 years by wildfires in the northern boreal peatlands. In addition, decomposition of unfrozen peatlands releases approximately 2.0 g C per m^2 per year, and a further 2.0 g C per m^2 per year is released by leaching of dissolved organic carbon (DOC), leading to a carbon decrease of approximately 4 Mt[**] annually, not including that released by wildfires (Figure 12.6). Note that these values are based on current measurements. However, rates of peat accumulation have varied during the past 5000–8000 years, with periods during which the rate of peat accumulation was much higher than at present.

The soils in the permafrost region of North America currently act as a net carbon sink.

Table 12.5 Average organic carbon content for soils in the various ecological regions (Tarnocai, 1998 and 2000).

Ecological regions	Average carbon content (kg per m^2)			
	Mineral soils[a]		Organic soils (peatlands)[b]	
	Frozen	Unfrozen	Frozen	Unfrozen
Arctic	49	12	86	43
Subarctic	61	17	129	144
Boreal	50	16	81	134

[a] For the 1-m depth.
[b] For the total depth of the peat deposit.

Table 12.6 Organic carbon mass in mineral soils in the various permafrost zones.

Permafrost zones	Carbon mass[a] (Gt)		
	Canada[b]	Alaska[c]	Total
Continuous	51.10	9.04	60.14
Discontinuous	10.33	4.82	15.15
Sporadic	9.15	0.75	9.90
Isolated Patches	13.59	0	13.59
Total	84.17	14.61	98.78

[a] Calculated for the 0–100 cm depth.
[b] Calculated using the Soil Carbon of Canada Database (Soil Carbon Database Working Group, 1993).
[c] Calculated using the Northern and Mid Latitudes Soil Database (Cryosol Working Group, 2001).

Table 12.7 Organic carbon mass in peatlands (organic soils) in the various permafrost zones.

Permafrost zones	Carbon mass[a] (Gt)		
	Canada[b]	Alaska[c]	Total
Continuous	21.82	1.46	23.28
Discontinuous	26.54	0.84	27.38
Sporadic	30.66	0.27	30.93
Isolated Patches	32.95	0	32.95
Total	111.97	2.57	114.54

[a] Calculated for the total depth of the peat deposit.
[b] Calculated using the Peatlands of Canada Database (Tarnocai *et al.*, 2005).
[c] Calculated using the Northern and Mid Latitudes Soil Database (Cryosol Working Group, 2001).

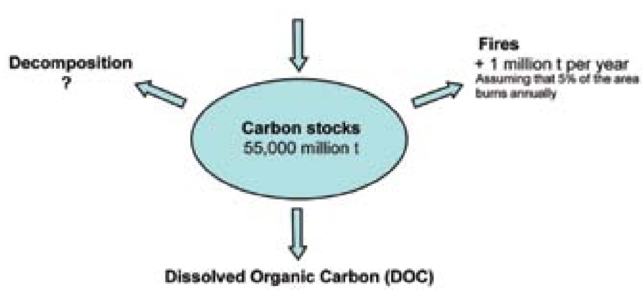


Figure 12.5 Carbon cycle in perennially frozen mineral soils in the permafrost region. Question marks represent data values that cannot be quantified.

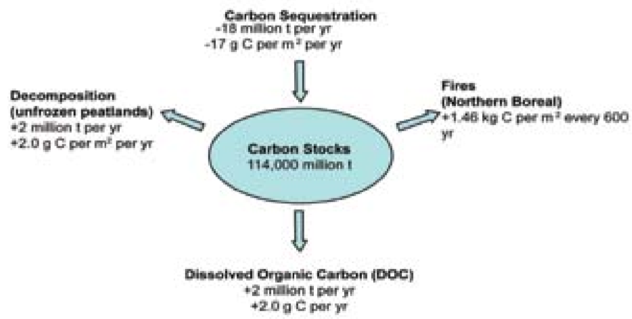


Figure 12.6 Carbon cycle in peatlands in the permafrost region.

12.4.3 Total Flux

Based on the limited data available for this vast, and largely inaccessible, area of the continent, approximately 27 Mt C per year** is deposited on the surface of mineral soils and peatlands (organic soils) in the permafrost region of North America. Approximately 8 Mt per year** of surface carbon (excluding vegetation) is released by decomposition and wildfires, and by leaching into the water systems. Thus, the soils in the permafrost region of North America currently act as a sink for approximately 19 Mt C per year** and as a source for approximately 8 Mt C per year** and are, therefore, a net carbon sink (Figures 12.5 and 12.6).

12.5 POSSIBLE EFFECTS OF GLOBAL CLIMATE CHANGE

The permafrost region is unique because the soils in this vast area contain large amounts of organic materials and much of the carbon has been actively sequestered by peat accumulation (organic soils) and cryoturbation (mineral soils) and stored in the permafrost for many thousands of years. Historical patterns of climate are responsible for the large amount of carbon found in the soils of the region today, but cryoturbation is a consequence of the region's current cool to cold climate and the effects of that climate on soil hydrology. As a result, patterns of climate and climate change are dominant drivers of carbon cycling in the region. Future climate change will determine the fate of that carbon and whether the region will remain a slow but significant carbon sink, or whether it will reverse and become a source, rapidly releasing large amounts of CO_2 and CH_4 to the atmosphere.

12.5.1 Peatlands

A model for estimating the sensitivity of peatlands to global climate change was developed using current climate (1x CO_2), vegetation, and permafrost data together with the changes in these variables expected in a 2x CO_2 environment (Kettles and Tarnocai, 1999). The data generated by this model were used to produce a peatland sensitivity map. Using geographic information system (GIS) techniques, this map was overlaid on the peatland map of Canada to determine both the sensitivity ratings of the various peatland areas and the associated organic carbon masses. The sensitivity ratings, or classes, used are no change, very slight, slight, moderate, severe, and extremely severe. Because global climate change is expected to have the greatest impact on the ecological processes and permafrost distribution in peatlands in the severe and extremely severe categories (Kettles and Tarnocai, 1999), the areas and carbon masses of peatlands in these two sensitivity classes are considered to be most vulnerable to climate change. The sensitivity ratings are determined by the degree of change in the ecological zonation combined with the degree of change in the permafrost zonation, with the greater the change, the more severe the sensitivity rating. For example, if a portion of the Subarctic becomes Boreal in ecology and the associated sporadic permafrost disappears (no permafrost remains in the region), the sensitivity of this region is rated as extremely severe. If however, a portion of the Boreal remains Boreal in ecology, but the discontinuous permafrost disappears (no permafrost remains in the region), the sensitivity of this region is rated as severe.

The peatland sensitivity model (Tarnocai, 2006) indicates that the greatest effect of global climate change will occur in the Subarctic region, where about 85% (314,270 km^2) of the peatland area and 78% (33.96 Gt) of the organic carbon mass will be severely or extremely severely affected by climate change, with 66% of the area and 57% of the organic carbon mass being extremely severely affected (Figure 12.7) †. The second largest effect will occur in the Boreal region, where about 49% (353,100 km^2) of the peatland area and 41% (40.20 Gt) of the organic carbon mass will be severely or extremely severely affected, with 10% of both the area and organic carbon mass being extremely severely affected.

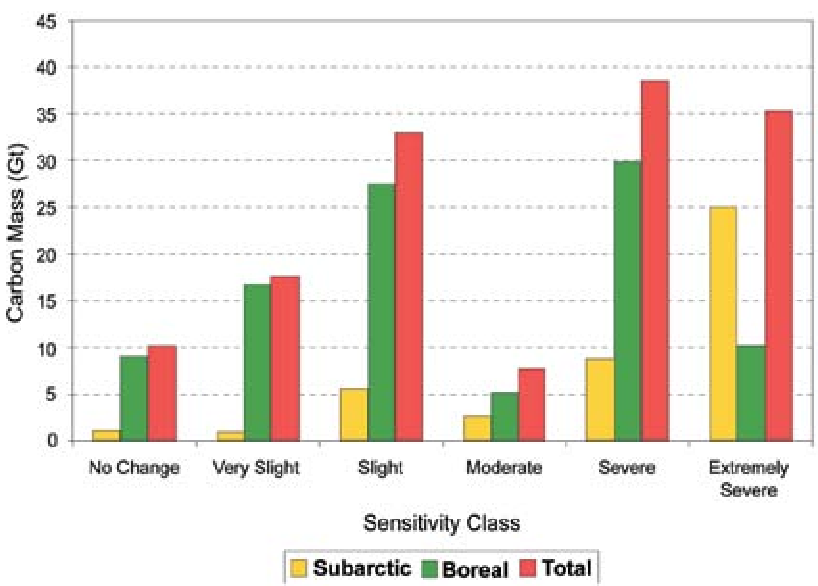


Figure 12.7 The organic carbon mass in the various sensitivity classes for the Subarctic and Boreal ecoclimatic provinces (ecological regions) (Tarnocai, 2006).

These two regions contain almost all (99%) of the Canadian peatland area and organic carbon mass that is predicted to be severely or extremely severely affected (Figure 12.7) (Tarnocai, 2006).

In the Subarctic region and the northern part of the Boreal region, where most of the perennially frozen peatlands occur, the increased temperatures are expected to cause increased thawing of the perennially frozen peat. Thawing of the ice-rich peat and the underlying mineral soil will initially result in water-saturated conditions. These water-saturated conditions, together with the higher temperatures, result in anaerobic decomposition, leading to the production of CH_4.

The greatest effect of global climate change will occur in the Subarctic region, where about 85% of the peatland area and 78% of the organic carbon mass will be severely or extremely severely affected by climate change.

In the southern part of the Boreal region, where the peatlands are generally unfrozen, the main impact is expected to be drought conditions resulting from higher summer temperatures and higher evapotranspiration. Under such conditions, peatlands become a net source of CO_2 because the oxygenated conditions lead to aerobic decomposition (Melillo *et al.*, 1990; Christensen, 1991). These dry conditions will likely also increase wildfires and, eventually, burning of peat, leading to the release of CO_2 to the atmosphere.

12.5.2 Permafrost-affected Mineral Soils

The same model described above was used to determine the effect of climate change on mineral permafrost-affected soils. The model suggests that approximately 21% (11.9 Gt)† of the total organic carbon in these soils could be severely or extremely severely affected by climate warming (Tarnocai, 1999). The model also suggests that the permafrost will probably disappear from the soils (the soils will become unfrozen) in the Sporadic and Isolated Patches permafrost zones. The main reason for the high sensitivity of mineral soils in these zones is that soil temperatures at both the 100- and 150-cm depths are only slightly below freezing (-0.3°C). The slightest disturbance or climate warming could initiate rapid thawing in these soils, with resultant loss of carbon (Tarnocai, 1999).

12.6 NON-CLIMATIC DRIVERS

Wildfires are an important part of the ecology of Boreal and Subarctic forests and are probably the major non-climatic drivers of carbon change in the permafrost region. There has been a rapid increase in both the frequency of fires and the area burned as a result of warmer and drier summers and increased human activity in the region. According to observations of natives, not only has the frequency of lightning strikes increased in the more southerly areas, but they have now appeared in more northerly areas where they were previously unknown. Because lightning is the major cause of wildfires in areas of little habitation, it is likely largely responsible for the increase in wildfires now being observed.

There has been a rapid increase in both the frequency of fires and the area burned as a result of warmer and drier summers and increased human activity in the region.

Increased human activity as a result of the construction of pipelines, roads, airstrips, and mines, expansion of agriculture, and development and expansion of town sites has disturbed the natural soil cover and exposed the organic-rich soil layers, leading to increased soil temperatures and, hence, decomposition of the exposed organic materials. Burgess and Tarnocai (1997), studying the Norman Wells Pipeline, provide some examples of the effect of pipeline construction on frozen peatlands and permafrost in Canada.

Shoreline erosion along rivers, lakes, and oceans and thermal erosion (thermokarst) are also common processes in the permafrost region, exposing the carbon-rich frozen soil layers to the atmosphere and making the organic materials available for decomposition. Along the 1957 km of the Beaufort Sea coast of Arctic Alaska, an estimated $1.8x10^5$ Mg C per year erodes into the Arctic Ocean due to thawing of permafrost. As a result, CO_2 and CH_4 are released directly into the atmosphere, but most of this carbon goes into the ocean as particulate organic matter, and a small fraction as dissolved organic carbon (Jorgenson and Brown, 2005; Ping *et al.*, 2006).

Large hydroelectric projects in northern areas, such as Southern Indian Lake in Manitoba and the James Bay region of Quebec, have flooded vast areas of peatlands and initiated permafrost degradation and decomposition of organic carbon, some of which is released into the atmosphere as CH_4. Of greater immediate concern, however, is the carbon that has entered the water system as dissolved organic carbon. These compounds include contaminants such as persistent organic pollutants (*e.g.*, Polychlorinated biphenyls [PCBs], Dichloro-Diphenyl-Trichloroethane [DDT], Hexachlorocyclohexanes [HCH], and chlorobenzene [AMAP, 2004]) that have been widely distributed in northern ecosystems over many years, much of it deposited by snowfalls, concentrated by cryoturbation, and stored in the organic soils. Of particular concern is the release of methylmercury because peatlands are net producers of this compound (Driscoll *et al.*, 1998; Suchanek *et al.*, 2000), which is a much greater health hazard than inorganic or elemental mercury. Natives in the regions where these hydroelectric developments have taken place have developed mercury poisoning after ingesting fish contaminated by this mercury, leading to serious health problems for many of the people. This is an example of what can happen when permafrost degrades as a result of human activities. When climate warming occurs, the widespread degradation of permafrost, with the resulting release of such dangerous pollutants into the water systems, could cause serious health problems for fish, animals, and humans that rely on such waters.

12.7 OPTIONS FOR MANAGEMENT OF CARBON IN THE PERMAFROST REGION

Although wildfires are the most effective mechanism for releasing carbon into the atmosphere, they are also an important factor in maintaining the integrity of northern ecosystems. Therefore, such fires are allowed to burn naturally and are controlled only if they are close to settlements or other man-made structures.

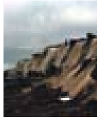
The construction methods currently used in permafrost terrain are designed to cause as little surface disturbance as possible and to preserve the permafrost. Thus, the construction of pipelines, airstrips, and highways is commonly carried out in the winter so that the heavy equipment used will cause minimal surface disturbance.

The greatest threat to the region is a warmer (and possibly drier) climate, which would drastically affect not only the carbon cycle, but also the biological systems, including human life. Unfortunately, we know very little about how to manage the natural systems in this new environment.

12.8 DATA GAPS AND UNCERTAINTIES

The permafrost environment is a very complex system, and the data available for it are very limited with numerous gaps and uncertainties. Information on the distribution of soils in the permafrost region is based on small-scale maps, and the carbon stocks calculated for these soils are derived from a relatively small number of datasets. Although there is some understanding of the carbon sinks and sources in these soils, the limited amount of data available make it very difficult, or impossible, to assign reliable values. Only limited amounts of flux data have been collected for the permafrost-affected soils and, in some cases, it has been collected on sites that are not representative of the overall landscape. This makes it very difficult to scale this information up for a larger area. As Davidson and Janssens (2006) state:

> "…the unresolved question regarding peatlands and permafrost is not the degree to which the currently constrained decomposition rates are temperature sensitive, but rather how much permafrost is likely to melt and how much of the peatland area is likely



> to dry significantly. Such regional changes in temperature, precipitation, and drainage are still difficult to predict in global circulation models. Hence, the climate change predictions, as much as our understanding of carbon dynamics, limit our ability to predict the magnitude of likely vulnerability of peat and permafrost carbon to climate change."

