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The First State of the Carbon Cycle Report (*SOCCR*): The North American Carbon Budget and Implications for the Global Carbon Cycle

U.S. Climate Change Science Program

Synthesis and Assessment Product 2.2

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Report by the U.S. Climate Change Science Program and the Subcommittee on Global Change Research

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[Note: The organization of this publication is subject to change]

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PREFACE

A primary objective of the U.S. Climate Change Science Program (CCSP) is to provide the best possible scientific information to support public discussion, as well as government and private sector decision-making, on key climate-related issues. To help meet this objective, the CCSP has identified an initial set of 21 synthesis and assessment products that address its highest priority research, observation, and decision-support needs.

This CCSP Report, which is one of the 21 products, provides a synthesis and integration of the current knowledge of the North American carbon budget and its context within the global carbon cycle. In a format useful to decision makers, it (1) summarizes our knowledge of carbon cycle properties and changes relevant to the contributions of and impacts[1] upon the United States and the rest of the world, and (2) provide scientific information for U.S. decision support focused on key issues for carbon management and policy. Consequently, this Report promises to be of significant value to decision-makers, and to the expert scientific and stakeholder communities. For example, we expect this Report to be a major contributor to the Intergovernmental Panel on Climate Change (IPCC) Fourth Assessment Report (due to be published in 2007).

This Report—Synthesis and Assessment Product (SAP) 2.2—addresses carbon emissions; natural reservoirs and sequestration; rates of transfer; the consequences of changes in carbon cycling on land and the ocean; effects of purposeful carbon management; effects of agriculture, forestry, and natural resource management on the carbon cycle; and the socio-economic drivers and consequences of changes in the carbon cycle. It covers North America's land, atmosphere, inland waters, and adjacent oceans, where "North America" is defined as Canada, the United States of America, and Mexico. The Report includes an analysis of North America's carbon budget that documents the state of knowledge and quantifies the best estimates (i.e., consensus, accepted, official) and uncertainties. This analysis provides a baseline against which future results from the North American Carbon Program (NACP) can be compared. SAP 2.2 will be coordinated with other CCSP synthesis and assessment products as appropriate, especially SAP 2.1 (*Scenarios of Greenhouse Gas Emissions and Atmospheric Concentrations and Review of Integrated Scenario Development and Application*) and SAP 3.1 (*Climate Models: An Assessment of Strengths and Limitations for User Applications*).

[1]The term "impacts" as used in this Report refers to specific effects of changes in the carbon cycle, such as acidification of the ocean, the effect of increased CO_2 on plant growth and survival, and changes in concentrations of carbon in the atmosphere. The term is not used as a shortened version of "climate impacts," as was adopted for the *Strategic Plan for the U.S.Climate Change Science Program*.

The focus of this Report follows the Prospectus developed by the Climate Change Science Program and posted on its website at www.climatescience.gov. More specifically, SAP 2.2 attempts to:

- Quantify current information on sources and sinks and associated uncertainties related to the buildup of carbon dioxide (CO_2) and methane (CH_4) in the atmosphere. For example, it provides the best available estimates of the contribution of carbon dioxide emissions from combustion of fossil fuels in North America to changes in global atmospheric concentrations of carbon dioxide for recent decades. Discussion of future changes in fossil fuel emissions are limited to existing scenarios because scenarios are the central element of the work being done under SAP 2.1.
- Discuss and assess current accepted projections of the future of the North American carbon budget, including uncertainties in projected fossil fuel emissions and the impact of policy and technology scenarios on those emissions.
- Provide current estimates, with the associated uncertainties, of the fractions of global and North American fossil-fuel carbon emissions being taken up by North America's ecosystems and adjacent oceans.
- Provide current, best available answers to specific questions about the North American carbon budget relevant to carbon management policy options. The key questions were identified through early and continuing dialogue with SAP 2.2 stakeholders. The answers include explicit characterization of uncertainties.
- Identify where NACP-supported research will reduce current uncertainties in the North American carbon budget and where future enhancements of NACP research can best be applied to further reduce critical uncertainties.
- Describe and characterize the carbon cycle as an integrated interactive system, using innovative graphics to depict the carbon cycle in ways that are easily understandable.

The audience for SAP 2.2 includes scientists, decisionmakers in the public sector (Federal, State, and local governments), the private sector (carbon-related industry, including energy, transportation, agriculture, and forestry sectors; and climate policy and carbon management interest groups), the international community, and the general public. This broad audience is indicative of the diversity of stakeholder groups interested in knowledge of carbon cycling in North America and of how such knowledge might be used to influence or make decisions. Not all the scientific information needs of this broad audience can be met in this first synthesis and assessment product, but the scientific information provided herein is designed to be understandable by all. The primary users of SAP 2.2 are likely to be officials involved in formulating climate policy, individuals responsible for managing carbon in the environment, and scientists involved in assessing the state of knowledge concerning carbon cycling and the carbon budget of North America.

It is envisioned that SAP 2.2 will be used (1) as a state-of-the-art assessment of our knowledge of carbon cycle properties and changes relevant to the contributions of and carbon-specific impacts upon the United States in the context of the rest of the world; (2) as a contribution to relevant national and international assessments; (3) to provide the scientific basis for decision support that will guide management and policy decisions that affect carbon fluxes, emissions, and sequestration; (4) as a means of informing policymakers and the public concerning the general state of our knowledge of the global carbon cycle with respect to the contributions of and impacts on the United States; and (5) as a statement of the carbon cycle science information needs of important stakeholder groups. For example, well-quantified regional and continental-scale carbon source and sink estimates, error terms, and associated uncertainties will be available for use in U.S. climate policy formulation and by resource managers interested in quantifying carbon emissions reductions or carbon uptake and storage. This Report is also intended for senior managers and members of the general public who desire to improve their overall understanding of the U.S. role in Earth's carbon budget and to gain perspective on what is and is not known.

The questions addressed by this Report include:

- What is the carbon cycle and why should we care?
- How do North American carbon sources and sinks relate to the global carbon cycle?
- What are the primary carbon sources and sinks in North America, and how are they changing and why?
- What are the direct, non-climatic effects of increasing atmospheric carbon dioxide or other changes in the carbon cycle on the land and oceans of North America?
- What are the options and measures implemented in North America that could significantly affect the North American and global carbon cycles (e.g., North American sinks and global atmospheric concentrations of carbon dioxide)?
- How can we improve the application of scientific information to decision support for carbon management and climate decision making?

These questions provide the basis for the five chapters in Part I of this Synthesis and Assessment Report. Part II of the Report focuses on the human-system components of the North American carbon cycle, and discusses the carbon "sources and sinks" aspects of (a) energy extraction and conversion, (b) the transportation sector, (c) industry and waste management, and (d) the buildings sector. Part III provides information about land and water systems, including human settlements, and their roles in the carbon cycle.

[NOTE TO REVIEWERS: The following items will also be included in the PREFACE, but have not yet been developed.]

- Structure and organization of this report; How to read this report
- Definition of basic terms, acronyms, units, etc.
- Treatment of carbon vs CO_2 vs CO_2 equivalents
- Treatment of CH_4
- Treatment of greenhouse gases
- Conventions for sources and sinks (i.e., positive and negative numbers)

U.S. Climate Change Science Program
Synthesis and Assessment Product 2.2
The First State of the Carbon Cycle Report (SOCCR):
North American Carbon Budget
and Implications for the Global Carbon Cycle

Executive Summary

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The Earth's carbon budget is in imbalance. Beginning with the Industrial Revolution in the 18th century, but most dramatically since World War II, the human use of coal, petroleum, and natural gas has released large amounts of carbon from geological deposits to the atmosphere, primarily as the combustion product carbon dioxide (CO_2). Clearing of forests and plowing of grasslands for agriculture has also released carbon from plants and soils to the atmosphere as CO_2. The combined rate of release is far larger than can be balanced by the biological and geological processes that naturally remove CO_2 from the atmosphere and store it in various terrestrial and marine reservoirs as part of the earth's carbon cycle. Although the oceans have taken up a large fraction of the CO_2 released through human activity, much of it has "piled up" in the atmosphere, as demonstrated by the dramatic increase in the atmospheric concentration of CO_2. The concentration has increased by 31% since 1750, and the present concentration is now higher than at any time in the past 420,000 years and perhaps the past 20 million years. Because CO_2 is an important greenhouse gas, this imbalance and buildup in the atmosphere has consequences for climate and climate change.

North America is a major contributor to this imbalance. Among all countries, the United States, Canada, and Mexico ranked, respectively, as the first, eighth, and eleventh largest emitters of CO_2 from fossil fuels in 2002. Combined, these three countries contributed more than a quarter (27%) of the world's entire fossil fuel emissions in 2002 and almost one third (32%) of the cumulative global fossil fuel emissions between 1751 and 2002. Emissions from parts of Asia are increasing at a growing rate and may surpass those of North America in the near future, but North America is incontrovertibly a major source of atmospheric CO_2, historically, at present, and in the immediate future.

North America may also be an important sink for carbon. Many lines of scientific evidence point to the vegetation and soils of the Northern Hemisphere as removing CO_2 from the atmosphere and to some degree mitigating fossil-fuel sources. The contribution of North America to that sink is highly uncertain, however. The mechanisms responsible for the sink are reasonably well known and include forest regrowth and sequestration of carbon in agricultural soils; but the relative contributions, magnitudes, and future fates of these mechanisms are highly uncertain.

Understanding the North American carbon budget, both sources and sinks, is critical to the U.S. Climate Change Science Program goal of providing the best possible scientific information to support public discussion, as well as government and private sector decision making, on key climate-related issues. In response, this Report provides a synthesis, integration and assessment of the current knowledge of the North American carbon budget and its context within the global carbon cycle. The Report is organized as a response to questions relevant to carbon management and to a broad range of stakeholders charged with understanding and managing energy and land use. The questions were identified through early and continuing dialogue with these stakeholders, including scientists, decision makers in the public and private sectors (e.g., Federal government, carbon-related industry, including energy, transportation, agriculture, and forestry sectors; and climate policy and carbon management interest groups).

The questions and the answers provided by this Report are summarized below. The reader is referred to the indicated chapters for further, more detailed, discussion. Unless otherwise referenced, all values, statements of findings and conclusions are taken from the chapters of this Report where the attribution and citation of the primary sources can be found.

What is the carbon cycle and why should we care? (Chapter 1)

The carbon cycle is the combination of many different physical, chemical and biological processes that transfer carbon between the major reservoirs: the atmosphere, plants, soils, freshwater systems, oceans, and geological sediments. Hundreds of millions of years ago, and over millions of years, this carbon cycle was responsible for the formation of coal, petroleum, and natural gas, the fossil fuels that are the primary sources of energy for our modern societies. Today, the cycling of carbon among atmosphere, land, freshwater and marine reservoirs is in a rapid transition—an imbalance. Over tens of years, the combustion of fossil fuels is releasing into the atmosphere quantities of carbon that were accumulated in the earth system over millions of years. Furthermore, forests that once held large quantities of carbon are being converted to agricultural lands, releasing additional carbon to the atmosphere as a result. It is not surprising, then, that the concentration of carbon dioxide (CO_2) is increasing in the atmosphere. Furthermore, these trends in fossil fuel use and deforestation are accelerating. The magnitude of the changes raises concerns about the future behavior of the carbon cycle. Will the carbon cycle continue to function as it has in recent history, or will a CO_2-caused warming cause further emissions of CO_2 and further warming?

The question is complicated because carbon dioxide is not the only substance in the atmosphere that affects the earth's surface temperature and climate. Other greenhouse gases include methane (CH_4), nitrous oxide, the halocarbons, and ozone, and all of these gases, together with aerosols, solar radiation, and properties of the earth's surface, are involved in the evolution of climate change. Carbon dioxide, alone, is responsible for 55-60% of the greenhouse forcing from gases, and methane, for another 20% (values are for the late 1990s with a relative uncertainty of 10%; IPCC, 2001). These two gases are the primary gases of the carbon cycle, with CO_2 being particularly important. Furthermore, the consequences of increasing atmospheric carbon dioxide extend beyond climate change alone. The accumulation of carbon in the oceans as a result of more than a century of fossil fuel use and deforestation has increased the acidity of the surface waters, with serious consequences for corals and other marine organisms that build their skeletons and shells from calcium carbonate.