To obtain more reliable estimates of the carbon sinks and sources in permafrost-affected soils, we need much more detailed data on the distribution and characteristics of these soils. Carbon stock estimates currently exist only for the upper 1 m of the soil. Limited data from the Mackenzie River Valley in Canada, Arctic coast of Alaska and the Kolyma Lowland of NE Russia indicate that a considerable amount of soil organic carbon occurs below the 1-m depth, even at the 3-m depth. Future estimates of carbon stocks should be extended to cover a depth of 0-2 m or, in some cases, even greater depths. More measurements of carbon fluxes and inputs are also needed if we are to understand the carbon sequestration process in these soils in the various permafrost zones. Our understanding of the effect that rapid climate warming will have on the carbon sinks and sources in these soils is also very limited. Future research should focus in greater detail on how the interactions of climate with the biological and physical environments will affect the carbon balance in permafrost-affected soils.

The changes that are occurring, and will occur, in the permafrost region are almost totally driven by natural forces and so are almost impossible for humans to manage on a large scale. Human activities, such as they are, are aimed at protecting the permafrost and, thus, preserving the carbon. Perhaps we humans should realize that there are systems (*e.g.*, glaciers, ocean currents, droughts, and rainfall) that will be impossible for us to manage. We simply must learn to accept them, and if possible, adapt.

Wetlands

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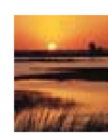
KEY FINDINGS

- North America is home to approximately 40% of the global wetland area, encompassing about 2.5 million square kilometers (965,000 square miles) with a carbon pool of approximately 223 billion tons, mostly in peatland soils.
- North American wetlands currently are a carbon dioxide sink of approximately 49 million tons of carbon per year, but that estimate has an uncertainty of greater than 100%. North American wetlands are also a source of approximately 9 million tons of methane, a more potent atmospheric heat-trapping gas. The uncertainty in that flux is also greater than 100%.
- Historically, the destruction of North American wetlands through land-use change has reduced carbon storage in wetlands by 15 million tons of carbon per year, primarily through the oxidation of carbon in peatland soils as they are drained and a more general reduction in carbon uptake and storage capacity of wetlands converted to other land uses. Methane emissions have also declined with the loss of wetland area.
- Projections of future carbon storage and methane emissions of North American wetlands are highly uncertain and complex, but the large carbon pools in peatlands may be at risk for oxidation and release to the atmosphere as carbon dioxide if they become substantially warmer and drier. Methane emissions may increase with warming, but the response will likely vary with wetland type and with changes in precipitation.
- Because of the potentially significant role of North American wetlands in methane production, the activities associated with the restoration, creation, and protection of wetlands are likely to focus on the ecosystem services that wetlands provide, such as filtering of toxics, coastal erosion protection, wildlife habitat, and havens of biological diversity, rather than on carbon sequestration, *per se*.
- Research needs to reduce the uncertainties in carbon storage and fluxes in wetlands to provide information about management options in terms of carbon uptake and storage and trace gas fluxes.

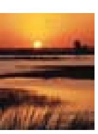


13.1 INTRODUCTION

While there are a variety of legal and scientific definitions of a wetland (National Research Council, 1995; National Wetlands Working Group, 1997), most emphasize the presence of waterlogged conditions in the upper soil profile during at least part of the growing season, and plant species and soil conditions that reflect these hydrologic conditions. Waterlogging tends to suppress microbial decomposition more than plant productivity, so wetlands are known for their ability to accumulate large amounts of soil carbon, most spectacularly seen in large peat deposits that are often many meters deep. Thus, when examining carbon dynamics, it is important to distinguish between freshwater wetlands with surface soil organic matter deposits greater than 40 cm thick (*i.e.*, peatlands) and those with lesser amounts of soil organic matter (*i.e.*, freshwater mineral-soil wetlands [FWMS]). Some wetlands have permafrost (fluxes and pools in wetlands with and without permafrost are discussed separately in Appendix F). We also differentiate between freshwater wetlands and estuarine wetlands (salt marshes, mangroves, and mud flats) with marine-derived salinity.

Peatlands occupy about 3% of the terrestrial global surface, yet they contain 16–33% of the total soil carbon pool (Gorham, 1991; Maltby and Immirzi, 1993)[1]. Most peatlands occur between 50 and 70º N, although significant areas occur at lower latitudes (Matthews and Fung, 1987; Aselmann and Crutzen, 1989; Maltby and Immirzi, 1993). Large areas of peatlands exist in Alaska, Canada, and in the northern midwestern, northeastern, and southeastern United States (Bridgham *et al*., 2000). Because this peat formed over thousands of years, these areas represent a large carbon pool, but with relatively slow rates of accumulation. By comparison, estuarine wetlands and some freshwater mineral-soil wetlands rapidly sequester carbon as soil organic matter due to rapid burial in sediments. Large areas of wetlands have been converted to other land uses, globally and in North America (Dugan, 1993; OECD, 1996), which may have resulted in a net flux of carbon to the atmosphere (Armentano and Menges, 1986; Maltby and Immirzi, 1993). Additionally, wetlands emit 92–237 million tons of methane (Mt CH_4) per year[1], which is a large fraction of the total annual global flux of about 600 Mt CH_4 per year (Ehhalt *et al*., 2001). This is important because CH_4 is a potent greenhouse gas (GHG), second in importance only to carbon dioxide (CO_2) (Ehhalt *et al*., 2001).

5.5% of the land area of the contiguous United States is wetlands. This represents just 48% of the original wetland area in the conterminous United States.



[1] The uncertainties for the numerical values cited in this chapter are presented and explained in Table 13.1 and Appendix F.

A number of previous studies have examined the role of peatlands in the global carbon balance (reviewed in Mitra *et al*., 2005), and Roulet (2000) focused on the role of Canadian peatlands in the Kyoto process. Here we augment these previous studies by considering all types of wetlands (not just peatlands) and integrate new data to examine the carbon balance in the wetlands of Canada, the United States, and Mexico. We also briefly compare these values to those from global wetlands. We limit this review to those components of the carbon budget that result in a net gaseous exchange with the atmosphere on an interannual basis and do not consider other internal carbon fluxes. We do not consider dissolved organic carbon (DOC) fluxes from wetlands, although they may be substantial (Moore, 1997), because the oxidation of the DOC would be counted as atmospheric carbon emissions in the receiving ecosystems downstream and we do not want to double-count fluxes.

Given that many undisturbed wetlands are a natural sink for CO_2 and a source of CH_4, a note of caution in interpretation of our data is important. Using the Intergovernmental Panel on Climate Change (IPCC) terminology, a radiative forcing denotes “an externally imposed perturbation in the radiative energy budget of the Earth’s climate system” (Ramaswamy *et al*., 2001). Thus, it is the change from a baseline condition in GHG fluxes in wetlands that constitute a radiative forcing that will impact climate change, and carbon fluxes in unperturbed wetlands are important only in establishing a baseline condition. For example, historical steady state rates of CH_4 emissions from wetlands have zero net radiative forcing, but an increase in CH_4 emissions due to climatic warming would constitute a positive radiative forcing. Similarly, steady state rates of soil carbon sequestration in wetlands have zero net radiative forcing, but the lost sequestration capacity and the oxidation of the soil carbon pool in drained wetlands are both positive radiative forcings.

13.2 INVENTORIES

13.2.1 Current Wetland Area and Rates of Loss

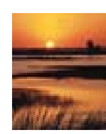
The current and original wetland area and rates of loss are the basis for all further estimates of pools and fluxes in this chapter. The loss of wetlands has caused the oxidation of their soil carbon, particularly in peatlands, reduced their ability to sequester carbon, and reduced their emissions of CH_4. The strengths and weakness of the wetland inventories of Canada, the United States, and Mexico are discussed in Appendix F.

The conterminous United States has 312,000 km^2 of FWMS wetlands, 93,000 km^2 of peatlands, and 25,000 km^2 of estuarine wetlands, which encompass 5.5% of the land area (Table 13.1). This represents just 48% of the original wetland area in the conterminous United States (Table F.1 in Appendix F).

Table 13.1 The area, carbon pool, net carbon balance, and methane flux from wetlands in North America and the world. Positive fluxes indicate net fluxes to the atmosphere, whereas negative fluxes indicate net fluxes into an ecosystem. Citations and assumptions in calculations are in the text and in Appendix F (see Box 13.1 for uncertainty conventions).

	Area[a] (km^2)	Carbon Pool[b] (Gt C)	Net Carbon Balance[c] (Mt C per year)	Historical Loss in Sequestration Capacity (Mt C per year)	Methane Flux (Mt CH_4 per year)
Canada					
Peatland	1,135,608****	152****	-19***	0.3*	3.2**
Freshwater Mineral	158,720**	4.9**	-2.7*	3.4*	1.2*
Estuarine	6,400***	0.1***	-1.3**	0.5*	0.0***
Total	1,300,728****	157****	-23**	4.2*	4.4*
Alaska					
Peatland	132,196****	15.9**	-2.0**	0.0****	0.3*
Freshwater Mineral	555,629****	27.1**	-9.4*	0.0****	1.4*
Estuarine	8,400****	0.1***	-1.9**	0.0****	0.0***
Total	696,224*****	43.2**	-13*	0.0****	1.7*
Conterminous United States					
Peatland	93,477****	14.4***	5.7*	1.2*	0.7**
Freshwater Mineral	312,193*****	6.2***	-9.8*	7.6*	2.4**
Estuarine	25,000*****	0.6*****	-5.4**	0.5*	0.0***
Total	430,670*****	21.2***	-9.5*	9.4*	3.1**
U.S. Total	1,126,895*****	64.3**	-23*	9.4*	4.8**
Mexico					
Peatland	10,000*	1.5*	-1.6*	ND*	0.1*
Freshwater Mineral	20,685*	0.4*	-0.4*	ND*	0.2*
Estuarine	5,000*	0.2*	-1.6*	1.0*	0.0*
Total	35,685*	2.0*	-3.6*	ND*	0.2*
North America					
Peatland	1,371,281****	184****	-17*	1.5*	4.3**
Freshwater Mineral	1,047,227****	39***	-22*	11*	5.1*
Estuarine	44,800***	0.9***	-10**	2.0*	0.1**
Total	2,463,308****	223****	-49*	15*	9.4*
Global					
Peatland	3,443,000***	462***	150**	16*	37**
Freshwater Mineral	2,315,000***	46***	-39*	45*	68**
Estuarine	203,000*	5.4*	-43*	21*	0.2**
Total	5,961,000***	513***	68*	82*	105**

[a] Estuarine includes salt marsh, mangrove, and mudflat, except for Mexico and global for which no mudflat estimates were available.
[b] Includes soil carbon and plant carbon, but overall soil carbon is 98% of the total pool.
[c] Includes soil carbon sequestration, plant carbon sequestration, and loss of carbon due to drainage of wetlands. Plant carbon sequestration and soil oxidative flux due to drainage are either unknown or negligible for North American wetlands except for the conterminous United States (see Appendix F).
ND indicates that no data are available.



BOX 13.1: CCSP SAP 2.2 Uncertainty Conventions

***** = 95% certain that the actual value is within 10% of the estimate reported,
**** = 95% certain that the estimate is within 25%,
*** = 95% certain that the estimate is within 50%,
** = 95% certain that the estimate is within 100%, and
* = uncertainty greater than 100%.

However, wetland losses in the United States have declined from 1855 km^2 per year in the 1950s–1970s to 237 km^2 per year in the 1980s–1990s (Dahl, 2000). Such data mask large differences in loss rates among wetland classes and conversion of wetlands to other classes (Dahl, 2000), with potentially large effects on carbon stocks and fluxes. For example, the majority of wetland losses in the United States have occurred in FWMS wetlands. As of the early 1980s, 84% of United States' peatlands were unaltered (Armentano and Menges, 1986; Maltby and Immirzi, 1993; Rubec, 1996), and, given the current regulatory environment in the United States, recent rates of loss are likely small.

Canada has 1,301,000 km^2 of wetlands, covering 14% of its land area, of which 87% are peatlands (Table 13.1). Canada has lost about 14% of its wetlands, mainly due to agricultural development of FWMS wetlands (Rubec, 1996), although the ability to estimate wetland losses in Canada is limited by the lack of a regular wetland inventory.

The wetland area in Mexico is estimated at 36,000 km^2 (Table 13.1), with an estimated historical loss of 16,000 km^2 (Table F.1 in Appendix F). However, given the lack of a nationwide wetland inventory and a general paucity of data, this number is highly uncertain.

> North America currently has about 43% of the global wetland area.

Problems with inadequate wetland inventories are even more prevalent in lesser developed countries (Finlayson *et al.*, 1999). We estimate a global wetland area of 6.0 × 10^6 km^2 (Table 13.1); thus, North America currently has about 43% of the global wetland area. It has been estimated that about 50% of the world's original wetlands have been converted to other uses (Moser *et al.*, 1996).

13.2.2 Carbon Pools

We estimate that North American wetlands have a current soil and plant carbon pool of 223 billion tons (Gt), of which approximately 98% is in the soil (Table 13.1). The majority of this carbon is in peatlands, with FWMS wetlands contributing about 18% of the carbon pool. The large amount of soil carbon (27 Gt) in Alaskan FWMS wetlands had not been identified in previous studies (see Appendix F).

13.2.3 Soil Carbon Fluxes

North American peatlands currently have a net carbon balance of about -17 million metric tons of carbon (Mt C) per year (Table 13.1), but several large fluxes are incorporated into this estimate. (Negative numbers indicate net fluxes into the ecosystem, whereas positive numbers indicate net fluxes into the atmosphere). Peatlands sequester -29 Mt C per year (Table F.2 in Appendix F). However, this carbon sink is partially offset by a net oxidative flux of 18 Mt C per year as of the early 1980s in peatlands in the conterminous United States that have been drained for agriculture and forestry (Armentano and Menges, 1986). Despite a substantial reduction in the rate of wetland loss since the 1980s (Dahl, 2000), drained organic soils continue to lose carbon over many decades, so the actual flux to the atmosphere is probably close to the 1980s estimate. There has also been a loss in sequestration capacity in drained peatlands of 1.5 Mt C per year (Table 13.1), so the overall soil carbon sink of North American peatlands is about 20 Mt C per year smaller than it would have been in the absence of disturbance.

Very little attention has been given to the role of FWMS wetlands in North American or global carbon balance estimates, with the exception of CH_4 emissions. Carbon sequestration associated with sediment deposition is a potentially large, but poorly quantified, flux in wetlands (Stallard, 1998; Smith *et al.*, 2001). We estimate that North American FWMS wetlands sequester -18 Mt C per year in sedimentation (Table F.2 in Appendix F). However, as discussed in Appendix F, wetland sedimentation rates are extremely variable. Moreover, almost no studies have placed sediment carbon sequestration in FWMS wetlands in a landscape context, considering allochthonous-derived (from on-site plant production) versus autochthonous-derived (imported from outside the wetland) carbon, replacement of carbon in terrestrial source areas, and differences in decomposition rates between sink and source areas (Stallard, 1998; Harden *et al.*, 1999; Smith *et al.*, 2001). However, it is clear that sedimentation in FWMS wetlands

is a potentially substantial carbon sink and an important unknown in carbon budgets. For example, agriculture typically increases sedimentation rates by 10- to 100-fold and 90% of sediments are stored within the watershed, amounting to about -40 Mt C per year in the conterminous United States (Stallard, 1998; Smith *et al.*, 2001). Our estimate of sediment carbon sequestration in FWMS wetlands seems quite reasonable in comparison to within-watershed sediment storage in North America. Moreover, Stallard (1998) and Smith *et al.* (2001) estimated a global sediment sink on the order of -1 Gt C per year.

Decomposition of soil carbon in FWMS wetlands that have been converted to other land uses appears to be responsible for only a negligible loss of soil carbon, currently (Table F.2 in Appendix F). However, due to the historical loss of FWMS wetland area, we estimate that they currently sequester 11 Mt C per year less than they did prior to disturbance (Table 13.1). This estimate has the same unknowns described in the previous paragraph on current sediment carbon sequestration in extant FWMS wetlands.

We estimate that estuarine wetlands currently sequester -10.2 Mt C per year (Table F.2 in Appendix F), with a historical reduction in sequestration capacity of 2.0 Mt C per year due to loss of area (Table 13.1). However, the reduction is almost certainly greater because our "original" area is only from the 1950s. Despite the relatively small area of estuarine wetlands, they currently contribute about 31% of total wetland carbon sequestration in the conterminous United States and about 18% of the North American total. Estuarine wetlands sequester carbon at a rate about 10 times higher on an area basis than other wetland ecosystems due to high sedimentation rates, high soil carbon content, and constant burial due to sea level rise. Estimates of sediment deposition rates in estuarine wetlands are reasonably robust, but the same 'landscape' issues of allochthonous versus autochthonous inputs of carbon, replenishment of carbon in source area soils, and differences in decomposition rates between sink and source areas exist as for FWMS wetlands. Another large uncertainty in the estuarine carbon budget is the area and carbon content of mud flats, particularly in Canada and Mexico.

Overall, North American wetland soils appear to be a substantial carbon sink with a net flux of -49 Mt C per year (with very large error bounds because of FWMS wetlands) (Table 13.1). The large-scale conversion of wetlands to upland uses has led to a reduction in the wetland soil carbon sequestration capacity of 15 Mt C per year from the likely historical rate (Table 13.1), but this estimate is driven by large losses of FWMS wetlands with their highly uncertain sedimentation carbon sink. Adding in the current net oxidative flux of 18 Mt C per year from conterminous United States' peatlands, we estimate that North American wetlands currently sequester 33 Mt C per year less than they did historically (Table F.2 in Appendix F). Furthermore, North American peatlands and FWMS wetlands have lost 2.6 Gt and 0.8 Gt of soil carbon, respectively, and collectively they have lost 2.4 Gt of plant carbon since approximately 1800. Very little data exist to estimate carbon fluxes for freshwater Mexican wetlands, but because of their small area, they will not likely have a large impact on the overall North American estimates.

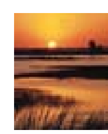
The global wetland soil carbon balance has only been examined in peatlands, which currently are a moderate source of atmospheric carbon of about 150 Mt C per year (Table 13.1), largely due to the oxidation of peat drained for agriculture and forestry and secondarily due to peat combustion for fuel (Armentano and Menges, 1986; Maltby and Immirzi, 1993). The cumulative historical shift in soil carbon stocks has been estimated to be 5.5 to 7.1 Gt C (Maltby and Immirzi, 1993). Although we are aware of no previous evaluation of the carbon balance of global FWMS and estuarine wetlands, using the assumption noted above, we estimate that they are a sink of approximately -39 and -43 Mt per year, respectively.

13.2.4 Methane and Nitrous Oxide Emissions

We estimate that North American wetlands emit 9.4 Mt CH_4 per year (Table 13.1). For comparison, a mechanistic CH_4 model yielded emissions of 3.8 and 7.1 Mt CH_4 per year for Alaska and Canada, respectively (Zhuang *et al.*, 2004). A regional inverse atmospheric modeling approach estimated total CH_4 emissions (from all sources) of 16 and 54 Mt CH_4 per year for boreal and temperate North America, respectively (Fletcher *et al.*, 2004b).

Despite the relatively small area of estuarine wetlands, they currently contribute about 31% of total wetland carbon sequestration in the conterminous United States and about 18% of the North American total.

Methane emissions are currently about 5 Mt CH_4 per year less than they were historically in North American wetlands (see Table F.4 in Appendix F) because of the loss of wetland area. We do not consider the effects of conversion of wetlands from one type to another (Dahl, 2000), which may have a significant impact on CH_4 emissions. Similarly, we estimate that global CH_4 emissions from natural wetlands are only about half of what they were historically due to loss of area (Table F.4 in Appendix F). However, this may be an overestimate because wetland losses have been higher in more developed countries than less developed countries (Moser *et al.*, 1996), and wetlands at lower latitudes have higher emissions on average (Bartlett and Harriss, 1993).

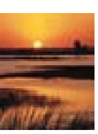


When we multiplied the very low published estimates of nitrous oxide emissions from natural and disturbed wetlands (Joosten and Clarke, 2002) by North American wetland area, the flux was insignificant (data not shown). However, nitrous oxide emissions have been measured in few wetlands, particularly in FWMS wetlands and wetlands with high nitrogen inputs (*e.g.*, from agricultural run-off) where emissions might be expected to be higher.

We use global warming potentials (GWPs) as a convenient way to compare the relative contributions of CO_2 and CH_4 fluxes in North American wetlands to the Earth's radiative balance. The GWP is the radiative effect of a pulse of a substance into the atmosphere relative to CO_2 over a particular time horizon (Ramaswamy *et al.*, 2001). However, it is important to distinguish between *radiative balance*, which refers to the static radiative effect of a substance, and *radiative forcing*, which refers to an externally imposed perturbation on the Earth's radiative energy budget (Ramaswamy *et al.*, 2001). Thus, changes in radiative balance lead to a radiative forcing, which subsequently leads to a change in the Earth's surface temperature. For example, wetlands have a large effect on the Earth's radiative balance through high CH_4 emissions, but it is only to the extent that emissions change through time that they represent a positive or negative radiative forcing and impact climate change.

Historically, the destruction of wetlands through land-use changes has had the largest effect on the carbon fluxes.

Methane has GWPs of 1.9, 6.3, and 16.9 CO_2-carbon equivalents on a mass basis across 500-year, 100-year, and 20-year time frames, respectively (Ramaswamy *et al.*, 2001)[2]. Depending upon the time frame and within the large confidence limits of many of our estimates in Table 13.1, the *net radiative balance* of North American wetlands as a whole currently are approximately neutral in terms of net CO_2-carbon equivalents to the atmosphere (note that we discuss *net radiative forcing* in *Trends and Drivers of Wetland Carbon Fluxes,* Section 13.3). The exception is estuarine wetlands, which are a net sink for CO_2-carbon equivalents because they support both rapid rates of carbon sequestration and low CH_4 emissions. However, caution should be exercised in using GWPs to draw conclusions about changes in the net flux of CO_2-carbon equivalents because GWPs are based upon a pulse of a gas into the atmosphere, whereas carbon sequestration is more or less continuous. For example, if one considers continuous CH_4 emissions and carbon sequestration in peat over time, most peatlands are a net sink for CO_2-carbon equivalents because of the long lifetime of CO_2 sequestered as peat (Frolking *et al.*, 2006).



Wetland ecosystems provide havens for biodiversity, recharge of groundwater, and reduction in flooding and fish nurseries.

[2] GWPs in Ramaswamy *et al.* (2001) were originally reported in CO_2-mass equivalents. We have converted them into CO_2-carbon equivalents so that the net carbon balance and CH_4 flux columns in Table 13.1 can be directly compared by multiplying CH_4 fluxes by the GWPs given here.

13.2.5 Plant Carbon Fluxes

We estimate that wetland forests in the conterminous United States currently sequester -10.3 Mt C per year as increased plant biomass (see Table F.3 in Appendix F). Sequestration in plants in undisturbed wetland forests in Alaska, many peatlands, and estuarine wetlands is probably minimal, although there may be substantial logging of Canadian forested peatlands that we do not have the data to account for.

13.3 TRENDS AND DRIVERS OF WETLAND CARBON FLUXES

While extensive research has been done on carbon cycling and pools in North American wetlands, to our knowledge, this is the first attempt at an overall carbon budget for all of the wetlands of North America, although others have examined the carbon budget for North American peatlands as part of global assessments (Armentano and Menges, 1986; Maltby and Immirzi, 1993; Joosten and Clarke, 2002). Historically, the destruction of wetlands through land-use changes has had the largest effect on the carbon fluxes and, consequently, the radiative forcing of North American wetlands. The primary effects have been a reduction in their ability to sequester carbon (a small to moderate increase in radiative forcing depending on carbon sequestration by sedimentation in FWMS and estuarine wetlands), oxidation of their soil carbon reserves upon drainage (a small increase in radiative forcing), and a reduction in CH_4 emissions (a small to large decrease in radiative forcing depending on actual emissions) (Table F.1 and Appendix F). Globally, the disturbance of peatlands appears to have shifted them into a net source of carbon to the atmosphere. Any positive effect of wetland loss due to a reduction in their CH_4 emissions, and hence radiative forcing, will be more than negated by the loss of the many ecosystem services they provide, such as havens for biodiversity, recharge of groundwater, reduction in flooding, fish nurseries, *etc.* (Zedler and Kercher, 2005).

A majority of the effort in examining future global change impacts on wetlands has focused on northern peatlands because of their large soil carbon reserves, although under current climate conditions they have modest CH_4 emissions (Moore and Roulet, 1995; Roulet, 2000; Joosten and Clarke, 2002, and references therein). The effects of global change

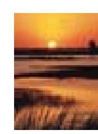
on carbon sequestration in peatlands are probably of minor importance as a global flux because of the relatively low rate of peat accumulation. However, losses of soil carbon stocks in peatlands drained for agriculture and forestry (Table F.2 in Appendix F) attest to the possibility of large losses from the massive soil carbon deposits in northern peatlands if they become substantially drier in a future climate. Furthermore, Turetsky *et al.* (2004) estimated that up to 5.9 Mt C per year are released from western Canadian peatlands by fire and predicted that increases in fire frequency may cause these systems to become net atmospheric carbon sources. We did not add this flux to our estimate of the net carbon balance of North American wetlands because historical oxidation of peat by fire should be integrated in the peat sequestration estimates and recent changes due to anthropogenic effects are highly uncertain.

Our compilation shows that attention needs to be directed toward understanding climate change impacts to FWMS wetlands, which collectively emit similar amounts of CH_4 and potentially sequester an equivalent amount of carbon than peatlands. The effects of changing water table depths are somewhat more tractable in FWMS wetlands than peatlands because FWMS wetlands have less potential for oxidation of soil organic matter. In forested FWMS wetlands, increased precipitation and runoff may increase radiative forcing by simultaneously decreasing wood production and increasing methanogenesis (Megonigal *et al*., 2005). The influence of changes in hydrology on CH_4 emissions, plant productivity, soil carbon preservation, and sedimentation will need to be addressed in order to fully anticipate climate change impacts on radiative forcing in these systems.

The effects of global change on estuarine wetlands is of concern because sequestration rates are rapid, and they can be expected to increase in proportion to the rate of sea level rise provided estuarine wetland area does not decline. Because CH_4 emissions from estuarine wetlands are low, this increase in sequestration capacity could represent a net decrease in radiative forcing, depending on how much of the sequestered carbon is autochthonous. Changes in tidal wetland area with sea-level rise will depend on rates of inland migration, erosion at the wetland-estuary boundary, and wetland elevation change. The rate of loss of tidal wetland area has declined in past decades due to regulations on draining and filling activities (Dahl, 2000). However, rapid conversion to open water is occurring in coastal Louisiana (Bourne, 2000) and Maryland (Kearney and Stevenson, 1991), suggesting that marsh area will decline with increased rates of sea level rise (Kearney *et al*., 2002). A multitude of human and climate factors are contributing to the current losses (Turner, 1997; Day Jr. *et al*., 2000; Day Jr. *et al*., 2001). Although it is uncertain how global changes in climate, eutrophication, and other factors will interact with sea level rise (Najjar *et al*., 2000), it is likely that increased rates of sea level rise will cause an overall decline in estuarine marsh area and soil carbon sequestration.

One of the greatest concerns is how climate change will affect future CH_4 emissions from wetlands because of their large GWP. Wetlands emit about 105 Mt CH_4 per year (Table 13.1), or 20% of the global total. Increases in atmospheric CH_4 concentrations over the past century have had the second largest radiative forcing (after CO_2) in human-induced climate change (Ehhalt *et al*., 2001). Moreover, CH_4 fluxes from wetlands have provided an important radiative feedback on climate over the geologic past (Chappellaz *et al*., 1993; Blunier *et al*., 1995; Petit *et al*., 1999). The large global warming observed since the 1990s may have resulted in increased CH_4 emissions from wetlands (Fletcher *et al*., 2004a; Wang *et al*., 2004; Zhuang *et al*., 2004).

It is likely that increased rates of sea level rise will cause an overall decline in estuarine marsh area and soil carbon sequestration.

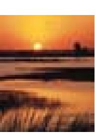


Data (Bartlett and Harriss, 1993; Moore *et al*., 1998; Updegraff *et al*., 2001) and modeling (Gedney *et al*., 2004; Zhuang *et al*., 2004) strongly support the contention that water table position and temperature are the primary environmental controls over CH_4 emissions. How this generalization plays out with future climate change is, however, more complex. For example, most climate models predict much of Canada will be warmer and drier in the future. Based upon this prediction, Moore *et al.* (1998) proposed a variety of responses to climate change in the carbon fluxes from different types of Canadian peatlands. Methane emissions may increase in collapsed former-permafrost bogs (which will be warmer and wetter) but decrease in fens and other types of bogs (warmer and drier). A CH_4-process model predicted that modest warming will increase global wetland emissions, but larger increases in temperature will decrease emissions because of drier conditions (Cao *et al*., 1998).

The direct, non-climatic effects of increasing atmospheric CO_2 on carbon cycling in wetland ecosystems has received far less attention than upland systems. Field studies have been done in tussock tundra (Tissue and Oechel, 1987; Oechel *et al*., 1994), bog-type peatlands (Hoosbeek *et al*., 2001), rice paddies (Kim *et al*., 2001), and a salt marsh (Rasse *et al*., 2005); and a somewhat wider variety of wetlands have been studied in small scale glasshouse systems. Temperate and tropical wetland ecosystems consistently respond to elevated CO_2 with an increase in photosynthesis and/or biomass (Vann and Megonigal, 2003). By comparison, the response of northern peatland plant communities has been inconsistent. A hypothesis that remains untested is that the elevated CO_2 response of northern peatlands will be limited by nitrogen availability. In an *in situ* study of tussock tundra, complete photosynthetic acclimation occurred when CO_2 was elevated, but acclimation was far less severe with both elevated CO_2 and a 4°C increase in air temperature (Oechel *et al*., 1994). It was hypothesized that soil warming relieved a severe nutrient limitation on photosynthesis by increasing nitrogen mineralization.

A consistent response to elevated CO_2-enhanced photosynthesis in wetlands is an increase in CH_4 emissions ranging from 50 to 350% (Megonigal and Schlesinger, 1997; Vann and Megonigal, 2003). It is generally assumed that the increased supply of plant photosynthate stimulates anaerobic microbial carbon metabolism, of which CH_4 is a primary end product. An increase in CH_4 emissions from wetlands due to elevated CO_2 constitutes a positive feedback on radiative forcing because CO_2 is rapidly converted to a more effective GHG (CH_4).

An elevated CO_2-induced increase in CH_4 emissions may be offset by an increase in carbon sequestration in soil organic matter or wood. Although there are very little data to evaluate this hypothesis, a study on seedlings of a wetland-adapted tree species reported that elevated CO_2 stimulated photosynthesis and CH_4 emissions, but not growth, under flooded conditions (Megonigal *et al*., 2005). It is possible that elevated CO_2 will stimulate soil carbon sequestration, particularly in tidal wetlands experiencing sea level rise, but a net loss of soil carbon is also possible due to priming effects (*i.e*., increased labile carbon inputs from elevated CO_2 enhance decomposition of the overall soil carbon pool) (Hoosbeek *et al*., 2004; Lichter *et al*., 2005). Elevated CO_2 has the potential to influence the carbon budgets of adjacent aquatic ecosystems by increasing export of dissolved organic carbon (Freeman *et al*., 2004) and dissolved inorganic carbon (Marsh *et al*., 2005).