Inevitably, any options or actions to prevent, minimize, or forestall future climate change, or to avoid damage to marine ecosystems from ocean acidification, will require management of the carbon cycle so as to influence or control concentrations of carbon dioxide in the atmosphere. That management involves both reducing sources of carbon dioxide to the atmosphere and enhancing sinks for carbon on land or in the oceans. Strategies may involve both short- and long-term solutions, where short-term solutions may help to gain time while longer-term solutions are developed. In any case, formulation of options by decision makers and successful management of the earth's carbon budget will require solid scientific understanding of the carbon cycle.

Understanding the current carbon cycle may not be enough, however. The concept of managing the carbon cycle carries with it the assumption that the carbon cycle will continue to operate as it has in recent centuries. A major concern is that the carbon cycle, itself, is vulnerable to change, and that the change could bring about additional sources of carbon to the atmosphere from either land or the oceans. Over recent decades both terrestrial ecosystems and the oceans have been natural sinks for carbon. If either, or both, of those sinks were to become sources, management of the carbon cycle could become unachievable. Thus, understanding the current global carbon cycle is necessary for managing carbon, but may not be sufficient. The scientific understanding must include confidence in projections of the future behavior of the carbon cycle in response to human activity and to climate and other environmental change.

But just as importantly, effective management of the carbon cycle requires much more than basic understanding of the current or future carbon cycle. It also requires cost-effective, feasible, and politically-palatable options for carbon management. Caring about the carbon cycle, and responding to those concerns, also involves investigation, understanding and evaluation of those options.

How do North American carbon sources and sinks relate to the global carbon cycle? (Chapter 2)

In recent years North America has been responsible for approximately 30% of the carbon dioxide emissions produced globally by fossil fuel combustion (Table ES-1). North America has also contributed approximately 30% to one-third of cumulative carbon dioxide emissions form fossil-fuel combustion (and cement manufacturing) since 1750. In 2002 the United States accounted for 85% of the North American total and approximately one quarter of the global total, ranking first among all nations in carbon dioxide emissions. These emission estimates are accurate, with 95% confidence, to within 5–10% or less, with variation among countries and data sources. Interestingly, despite accounting for 30% of global emissions, North America accounted for only 10% of the global extraction of fossil fuels; that is, North America imported more than 50% of fossil fuels it used.

Table ES-1. North American contribution to the global carbon budget of approximately the 1990s.

The carbon budget of North America is dominated by fossil fuel emissions; however, the vegetation and soils of North America are, in recent years, a net sink, and the surrounding coastal oceans are a small source. The terrestrial sink in the late 1990s and early 21st century of ~600 Mt C yr^{-1}, with a 95% confidence limit of ±300 Mt C yr^{-1}, offsets approximately 30% (15–45%) of the North American fossil fuel emissions. Most of that sink is in relatively young forests in the United States and Canada, growing on lands that were once farmed. The *global* terrestrial sink is quite uncertain, averaging somewhere in the range of 0 to 3800 Mt C yr^{-1} during the 1980s (the land might have even been a small source of 300 Mt C yr^{-1} during that period) (IPCC, 2001) and in the range of 1000 to 3600 Mt C yr^{-1} in the 1990s (IPCC, 2000). As this global sink is predominantly in northern lands [the sink north of 30° N alone is estimated to be 600 to 2300 Mt C yr^{-1} for the 1980s; IPCC (2001)], the sink of ~600 Mt C yr^{-1} in North America is consistent with the fraction of northern land area in North America (37%), as opposed to Eurasia (63%). The uncertainties are very large, but the North American carbon sink reasonably represents approximately 30% of the global terrestrial sink as a most likely estimate (Table ES-1).

It is clear that the global carbon cycle of the 21st century will continue to be dominated by large fossil fuel emissions from North America. The future trajectory of carbon sinks in North America, and their contribution to the global terrestrial sink is less certain, in part because the role of regrowing forests is likely to decline as the forests mature, and in part because the response of forests and other ecosystems to future climate change and increases in atmospheric CO_2 concentrations is uncertain. The variation among model projections and scenarios of where and how future climate will change contribute to that uncertainty. Additionally, response to a particular future change will likely vary among ecosystems and the response will depend on a variety of incompletely understood environmental factors.

Because North America's carbon budget is such a substantial part of the global carbon budget, management of the North American carbon budget will have important global consequences. North America has many opportunities for decreasing emissions, including changes to the energy system and increasing energy efficiency, as well as for increasing sinks, such as investments in forest planting and agricultural soil management, biomass energy, and geological sequestration. Implementation of policies to deploy these technologies and practices is best achieved by national governments through international cooperation. National programs provide maximum coverage of CO_2 emissions and carbon sinks. They also allow better allocation of resources for technology research and development.

What are the primary carbon sources and sinks in North America, and how are they changing and why? (Chapter 3, Part II Overview, Chapters 6–9, Part III Overview, and Chapters 10–15)

The Sources

The primary source of carbon in North America is the release of CO_2 during the combustion of fossil fuels (Figure ES-1). Fossil fuel carbon emissions in the United States, Canada and Mexico totaled 1856 Mt C in 2003 (with 95% confidence that the actual value lies within 10% of that estimate) and have increased at an average rate of approximately 1% per year for the last 30 years. The United States was responsible for approximately 85% of North America's fossil fuel emissions in 2003, Canada for 9% and Mexico 6%. The 1% growth in U.S. emissions masks faster than 1% growth in some sectors (e.g., transportation) and slower growth in others (e.g., increased manufacturing energy efficiency).

Figure ES-1. North American carbon sources and sinks (Mt C yr^{-1}) circa 2003. Height of a bar indicates a best estimate for net carbon exchange between the atmosphere and the indicated element of the North American carbon budget. Error bars indicate the uncertainty in that estimate, and define the range of values that include the actual value with 95% certainty. See Chapter 3 and Chapters 6-15 of this report for details and discussion of these sources and sinks.

Despite an average growth rate of ~1.0% per year in U.S. emissions, per capita emissions have been roughly constant for the past 30 years, and carbon intensity (carbon emitted/dollar of GDP) has decreased at a rate of ~2% per year. The decrease is in large part caused by the comparatively rapid growth of the service sector (3.6% per year), which now dominates the economy (roughly three-fourths of GDP) and has a carbon intensity only 15% that of manufacturing. Increasing emissions and declining carbon intensity imply that emissions growth is to a large extent decoupled from economic growth. Also, because the service sector is likely to continue to grow more rapidly than other sectors of the economy, carbon intensity may continue to decline.

Electricity generation is the single largest contributor to the North American fossil-fuel source, accounting for approximately 40% of North American fossil emissions. Again, U.S. emissions dominate. In 2003, electricity generation in the United States alone accounted for 35% of total North American fossil fuel emissions.

The transportation sector of North America accounted for 31% of total North American emissions in 2003, most (87%) of it from the United States. The growth in transportation and associated CO_2 emissions has been steady during the past forty years and has been most rapid in Mexico, the country most dependent upon road transport. The growth of transportation is driven by population, per capita income, and economic output and is expected to increase by 46% in North America between 2003 and 2025.

More than half of electricity produced in North America (67% in the United States) is consumed in buildings, making that single use the third largest carbon source in North America (25% of the total). In fact, the CO_2 emissions from U.S. buildings alone were greater than total CO_2 emissions of any country in the world, except China. Energy use in buildings in the United States and Canada (including the use of natural gas, wood, and other fuels as well as electricity) has increased by 30% since 1990, corresponding to an annual growth rate of 2.1%. In the U.S., the major drivers of energy consumption in the buildings sector are growth in commercial floor space and increase in the size of the average home. Carbon emissions from buildings are expected to grow with population and income. Furthermore, the shift from family to single-occupant households means that the number of households will increase faster than population growth—each household with its own heating and cooling systems and electrical appliances. Certain electrical appliances (such as air-conditioning equipment) once considered a luxury are now becoming commonplace. Technology- and market-driven improvements in the efficiency of appliances are expected to continue, but the improvements will probably not be sufficient to curtail emissions growth in the buildings sector without government intervention.

Emissions from North American industry (not including fossil fuel mining and processing or electricity generation) are a relatively small (12%) and declining component of North America's emissions. Emissions decreased nearly 11% between 1990 and 2002, while energy consumption in the U.S. and Canada increased by 8-10% during that period. In both countries, a shift in production toward less energy-intensive industries and dissemination of more energy efficient equipment has kept the rate of growth in energy demand lower than the rate of growth of industrial GDP.

The Sinks

Approximately 30% of North American fossil fuel emissions are offset by a sink of ~600 (±300) Mt C yr^{-1} caused by a variety of factors, including forest regrowth, fire suppression, and agricultural soil conservation (Figure ES-1). The sink is currently ~500 (±250)[1] Mt C yr^{-1} in the U.S. and ~150 (±150) Mt C yr^{-1} in Canada. Mexican ecosystems are a net source of ~50 (±50 or greater) Mt C yr^{-1}, mostly as a consequence of ongoing deforestation. The coastal ocean surrounding North America is also a small net source of carbon to the atmosphere [~20 (±20) Mt C yr^{-1}][2].

The primary carbon sink in North America (approximately 50%) is in the forests of the U.S. and Canada (Figure ES-1). These forests are still growing (accumulating carbon) after their re-colonization of farmland 100 or more years ago. Forest regrowth takes carbon out of the atmosphere and stores most of it in aboveground vegetation (wood), with as much as a third of it in soils. The suppression of forest fires also increases a net accumulation of carbon in forest biomass. As the recovering forests mature, however, the rate of net carbon uptake (the sink) declines. In Canada, the estimated forest sink declined by nearly a third between 1990 and 2004, but with high interannual variability. Over that period, the annual changes in aboveground carbon stored in managed Canadian forests varied from between a sink of approximately 50 Mt C yr^{-1} to a source of approximately 40 Mt C yr^{-1}. Years when the forests were a source were generally years with high forest fire activity.

Woody encroachment, the invasion of woody plants into grasslands or of trees into shrublands, is a potentially large, but highly uncertain carbon sink. It is caused by a combination of fire suppression and grazing. Fire inside the United States has been reduced by more than 95% from the pre-settlement levels, and this reduction favors shrubs and trees in competition with grasses. The sink may be as large as 20% of

[1]With 95% certainty that the actual value is within this range of the estimate.

[2]The variation here is one standard deviation of the measurements used for analysis and represents primarily seasonal variability rather than uncertainty in the estimate of the mean (see Chapter 15).

the North American sink, but it may also be negligible. The uncertainty of this estimate is greater than 100%.

Wood products and wetlands are each thought to account for about 12% of the total North American sink. The uncertainty in this sink is ±50%. Wood products are a sink because they are increasing, both in use (e.g., furniture, house frames, etc.) and in landfills. The wetland sink is in both the peats of Canada's extensive frozen and unfrozen wetlands and the mineral soils of Canadian and U.S. wetlands. Drainage of peatlands in the U.S. has released carbon to the atmosphere, and the very large reservoir of carbon in North American wetlands (the single largest carbon reservoir of any North American ecosystem) is vulnerable to release in response to both climate change and the further drainage of wetlands for development. Either change might shift the current moderate sink to a potentially large source.

The carbon balance of agricultural lands is determined by two processes: management and changes in the environment. The effects of 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 current losses in Mexico), albeit with some increased sequestration in recent years. Agricultural lands in North America are nearly neutral with respect to carbon, with mineral soils sequestering carbon and organic soils releasing it. The effects of climate on this balance are not well known.