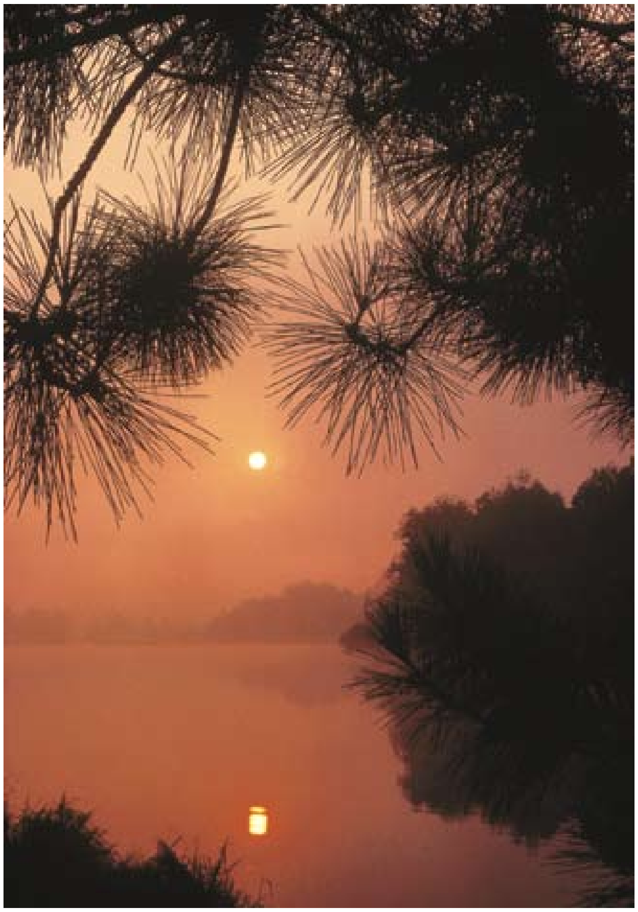
Other important anthropogenic forcing factors that will affect future CH_4 emissions include atmospheric sulfate deposition (Vile *et al*., 2003; Gauci *et al*., 2004) and nutrient additions (Keller *et al*., 2005). These external forcing factors in turn will interact with internal ecosystem constraints such as pH and carbon quality (Moore and Roulet, 1995; Bridgham *et al*., 1998), anaerobic carbon flow (Hines and Duddleston, 2001), and net ecosystem productivity and plant community composition (Whiting and Chanton, 1993; Updegraff *et al*., 2001; Strack *et al*., 2004) to determine the actual response.

13.4 OPTIONS FOR MANAGEMENT

Wetland policies in the United States and Canada are driven by a variety of federal, state or provincial, and local laws and regulations in recognition of the many wetland ecosystem services and large historical loss rates (Lynch-Stewart *et al*., 1999; National Research Council, 2001; Zedler and Kercher, 2005). Thus, any actions to enhance the ability of wetlands to sequester carbon, or reduce their CH_4 emissions, must be implemented within the context of the existing regulatory framework. The most important option in the United States has already been largely achieved, and that is to reduce the historical rate of peatland losses with their accompanying large oxidative losses of the stored soil carbon. Decreases in the rates of loss of all wetlands have helped to maintain their soil carbon sequestration potential.

There has been strong interest expressed in using carbon sequestration as a rationale for wetland restoration and creation in the United States, Canada, and elsewhere (Wylynko, 1999; Watson *et al*., 2000). However, high CH_4 emissions from conterminous United States' wetlands suggest that creating and restoring wetlands may increase net radiative forcing, although adequate data do not exist to fully evaluate this possibility. Roulet (2000) came to a similar conclusion concerning the restoration of Canadian wetlands. Net radiative forcing from restoration will likely vary among different kinds of wetlands and the specifics of their carbon budgets. The possibility of increasing radiative forcing by creating or restoring wetlands does not apply to estuarine wetlands, which emit relatively little CH_4 compared to the carbon they sequester. Restoration of drained peatlands may stop the rapid loss of their soil carbon, which may compensate for increased CH_4 emissions. However, Canadian peatlands restored from peat extraction operations increased their net emissions of carbon because of straw addition during the restoration process, although it was assumed that they would eventually become a net sink (Cleary *et al*., 2005).

Regardless of their internal carbon balance, the area of restored wetlands is currently too small to form a significant carbon sink at the continental scale. Between 1986 and 1997,



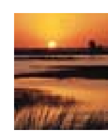
only 4157 km^2 of uplands were converted into wetlands in the conterminous United States (Dahl, 2000). Using the soil carbon sequestration rate of 3.05 Mg C per hectare per year found by Euliss *et al.* (2006) for restored prairie pothole wetlands[3], we estimate that wetland restoration in the United States would have sequestered 1.3 Mt C over this 11-year period. However, larger areas of wetland restoration may have a significant impact on carbon sequestration. A simulation model of planting 20,000 km^2 into bottomland hardwood trees as part of the Wetland Reserve Program in the United States showed a sequestration of 4 Mt C per year through 2045 (Barker *et al.*, 1996). Euliss *et al.* (2006) estimated that if all cropland on former prairie pothole wetlands in the United States and Canada (162,244 km^2) were restored that 378 Mt C would be sequestered over 10 years in soils and plants. However, neither study accounted for the GWP of increased CH_4 emissions.

Potentially more significant is the conversion of wetlands from one type to another; for example, 8.7% (37,200 km^2) of the wetlands in the conterminous United States in 1997 were in a previous wetland category in 1986 (Dahl, 2000). The net effect of these conversions on wetland carbon fluxes is unknown. Similarly, Roulet (2000) argued that too many uncertainties exist to include Canadian wetlands in the Kyoto Protocol.

In summary, North American wetlands form a very large carbon pool, primarily because of storage as peat, and are a small-to-moderate carbon sink (excluding CH_4 effects). The largest unknown in the wetland carbon budget is the amount and significance of sedimentation in FWMS and estuarine wetlands, and CH_4 emissions in freshwater wetlands. With the exception of estuarine wetlands, CH_4 emissions from wetlands may largely offset any positive benefits of carbon sequestration in soils and plants. Given these conclusions, it is probably unwarranted to use carbon sequestration as a rationale for the protection and restoration of FWMS wetlands, although the many other ecosystem services that they provide justify these actions. However, protecting and restoring peatlands will stop the loss of their soil carbon (at least over the long term) and estuarine wetlands are an important carbon sink given their limited areal extent and low CH_4 emissions.

The most important areas for further scientific research in terms of current carbon fluxes in the United States are to establish an unbiased, landscape-level sampling scheme to determine sediment carbon sequestration in FWMS and estuarine wetlands and additional measurements of annual CH_4 emissions to better constrain these important fluxes. It would also be beneficial if the approximately decadal National Wetland Inventory (NWI) status and trends data were collected in sufficient detail with respect to the Cowardin *et al.* (1979) classification scheme to determine changes among mineral-soil wetlands and peatlands.

Larger areas of wetland restoration may have a significant impact on carbon sequestration, but may also increase methane emissions offsetting any positive greenhouse gas effects.



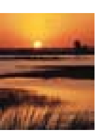
Canada lacks any regular inventory of its wetlands, and thus, it is difficult to quantify land-use impacts upon their carbon fluxes and pools. While excellent scientific data exists on most aspects of carbon cycling in Canadian peatlands, Canadian FWMS and estuarine wetlands have been relatively poorly studied, despite having suffered large proportional losses to land-use change. Wetland data for Mexico is almost entirely lacking. Thus, anything that can be done to improve upon this would be helpful. All wetland inventories should consider the area of estuarine mud flats, which have the potential to sequester considerable carbon and are poorly understood with respect to carbon sequestration.

[3] Euliss *et al.* (2006) regressed surface soil carbon stores in 27 restored semi-permanent prairie pothole wetlands against years since restoration to derive this estimate ($r^2 = 0.31$, $P = 0.002$). However, there was no significant relationship in seasonal prairie pothole wetlands ($r^2 = 0.04$, $P = 0.241$).

The greatest unknown is how global change will affect the carbon pools and fluxes of North American wetlands. We will not be able to accurately predict the role of North American wetlands as potential positive or negative feedbacks to anthropogenic climate change without knowing the integrative effects of changes in temperature, precipitation, atmospheric CO_2 concentrations, and atmospheric deposition of nitrogen and sulfur within the context of internal ecosystem drivers of wetlands. To our knowledge, no manipulative experiment has simultaneously measured more than two of these perturbations in any North American wetland, and few have been done at any site. Modeling expertise of the carbon dynamics of wetlands has rapidly improved in the last few years (Frolking *et al.*, 2002; Zhuang *et al.*, 2004, and references therein), but this needs even further development in the future, including for FWMS and estuarine wetlands.

ACKNOWLEDGMENTS

Steve Campbell (U.S. Department of Agriculture [USDA] Natural Resource Conservation Service [NRCS], OR) synthesized the National Soil Information database so that it was useful to us. Information on wetland soils within specific states was provided by Joseph Moore (USDA NRCS, AK), Robert Weihrouch (USDA NRCS, WI), and Susan Platz (USDA NRCS, MN). Charles Tarnocai provided invaluable data on Canadian peatlands. Thomas Dahl (U.S. Fish and Wildlife Service) explored the possibility of combining NWI data with United States' soils maps. Nigel Roulet (McGill University) gave valuable advice on recent references. R. Kelman Wieder provided useful initial information on peatlands in Canada. Comments of two anonymous reviewers and Shuguang Liu (USGS Center for Earth Resources Observation and Science) greatly improved this manuscript.



CHAPTER 14

Human Settlements and the North American Carbon Cycle

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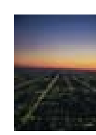
KEY FINDINGS

- Human settlements occupy almost 5% of the North American land area.
- There is currently insufficient information to determine the complete carbon balance of human settlements in North America. Fossil-fuel emissions, however, very likely dominate carbon fluxes from settlements.
- An estimated 410 to 1679 million tons of carbon are currently stored in the urban tree component of North American settlements. The growth of urban trees in North America produces a sink of approximately 16 to 49 million tons of carbon per year, which is 1 to 3% of the fossil-fuel emissions from North America in 2003.
- Estimates of historical trends of the net carbon balance of North American settlements are not available. Fossil-fuel emissions have likely gone up with the growth of urban lands, but the net balance of carbon loss during conversion of natural to urban or suburban land cover and subsequent uptake by lawns and urban trees is highly uncertain.
- The density and development patterns of human settlements are drivers of fossil-fuel emissions, especially in the residential and transportation sectors. Biological carbon gains and losses are influenced by type of predevelopment land cover, post-development urban design and landscaping choices, soil and landscape management practices, and the time since land conversion.
- Projections of future trends in the net carbon balance of North American settlements are not available. However, the projected expansion of urban areas in North America will strongly impact the future North American carbon cycle as human settlements affect (1) the direct emission of carbon dioxide from fossil-fuel combustion, (2) plant and soil carbon cycling in converting wildlands to residential and urban land cover.
- A number of municipalities in Canada, Mexico, and the United States have made commitments to voluntary greenhouse gas emission reductions under the Cities for Climate Protection program of International Governments for Local Sustainability (formerly the International Council for Local Environmental Initiatives [ICLEI]). Reductions have in some cases been associated with improvements in air quality.
- Research is needed to improve comprehensive carbon inventories for settled areas, to improve understanding of how development processes relate to driving forces for the carbon cycle, and to improve linkages between understanding of human and environmental systems in settled areas.



14.1 BACKGROUND

Activities in human settlements form the basis for much of North America's contribution to global carbon dioxide (CO_2) emissions. Settlements such as cities, towns, and suburbs vary widely in density, form, and distribution. Urban settlements, as they have been defined by the census bureaus of the United States, Canada, and Mexico, make up approximately 75 to 80% of the population of the continent, and this proportion is projected to continue to increase (United Nations, 2004). The density and forms of new development will strongly impact the future trajectory of the North American carbon cycle as human settlements affect the carbon cycle by (1) direct emission of CO_2 from fossil-fuel combustion, (2) alterations to plant and soil carbon cycles in conversion of wildlands to residential and urban land cover, and (3) indirect effects of residential and urban land cover on energy use and ecosystem carbon cycling.

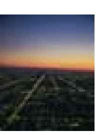
14.2 CARBON INVENTORIES OF HUMAN SETTLEMENTS

Conversion of agricultural and wildlands to settlements of varying densities is occurring at a rapid rate in North America, faster, in fact, than the rate of population growth. For example, according to U.S. Census Bureau estimates, urban land in the coterminous United States increased by 23% in the 1990s (Nowak *et al.*, 2005) while the population increased by 13%.[1]† Given these trends, it is important to determine the carbon balance of different types of settlements and how future urban policy and planning may impact the magnitude of CO_2 sources and sinks at regional, continental, and global scales. However, unlike many other types of common land cover, complete carbon inventories including fossil-fuel emissions and biological sources and sinks of carbon have been conducted only rarely for settlements as a whole. Assessing the carbon balance of settlements is challenging, as they are characterized by large CO_2 emissions from fuel combustion and decomposition of organic waste as well as transformations to vegetation and soil that affect carbon sources and sinks.

Conversion of agricultural and wildlands to settlements of varying densities is occurring at a rapid rate in North America, faster, in fact, than the rate of population growth.

Determining the extent of human settlements across North America also presents a challenge, as definitions of "developed," "built-up," and "urban" land vary greatly, particularly among nations. The U.S., Canadian, and Mexican census definitions are not consistent; in addition, several other classification schemes for defining and mapping settlements have been developed, such as the U.S. Department of Agriculture's National Resource Inventory categorization of developed land, which uses a variety of methods based on satellite imagery and ground-based information. One method of classifying settled land cover that has been consistently applied at a continental scale is the Global Rural-Urban Mapping Project conducted by a consortium of institutions, including Columbia University and the World Bank (CIESIN *et al.*, 2004). This estimate, which is based on nighttime lights satellite imagery, is 1,039,450 km^2, almost 5% of the total continental land area (Figure 14.1).†

Currently, there is insufficient information to determine the complete current or historical carbon balance of total continental land area. Fossil-fuel emissions very likely dominate carbon fluxes from settlements, just as settlement-related emissions likely dominate total fossil-fuel consumption in North America. However, specific estimates of the proportion of total fossil-fuel emissions directly attributable to settlements are difficult to make given current inventory methods, which are often conducted on a state or province-wide basis. In addition, the biological component of the carbon balance of settlements is highly uncertain, particularly with regard to the influence of urbanization on soil carbon pools and biogenic greenhouse gas emissions.



For the urban tree component of the settlement carbon balance, carbon stocks and sequestration have been estimated for urban land cover (as defined by the U.S. Census Bureau) in the coterminous United States to be on the order of 700 million tons (Mt) (335-980 million metric tons of carbon [Mt C]) with sequestration rates of 22.8 Mt C per year (13.7-25.9 Mt C per year) (Nowak and Crane, 2002). These estimates

[1] † A dagger symbol indicates that the magnitude and/or range of uncertainty for the given numerical value(s) is not provided in the references cited.

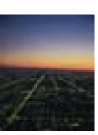
Figure 14.1 North America urban extents.

encompass a great deal of regional variability and contain some uncertainty about differences in carbon allocation between urban and natural trees, as urban trees have been less studied. However, to a first approximation, these estimates can be used to infer a probable range of urban tree carbon stocks and gross sequestration on a continental basis. Nowak and Crane (2002) estimated that urban tree carbon storage in the Canadian border states (excluding semi-arid Montana, Idaho, and North Dakota) ranged from 24 to 45 tons of carbon per hectare (t C per ha), and carbon sequestration ranged from 0.8 to 1.5 t C per ha per year. Applying these values to a range of estimates of the extent of urban

land in Canada (28,045 km^2 from the 1996 Canadian Census and 131,560 km^2 from CIESIN *et al.*, 2004), Canadian urban forest carbon stocks are between 67 and 592 Mt while carbon sequestration rates are between 2.2 and 19.7 Mt C per year. Similarly, for Mexico, Nowak and Crane (2002) estimated that urban carbon storage and sequestration in the U.S. southwestern states varied from 4.4 to 10.5 t per ha and 0.1 to 0.3 t per ha per year, respectively, leading to estimates of 10 to 107 Mt C stored in urban trees in Mexico and 0.2 to 3.1 Mt C per year sequestered. In this analysis, urban "trees" were defined as vegetation with woody stems greater than 1 inch diameter as measured 4.5 feet from the ground; carbon storage of other types of urban vegetation is not included in these estimates. Estimates of historical trends are not available.