Conversion of agricultural and wildlands to cities and other human settlements reduces carbon stocks, while the growth of urban and suburban trees increases them. However, the rates of carbon sequestration in the vegetation and soils of settlements, while poorly quantified, are probably relatively small, certainly in comparison to fossil fuel emissions from these areas. Thus, settlements in North America are almost certainly a source of atmospheric carbon, yet the density and development patterns of human settlements are drivers of fossil fuel emissions, especially in the important residential and transportation sectors.

What are the direct, non-climatic effects of increasing atmospheric CO_2 or other changes in the carbon cycle on the land and oceans of North America? (Chapters 2–3, Chapters 10–15)

The potential impacts of increasing concentrations of atmospheric CO_2 (and other greenhouse gases) on the earth's climate are well documented (IPCC, 2001) and are the dominant reason for societal interest in the carbon cycle. However, the consequences of a carbon cycle imbalance and the buildup of CO_2 in the atmosphere extend beyond climate change alone. Ocean acidification and "CO_2 fertilization" of land plants are foremost among these direct, non-climatic effects.

The uptake of carbon by the world's oceans as a result of human activity over the last century has made them more acidic. This acidification negatively impacts corals and other marine organisms that build their skeletons and shells from calcium carbonate. Future changes could dramatically alter the composition of ocean ecosystems of North America and elsewhere, possibly eliminating coral reefs by 2100.

Rates of photosynthesis of many plant species often increase in response to elevated concentrations of carbon dioxide, thus potentially increasing plant growth and even agricultural crop yields in the future. There is, however, continuing scientific debate about whether such "CO_2 fertilization" will continue into the future with prolonged exposure to elevated carbon dioxide, and whether the fertilization of photosynthesis will translate into increased plant growth and net uptake and storage of carbon by terrestrial ecosystems. Recent studies include many examples in which experimental treatment with elevated CO_2 leads to consistent increases in plant growth, but others in which elevated CO_2 has little effect on plant growth, leads to an initial stimulation but limited long-term effects, or increases carbon losses as well as gains. Moreover, it is unclear how plants and ecosystem might respond simultaneously to both "CO_2 fertilization" and climate change. While there is some experimental evidence that plants may use less water when exposed to elevated CO_2, it seems likely that extended deep drought or other unfavorable climatic conditions could mitigate the positive effects of elevated CO_2 on plant growth. Thus, it is far from clear that elevated concentrations of atmospheric CO_2 have led to terrestrial carbon sequestration or will do so over large areas in the future. Moreover, elevated CO_2 is known to increase methane emissions from wetlands, further increasing the uncertainty in how plant response to elevated CO_2 will affect the global atmosphere and climate.

The carbon cycle also intersects with a number of critical earth system processes, including the cycling of both water and nitrogen. Virtually any change in the lands or waters of North America as part of purposeful carbon management will consequently affect these other processes and cycles. Some interactions may be beneficial. For example, an increase in organic carbon in soils is likely to increase the availability of nitrogen for plant growth and enhance the water holding capacity of the soil. Other interactions, such as nutrient limitation, fire, insect attack, increased respiration from warming, may be detrimental. However, very little is known about the complex web of interactions between carbon and other systems at continental scales, and the direct, non-climatic effects of management on the interwoven systems of the earth system is essentially unknown.

What are the options and measures implemented in North American that could significantly affect the North American and global carbon cycles (e.g., North American sinks and global atmospheric CO_2 concentrations)? (Chapter 4)

Addressing imbalances in the North American and global carbon cycles requires options and measures focused on reducing carbon emissions. Measures refer to actions and activities designed to reduce carbon emissions or otherwise manage the carbon budget. Options refer to choices among those possible measures. Options and measures focused on enhancing carbon sinks in soils and biomass can contribute as well, but their potential is far from sufficient to deal with the magnitude of current imbalances. Furthermore, carbon sinks are more vulnerable to disturbances and to changes in climate than reduced emissions because, for example, the carbon buried in fossil fuels is more secure than the carbon stored in forests.

Options for reducing carbon emissions include:

- Reducing emissions from the transportation sector through efficiency improvement, higher prices for carbon-based fuels, liquid fuels derived from biomass, and in the longer run (after 2025), hydrogen generated from non-fossil sources of energy;
- Reducing the carbon emission impact of buildings through efficiency improvements and energy-saving passive design measures;
- Reducing emissions from the industrial sector through efficiency improvement, fuel-switching, and innovative process designs; and
- Reducing emissions from energy extraction and conversion through efficiency improvement, fuel-switching, technological change (including carbon sequestration and capture) and reduced demands due to increased end-use efficiency.

In many cases, significant progress with such options would require a combination of technology research and development, policy interventions, and information and education programs

Opinions differ about the relative mitigation impact of cost-effective emission reduction vs. carbon sequestration at modest cost increases per metric ton of CO_2 emitted. Some economic analyses suggest that the potential mitigation is greater at relatively low prices for agricultural soil carbon sequestration than from fossil fuel use reduction. In addition, analyses suggest that carbon emission cap and trading policies could reduce carbon emissions significantly without a major net economic cost by providing incentives to use the least-cost combination of mitigation/sequestration alternatives.

Many options and measures that reduce emissions and increase sequestration have significant co-benefits in terms of economic efficiency, environmental management, and energy security. At the same time, actions focused on one greenhouse gas or one mitigation pathway can have unintended consequences. For instance, carbon sequestration strategies such as reduced tillage can increase emissions of methane and nitrous oxide, which are also greenhouse gases. Strategies for dealing with climate change will have to consider these other gases as well as other components of the climate systems, such as aerosols and the physical aspects of plant communities, although these components are not considered here.

Options and measures can be implemented in a variety of ways at a variety of scales, not only at international or national levels. For example, a number of municipalities, state governments, and private firms in North America have made commitments to voluntary GHG emission reductions. For cities, one focus has been the Cities for Climate Protection program of International Governments for Local Sustainability (formerly ICLEI). For states, the Regional Greenhouse Gas (Cap and Trade) Initiative is nearing implementation. For industry, one focus has been membership in the Pew Center and in the EPA Climate Leaders Program.

How can we improve the application of scientific information to decision support for carbon management and climate decision making? (Chapter 5)

Effective carbon management requires that relevant, appropriate science be communicated to the wide variety of people whose decisions affect carbon cycling. Because the field is relatively new and the demand for policy-relevant information has been limited, carbon cycle science has rarely been organized or conducted to inform carbon management. To generate information that can systematically inform carbon management decisions, scientists and decision makers need to clarify what information would be most relevant in specific sectors and arenas for carbon management, adjust research priorities as necessary, and develop mechanisms that enhance the credibility and legitimacy of the information being generated.

In the United States, the Federal carbon science enterprise does not yet have many mechanisms to assess emerging demands for carbon information across scales and sectors. Federally funded carbon science has focused predominantly on basic research to reduce uncertainties about the carbon cycle. Initiatives are now underway to promote coordinated, interdisciplinary research that is strategically prioritized to address societal needs. The need for this type of research is increasing. Interest in carbon management across sectors suggests that there may be substantial demand for information in the energy, transportation, agriculture, forestry and industrial sectors, at scales ranging from local to global.

To ensure that carbon science is as useful as possible for decision making, carbon scientists and carbon managers need to create new forums and institutions for communication and coordination. Research suggests that in order to make a significant contribution to management, scientific and technical information intended for decision making must be perceived not only as credible (worth believing), but also as salient (relevant to decision making on high priority issues) and legitimate (conducted in a way that they believe is fair, unbiased and respectful of divergent views and interests). To generate information that meets these tests, carbon stakeholders and scientists need to collaborate to develop research questions, design research strategies, and review, interpret and disseminate results. Transparency and balanced participation are important for guarding against politicization and enhancing usability.

To make carbon cycle science more useful to decision makers in the United States and elsewhere in North America, we suggest that leaders in the carbon science community take the following steps:

- Identify specific categories of decision makers for whom carbon cycle science is likely to be salient, focusing on policy makers and private sector managers in carbon-intensive sectors (energy, transport, manufacturing, agriculture and forestry);
- Identify and evaluate existing information about carbon impacts of decisions and actions in these arenas, and assess the need and demand for additional information. In some cases, demand may need to be nurtured and fostered through a two-way interactive process;
- Encourage scientists and research programs to experiment with new and different ways of making carbon cycle science more salient, credible, and legitimate to carbon managers;
- Involve not just physical or biological disciplines in scientific efforts to produce useable science, but also social scientists, economists, and communication experts; and
- Consider initiating participatory pilot research projects and identifying existing "boundary organizations" (or establishing new ones) to bridge carbon management and carbon science.

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Table ES-1. North American contribution to the global carbon budget of approximately the 1990s. Global values are for the 1990s (IPCC, 2001); the North American terrestrial sink estimate, from this report span the 1990s and first years of the 21st century. Values are in Mt C yr^{-1}, with positive values flues to the atmosphere and negative value are uptake from the atmosphere.

	Global[a] (Mt C yr^{-1})	**North America[b] (Mt C yr^{-1})**	**North American fraction of global (percent of means)**
Atmospheric increase	3200 ± 100	NA	NA
Emissions (fossil, fuel, cement)	6300 ± 400	1640 ± 164[c]	26%
Ocean-atmosphere flux	–1700 ± 500	20 ± 20[d]	1%
Emissions from land-use change	1600 ± 800[e]	–37[f]	2%
Terrestrial Sink	–2300 ± 1300[g]	–600 ± 300[h]	26%

NA indicates "Not Applicable"

[a] Global uncertainties are ± 1 standard error (67% confidence intervals) (IPCC, 2001).
[b] North American uncertainties are 95% confidence intervals. See Chapter 3 in this report.
[c] Average emissions for 1990–1999 (Marland *et al*., 2006).
[d] The variation here is one standard deviation of the measurements used for analysis and represents primarily seasonal variability rather than uncertainty in the estimate of the mean (see Chapter 15 in this report).
[e] Estimate for the years 1989–1995 (IPCC, 2000).
[f] United States only; values for the 1980s (Houghton *et al*., 1999).
[g] Residual calculated as the difference between combined fossil-fuel and land-use emissions minus ocean uptake and increase in the atmosphere (IPCC, 2001)
[h] Estimated from changes in inventories of carbon stored in plants and soils (see Chapter 3 in this report).

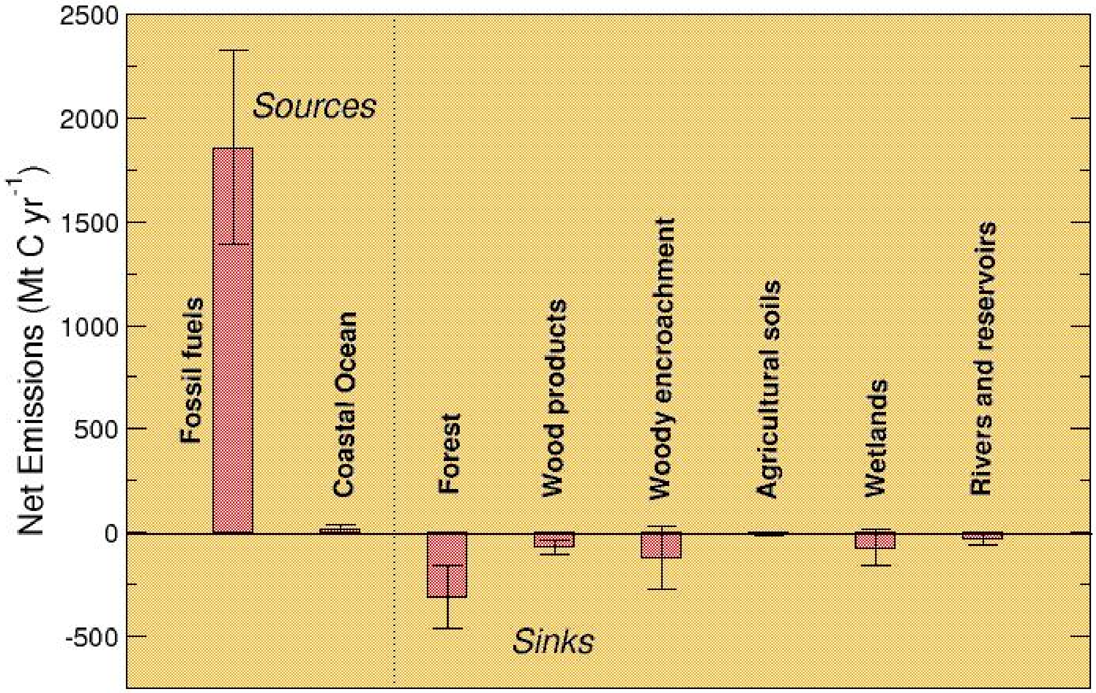

Figure ES-1. North American carbon sources and sinks (Mt C yr^{-1}) circa 2003. Height of a bar indicates a best estimate for net carbon exchange between the atmosphere and the indicated element of the North American carbon budget. Error bars indicate the uncertainty in that estimate, and define the range of values that include the actual value with 95% certainty. See Chapter 3 and Chapters 6-15 of this report for details and discussion of these sources and sinks.

Chapter 1. What is the Carbon Cycle and Why Do We Care?

Lead Authors: SOCCR Coordinating Team

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WHY A REPORT ON THE CARBON CYCLE?

The concept of a *carbon budget* or *carbon cycle* is unfamiliar to many decision makers and other citizens. We are familiar with a *water cycle*, where precipitation falls on the earth to supply water bodies and evaporation returns water vapor to the earth's clouds, which then renew the cycle through precipitation. Similarly, carbon—a fundamental requirement for life on earth—cycles through exchanges between pools of carbon on and near the earth's surface (mainly in plants and soils), in the atmosphere, and in water and sediments in the ocean. Stated in oversimplified terms, plants consume carbon dioxide (CO_2) from the atmosphere through photosynthesis and create sugars and other carbohydrates, which animals and humans use for food, shelter and energy to sustain life. Emissions from plants, other natural systems, and human activities return carbon to the atmosphere, which renews the cycle (Fig. 1-1).

Figure 1-1. The global carbon cycle. Carbon cycles through pools or reservoirs of carbon on land, in the ocean, and in sedimentary rock formations over daily, seasonal, annual, millennial and geological time scales. See the accompanying text box.

All of the components of this cycle—the atmosphere, the terrestrial vegetation, soils, freshwater lakes and rivers, the ocean, and geological sediments—are reservoirs of carbon. As carbon cycles through the system, it is exchanged between reservoirs, transferred from one to the next, with exchanges often in both directions. The carbon *budget* is an accounting of the balance of exchanges of carbon among the reservoirs: how much carbon is stored in a reservoir at a particular time, how much is coming in from other reservoirs, and how much is going out. When the inputs to a reservoir (the sources) exceed the outputs (the sinks), the amount of carbon in the reservoir increases. The myriad physical, chemical, and biological processes that transfer carbon among reservoirs, and transform carbon among its various molecular forms during those transfers, are responsible for the cycling of carbon through reservoirs. That cycling determines the balance of the carbon budget observed at any particular time. Examining the carbon budget not only reveals whether the budget is in balance, and if it is unbalanced can provide insights about why such a condition exists and how it might be managed. Currently, the global carbon budget is out of balance, and human use of coal, petroleum, and natural gas to fuel economies is primarily responsible (IPCC, 2001). Ongoing tropical deforestation also contributes, transferring carbon from plants and soils to the atmosphere as carbon dioxide (Houghton, 1999).

If vast quantities of water had been trapped underground for millennia and then, in recent centuries, released to trigger unprecedented rates of evaporation—and thus significant changes in cloud formation and precipitation patterns—there might be concerns about possible imbalances in the water cycle. Although this has not happened for water, it has happened for carbon. Over the millennia, vast quantities of carbon were stored in residues from dead plant and animal life that sank into the earth and became fossilized. With the expansion of the Industrial Revolution in the 19th and 20th centuries, human societies found that these fossils had great value as energy sources for economic growth; and the 20th century saw a dramatic rise in the combustion of these “fossil fuels” (e.g., coal, petroleum, and natural gas), releasing into the atmosphere over *decades* quantities of carbon that had been stored in the earth system over *millennia*. During this same time, forests that had once absorbed very large quantities of carbon dioxide each year shrank in their extent, and continue to do so in tropical regions.

It is not surprising, then, that measurements of carbon dioxide and other carbon compounds in the earth’s atmosphere, such as methane, have shown steady increases in concentrations. This fact, together with patterns of human activity that continue trends in fossil fuel use and deforestation, raises concerns about imbalances in the carbon cycle and their implications.

The Carbon Cycle and Climate Change

Most of the carbon in the earth’s atmosphere is in the form of carbon dioxide and methane (CH_4). Both carbon dioxide and methane are important “greenhouse gases.” Along with water vapor, and other “radiatively active” gases in the atmosphere, they absorb heat radiated from the earth’s surface, heat that would otherwise be lost into space. As a result, these gases help warm the earth’s atmosphere. Rising concentrations of atmospheric carbon dioxide and other greenhouse gases can alter the earth’s radiant energy balance. The earth’s energy budget determines the global circulation of heat and water through the atmosphere and the patterns of temperature and precipitation we experience as weather and climate. Thus, the human disturbance of the earth’s global carbon cycle during the Industrial era and the resulting imbalance in the earth’s carbon budget and buildup of carbon dioxide in the atmosphere have consequences for climate and climate change. According to the Strategic Plan of the U.S. Climate Change Science Program, carbon dioxide is the largest single forcing agent of climate change (CCSP, 2003).

In addition to the relationship between climate change and atmospheric carbon dioxide as a greenhouse gas, research is beginning to reveal the feedbacks between a changing carbon cycle and changing climate and what that implies for future climate change. Simulations with climate models that include an interactive global carbon cycle indicate a positive feedback between climate change and atmospheric carbon dioxide concentrations., The magnitude of the feedback varies considerably among models; but in all cases, future atmospheric carbon dioxide concentrations are higher and temperature increases are larger in the coupled climate-carbon cycle simulations than in simulations without the coupling and feedback between climate change and changes in the carbon cycle (Friedlingstein *et al*., 2006). The research is in its early stages, but 8 of the 11 models in a recent comparison among models (Friedlingstein *et al*., 2006) attributed most of the feedback to changes in land carbon, with the majority locating those changes in the Tropics. Differences among models in almost every aspect of plant and soil response to climate were responsible for the differences in model results, including plant growth in response to atmospheric carbon dioxide concentrations and climate and accelerated decomposition of dead organic matter in response to warmer temperatures.

Invariably, any options or actions to prevent, minimize, or forestall future climate change will require management of the carbon cycle and concentrations of carbon dioxide in the atmosphere. That management involves both reducing sources of atmospheric carbon dioxide such as the combustion of fossil fuels and enhancing sinks such as uptake and storage or sequestration in vegetation and soils. In either case, the formulation of options by decision makers and successful management of the earth's carbon budget requires solid scientific understanding of the carbon cycle and the "ability to account for all carbon stocks, fluxes, and changes and to distinguish the effects of human actions from those of natural system variability" (CCSP, 2003). In short, because people care about the potential consequences of global climate change, they also necessarily care about the carbon cycle, the atmospheric imbalance in the carbon budget, and the balance between sources and sinks of atmospheric carbon on land and in the ocean.

Other Implications of an Imbalance in the Carbon Budget

We do not yet have a full understanding of the consequences of an unbalanced carbon budget with carbon accumulating in the atmosphere as carbon dioxide and methane, but we do know that they extend beyond climate change alone. Experimental studies, for example, tell us that, for many plant species, rates of photosynthesis often increase in response to elevated concentrations of carbon dioxide, thus potentially increasing plant growth and even agricultural crop yields in the future. There is, however, considerable uncertainty about whether such "CO_2 fertilization" will continue into the future with prolonged exposure to elevated carbon dioxide; and, of course, its potential beneficial effects on plants presume climatic conditions that are also favorable to plant and crop growth.

It is also increasingly evident that atmospheric carbon dioxide concentrations are responsible for increased acidity of the surface ocean (Caldeira and Wickett, 2003), with potentially dire future consequences for corals and other marine organisms that build their skeletons and shells from calcium carbonate. Ocean acidification is a powerful reason, in addition to climate change, to care about the carbon cycle and the accumulation of carbon dioxide in the atmosphere (Orr *et al*., 2005).

It is clear that we need to appreciate the importance of the earth's carbon cycle, its implications for our well-being in North America, and the challenge of clarifying what we know versus what we do not know about the carbon cycle. The reason is that any sustained imbalance in the earth's carbon cycle could be serious business indeed for North America, as it could be for any other part of the world.

Why the Carbon Budget of North America?

The continent of North America has been identified as both a significant source and a significant sink of atmospheric carbon dioxide (Wofsy and Harriss, 2002). More than a quarter (27%) of global carbon emissions from the combination of fossil fuel and cement manufacturing are attributable to North America (United States, Canada, and Mexico) (Marland *et al*., 2003). North American plants remove carbon dioxide from the atmosphere and store it as carbon in plant biomass and soil organic matter, mitigating to some degree the anthropogenic sources. The magnitude of the "North American sink" has been estimated at anywhere from less than 100 Mt C yr^{-1} to slightly more than 2000 Mt C y^{-1} (Turner *et al*., 1995; Fan *et al*., 1998), with a value near 350 to 750 Mt C yr^{-1} perhaps most likely (Houghton *et al*., 1999; Goodale *et al*., 2002; Gurney *et al*., 2002). In Chapter 3 of this report the sink is estimated to be 592 Mt C yr^{-1}. The North American sink is thus a substantial, if highly uncertain fraction, from 15% to essentially 100%, of the extra-tropical Northern Hemisphere terrestrial sink estimated to be in the range of 600 to 2300 Mt C yr^{-1} during the 1980s (IPCC, 2001). It is also a reasonably large fraction (perhaps near 30%) of the global terrestrial sink estimated at 1900 Mt C yr^{-1} for the 1980s (but with a range of uncertainty from a large sink of 3800 Mt C yr^{-1} to a small source of 300 Mt C yr^{-1} (IPCC, 2001). The global terrestrial sink is responsible for about a quarter to a half of the carbon added to the atmosphere by human actions that was subsequently transferred to oceans and land by carbon cycle processes. This is carbon that did not contribute to the accumulation and increase of carbon dioxide in the atmosphere. Global atmospheric carbon concentrations would be substantially higher than they are without the partially mitigating influence of the sink in North America.

Some mechanisms that might be responsible for the North American terrestrial sink are reasonably well known. These mechanisms include, but are not limited to, the re-growth of forests following abandonment of agriculture, changes in fire and other disturbance regimes, historical climate change, and fertilization of ecosystem production by nitrogen deposition and elevated atmospheric carbon dioxide (Dilling *et al*., 2003). Recent studies have indicated that some of these processes are likely more important than others for the current North American carbon sink, with regrowth of forests on former agricultural generally considered to be a major contributor, and with perhaps a significant contribution from enhanced plant growth in response to higher concentrations of atmospheric carbon dioxide (CO_2 fertilization) (Caspersen *et al*., 2000; Schimel *et al*., 2000; Houghton 2002). But significant uncertainties remain (Caspersen *et al.*, 2000; Schimel *et al*., 2000; Houghton 2002), with some arguing that even the experimental evidence for CO_2 fertilization is equivocal at the larger spatial scales necessary for a significant terrestrial sink (e.g., Nowak *et al*., 2004; Friedlingstein *et al*., 2006). The future of the current North American terrestrial sink is highly uncertain, and it depends on which mechanisms are the dominant drivers now and in the future.