Projections for increases in the extent of developed, nonfederal land cover in the United States in the next 25 years would increase the proportion of developed land from 5.2% to 9.2% of total land cover.

While complete national or continental-scale estimates of the carbon budget of settlements including fossil fuels, vegetation, and soils are not available, several methods are available to assess the full carbon balance of individual settlements and can be applied in the next several years toward constructing larger-scale inventories. Atmospheric measurements can be used to determine the net losses of carbon from settlements and urbanizing regions (Grimmond *et al.*, 2002; Grimmond *et al.*, 2004; Nemitz *et al.*, 2002; Soegaard and Moller-Jensen, 2003). Specific sources of CO_2 can be determined from unique isotopic signatures (Pataki *et al.*, 2003; Pataki *et al.*, 2006b) and from the relationship between CO_2 and carbon monoxide (Lin *et al.*, 2004). Many of these techniques have been commonly applied to natural ecosystems and may be easily adapted for settled regions. In addition, there have been several attempts to quantify the "metabolism" of human settlements in terms of their inputs and outputs of energy, materials, and wastes (Decker *et al.*, 2000) and the "footprint" of settlements in terms of the land area required to supply their consumption of resources and to offset CO_2 emissions (Folke *et al.*, 1997). Often these calculations include local flows and transformations of materials as well as upstream energy use and carbon appropriation, such as remote electrical power generation and food production.



To conduct metabolic and footprint analyses of specific settlements, energy and fuel use statistics are needed for individual municipalities, and these data are seldom made available at that scale. Consequently, metabolic and footprint analyses of carbon flows and conversions associated with metropolitan regions have been conducted for a relatively small number of cities. A metabolic analysis of the Toronto metropolitan region showed per capita net CO_2 emissions of 14 t CO_2 per year† (Sahely *et al.*, 2003), higher than analyses of other large metropolitan areas in developed countries (Newman, 1999; Pataki *et al.*, 2006a; Warren-Rhodes and Koenig, 2001). In contrast, an analysis of Mexico City estimated per capita CO_2 emissions of 3.4 t CO_2 per year† (Romero Lankao *et al.*, 2004). Local emissions inventories can provide useful supplements to national and global inventories in order to ensure that emissions reductions policies are applied effectively and equitably (Easterling *et al.*, 2003). A detailed review of methodological uncertainties and research needs is given in Pataki *et al.* (2006b).

Current projections for urban land development in North America highlight the importance of improving carbon inventories of settlements and assessing patterns and impacts of future urban and rural development. Projections for increases in the extent of developed, nonfederal land cover in the United States in the next 25 years are as high as 79%, which would increase the proportion of developed land from 5.2% to 9.2% of total land cover (Alig *et al.*, 2004). The potential consequences of this increase for the carbon cycle are significant in terms of CO_2 emissions from an expanded housing stock and transportation network as well as from conversion of agricultural land, forest, rangeland, and other ecosystems to urban land cover. Because the dynamics of carbon cycling in settled areas encompass a range of physical, biological, social, and economic processes, studies of the potential impacts of future development on the carbon cycle must be interdisciplinary. Large-scale research on what has been called the study "of cities as ecosystems" (Pickett *et al.*, 2001) has begun only relatively recently, pioneered

Table 14.1 Increases in number of households and the total population of the United States, Canada, and Mexico between 1985 and and 2000. (United Nations, 2002; United Nations Habitat, 2003).

	Total Population (%)	Households (%)
Canada	19	39
Mexico	33	60
United States	15	25

by interdisciplinary studies such as the National Science Foundation's Long-Term Ecological Research sites in the central Arizona-Phoenix area and in Baltimore (Grimm *et al.*, 2000). Although there is not yet sufficient data to construct a complete carbon inventory of settlements across North America, it is a feasible research goal to do so in the next several years if additional studies in individual municipalities are conducted in a variety of urbanizing regions.

14.3 TRENDS AND DRIVERS

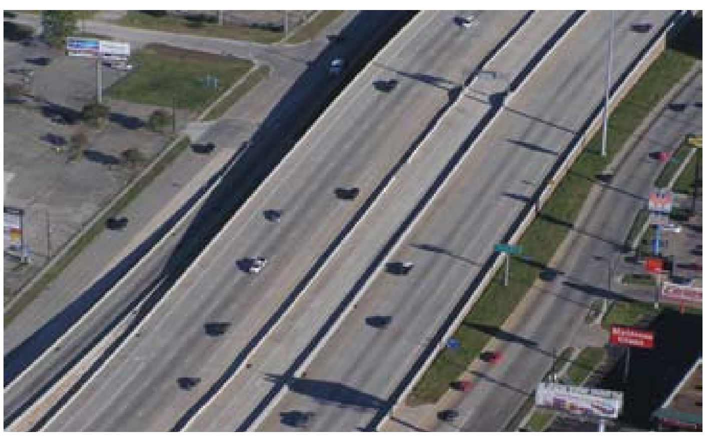
Drivers of change in the carbon cycle associated with human settlements include (1) factors that influence the rate of land conversion and urbanization, such as population growth and density, household size, economic growth, and transportation infrastructure; (2) additional factors that influence fossil-fuel emissions, such as climate, residence and building characteristics, transit choices, and affluence; and (3) factors that influence biological carbon gains and losses, including the type of predevelopment land cover, post-development urban design and landscaping choices, soil and landscape management practices, and the time since land conversion.

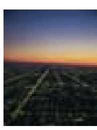
14.3.1 Fossil-fuel Emissions

The density and patterns of development of human settlements (*i.e.*, their "form") are drivers of the magnitude of the fossil-fuel emissions component of the carbon cycle. The size and number of residences and households influence CO_2 emissions from the residential sector, and the spatial distribution of residences, commercial districts, and transportation networks is a key influence in the vehicular and transportation sectors. Many of the attributes of urban form that influence the magnitude of fossil-fuel emissions are linked to historical patterns of economic development, which have differed in Canada, the United States, and Mexico. The future trajectory of development and associated levels of affluence and technological and social change will strongly influence key aspects of urban form such as residence size, vehicle miles traveled, and investment in urban infrastructure, along with associated fossil-fuel emissions. Whereas emissions from the transportation and residential sectors are discussed in detail in Chapters 7 and 9, respectively, this chapter discusses specific aspects of the form of human settlements that affect the current continental carbon balance and its possible future trajectories.

Although there is not yet sufficient data to construct a complete carbon inventory of settlements across North America, it is a feasible research goal to do so in the next several years.

Household size in terms of the number of occupants per household has been declining in North America (Table 14.1) while the average size of new residences has been increasing. For example, the average size of new, single family homes in the United States increased from 139 m^2 (1500 ft^2) to more than 214 m^2 (2300 ft^2) between 1970 and 2004 (NAHB, 2005). These trends have contributed to increases in per capita CO_2 emissions from the residential sector as well as increases in the consumption of land for residential and urban development (Alig *et.al.*, 2003; Ironmonger *et al.*, 1995; Liu *et al.*, 2003; MacKellar *et al.*, 1995). In addition, when considering total emissions from settlements, the trajectory of the transportation and residential sectors may be linked. There have been a number of qualitative discussions of the role of "urban sprawl" in influencing fossil-fuel and pollutant emissions from cities (CEC, 2001; Gonzalez, 2005), although definitions of urban sprawl vary (Ewing *et al.*, 2003). Quantitative linkages between urban form and energy use have been attempted by comparing datasets for a variety of cities, but the results have been difficult to interpret due to the large number of factors that may affect transportation patterns and energy consumption (Anderson *et al.*, 1996). For example, in a seminal analysis of data from a variety of cities, Kenworthy and Newman (1990) found a negative correlation between population density and per capita energy use in the transportation sector. However, their data have been reanalyzed and reinterpreted in a number of subsequent studies that





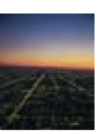
have highlighted other important driving variables, such as income levels, employment density, and transit choice (Gomez-Ibanez, 1991; Gordon and Richardson, 1989; Mindali *et al.*, 2004).

Quantifying the nature and extent of the linkage between development patterns of human settlements and greenhouse gas emissions is critical from the perspective of evaluating the potential impacts of land-use policy. One way forward is to further the application of integrated land-use and transportation models that have been developed to analyze future patterns of urban development in a variety of cities (Agarwal *et al.*, 2000; EPA, 2000; Hunt *et al.*, 2005). Only a handful have been applied to date for generating fossil-fuel emissions scenarios from individual metropolitan areas (Jaccard *et al.*, 1997; Pataki *et al.*, 2006a), such that larger-scale national or continental projections for human settlements are not currently available. However, there is potential to add a carbon cycle component to these models that would assess the linkages between land-use and land-cover change, residential and commercial energy use and emissions, emissions from the transportation sector, and net carbon gains and losses in biological sinks following land conversion. A critical feature of these models is that they may be used to evaluate future scenarios and the potential impacts of policies to influence land-use patterns and transportation networks in individual settlements and developing regions.

Quantifying the nature and extent of the linkage between development patterns of human settlements and greenhouse gas emissions is critical from the perspective of evaluating the potential impacts of land-use policy.

14.3.2 Vegetation and Soils in Human Settlements

Human settlements contain vegetation and soils that are often overlooked in national inventories, as they fall outside common classification schemes. Nevertheless, patterns of development affect the carbon balance of biological systems, both in the replacement of natural ecosystems with rural, residential, or urban land cover and in processes within settlements that affect constructed and managed land cover. In the United States, satellite data and ecosystem modeling for the mid-1990s suggested that urbanization occurred largely on productive agricultural land and therefore caused a net loss of carbon fixed by photosynthesis of 40 Mt C per year † (Imhoff *et al.*, 2004).



Urban trees generally result in net reductions in energy use.

Urban forests and vegetation sequester carbon directly as described under carbon inventories. In addition, urban trees influence the carbon balance of municipalities indirectly through their effects on energy use. Depending on their placement relative to buildings, trees may cause shading and windbreak effects, as well as evaporative cooling due to transpiration (Akbari, 2002; Oke, 1989; Taha, 1997). These effects have been estimated in a variety of studies, mostly involving model calculations that suggest that urban trees generally result in net reductions in energy use (Akbari, 2002; Akbari and Konopacki, 2005; Akbari *et al.*, 1997; Akbari and Taha, 1992; Huang *et al.*, 1987). Taking into account CO_2 emissions resulting from tree maintenance and decomposition of removed trees, "avoided" emissions from energy savings were responsible for approximately half of the total net reduction in CO_2 emissions from seven municipal urban forests, with the remainder attributable to direct sequestration of CO_2 (McPherson *et al.*, 2005). Direct measurements of meteorological fluxes that quantify the contribution of vegetation are needed to validate these estimates.

Like natural ecosystems, soils in human settlements contain carbon, although rates of sequestration are much more uncertain in urban soils than in natural soils. In general, soil carbon is lost following disturbances associated with conversion from natural to urban or suburban land cover (Pouyat *et al.*, 2002). Soil carbon pools may subsequently increase at varying rates, depending on the soil and land cover type, local climate, and management intensity (Golubiewski, 2006; Pouyat *et al.*, 2002; Qian and Follet, 2002). In ecosystems with low rates of carbon sequestration in native soil such as arid and semi-arid ecosystems, conversion to highly managed, settled land cover can result in higher rates of carbon sequestration and storage than pre-settlement due to large inputs of water, fertilizer, and organic matter (Golubiewski, 2006). Pouyat *et al.* (2006) used urban soil organic carbon measurements to estimate the total above- and below-ground carbon storage, including soil carbon, in U.S. urban land cover to be 2,640 Mt (1,890 to 3,300 Mt). This range does not include the uncertainty in classifying urban land cover,

but applies the range of uncertainty in above-ground urban carbon stocks reported in Nowak and Crane (2002) and the standard deviation of urban soil carbon densities reported in Pouyat *et al.* (2006). In addition, irrigated and fertilized urban soils have been associated with higher emissions of CO_2 and the potent greenhouse gas nitrous oxide (N_2O) relative to natural soils, offsetting some potential gains of sequestering carbon in urban soils (Kaye *et al.*, 2004; Kaye *et al.*, 2005; Koerner and Klopatek, 2002). Finally, full carbon accounting that incorporates fossil-fuel emissions associated with soil management (*e.g.*, irrigation and fertilizer production and transport) has not yet been conducted. In general, additional data on soil carbon balance in human settlements are required to assess the potential for managing urban and residential soils for carbon sequestration.

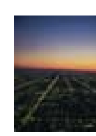
14.4 OPTIONS FOR MANAGEMENT

A number of municipalities in Canada, the United States, and Mexico have committed to voluntary programs of greenhouse gas emissions reductions. Under the Cities for Climate Protection program (CCP) of International Governments for Local Sustainability (ICLEI, formerly the International Council of Local Environmental Initiatives) 269 towns, cities, and counties in North America have committed to conducting emissions inventories, establishing a target for reductions, and monitoring the results of reductions initiatives (the current count of the number of municipalities participating in voluntary greenhouse gas reduction programs may be found on-line at http://www.iclei.org). Emissions reductions targets vary by municipality, as do the scope of reductions, which may apply to the municipality as a whole or only to government operations (*i.e.*, emissions related to operation of government-owned buildings, facilities, and vehicle fleets).

Kousky and Schneider (2003) interviewed representatives from 23 participating CCP municipalities in the United States who indicated that cost savings and other co-benefits of greenhouse gas reductions in cities and towns were the most commonly cited reasons for participating in voluntary greenhouse gas reductions programs. Potential cost savings include reductions in energy and fuel costs from energy efficiency programs in buildings, street lights, and traffic lights; energy cogeneration in landfills and sewage treatment plants; mass transit programs; and replacement of municipal vehicles and buses with alternative fuel or hybrid vehicles (ICLEI, 1993; 2000). Other perceived co-benefits include reductions in emissions of particulate and oxidant pollutants, alleviation of traffic congestion, and availability of lower-income housing in efforts to curb urban sprawl. These co-benefits are often "perceived" because many municipalities have not attempted to quantify them as part of their emissions reductions programs (Kousky and Schneider, 2003); however, it has been suggested that they play a key role in efforts to promote reductions of municipal-scale greenhouse gas emissions because local constituents regard them as an issue of interest (Betsill, 2001).

Of the co-benefits of municipal programs to reduce CO_2 emissions, improvements in air quality are perhaps the most well studied. Cifuentes (2001) analyzed the benefits of reductions in atmospheric particulate matter measuring less than 10 micrometers (μm) in diameter (PM10) and ozone concentrations in four cities in North and South America. Using a greenhouse gas reduction of 13% of 2000 levels by 2020 from energy efficiency and fuel substitution programs, Cifuentes (2001) estimated that PM10 and ozone concentrations would decline by 10% of 2000 levels. Estimated health benefits from such a reduction included avoidance of 64,000 (18,000-116,000) premature deaths associated with air quality-related heath problems as well as avoidance of 91,000 (28,000-153,000) hospital admissions and 787,000 (136,000-1,430,000) emergency room visits. However, using calculations for co-control of CO_2 and air pollutants in Mexico City, West *et al.* (2004) found that in practice, if electrical energy is primarily generated in remote locations relative to the urban area, cost-effective energy efficiency programs may have a relatively small effect on air quality. In that case, options for reducing greenhouse gas emissions would have to be implemented primarily in the transportation sector to appreciably affect air quality.