Estimates of coastal carbon cycling and input of carbon from the land are equally uncertain (JGOFS, 2001). Coastal processes are also difficult to parameterize in global carbon cycle models, which are often used to derive best-guess estimates for regional carbon budgets (JGOFS, 2001). It is very important to quantify carbon fluxes in coastal margins of the area adjacent to the North American continent, lest regional budgets of carbon on land be mis-attributed.

Whether as source or sink, North America is a major player in the global carbon cycle. The scientific understanding of the global carbon cycle required for successful carbon management strategies and by decision makers searching for options to stabilize or mitigate concentrations of greenhouse gases in the atmosphere (CCSP, 2003) requires an understanding of the North American carbon budget.

In the absence of explicit and specific carbon management targets it is difficult to address the question of just how well, with what precision, the North American carbon budget must be known to achieve carbon management goals. It is clear, however, that a terrestrial sink generated by “natural” processes is an ecosystem service worth billions of dollars if purchased or realized through direct human economic and technological intervention (Pep Canadell, personal communication, 2006). Its existence will influence carbon management decision making, and it is important that its magnitude and its dynamics be well understood.

It is particularly important to understand the likely future behavior of the carbon cycle, including terrestrial and oceanic sources and sinks. Decisions made about future carbon management with expectations of the future behavior of the carbon cycle that proved to be significantly in error, could be costly. For example, the response of the carbon cycle to future climate-carbon feedbacks could change the strength of terrestrial sinks and put further pressure on emission reductions to achieve, for example, atmospheric stabilization targets (Pep Canadell, personal communication, 2006). The future can't be known, but understanding it's past and present will increase confidence in projections of future carbon cycle behavior for appropriate consideration by decision makers.

CARBON CYCLE SCIENCE IN SUPPORT OF CARBON MANAGMENT DECISIONS

Beyond understanding the science of the North American carbon budget and its drivers, increasing attention is now being given to deliberate management strategies for carbon (DOE, 1997, Hoffert *et al.*, 2002; Dilling *et al.*, 2003). Carbon management is now being considered at a variety of scales in North America. There are tremendous opportunities for carbon cycle science to improve decision-making in this arena, whether in reducing carbon emissions from the use of fossil fuels, or in managing terrestrial carbon sinks. Many decisions in government, business, and everyday life are connected with the carbon cycle. They can relate to *driving forces* behind changes in the carbon cycle (such as consumption of fossil fuels) and strategies for managing them and/or *impacts* of changes in the carbon cycle (such as climate change or ocean acidification) and responses to reduce their severity. Carbon cycle science can help to inform these decisions by providing timely and reliable information about facts, processes, relationships, and levels of confidence.

In seeking ways to more effectively use scientific information in decision-making, we must pay particular attention to the importance of developing constructive scientist–stakeholder interactions. Studies of these interactions all indicate that neither scientific research nor assessments can be assumed to be relevant to the needs of decision-makers if conducted in isolation from the context of those users needs (Cash and Clark, 2001; Cash *et al*., 2003; Dilling *et al*., 2003; Parson, 2003). Carbon cycle science's support of decision-making is more likely to be effective if the science is connected with communication structures that are considered by both scientists and users to be legitimate and credible. Well designed scientific assessments can be one of these effective communication media.

The U.S. climate and carbon research community, and a diverse range of stakeholders, recognize the need for an integrated synthesis and assessment focused on North America to (a) summarize what is known and what is known to be unknown, documenting the maturity as well as the uncertainty of this knowledge; (b) convey this information among scientists and to the larger community; and (c) ensure that our studies are addressing the questions of concern to society and decision-making communities. As the most comprehensive treatment to date of carbon cycle facts, directions, and issues for North America, incorporating stakeholder interactions throughout, this report, the *First State of the Carbon Cycle Report (SOCCR)*, focused on *The North American Carbon Budget and Implications for the Global Carbon Cycle* is intended as a step in that direction.

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The Global Carbon Cycle

The burning of fossil fuels transfers carbon from geological reservoirs of coal, oil and gas and releases carbon dioxide into the atmosphere. Tropical deforestation and other changes in land-use also release carbon to the atmosphere as vegetation is burned and dead material decays. Photosynthesis transfers carbon dioxide from the atmosphere and the carbon is stored in wood and other plant tissues. The respiration that accompanies plant metabolism transfers some of the carbon back to the atmosphere as carbon dioxide. When plants die, their decay also releases carbon dioxide to the atmosphere. A fraction of the dead organic material is resistance to decay and that carbon accumulates in the soil. Chemical and physical processes are responsible for the exchange of carbon dioxide across the sea surface. The small difference between the flux in to and out of the surface ocean is responsible for net uptake of carbon dioxide by the ocean. Phytoplankton, small plants floating in the surface ocean, use carbon dissolved in the water to build tissue and calcium carbonate shells. When they die, they begin to sink and decay. As they decay, most of the carbon is redissolved into the surface water, but a fraction sinks into the deeper ocean, the so-called “biological pump”, eventually reaching he ocean sediments. Currents within the ocean also circulate carbon from surface waters to Deep Ocean and back. Carbon accumulated in soils and ocean sediments millions of years of ago was slowly transformed to produce the geological reservoirs of today’s fossil fuels.

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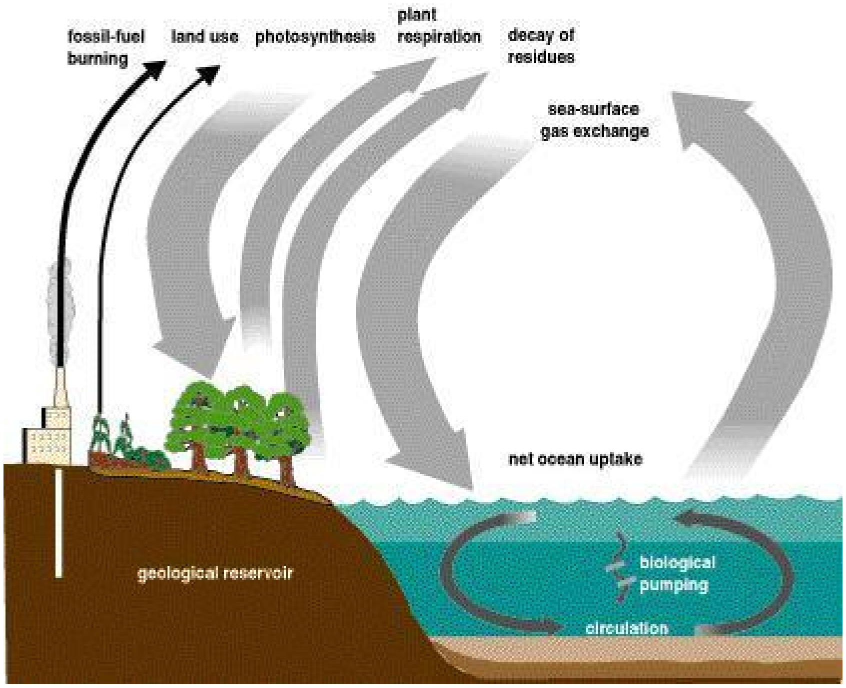

Figure 1-1. The global carbon cycle. Carbon cycles through pools or reservoirs of carbon on land, in the ocean, and in sedimentary rock formations over daily, seasonal, annual, millennial and geological time scales. See the accompanying text box.

Chapter 2. The Carbon Cycle of North America in a Global Context

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KEY FINDINGS

- Human activity over the last two centuries, including combustion of fossil fuel and clearing of forests, has led to a dramatic increase in the concentration of atmospheric carbon dioxide. Global atmospheric CO_2 concentrations have risen by 31% since 1850, and they are now higher than they have been for 420,000 years.
- North America is responsible for approximately 27% of the emissions produced globally by fossil-fuel combustion, with the United States accounting for 86% of the North American total.
- Anthropogenic emissions (a carbon source) dominate the carbon budget of North America. Largely unmanaged, unintentional processes lead to a smaller carbon sink (uptake of carbon). The sink is approximately 30% of the North American emissions, 9% of global emissions, and approximately 50% of the global terrestrial sink inferred from global budget analyses and atmospheric inversions.
- While the future trajectory of carbon sinks in North America is uncertain (substantial climate change could convert current sinks into sources), it is clear that the carbon cycle of the next few decades will be dominated by the large sources from fossil-fuel emissions.
- Because North American carbon emissions are at least a quarter of global emissions, a reduction in North American emissions would have global consequences.

THE GLOBAL CYCLE

The modern global carbon cycle is a collection of many different kinds of processes, with diverse drivers and dynamics, that transfer carbon among major pools in rocks, fossil fuels, the atmosphere, the oceans, and plants and soils on land (Sabine *et al*., 2004b) (Fig. 2-1). During the last two centuries, human actions, especially the combustion of fossil fuel and the clearing of forests, have altered the global carbon cycle in important ways. Specifically, these actions have led to a rapid, dramatic increase in the concentration of carbon dioxide (CO_2) in the atmosphere (Fig. 2-2), changing the radiation balance of the Earth (Hansen *et al*., 2005), and most likely warming the planet (Mitchell *et al*., 2001). The cause of the recent increase in atmospheric CO_2 is confirmed beyond a reasonable doubt (Prentice, 2001). This does not imply, however, that the other components of the carbon cycle have remained unchanged during this period. The background or unmanaged parts of the carbon cycle have, in fact, changed dramatically over the past two centuries. The consequence of these changes is that only about 40% ± 15% of the carbon dioxide emitted to the atmosphere from fossil-fuel combustion and forest clearing has remained there (with most of the uncertainty in this number due to the uncertainty in carbon lost from forest clearing) (Sabine *et al*., 2004b). In essence, human actions have received a large subsidy from the unmanaged parts of the carbon cycle. This subsidy has sequestered, or hidden from the atmosphere, approximately 279 ± 160 Gt of carbon. [Throughout this chapter, we will present the pools and fluxes in the carbon cycle in Gt C (1 Gt = 1 billion tons or 1×10^{15} g). The mass of CO_2 is greater than the mass of carbon by the ratio of their molecular weights, 44/12 or 3.67 times; 1 km^3 of coal contains approximately 1 Gt C.]

Figure 2-1. Schematic representation of the components of the global carbon cycle. The three panels show (A) the overall cycle, (B) the details of the ocean cycle, and (C), and the details of the land cycle. For all panels, carbon stocks are in brackets, and fluxes have no brackets. Pre-anthropogenic stocks and fluxes are in black. Anthropogenic perturbations are in red. For stocks, the anthropogenic perturbations are the cumulative total since 1850. Anthropogenic fluxes are means for the 1990s. Redrawn from (Sabine *et al*., 2004b) with updates as discussed in the text.

Figure 2-2. Atmospheric CO_2 concentration from 1850 to 2005. The data prior to 1957 (red circles) are from the Siple ice core (Friedli *et al*., 1986). The data since 1957 (blue circles) are from continuous atmospheric sampling at the Mauna Loa Observatory (Hawaii) (Keeling *et al*., 1976;, Thoning *et al*., 1989) (with updates available at http://cdiac.ornl.gov/trends/co2/sio-mlo.htm).

The recent subsidy or sequestration of carbon by the unmanaged parts of the carbon cycle makes them critical for an accurate understanding of climate change. Future increases in carbon uptake in the unmanaged parts of the cycle could moderate the risks from climate change, while decreases or transitions from uptake to release could amplify the risks, perhaps dramatically.

In addition to its role in the climate, the carbon cycle intersects with a number of critical earth system processes. Because plant growth is essentially the removal of carbon dioxide from the air through photosynthesis, agriculture and forestry contribute important fluxes. Wildfire is a major release of carbon from plants and soils to the atmosphere (Sabine *et al*., 2004b). The increasing concentration of CO_2 in the atmosphere has already made the world's oceans more acid (Caldeira and Wickett, 2003). Future changes could dramatically alter the composition of ocean ecosystems (Feely *et al*., 2004; Orr *et al*., 2005).

The Unmanaged Global Carbon Cycle

The modern background, or unmanaged, carbon cycle includes the processes that occur in the absence of human actions. These processes are, however, currently so altered by human influences on the carbon cycle that it is not appropriate to label them natural. This background part of the carbon cycle is dominated by two pairs of gigantic fluxes with annual uptake and release that are close to balanced (Sabine *et al*., 2004b) (Fig. 2-1). The first of these comprises the terrestrial carbon cycle: plant growth on land annually fixes about 57 ± 9 Gt of atmospheric carbon, approximately ten times the annual emission from fossil-fuel combustion, into carbohydrates. Respiration by land plants, animals, and microorganisms, which provides the energy for growth, activity, and reproduction, returns a slightly smaller amount to the atmosphere. Part of the difference between photosynthesis and respiration is burned in wildfires, and part is stored as plant biomass or soil organic carbon. The second comprises the ocean carbon cycle: about 92 Gt of atmospheric carbon dissolves annually in the oceans, and about 90 Gt yr^{-1} moves from the oceans to the atmosphere (While the gross fluxes have a substantial uncertainty, the difference is known to within ± 0.3 Gt) . These air-sea fluxes are driven by internal cycling within the oceans that governs exchanges between pools of dissolved CO_2, bicarbonate (HCO_3^-), and carbonate ($CO_3^=$); organic matter; and calcium carbonate.

Before the beginning of the industrial revolution, carbon uptake and release through these two pairs of large fluxes were almost balanced, with carbon uptake on land of approximately 0.55 ± 0.15 Gt C yr^{-1} transferred to the oceans by rivers and released from the oceans to the atmosphere. As a consequence, the level of carbon dioxide in the atmosphere varied by less than 25 ppm in the 10,000 years prior to 1850 (Joos and Prentice, 2004). But atmospheric CO_2 was not always so stable. During the preceding 420,000 years, atmospheric CO_2 was 180–200 ppm during ice ages and approximately 275 ppm during interglacials (Petit *et al*., 1999). The lower ice-age concentrations in the atmosphere most likely reflect a transfer of carbon from the atmosphere to the oceans, possibly driven by changes in ocean circulation and sea-ice cover (Sigman and Boyle, 2000; Keeling and Stephens, 2001). Enhanced biological activity in the oceans, stimulated by increased delivery of iron-rich terrestrial dust, may have also contributed to this increased uptake (Martin, 1990).

In the distant past, the global carbon cycle was out of balance in a different way. Fossil fuels are the product of prehistorically sequestered plant growth, especially 354 to 290 million years ago in the Carboniferous period. During this time, luxuriant plant growth and geological activity combined to bury a small fraction of each year's growth. Over millions of years, this gradual burial led to the accumulation of vast stocks of fossil fuel. The total accumulation of fossil fuels is uncertain, but probably in the range of 6000 ± 3000 Gt (Sabine *et al*., 2004b). It also led to a near doubling of atmospheric oxygen (Falkowski *et al*., 2005).

Anthropogenic Perturbations

Since the beginning of the industrial revolution, there has been a massive release of carbon from fossil-fuel combustion and deforestation. Cumulative carbon emissions from fossil-fuel combustion, natural gas flaring, and cement manufacture from 1751 through 2003 are 304 ± 30 Gt (Marland and Rotty, 1984; Andres *et al*., 1999) (with updates through 2003 online at http://cdiac.ornl.gov/trends/emis/tre_glob.htm). Land use change from 1850 to 2003, mostly from the clearing of forests, added another 162 ± 160 Gt (DeFries *et al*., 1999; Houghton, 1999a)(with updates through 2000 online at http://cdiac.ornl.gov/trends/landuse/houghton/houghton.html. We extrapolated the total through 2003 based on the assumption that the fluxes in 2001-2003 were the same as that in 2000.) . The rate of fossil-fuel consumption in any recent year would have required, for its production, more than 400 times the current global primary production (total plant growth) of the land and oceans combined (Dukes, 2003). This has led to a rapid increase in the concentration of CO_2 in the atmosphere since the mid-nineteenth century, with atmospheric CO_2 rising by 31% (i.e., from 287 ppm to 375 ppm in 2003; the increase from the mid-eighteenth century was 35%).

In 2003 the three major countries of North America (Canada, Mexico, and the United States) together accounted for carbon emissions from fossil-fuel combustion of approximately 1.86 ± 0.2 Gt C, or about 27% of the global total. The United States, the world's largest emitter of carbon dioxide, was responsible for 86% of the North American total. Per capita emissions in 2003 were 5.4 ± 0.5 metric ton in the United States, 5.0 ± 0.55 metric ton in Canada, and 0.9 ± 0.1 metric ton in Mexico. Per capita emissions in the United States were nearly 5 times the world average, 2.5 times the per capita emissions for Western Europe, and more than 8 times the average for Asia and Oceania (DOE EIA, 2005). The world's largest countries, China and India, have total carbon emissions from fossil-fuel combustion and the flaring of natural gas that are substantially lower than those in the United States. The 2003 total for China was 61% of that in the United States, and the total for India was 18% that of the United States. Per capita emissions for China and India in 2003 were 14% and 5%, respectively, of the U.S. rate (DOE EIA, 2005).

ASSESSING GLOBAL AND REGIONAL CARBON BUDGETS

Changes in the carbon content of the oceans and plants and soils on land can be evaluated with at least five different approaches—flux measurements, inventories, inverse estimates based on atmospheric CO_2, process models, and calculation as a residual. The first method, direct measurement of carbon flux, is well developed over land for measurements over the spatial scale of up to 1 km^2, using the eddy flux technique (Wofsy *et al*., 1993;, Baldocchi and Valentini, 2004). Although eddy flux measurements are now collected at more than 100 networked sites, spatial scaling presents formidable challenges due to spatial heterogeneity. To date, estimates of continental-scale fluxes based on eddy flux must be regarded as preliminary. Over the oceans, eddy flux is possible (Wanninkhof and McGillis, 1999), but estimates based on air-sea CO_2 concentration difference are more widely used (Takahashi *et al*., 1997).

Inventories, based on measuring trees on land (Birdsey and Heath, 1995) or carbon in ocean-water samples (Takahashi *et al*., 2002;, Sabine *et al*., 2004a), can provide useful constraints on changes in the size of carbon pools, though their utility for quantifying short-term changes is limited. Inventories were the foundation of the recent conclusion that 118 Gt of anthropogenic carbon has entered the oceans (Sabine *et al*., 2004a) and that forests in the mid-latitudes of the Northern Hemisphere sequestered 0.6 to 0.7 Gt C yr^{-1} in the 1990s (Goodale *et al*., 2002). Changes in the atmospheric inventory of O_2 (Keeling *et al*., 1996) and ^{13}C in CO_2 (Siegenthaler and Oeschger, 1987) provide a basis for partitioning CO_2 flux into land and ocean components.

Process models and inverse estimates based on atmospheric CO_2 (or CO_2 in combination with ^{13}C or O_2) also provide useful constraints on carbon stocks and fluxes. Process models build from understanding the underlying principles of atmosphere/ocean or atmosphere/ecosystem carbon exchange to make estimates over scales of space and time that are relevant to the global carbon cycle. For the oceans, calibration against observations with tracers (Broecker *et al*., 1980) (^{14}C and chlorofluorocarbons) tends to nudge a wide range of models toward similar results. Sophisticated models with detailed treatment of the ocean circulation, chemistry, and biology all reach about the same estimate for the current ocean carbon sink, 1.5 to 1.8 Gt C yr^{-1} (Greenblatt and Sarmiento, 2004), and while uncertainties on these estimates are about ±50%, they are in quantitative agreement with data-inventory approaches. Models of the land carbon cycle take a variety of approaches. They differ substantially in the data used as constraints, in the processes simulated, and in the level of detail (Cramer *et al*., 1999; Cramer *et al*., 2001). Models that take advantage of satellite data have the potential for comprehensive coverage at high spatial resolution (Running *et al*., 2004), but only over the time domain with available satellite data. Flux components related to human activities, for example deforestation, have been modeled based on historical land use (Houghton, 1999b). At present, model estimates are uncertain enough that they are often used most effectively in concert with other kinds of estimates (e.g., Peylin *et al*., 2005).

Inverse estimates based on atmospheric gases (CO_2, ^{13}C in CO_2, or O_2) infer surface fluxes based on the spatial and temporal pattern of atmospheric concentration, coupled with information on atmospheric transport (Newsam and Enting, 1988). The atmospheric concentration of CO_2 is now measured with high precision at approximately 100 sites worldwide, with many of the stations added in the last decade (Masarie and Tans, 1995). The ^{13}C in CO_2 and high-precision O_2 are measured at far fewer sites. The basic approach is a linear Bayesian inversion (Tarantola, 1987; Enting, 2002), with many variations in the time scale of the analysis, the number of regions used, and the transport model. Inversions have more power to resolve year-to-year differences than mean fluxes (Rodenbeck *et al*., 2003; Baker *et al*., 2006). Limitations in the accuracy of atmospheric inversions come from the limited density of concentration measurements, especially in the tropics, uncertainty in the transport, and errors in the inversion process (Baker *et al*. 2006). Recent studies that use a number of sets of CO_2 monitoring stations (Rodenbeck *et al*. 2003), models (Gurney *et al*., 2003; Law *et al*., 2003; Gurney *et al*., 2004; Baker *et al*., 2006), temporal scales, and spatial regions (Pacala *et al*., 2001), highlight the sources of the uncertainties and appropriate steps for managing them.

A final approach to assessing large-scale CO_2 fluxes is solving as a residual. At the global scale, the net flux to or from the land is often calculated as the residual left after accounting for fossil emissions, atmospheric increase, and ocean uptake (Post *et al*., 1990). Increasingly, the need to treat the land as a residual is receding, as the other methods improve. Still, the existence of constraints at the level of the overall budget injects an important connection with reality.

RECENT DYNAMICS OF THE UNMANAGED CARBON CYCLE

Of the approximately 466 ± 160 Gt carbon added to the atmosphere by human actions since 1850, only about 187 ± 5 Gt remain. The “missing carbon” must be stored, at least temporarily, in the oceans and in ecosystems on land. Based on a recent ocean inventory, 118 ± 19 Gt of the missing carbon has now been identified in the oceans (Sabine *et al*., 2004a). This leaves about 161 ± 160 Gt that must be stored on land (with most of the uncertainty due to the uncertainty in emissions from land use). Identifying the processes responsible for the uptake on land, their spatial distribution, and their likely future trajectory has been one of the major goals of carbon cycle science over the last decade.

Much of the recent research on the global carbon cycle has focused on annual fluxes and their spatial and temporal variation. The temporal and spatial patterns of carbon flux provide a pathway to understanding the underlying mechanisms. Based on several different approaches, carbon uptake by the oceans averaged 1.7 ± 0.3 Gt C yr^{-1} for the period from 1992–1996 (Takahashi *et al*., 2002; Gloor *et al*., 2003; Gurney *et al*., 2003; Matear and McNeil, 2003; Matsumoto *et al*., 2004). The total anthropogenic flux is this amount, plus 0.45 Gt yr^{-1} of preindustrial outgasing, for a total of 2.2 ± 0.4 Gt yr^{-1}. This rate represents an integral over large areas that are gaining carbon and the tropics, which are losing carbon (Takahashi *et al*., 2002; Gurney *et al*., 2003; Gurney *et al*., 2004; Jacobson *et al*., 2006). Interannual variability in the ocean sink for CO_2, though substantial (Greenblatt and Sarmiento, 2004), is much smaller than interannual variability on the land (Baker *et al*., 2006).

In the 1990s, carbon releases from land-use change were more than balanced by ecosystem uptake, leading to a net sink on land (without accounting for fossil-fuel emissions) of approximately 1.1 Gt C yr^{-1}

(Schimel *et al*., 2001; Sabine *et al*., 2004b). The dominant sources of recent interannual variation in the net land flux were El Niño and the eruption of Mt. Pinatubo in 1991 (Bousquet *et al*., 2000; Rodenbeck *et al*., 2003; Baker *et al*., 2006), with most of the year-to-year variation in the tropics (Fig. 2-3). Fire likely plays a large role in this variability (van der Werf *et al*., 2004).