Two hundred and sixty nine towns, cities, and counties in North America have committed to conducting emissions inventories, establishing a target for reductions, and monitoring the results of reductions initiatives.

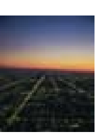


14.5 RESEARCH NEEDS

Additional studies of the carbon balance of settlements of varying densities, geographical location, and patterns of development are needed to quantify the potential impacts of various policy and planning alternatives on net greenhouse gas emissions. While it may seem intuitive that policies to curb urban sprawl or enhance tree planting programs will result in emissions reductions, different aspects of urban form (*e.g.*, housing density, availability of public transportation, type and location of forest cover) may have different net effects on carbon sources and sinks, depending on the location, affluence, economy, and geography of various settlements. It is possible to develop quantitative tools to take many of these factors into account. To facilitate development and application of integrated urban carbon cycle models and to extrapolate local studies to regional, national, and continental scales, useful additional data include:

- common land cover classifications appropriate for characterizing a variety of human settlements across North America,
- emissions inventories at small spatial scales such as individual neighborhoods and municipalities,
- expansion of the national carbon inventory and flux measurement networks to include land cover types within human settlements,
- comparative studies of processes and drivers of development in varying regions and nations, and
- interdisciplinary studies of land-use change that evaluate socioeconomic as well as biophysical drivers of carbon sources and sinks.

In general, there has been a focus in carbon cycle science on measuring carbon stocks and fluxes in natural ecosystems, and consequently highly managed and human-dominated systems such as settlements have been underrepresented in many regional and national inventories. To assess the full carbon balance of settlements ranging from rural developments to large cities, a wide range of measurement techniques and scientific, economic, and social science disciplines are required to understand the dynamics of urban expansion, transportation, economic development, and biological sources and sinks. An advantage to an interdisciplinary focus on the study of human settlements from a carbon cycle perspective is that human activities and biological impacts in and surrounding settled areas encompass many aspects of perturbations to atmospheric CO_2, including a large proportion of national CO_2 emissions and changes in carbon sinks resulting from land-use change.



Coastal Oceans

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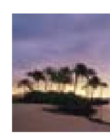
KEY FINDINGS

- The combustion of fossil fuels has increased carbon dioxide in the atmosphere, and the oceans have annually absorbed an equivalent of 20-30% of the carbon dioxide in fossil-fuel emissions. The present annual uptake by the oceans of approximately 1.8 billion tons of carbon (26% of global fossil-fuel emissions in 2003) is well constrained, has slightly acidified the oceans and may ultimately affect ocean ecosystems in unpredictable ways.
- The carbon budgets of ocean margins (coastal regions) are not as well-characterized due to lack of observations coupled with complexity and highly localized geographic variability. Existing data are insufficient, for example, to estimate the amount of carbon derived from human activity stored in the coastal regions of North America or to predict future scenarios.
- New air-sea carbon flux observations reveal that on average, waters within about 100 km (60 miles) of the shores surrounding North America are neither a source nor a sink of carbon dioxide to the atmosphere. A small net source of carbon dioxide to the atmosphere of 19 million tons of carbon per year (with large uncertainty) is estimated mostly from waters around the Gulf of Mexico and the Caribbean Sea. This is equivalent to about 1% of the global ocean uptake.
- With the exception of one or two time-series sites, almost nothing is known about historical trends in air-sea fluxes and the source-sink behavior of North America's coastal oceans.
- The Great Lakes and estuarine systems of North America may be net sources of carbon dioxide where terrestrially-derived organic material is decomposing, while reservoir systems may be storing carbon through sediment transport and burial.
- Options for sequestering carbon in the ocean include iron fertilization in sunlit surface waters and injection of carbon dioxide in subsurface coastal waters. However, sequestration capacity and potential adverse effects on marine environments need to be investigated.
- Highly variable air-sea carbon dioxide fluxes in coastal areas may introduce errors in North American carbon dioxide fluxes calculated by atmospheric inversion methods. Reducing these errors and the uncertainties regarding the variability of carbon cycling in coastal oceans will require observation systems utilizing fixed and mobile platforms, novel instrumentation to measure critical stocks and fluxes, and coordinated national and international research programs. Experimental studies involving coastal carbon cycling should be encouraged.



15.1 INVENTORIES (STOCKS AND FLUXES, QUANTIFICATION)

BOX 15.1: CCSP SAP 2.2 Uncertainty Conventions

*****	= 95% certain that the actual value is within 10% of the estimate reported,
****	= 95% certain that the estimate is within 25%,
***	= 95% certain that the estimate is within 50%,
**	= 95% certain that the estimate is within 100%, and
*	= uncertainty greater than 100%.
†	= The magnitude and/or range of uncertainty for the given numerical value(s) is not provided in the references cited.

Climate-driven changes in ocean circulation, chemical properties or biological rates could result in strong feedbacks to the atmosphere.

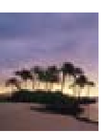
This chapter first introduces the role the oceans play in modulating atmospheric carbon dioxide (CO_2), then quantifies air-sea CO_2 fluxes in coastal waters[1] surrounding North America and considers how the underlying processes affect the air-sea fluxes. Stocks of living organisms in marine environments are small relative to those on land, but turnover rates are very high. In addition, aquatic stocks are not well characterized because of their spatial and temporal variability, the complexity of carbon compound transformations, and limited data on these processes. The oceans act as a huge reservoir for inorganic carbon, containing about 50 times as much CO_2 as the atmosphere. The ocean's biological pump converts CO_2 to organic particulate carbon by photosynthesis, transports the organic carbon from the surface by sinking, and therefore plays a critical role in removing atmospheric CO_2 in combination with physical and chemical processes (Gruber and Sarmiento, 2002; Sarmiento and Gruber, 2006). Atmospheric concentration of CO_2 would be much higher in the absence of current ocean processes implying that climate-driven changes in ocean circulation, chemical properties or biological rates could result in strong feedbacks to the atmosphere.

The uptake of this human-caused CO_2 by the oceans is, on average, turning them more acidic with negative and potentially catastrophic effects on some biota.

The release of CO_2 into the atmosphere by the combustion of fossil fuels has increased pre-industrial concentrations from around 280 ppm to present day levels of nearly 380 ppm in 2005. This increase in atmospheric concentrations is driving CO_2 into the ocean with the present net air-sea CO_2 flux from the atmosphere into the ocean well constrained to about 1800 million metric tons of carbon (Mt C, See Box 15.1)**** per year (or 1.8 billion tons of carbon [Gt C]**** per year) (Figure 15.1 and Table 15.1) (Chapter 2 for a description of how ocean carbon fluxes relate to the global carbon cycle). The uptake of this human-caused CO_2 by the oceans is, on average, turning them more acidic with negative and potentially catastrophic effects on some biota (Kleypas *et al.*, 2006). The atmosphere is well mixed and nearly homogenous so the large spatial variability in air-sea CO_2 fluxes shown in Figure 15.1 is driven by a combination of physical, chemical, and biological processes in the ocean. The flux over the coastal margins has neither been well characterized (Liu *et al.*, 2000) nor integrated into global calculations because there are large variations over small spatial and temporal scales, and observations have been limited. The need for higher spatial

Table 15.1 Climatological mean distribution of the net air-sea CO_2 flux (in Gt C per year) over the global ocean regions (excluding coastal areas) in reference year 1995. The fluxes are based on about 1.75 million partial pressure measurements for CO_2 in surface ocean waters, excluding the measurements made in the equatorial Pacific (10°N- 10°S) during El Niño periods (see Takahashi *et al.*, 2002). The NCAR/NCEP 42-year mean wind speeds and the (wind speed)² dependence for air-sea gas transfer rate are used (Wanninkhof, 1992). Plus signs indicate that the ocean is a source for atmospheric CO_2, and negative signs indicate that ocean is a sink. The ocean uptake has also been estimated on the basis of the following methods: temporal changes in atmospheric oxygen and CO_2 concentrations (Keeling and Garcia, 2002; Bender *et al.*, 2005), $^{13}C/^{12}C$ ratios in sea and air (Battle *et al.*, 2000; Quay *et al.*, 2003), ocean CO_2 inventories (Sabine *et al.*, 2004), and coupled carbon cycle and ocean general circulation models (Sarmiento *et al.*, 2000; Gruber and Sarmiento, 2002). The consensus is that the oceans take up 1.3 to 2.3 Gt C per year.

Latitude bands	Pacific	Atlantic	Indian	Southern Ocean	Global
N of 50°N	+0.01	−0.31			−0.30
14°N-50°N	−0.49	−0.25	+0.05		−0.69
14°N-14°S	+0.65	+0.13	+0.13		+0.91
14°S-50°S	−0.39	−0.21	−0.52		−1.12
S of 50°S				−0.30	−0.30
Total flux	−0.23	−0.64	−0.34	−0.30	−1.50
% of flux	15	42	23	20	100
Area (10⁶ km²)	152.0	74.6	53.0	41.1	320.7
% of area	47	23	17	13	100



[1] "Coastal waters" are the region within 100 km from shore in which processes unique to coastal marine environments influence the partial pressure of CO_2 in surface sea waters.

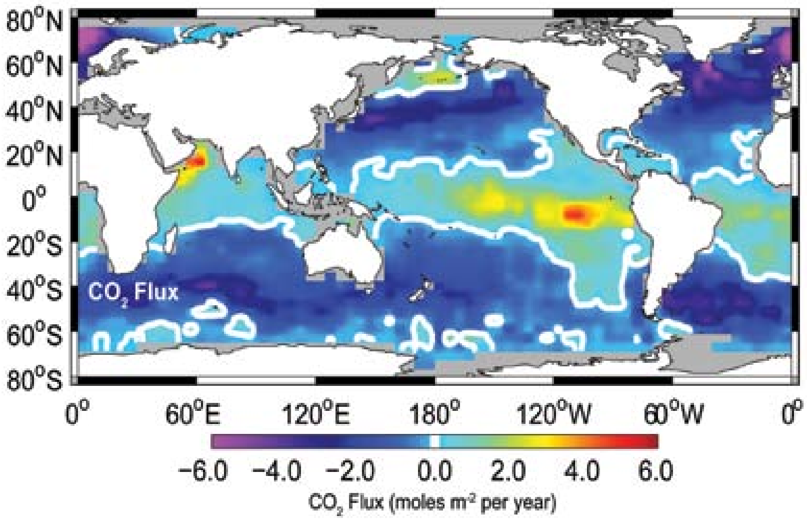


Figure 15.1 Global distribution of sea-air CO_2 flux. The source areas (cyan-green-yellow-orange) are primarily in the tropics with a few high latitiude areas where deep mixing occurs in winter. The sink areas (blue-magenta) are located in mid to high latitiudes. The white line represents zero flux. Updated from Takahashi *et al.* (2002).

resolution to resolve the coastal variability has hampered modeling efforts. In the following sections we review existing information on the coastal ocean carbon cycle and its relationship to the global ocean, and we present the results of a new analysis of about a half million observations of air-sea flux of CO_2 in coastal waters surrounding the North American continent.

15.1.1 Global Coastal Ocean Carbon Fluxes

The carbon cycle in coastal oceans involves a series of processes, including runoff from terrestrial environments, upwelling and mixing of high CO_2 water from below, photosynthesis at the sea surface, sinking of organic particles, respiration, production and consumption of dissolved organic carbon, and air-sea CO_2 fluxes (Figure 15.2). Although fluxes in the coastal oceans are large relative to surface area (Muller-Karger *et al.*, 2005), there is disagreement as to whether these regions are a net sink or a net source of CO_2 to the atmosphere (Tsunogai *et al.*, 1999; Cai and Dai, 2004; Thomas *et al.*, 2004). Great uncertainties remain in coastal carbon fluxes, which are complex and dynamic, varying rapidly over short distances and at high frequencies. Only recently have new technologies allowed for the measurement of these rapidly changing fluxes (Friederich *et al.*, 1995 and 2002; Hales and Takahashi, 2004).

Carbon is transported from land to sea mostly by rivers in four components: CO_2 dissolved in water, organic carbon dissolved in water, particulate inorganic carbon (*e.g.*, calcium carbonate [$CaCO_3$]), and particulate organic carbon. The global rate of river input has been estimated to be 1000 Mt C*** per year, about 38% of it as dissolved CO_2 (or 384 Mt C per year), 25% as dissolved organic matter, 21% as organic particles, and 17% as $CaCO_3$ particles (Gattuso *et al.*, 1998). Estimates for the riverine dissolved CO_2 flux vary from 385 to 429 Mt C per year (Sarmiento and Sundquist, 1992). The Mississippi River, the seventh-largest in freshwater discharge in the world, delivers about 13 Mt C*** per year as dissolved CO_2 (Cai, 2003). Organic matter in continental

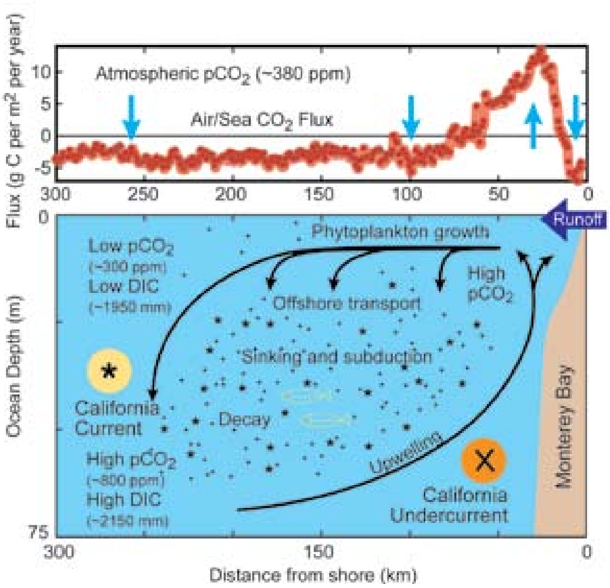


Figure 15.2 Mean air-sea CO_2 flux as calculated from shipboard measurements on a line perpendicular to the central California coast (top panel). Flux within Monterey Bay (~0-20 km offshore) is into the ocean, flux across the active upwelling region (~20-75 km offshore) is from the ocean, and flux in the California Current (75-300 km) is on average into the ocean. These fluxes result from the processes shown in the bottom panel. California Undercurrent water, which has a high CO_2 partial pressure, upwells near shore, and is advected offshore into the California Current and into Monterey Bay. Phytoplankton growing in the upwelled water use CO_2 as a carbon source, and CO_2 is drawn to low levels in those areas. Phytoplankton carbon eventually sinks or is subducted below the euphotic zone, where it decays, elevating the CO_2 levels of subsurface waters. Where the level of surface CO_2 is higher than the level of atmospheric CO_2, diffusion drives CO_2 into the atmosphere. Conversely, where the level of surface CO_2 is lower than that of atmospheric CO_2, diffusion drives CO_2 into the ocean. The net air-sea flux on this spatial scale is near zero. DIC = concentration of inorganic carbon (*i.e.*, all CO_2 species) dissolved in seawater. Updated from Pennington *et al.* (in press).

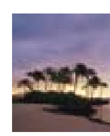


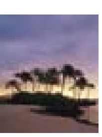
Table 15.2 Variability of CO_2 distributions and fluxes in U.S. coastal waters from regional surveys and moored measurements (from Doney *et al.*, 2004).

Location	Surface seawater pCO_2 (µatm)	Instantaneous CO_2 flux (mol/ per m^2 per year)	Annual average (mol per m^2 per year)	Sampling method	Reference
New Jersey Coast	211–658	−17 to +12	−0.65	Regional survey	Boehme *et al.* (1998)
Cape Hatteras, North Carolina	ND	−1.0 to +1.2	ND	Moored measurements	DeGrandpre *et al.* (1997)
Middle Atlantic Bight, inner shelf	150–620	ND	−0.9	Regional survey	DeGrandpre *et al.* (2002)
Middle Atlantic Bight, middle shelf	220–480	ND	−1.6	Regional survey	DeGrandpre *et al.* (2002)
Middle Atlantic Bight, outer shelf	300–430	ND	−0.7	Regional survey	DeGrandpre *et al.* (2002)
Florida Bay, Florida	325–725	ND	ND	Regional survey	Millero *et al.* (2001)
Southern California Coastal Fronts	130–580	ND	ND	Regional survey	Simpson (1985)
Coastal Calif. (M-1; Monterey Bay)	245–550	−8 to +50	1997–98: −1.0 1998–99: +1.1	Moored measurements	Friederich *et al.* (2002)
Oregon Coast	250–640	ND	ND	Regional survey	van Geen *et al.* (2000)
Bering Sea Shelf in spring (April–June)	130–400	−8 to −12	−8	Regional survey	Codispoti *et al.* (1986)
South Atlantic Bight	300–1200	ND	2.5	Regional survey	Cai *et al.* (2003)
Miss. River Plume (summer)	80–800	ND	ND	Regional survey	Cai *et al.* (2003)
Bering Sea (Aug–Sep.)	192–400	ND	ND	Regional survey	Park *et al.* (1974)

ND indicates that no data are available.

To convert from "mol" to "grams," multiply the numerical "mol" value by 12.

shelf sediments exhibits only weak isotope and chemical signatures of terrestrial origin, suggesting that riverine organic matter is reprocessed in coastal environments on a time scale of 20 to 130 years (Hedges *et al*., 1997; Benner and Opsahl, 2001). Of the organic carbon, about 30% is accumulating in estuaries, marshes, and deltas, and a large portion (20% to 60%) of the remaining 70% is readily and rapidly oxidized in coastal waters (Smith and Hollibaugh, 1993). Only about 10% is estimated to be contributed by human activities, such as agriculture and forest clearing (Gattuso *et al*., 1998), and the rest is a part of the natural carbon cycle.



One of the major differences between coastal and open ocean systems is the activity of the biological pump. In coastal environments, the pump operates much more efficiently, leading to rapid reduction of surface CO_2 and thus complicating the accurate quantification of air-sea CO_2 fluxes. For example, Ducklow and McCallister (2004) constructed a carbon balance for the coastal oceans using the framework of the ocean carbon cycle of Gruber and Sarmiento (2002) and estimated a net CO_2 removal by primary productivity of 1200 Mt C per year and a large CO_2 sink of 900 Mt C per year for the atmosphere. In contrast, Smith and Hollibaugh (1993) estimated a biological pump of about 200 Mt C per year and concluded that the coastal oceans are a weak CO_2 sink of 100 Mt C per year, about one-ninth of the estimate by Ducklow and McCallister (2004). Since the estimated air-sea CO_2 flux depends on quantities that are not well constrained, the mass balance provides widely varying results. For this reason, in this chapter, the net air-sea flux over coastal waters is estimated on the basis of direct measurements of the air-sea difference of partial pressure of CO_2 (pCO_2).

15.1.2 North American Coastal Carbon

Two important types of North American coastal ocean environments can be identified: (1) river-dominated coastal

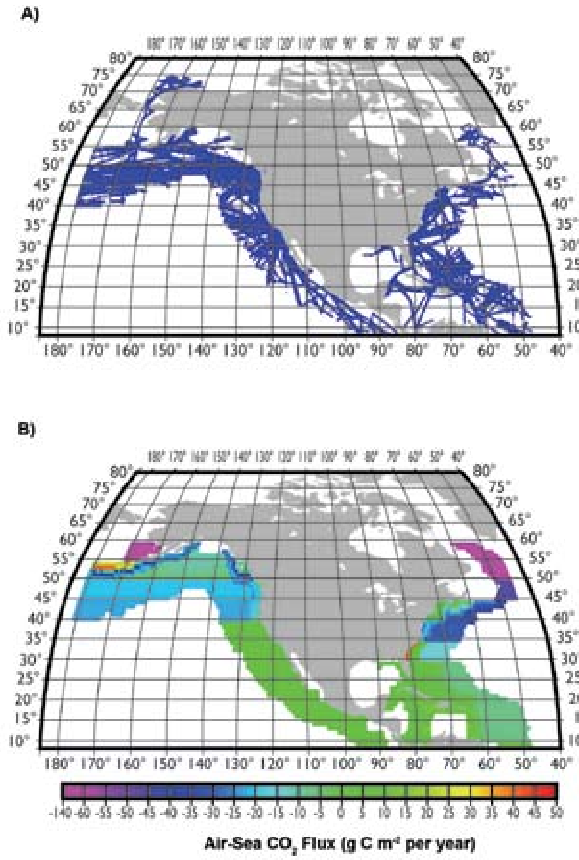


Figure 15.3 (A) Distribution of coastal surface water CO_2 partial pressure measurements made between 1979 and 2004. (B) The distribution of the annual mean air-sea net CO_2 flux over 1° × 1° pixel areas (N-S 100 km, E-W 80 km) around North America. The flux (g C per m2 per year) represents the climatological mean over the 25-year period. The magenta-blue colors indicate that the ocean water is a sink for atmospheric CO_2, and the green-yellow-orange colors indicate that the sea is a CO_2 source. The data were obtained by the authors and collaborators of this chapter and are archived at the Lamont-Doherty Earth Observatory (www.ldeo.columbia.edu/res/pi/CO_2).

margins with large inputs of fresh water, organic matter, and nutrients from land (*e.g.*, Mid- and South-Atlantic Bights) (Cai *et al.*, 2003) and (2) coastal upwelling zones (*e.g.*, the California-Oregon-Washington coasts, along the eastern boundary of the Pacific) where physical processes bring cool, high-nutrient, and high-CO_2 waters to the surface. In both environments, the biological uptake of CO_2 plays an important role in determining whether an area becomes a sink or a source for the atmosphere.

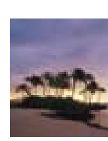
High biological productivity fueled by nutrients added to coastal waters can lead to seawater becoming a CO_2 sink during the summer growing season, as observed in the Bering Sea Shelf (Codispoti and Friederich, 1986) and the northwest waters off Oregon and Washington (van Geen *et al.*, 2000; Hales *et al.*, 2005). Similar CO_2 draw-downs may occur in the coastal waters of the Gulf of Alaska and in the Gulf of Mexico near the Mississippi River outflow. Coastal upwelling results in a very high concentration of CO_2 for the surface water (as high as 1000 μatm), and, hence, the surface water becomes a strong CO_2 source. This is followed by rapid biological uptake of CO_2, which causes the water to become a strong CO_2 sink (Friederich *et al.*, 2002; Hales *et al.*, 2005).

A review of North American coastal carbon fluxes has been carried out by Doney *et al.* (2004) (Table 15.2). The information reviewed was very limited in space (only 13 locations) and time, leading Doney *et al.* to conclude that it was unrealistic to reliably estimate an annual flux for North American coastal waters. Measurement programs have increased recently, and we have used the newly available data to calculate annual North American coastal air-sea fluxes for the first time.

15.1.3 Synthesis of Available North American Air-Sea Coastal CO_2 Fluxes

A large data set consisting of 550,000 measurements of the pCO_2 in surface waters has been assembled and analyzed (Figure 15.3; see Appendix G for details). Partial pressure of CO_2 is measured in a carrier gas equilibrated with seawater and, as such, it is a measure of the outflux/influx tendency of CO_2 from the atmosphere. Carbon dioxide reacts with seawater and 99.5% of the total amount of CO_2 dissolved in seawater is in the form of bicarbonate (HCO_3^-) and carbonate ions ($CO_3^=$), which do not exchange with the overlying atmosphere. Only CO_2 molecules, which constitute about 0.5% of the total dissolved CO_2, exchange with the atmosphere. This is expressed as pCO_2, which is affected by physical and biological processes; pCO_2 increases as seawater warms and decreases when photosynthesis is stimulated. The data were obtained by the authors and collaborators, quality-controlled, and assembled in a uniform electronic format for analysis (available at www.ldeo.columbia.edu/res/pi/CO2). Observations in each 1° × 1° pixel area were compiled into a single year and were analyzed for time-space variability. Seasonal and interannual variations were not well characterized except in a few locations (Friederich *et al.*, 2002). The annual mean air-sea pCO_2 difference (ΔpCO_2) was computed for 5°-wide zones along the North American continent and was plotted as a function of latitude for four regions (Figure 15.4): North Atlantic, Gulf of Mexico/Caribbean, North Pacific, and Bering/Chukchi Seas. Figure 15.4A shows the fluxes in the first nearshore band, and Figure 15.4B shows the fluxes for a band that is several hundred kilometers from shore. The average fluxes for them and for the intermediate bands are given in Table 15.3. The flux and area data are listed in Table

> The open ocean Pacific waters south of 30°N are, on the annual average, a CO_2 source to the atmosphere, whereas the area north of 40°N is a sink.



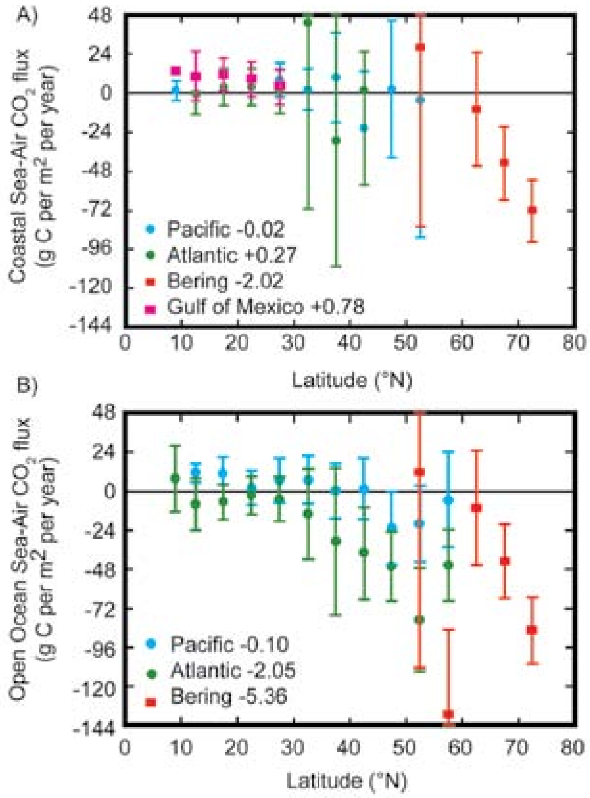
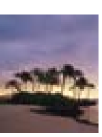


Figure 15.4 Estimated air-sea CO_2 fluxes (g C per per m^2 per year) from 550,000 seawater CO_2 partial pressure (pCO_2) observations made from 1979 to 2004 in ocean waters surrounding the North American continent. (A) Waters within one degree (about 80 km) of the coast and (B) open ocean waters between 300 and 900 km from the shore (see Figure 15.3B). The annual mean air-sea pCO_2 difference (Δ pCO_2) values were calculated from the weekly mean atmospheric CO_2 concentrations in the GLOBALVIEW-CO_2 database (2004) over the same pixel area in the same week and year as the seawater pCO_2 was measured. The monthly net air-sea CO_2 flux was computed from the mean monthly wind speeds in the National Centers for Environmental Prediction/National Center for Atmospheric Research (NCEP/NCAR) database in the (wind speed)^2 formulation for the air-sea gas transfer rate by Wanninkhof (1992). The ± uncertainties represent one standard deviation.

15.4. A full complement of seasonal observations are lacking in the Arctic Sea, including Hudson Bay, the northern Labrador Sea, and the Gulf of St. Lawrence; the northern Bering Sea; the Gulf of Alaska; the Gulf of California; and the Gulf of Mexico and the Caribbean Sea.



In contrast to the Pacific coast, the latitude where Atlantic coastal waters become a CO_2 sink is located further north.

The offshore patterns follow the same general trend found in the global open ocean data set shown in Figure 15.1. On an annual basis the lower latitudes tend to be a source of CO_2 to the atmosphere, whereas the higher latitudes tend to be sinks (Figures 15.3B and 15.4B). The major difference in the coastal waters is that the latitude where CO_2 starts to enter the ocean is further north than it is in the open ocean, particularly in the Atlantic. A more detailed region-by-region description follows.

15.1.4 Pacific Ocean

Observations made in waters along the Pacific coast of North America illustrate how widely coastal waters vary in space and time, in this case driven by upwelling and relaxation (Friederich *et al.*, 2002). Figure 15.5A shows a summertime quasi-synoptic distribution of temperature, salinity, and pCO_2 in surface waters based on measurements made in July through September 2005. The effects of the Columbia River plume emanating from ~46°N are clearly seen (colder temperature, low salinity, and low pCO_2), as are coastal upwelling effects off Cape Mendocino (~40°N) (colder, high salinity, and very high pCO_2). These coastal features are confined to within 300 km from the coast. The 1997-2005 time-series data for surface water pCO_2 observed off Monterey Bay (Figure 15.5B) show the large, rapidly fluctuating air-sea CO_2 fluxes during the summer upwelling season in each year, as well as the low-pCO_2 periods during the 1997-1998 and 2002-2003 El Niño events. In spite of the large seasonal variability, ranging from 200 to 750 µatm, the annual mean air-sea pCO_2 difference and the net CO_2 flux over the waters off Monterey Bay areas (~37°N) are close to zero (Pennington *et al.*, in press). The seasonal amplitude decreases away from the shore and in the open ocean bands, where the air-sea CO_2 flux changes seasonally in response to seawater temperature (out of the ocean in summer and into the ocean in winter).

The open ocean Pacific waters south of 30°N are, on the annual average, a CO_2 source to the atmosphere, whereas the area north of 40°N is a sink, and the zone between 30° and 40°N is neutral (Takahashi *et al.*, 2002). Coastal waters in the 40°N through 45°N zone (northern California-Oregon coasts) are even a stronger CO_2 sink, associated with nutrient input and stratification by fresh water from the Columbia River (Hales *et al.*, 2005). On the other hand, coastal pCO_2 values in the 15°N through 40°N zones have pCO_2 values similar to open ocean values and to the atmosphere. In the zones 15°N through 40°N, the annual mean values for the net air-sea CO_2 flux are nearly zero, consistent with the finding by Pennington *et al.* (in press).

15.1.5 Atlantic Ocean

With the exception of the 5°N-10°N zone, the open ocean areas are an annual net sink for atmospheric CO_2 with stronger sinks at high latitudes, especially north of 35°N (Figure 15.3B). In contrast, the nearshore waters are a CO_2 source between 15°N and 45°N. Accordingly, in contrast to the Pacific coast, the latitude where Atlantic coastal waters become a CO_2 sink is located further north. In the areas north of 45°N, the open ocean waters are a strong CO_2 sink, due primarily to the cold Labrador Sea waters.

Table 15.3 Climatological mean annual air-sea CO_2 flux (g C per m^2 per year) over the oceans surrounding North America. Negative values indicate that the ocean is a CO_2 sink for the atmosphere. N is the number of seawater pCO_2 measurements. The ± uncertainty is given by one standard deviation of measurements used for analysis and represents primarily the seasonal variability.