Figure 2-3. The 13-model mean CO_2 flux interannual variability (Gt C yr^{-1}) for several continents (solid lines) and ocean basins (dashed lines). (A) North Pacific and North America, (B) Atlantic north of 15ºN and Eurasia, (C) Australasia and Tropical Pacific, (D) Africa, and (E) South America (note the different scales for Africa and South America) (from Baker *et al*., 2006).

On a time scale of thousands of years, the ocean will be the sink for more than 90% of the carbon released to the atmosphere by human activities (Archer *et al*., 1998). The rate of CO_2 uptake by the oceans is, however, limited. CO_2 enters the oceans by dissolving in seawater. The rate of this process is determined by the concentration difference between the atmosphere and the surface waters and by an air-sea exchange coefficient related to wave action, wind, and turbulence (Le Quéré and Metzl, 2004). Because the surface waters represent a small volume with limited capacity to store CO_2, the major control on ocean uptake is at the level of moving carbon from the surface to intermediate and deep waters. Important contributions to this transport come from the large scale circulation of the oceans, especially the sinking of cold water in the Southern Ocean and, to a lesser extent, the North Atlantic.

On land, numerous processes contribute to carbon storage and carbon loss. Some of these are directly influenced through human actions (e.g., the planting of forests, conversion to no-till agriculture, or the burying of organic wastes in landfills). The human imprint on others is indirect. This category includes ecosystem responses to climate change (e.g., warming and changes in precipitation), changes in the composition of the atmosphere (e.g., increased CO_2 and increased tropospheric ozone), and delayed consequences of past actions (e.g., regrowth of forests after earlier harvesting). Early analyses of the global carbon budget (e.g., Bacastow and Keeling, 1973) typically assigned all of the net flux on land to a single mechanism, especially fertilization of plant growth by increased atmospheric CO_2. Recent evidence emphasizes the diversity of mechanisms.

The Carbon Cycle of North America

By most estimates, the land area of North America is currently a sink for carbon, in the absence of emissions from fossil-fuel combustion. This conclusion for the continental scale is based mainly on the results of atmospheric inversions. Several studies address the carbon balance of particular ecosystem types [e.g., forests (Kurz and Apps, 1999; Goodale *et al*., 2002; Chen *et al*., 2003)]. Pacala and colleagues

(Pacala *et al*., 2001) used a combination of atmospheric and land-based techniques to estimate that the 48 contiguous U.S. states are currently a carbon sink of 0.3 to 0.6 Gt C yr^{-1}. This estimate and a discussion of the processes responsible for recent sinks in North America are updated in chapter 3. Based on inversions using 13 atmospheric transport models, North America was a carbon sink of 0.97 Gt C yr^{-1} from 1991–2000 (Baker *et al*., 2006). Over the area of North America, this amounts to an annual carbon sink of 39.6 g C m^{-2} yr^{-1}, similar to the sink inferred for all northern lands (North America, Europe, Boreal Asia, and Temperate Asia) of 32.5 g C m^{-2} yr^{-1} (Baker *et al*., 2006).

Very little of the current carbon sink in North America is a consequence of deliberate action to sequester carbon. Some is a collateral benefit of steps to improve land management, for increasing soil fertility, improving wildlife habitat, etc. Much of the current sink is unintentional, a consequence of historical changes in technologies and preferences in agriculture, transportation, and urban design.

CARBON CYCLE OF THE FUTURE

The future trajectory of carbon sinks in North America is very uncertain. Several trends will play a role in determining the sign and magnitude of future changes. One important controller is the magnitude of future climate changes. If the climate warms significantly, much of the United States could experience a decrease in plant growth and an increase in the risk of wildfire (Bachelet *et al*., 2003), especially if the warming is not associated with substantial increases in precipitation. Exactly this pattern—substantial warming with little or no change in precipitation—characterizes North America in many of the newer climate simulations (Rousteenoja *et al*., 2003). If North American ecosystems are sensitive to elevated CO_2, nitrogen deposition, or warming, plant growth could increase (Schimel *et al*., 2000). The empirical literature on CO_2 and nitrogen deposition is mixed, with some reports of substantial growth enhancement (Norby *et al*., 2005) and others reporting small or modest effects (Oren *et al*., 2001; Shaw *et al*., 2002; Heath *et al*., 2005).

Overall, the carbon budget of North America is dominated by carbon releases from the combustion of fossil fuels. Recent sinks, largely from carbon uptake in plants and soils, may approach 50% of the recent fossil fuel source (Baker *et al*., 2006). Most of this uptake appears to be a rebound, as natural and managed ecosystems recover from past disturbances. Little evidence supports the idea that these ecosystem sinks will increase in the future. Substantial climate change could convert current sinks into sources (Gruber *et al*., 2004).

In the future, trends in the North American energy economy may intersect with trends in the natural carbon cycle. A large-scale investment in afforestation could offset substantial future emissions (Graham, 2003). Costs of this kind of effort would, however, include the loss of the new forested area from its previous uses, including grazing or agriculture, plus the energy costs of managing the new forests, plus any increases in emissions of non-CO_2 greenhouse gases from the new forests. Large-scale investments in biomass energy would have similar costs but would result in offsetting emissions from fossil-fuel combustion, rather than sequestration (Giampietro *et al*., 1997). The relative costs and benefits of investments in afforestation and biomass energy will require careful analysis (Kirschbaum, 2003). Investments in other energy technologies, including wind and solar, will require some land area, but the impacts on the natural carbon cycle are unlikely to be significant or widespread (Hoffert *et al*., 2002; Pacala and Socolow, 2004).

Like the present, the carbon cycle of North America during the next several decades will be dominated by fossil emissions. Geological sequestration may become an increasingly important component of the budget sheet. Still, progress in controlling the net release to the atmosphere must be centered on the production and consumption of energy rather than the processes of the unmanaged carbon cycle. North America has many opportunities to decrease emissions (Chapter 4). Nothing about the status of the unmanaged carbon cycle provides a justification for assuming that it can compensate for emissions from fossil fuel combustion.

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Table 1. Sinks of carbon for 1980–90 in the coterminous United States (in Gt C yr^{-1}).

Category	Low	High	Land area 1980–90 (10^6 ha)	Houghton *et al.* (8)	Birdsey and Heath (12)
Forest trees	0.11	0.15	247–247	0.06 [a]	0.11
Other forest organic matter	0.03	0.15	247–247	– 0.01	0.18
Cropland soils	0.00	0.04	185–183	0.14	—
Nonforest, non-cropland (woody encroachment)	0.12 [b]	0.13 [b]	334–336 [c]	0.12	—
Wood products	0.03	0.07	—	0.03	0.03
Reservoirs, alluvium, colluvium	0.01	0.04	—	—	—
Exports minus imports of food, wood	0.04	0.09	—	—	—
Fixed in the United States but exported by rivers	0.03	0.04	—	—	—
"Apparent"[d] U.S. sink without woody encroachment	0.25	0.58	766	0.15–0.23 [e]	0.31
"Apparent"[d] U.S. sink including woody encroachment	0.37	0.71	766	0.15–0.35 [e]	—
Sink [f]	0.03	0.58	766	0.15–0.35 [e]	0.31

[a] Assumes that the 0.05 Gt C yr^{-1} estimated in (8) to be accumulating in western pine woodlands as a result of the suppression is assigned to forest instead of row 4.
[b] These numbers are not bounds, but rather the only two existing estimates.
[c] Total area for all lands other than forest and croplands. Possible woody encroachment because of fire suppression on up to about two-thirds of this land (10,16).
[d] By "apparent" sink, we mean the net flux from the atmosphere to the land that would be estimated in an inversion. It includes all terms in the table.
[e] Lower bound reflects uncertainty in the estimates for the effects of fire suppression.
[f] Excludes sinks caused by the export/import imbalance for food and wood products and river exports because these create corresponding sources outside the United States.
Source: Pacala *et al*. (2001)

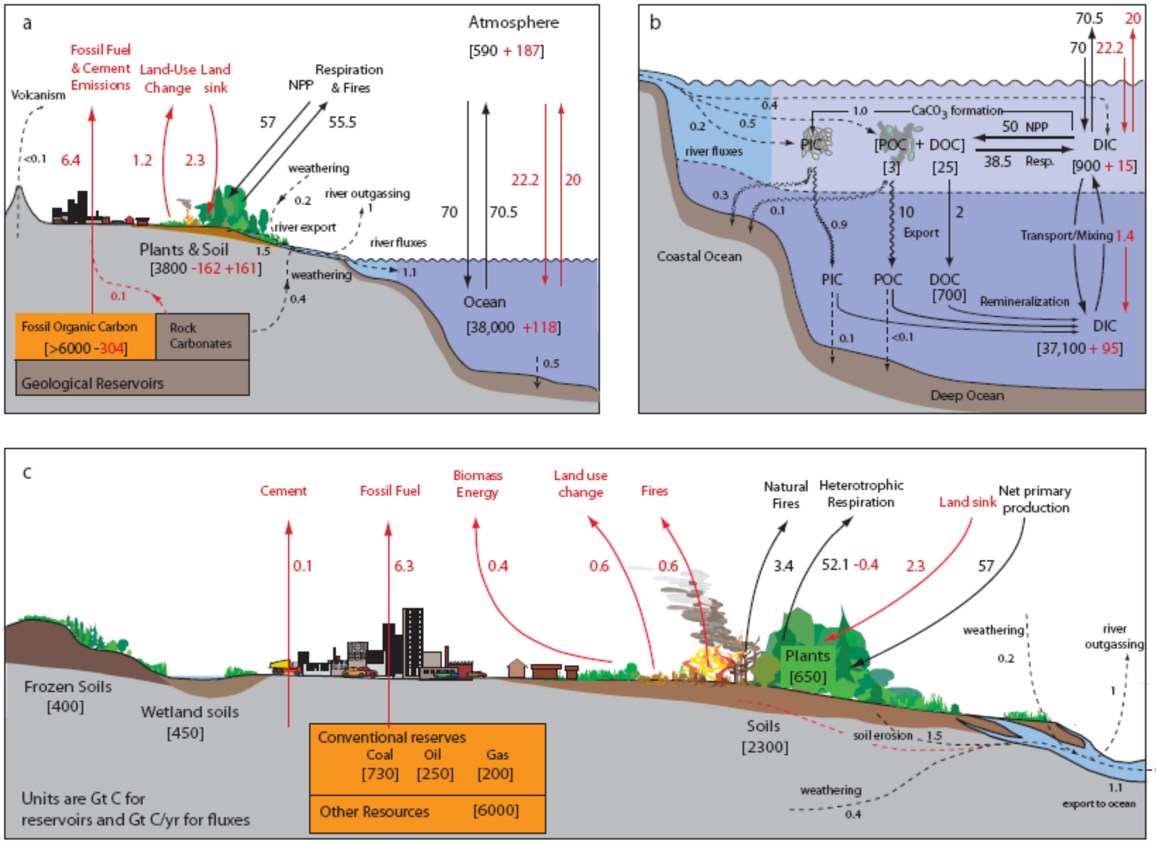

Figure 2-1. Schematic representation of the components of the global carbon cycle. The three panels show (A) the overall cycle, (B) the details of the ocean cycle, and (C) and the details of the land cycle. For all panels, carbon stocks are in brackets, and fluxes have no brackets. Pre-anthropogenic stocks and fluxes are in black. Anthropogenic perturbations are in red. For stocks, the anthropogenic perturbations are the cumulative total since 1850. Anthropogenic fluxes are means for the 1990s. Redrawn from Sabine *et al*. (2004b) with updates as discussed in the text.