Ocean regions	Coastal boxes [a]		First offshore [a]		Second offshore [a]		Third offshore [a]		Open ocean [a]	
	Flux	N	Flux	N	Flux	N	Flux	N	Flux	N
North Atlantic	3.2±142	80,417	–1.4±94	65,148	–7.3±57	35,499	–10.4± 76.4	15,771	–26±83	37,667
North Pacific	–0.2±105	164,838	–6.0±81	69,856	–4.3±66	32,045	–5.3±60	16,174	–1.2±56	84,376
G. Mexico Caribbean	9.4±24	75,496	8.4±23	61,180	11.5±17.0	8,410	13±20	1,646		
Bering/ Chukchi	28.0±110	892	–28±128	868	–44±104	3,399	–53±110	1,465	–63±130	1,848

[a] The pCO_2 data are binned into 1° latitude x 1° longitude box areas. The boxes that include shorelines are named "coastal boxes," and the 1° x 1° boxes located on the ocean side of these "coastal boxes" are called "first offshore" boxes. The next two rows of ocean side boxes are called respectively the "second offshore" and the "third offshore" boxes.

In the coastal zone very high pCO_2 values (up to 2600 µatm) are observed occasionally in areas within 10 km offshore of the barrier islands (see small red dots off the coasts of Georgia and the Carolinas in Figure 15.3B). These waters which have salinities around 20 and high total CO_2 concentrations appear to represent outflow of estuarine/marsh waters rich in carbon (Cai *et al*., 2003). The large contribution of fresh water that is rich in organic matter relative to the Pacific contributes to this small coastal Atlantic source. Offshore fluxes are in phase with the seasonal cycle of warming and cooling; fluxes are out of the ocean in summer and fall and are the inverse in winter and spring.

15.1.6 Bering and Chukchi Seas

Although measurements in these high-latitude waters are limited, the relevant data for the Bering Sea (south of 65°N) and Chukchi Sea (north of 65°N) are plotted as a function of the latitude in Figure 15.4. The values for the areas north of 55°N are for the summer months only; CO_2 observations are not available during winter seasons. Although data scatter widely, the coastal and open ocean waters are a strong CO_2 sink during the summer months due to photosynthetic drawdown of CO_2. The data in the 70°-75°N zone are from the shallow shelf areas in the Chukchi Sea. These waters are a very strong CO_2 sink (air-sea pCO_2 differences ranging from –80 to -180 µatm) with little changes between the coastal and open ocean areas. The air-sea CO_2 flux during winter months is not known but the summer fluxes are shown in Figure 15.4 for comparison. Bates (2006) estimated a mean-annual air-to-sea CO_2 flux[2] of 39 Mt C[***] per year over the Chukchi shelf using data from spring and summer of 2002 that suggested that remnant winter waters were as strong a CO_2 sink as summer waters (with air-sea pCO_2 differences of -60 to -160 µatm).

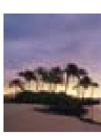
15.1.7 Gulf of Mexico and Caribbean Sea

Although observations are limited, available data suggest that these waters are a strong CO_2 source (Figure 15.4 and Table 15.3). A subsurface anoxic zone has been formed in the Texas-Louisiana coast as a result of the increased addition of anthropogenic nutrients and organic carbon by the Mississippi River (*e.g.*, Lohrenz *et al*., 1999). The carbon-nutrient cycle in the northern Gulf of Mexico is also being investigated (*e.g.*, Cai, 2003), and the studies suggest that at times those waters are locally a strong CO_2 sink due to high biological production.

15.2 SYNTHESIS

An analysis of half a million measurements of air-sea flux of CO_2 shows that the nearshore (< 100 km) coastal waters surrounding North America are a net CO_2 source for the atmosphere on an annual average of about 19±22 Mt C per year[3] (Table 15.4). Most of the flux (14±9 Mt C per year)[3] occurs in the Gulf of Mexico

An analysis of half a million measurements shows that the nearshore (< 100 km) coastal waters surrounding North America are a weak net CO_2 source for the atmosphere, the open oceans are a net CO_2 sink on an annual average.



[2] The flux was estimated on the basis of measurements made only during the spring and summer months of 2002 at several stations located in a limited area of the Chukchi Sea. The uncertainty of ± 7 Mt C given in the original paper represents one standard deviation of measured pCO_2, but does not include uncertainties in the sea-air gas transfer coefficient estimated on the basis of wind speeds and those from limited time-space coverage.

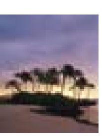
[3] The specified uncertainty is ± one standard deviation around the mean.

Table 15.4 Areas (km²) and mean annual air-sea CO_2 flux (Mt C per year) over four ocean regions surrounding North America. Since the observations in the areas north of 60°N in the Chukchi Sea were made only during the summer months, the fluxes from that area are not included. The ± uncertainty is given by one standard deviation of measurements used for analysis and represents primarily the seasonal variability.

Ocean areas (km²)					Mean air-sea CO_2 flux (10^{12} grams or Mt C per year)				
Coastal boxes	First offshore	Second offshore	Third offshore	Open ocean	Coast box	First offshore	Second offshore	Third offshore	Open ocean
North Atlantic coast (8° N to 45°N)									
625,577	651,906	581,652	572,969	3,388,500	2.7±9.5	-0.5±9.3	-4.0±4.9	-6.5±6.3	-41.5±28.1
North Pacific coast (8°N to 55°N)									
1,211,555	855,626	874,766	646,396	7,007,817	2.1±17.1	-7.0±14.1	-4.8±12.5	-3.7±5.3	-53.8±60.7
Gulf of Mexico and Caribbean Sea (8°N to 30°N)									
1,519,335	1,247,413	935,947	1,008,633		13.6±8.9	10.9±7.5	6.8±5.00	6.6±5.0	
Bering and Chukchi Seas (50°N to 70°N)									
481,872	311,243	261,974	117,704	227,609	0.8±3.1	-6.2±9.5	-5.3±7.5	-3.7±3.0	-9.8±3.7
Total ocean areas surrounding North America									
3,838,339	3,066,188	2,654,339	2,300,702	10,623,926	19.1±21.8	-2.8±20.7	-7.4±16.2	-7.3±10.1	-105.2±67.0

and Caribbean Sea. The open oceans are a net CO_2 sink on an annual average (Table 15.4; Takahashi *et al.*, 2002). The reported uncertainties reflect the time-space variability but do not reflect uncertainties due to lack of observations in some portions of the Arctic Sea, Bering Sea, Gulf of Alaska, Gulf of Mexico, or Caribbean Sea. Observations in these areas will be needed to improve estimates. If the estimate of 39 Mt C[***] per year sink for the Chukchi Sea (Bates, 2006) is included, the North American coastal waters might be a small CO_2 sink. These results are consistent with recent global estimates that suggest that nearshore areas receiving terrestrial organic carbon input are sources of CO_2 to the atmosphere and that marginal seas are sinks (Borges, 2005; Borges *et al.*, in press). Hence, the net contribution from North American ocean margins is small and difficult to distinguish from zero. It is not clear how much of the open ocean sink results from photosynthesis driven by nutrients of coastal origin.

15.3 TRENDS AND DRIVERS



The sea-to-air CO_2 flux from the coastal zone is small (about 1%) compared with the global ocean uptake flux, which is about 1800 Mt C per year (or 1.8 Gt C per year), and hence does not influence the global air-sea CO_2 budget. However, coastal waters undergo large variations in air-sea CO_2 flux on daily to seasonal time scales and on small spatial scales (Figure 15.5). Fluxes can change on the order of 250 g C per m² per year or 0.7 g C per m² per day on a day to day basis (Figure 15.5). These large fluctuations can significantly modulate atmospheric CO_2 concentrations over the adjacent continent and need to be considered when using the distribution of CO_2 in calculations of continental fluxes.

Freshwater bodies have not been treated in this analysis except to note the large surface pCO_2 resulting from estuaries along the east coast. The Great Lakes and rivers also represent net sources of CO_2 as, in the same manner as the estuaries, organic material from the terrestrial environment is oxidized so that respiration exceeds photosynthesis. Interestingly, the effect of fresh water is opposite along the coast of the Pacific northwest, where increased stratification and iron inputs enhance photosynthetic activity (Ware and Thomson, 2005), resulting in a large sink for atmospheric CO_2 (Figure 15.3). A similar process may be at work at the mouth of the Amazon (Körtzinger, 2003). This emphasizes once again the important role of biological processes in controlling the air-sea fluxes of CO_2.

The air-sea fluxes and the underlying carbon cycle processes that determine them (Figure 15.2) vary seasonally, interannually, and on longer time scales. The eastern Pacific, including the United States' west coast, is subject to changes associated with large-scale climate oscillations such as El Niño (Chavez *et al.*, 1999; Feely *et al.*, 2002; Feely *et al.*, 2006) and the Pacific Decadal Oscillation (PDO) (Chavez *et al.*, 2003; Hare and Mantua, 2000; Takahashi *et al.*, 2003). These climate patterns, and others, like the North Atlantic Oscillation (NAO), alter the oceanic CO_2 sink/source conditions directly through seawater temperature changes as well as ecosystem variations that occur via complex physical-

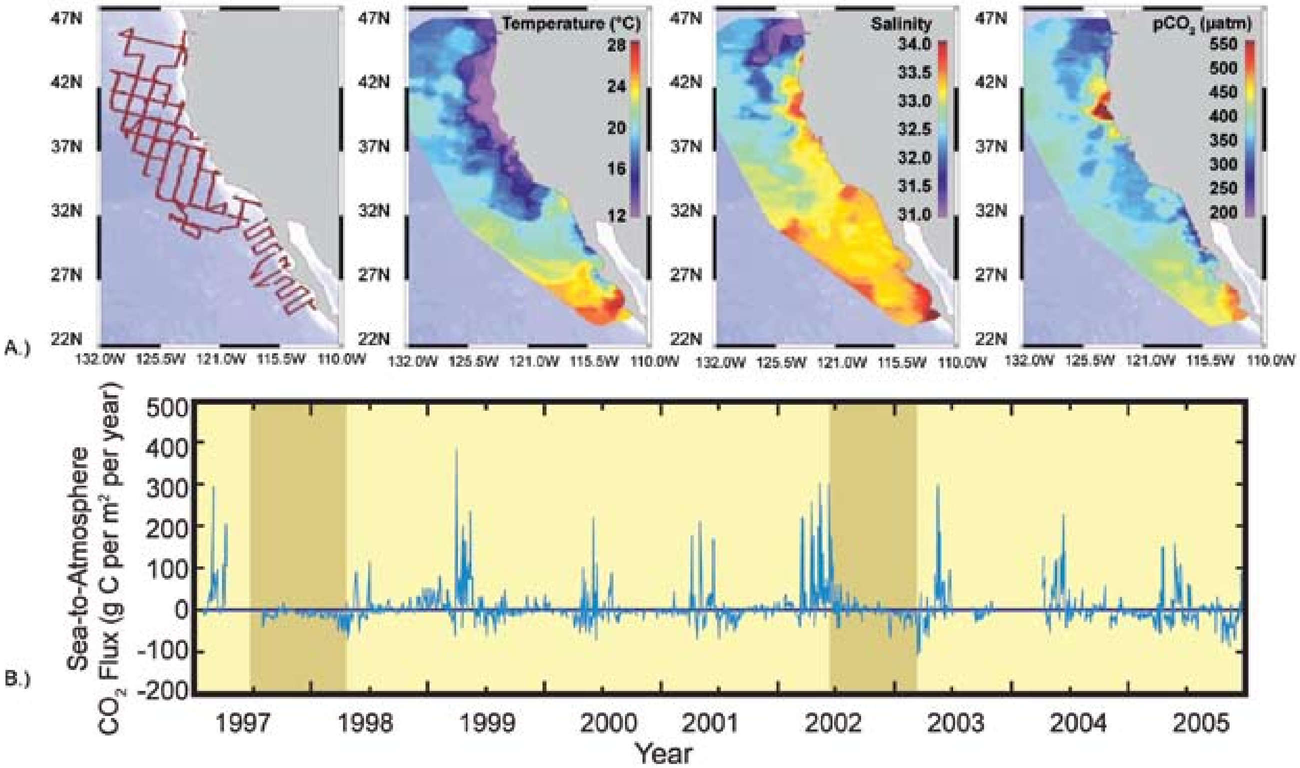


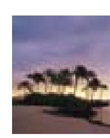
Figure 15.5 Time-space variability of coastal waters off the west coast of North America. (A) Quasi-synoptic distribution of the temperature, salinity, and pCO2 in surface waters during July-September 2005. The Columbia River plume (~46°N) and the upwelling of deep waters off Cape Mendocino (~40°N) are clearly seen. (B) 1997-2005 time-series data for air-sea CO_2 flux from a mooring off Monterey Bay, California. Seawater is a CO_2 source for the atmosphere during the summer upwelling events, but biological uptake reduces levels very rapidly. These rapid fluctuations can affect atmospheric CO_2 levels. For example, if CO_2 from the sea is mixed into a static column, a 500 m thick planetary boundary layer over the course of one day, atmospheric CO_2 concentration would change by 2.5 µatm. If the column of air is mixed vertically through the troposphere to 500 mbar, a change of about 0.5 µatm would occur. The effects would be diluted as the column of air mixes laterally. However, this demonstrates that the large fluctuations of air-sea CO_2 flux observed over coastal waters could affect the concentration of CO_2 significantly enough to affect estimates of air-land flux based on the inversion of atmospheric CO_2 data. Air-sea CO_2 flux was low during the 1997-1998 and 2002-2003 El Niño periods. The shaded areas indicate the 1997-1998 and 2002-2003 El Niño episodes. The greatest El Niño anomalies occur in the winter which is the period of lowest air-sea fluxes.

biological interactions (Hare and Mantua, 2000; Chavez *et al.*, 2003; Patra *et al.*, 2005). For example, during El Niño, upwelling of high CO_2 waters is dramatically reduced along central California (Figure 15.5) so that flux out of the ocean is reduced. At the same time, photosynthetic uptake of CO_2 is also reduced (Chavez *et al.* 2002), reducing ocean uptake. The net effect of climate variability on air-sea fluxes therefore remains uncertain and depends on the time-space integral of the processes.

15.4 OPTIONS FOR MANAGEMENT

Two options for carbon sequestration have been proposed: (1) injection of CO_2 in deep subsurface waters (Brewer, 2003) and (2) ocean iron fertilization (Martin, 1990). The first might be applicable in waters surrounding North America, although potential biological side effects are unresolved. The largest potential for iron fertilization resides in the high nutrient waters of the equatorial Pacific, subarctic Pacific, and Southern Ocean. Offshore waters of coastal upwelling systems have also been considered to be iron limited. However, efficiency and capacity of sequestration remain unresolved (Bakker *et al.*, 2001; Boyd *et al.*, 2000; Coale *et al.*, 2004; Gervais *et al.*, 2002) as do environmental perturbations that could be induced by fertilization (Chisholm *et al.*, 2001).

15.5 RESEARCH AND DEVELOPMENT NEEDS *VIS-À-VIS* OPTIONS



Waters with highly variable air-sea CO_2 fluxes are located primarily within 100 km of the coast (Figure 15.5). With the exception of a few areas, the available observations are grossly inadequate to resolve the high-frequency, small-spatial-scale variations. These high intensity air-sea CO_2 flux events may introduce errors in continental CO_2 fluxes calculated by atmospheric inversion methods. Achieving a comprehensive understanding of the carbon cycle in waters surrounding the North American continent will

require development of advanced technologies and sustained and inter-disciplinary research efforts. Both of these seem to be on the horizon with (1) the advent of ocean observatories that include novel fixed and mobile platforms together with developing instrumentation to measure critical stocks and fluxes and (2) national and international research programs that include the Integrated Ocean Observing System (IOOS) and Ocean Carbon and Climate Change (OC^3). A more comprehensive understanding will require the development of a robust observing program that incorporates time series observations of air-sea and sinking-particulate carbon fluxes in the coastal and open ocean. Our present estimates suggest that the carbon that reaches the bottom over continental margins may be responsible for upwards of 40%*** of the carbon reaching the ocean seafloor (Muller-Karger *et al*., 2005). Given the importance of aquatic systems to atmospheric CO_2 concentrations, these developing efforts must be strongly encouraged. Ocean carbon sequestration studies should also be continued.