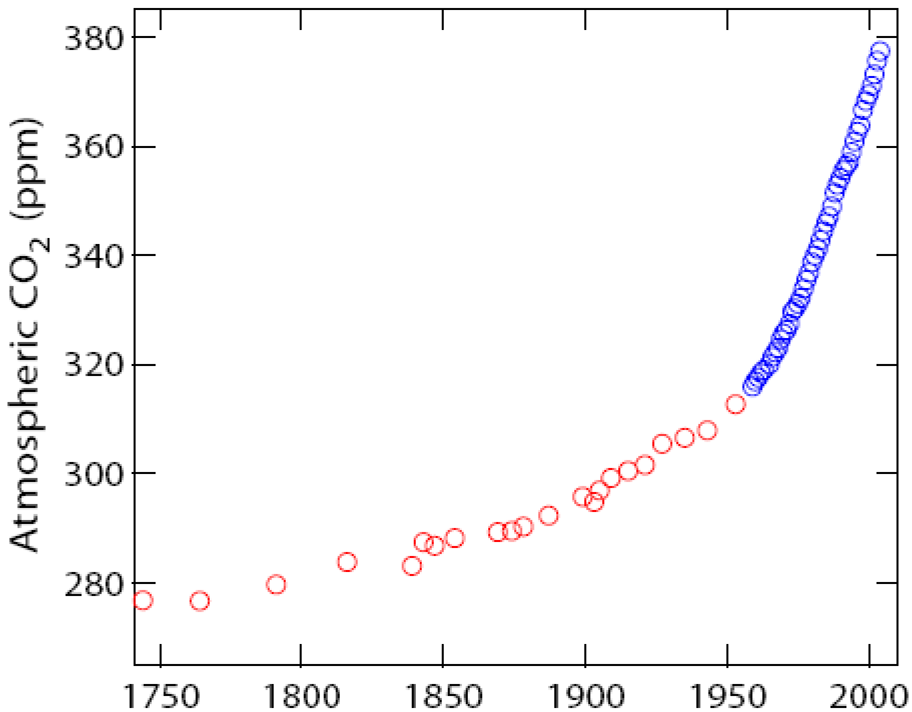

Fig. 2-2. Atmospheric CO_2 concentration from 1850 to 2005. The data prior to 1957 (red circles) are from the Siple ice core (Friedli *et al.*, 1986). The data since 1957 (blue circles) are from continuous atmospheric sampling at the Mauna Loa Observatory (Hawaii) (Keeling *et al.*, 1976; Thoning *et al.*, 1989) (with updates available at http://cdiac.ornl.gov/trends/co2/sio-mlo.htm).

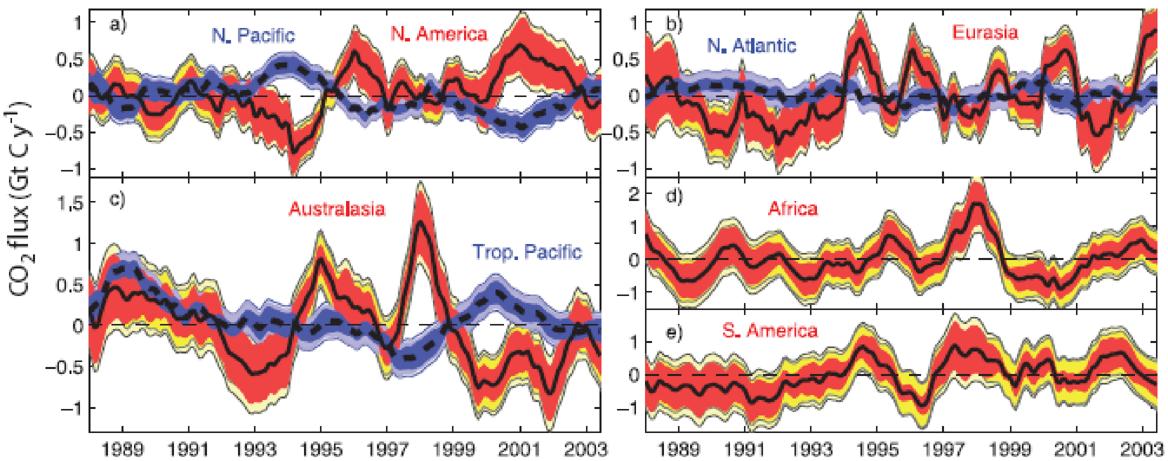

Figure 2-3. The 13-model mean CO_2 flux interannual variability (Gt C yr^{-1}) for several continents (solid lines) and ocean basins (dashed lines). (A) North Pacific and North America, (B) Atlantic north of 15ºN and Eurasia, (C) Australasia and Tropical Pacific, (D) Africa, and (E) South America (note the different scales for Africa and South America) [from (Baker *et al*., 2006)].

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Chapter 3. The North American Carbon Budget Past and Present

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KEY FINDINGS

- Fossil fuel carbon emissions in the United States, Canada, and Mexico totaled 1856 Mt C yr^{-1} in 2003 This represents 27% of global fossil fuel emissions.
- Approximately 30% of North American fossil fuel emissions are offset by a natural sink of 592 Mt C yr^{-1} caused by a variety of factors, including forest regrowth, fire suppression, and agricultural soil conservation.
- North American carbon dioxide emissions from fossil fuel have increased at an average rate of approximately 1% per year for the last 30 years.
- The growth in emissions accompanies the historical growth in the industrial economy and Gross Domestic Product (GDP) of North America. However, at least in the United States and Canada the rate of emissions growth is less than the growth in GDP, reflecting a decrease in the carbon intensity of these economies.
- Historically the plants and soils of the United States and Canada were sources for atmospheric CO2, primarily as a consequence of the expansion of croplands into forests and grasslands. In recent

decades the terrestrial carbon balance of these regions have shifted from source to sink as forests recover from agricultural abandonment, fire suppression and reduced logging and, as a result, are accumulating carbons. In Mexico, emissions of carbon continue to increase from net deforestation.

- Fossil fuel emissions from North America are expected to continue to grow, but will also continue to grow more slowly than GDP.
- The future of the North American carbon sink is highly uncertain. The contribution of recovering forests to this sink is likely to decline as these forests mature, but we do not know how much of the sink is due to fertilization of the ecosystems by nitrogen in air pollution and by increasing CO2 concentrations in the atmosphere, nor do we understand the impact of tropospheric ozone or how the sink will change as the climate changes.
- The magnitude of the North American sink offers the possibility that significant mitigation of fossil fuel emissions could be accomplished by managing forests, rangelands, and croplands to increase the carbon stored in them. However, the range of uncertainty in these estimates is at least as large as the estimated values themselves.
- Current trends towards lower carbon intensity of U.S. and Canadian economies increase the likelihood that a portfolio of carbon management technologies will be able to reduce the 1% annual growth in fossil fuel emissions. This same portfolio might be insufficient if carbon emissions were to begin rising at the approximately 3% growth rate of GDP.

Fossil Fuel

Fossil fuel carbon emissions in the United States, Canada, and Mexico totaled 1856 Mt C yr^{-1} in 2003 and have increased at an average rate of approximately 1% per year for the last 30 years (United States = 1582, Canada = 164, Mexico = 110 Mt C yr^{-1}, see Fig. 3-1). This represents 27% of global emissions, from a continent with 7% of the global population, and 25% of global GDP (EIA, 2005).

Figure 3-1. Historical carbon emissions from fossil fuel in the United States, Canada, and Mexico. Data from the US Energy Information Administration (EIA 2005).

The United States is the world's largest emitter in absolute terms. Its per capita emissions of 5.4 t C yr^{-1} are among the largest in the world, but the carbon intensity of its economy (emissions per unit GDP) at 0.15 metric tons of emitted carbon per dollar of GDP is close to the world's average of 0.14 t C/$ (EIA, 2005). Total U.S. emissions have grown at close to the North American average rate of about 1.0% per year over the past 30 years, but U.S. per capita emissions have been roughly constant, while the carbon intensity of the U.S. economy has decreased at a rate of about 2% per year (see Figs. 3-1 to 3-5).

Absolute emissions grew at 1% per year even though per capita emissions were roughly constant simply because of population growth at an average rate of 1%. The constancy of U.S. per capita values masks faster than 1% growth in some sectors (e.g., transportation) that was balanced by slower growth in others (e.g., increased manufacturing energy efficiency) (Fig. 3-3, 3-4 and 3-5).

Historical decreases in U.S. carbon intensity began early in the 20th century and continue despite the approximate stabilization of per capita emissions (Fig. 3-2). Why has the U.S. carbon intensity declined? This question is the subject of the extensive literature on the so-called structural decomposition of the energy system and on the relationship between GDP and environment (i.e., Environmental Kuznets Curves; Grossman and Krueger, 1995; Selden and Song, 1994). See for example Greening *et al*. (1997, 1998), Casler and Rose (1998), Golove and Schipper (1998), Rothman (1998), Suri and Chapman (1998), Greening *et al*. (1999), Ang and Zhang (2000), Greening *et al*. (2001), Davis *et al*. (2002), Kahn (2003), Greening (2004), Lindmark (2004), Aldy (2005), and Lenzen *et al*. (2006).

Possible causes of the decline in U.S. carbon intensity include structural changes in the economy, technological improvements in energy efficiency, behavioral changes by consumers and producers, the growth of renewable and nuclear energy, and the displacement of oil consumption by gas, or coal by oil and gas (if we produce the same amount of energy from coal, oil, and gas, then the emissions from oil are only 80% of those from coal, and from gas only 75% of those from oil) (Casler and Rose, 1998; Ang and Zhang, 2000). The last two items on this list are not dominant causes because we observe that both primary energy consumption and carbon emissions grew at close to 1% per year over the past 30 years (EIA, 2005). At least in the United States, there has been no significant decarbonization of the energy system during this period. However, all of the other items on the list play a significant role. The economy has grown at an annual rate of 2.8% over the last three decades because of 3.6% growth in the service sector; manufacturing grew at only 1.5% per year (Fig. 3-4). Because the service sector has a much lower carbon intensity than manufacturing (a factor of 6.5 in 2002; compare Figs. 3-4 and 3-5), this faster growth of services reduces the country's carbon intensity. If all of the growth in the service sector had been in manufacturing from 1971 to 2001, then the emissions would have grown at 2% per year instead of 1%. So, structural change is at least one-half of the answer. Because the service sector is likely to continue to grow more rapidly than other sectors of the economy, we expect that carbon emissions will continue to grow more slowly than GDP. This is important because it implies that emissions growth is essentially decoupled from economic growth and speaks to the issue of our technological readiness to achieve an emissions target. For example, a portfolio of technologies able to convert the 1% annual growth in emissions into a 1% annual decline, might be insufficient if carbon emissions were to begin rising at the ~3% growth rate of GDP (Pacala and Socolow, 2004).

However, note that emissions from manufacturing are approximately constant despite 1.5% economic growth, while those of services grew at 2.1% despite 3.6% economic growth (Figs. 3-3 and 3-4). The decrease in the carbon intensity within these sectors is caused both by within-sector structural shifts (i.e., from heavy to light manufacturing) and by technological improvements (See Part II of this report). Emissions from the residential sector are growing at roughly the same rate as the population (Fig. 3-4; 30-year average of 1.0% per year), while emissions from transportation are growing faster than the population but slower than GDP (Fig. 3-4; 30-year average of 1.4% per year). The difference between the 3% growth rate of GDP and the 1.6% growth in emissions from transportation is not primarily due to technological improvement because carbon emissions per mile traveled have been level or increasing over the period (Chapter 7).

Figure 3-2. The historical relationship between U.S. per capita GDP and U.S. carbon intensity (green symbols, kg CO_2 emitted per 1995 dollar of GDP) and per capita carbon emissions (blue symbols, kg CO_2 per person). Each symbol shows a different year and each of the two time series progresses roughly chronologically from left (early) to right (late) and ends in 2002. *Source*: Maddison (2003), Marland *et al*. (2005). Thus, the red square farthest to the right shows U.S. per capita CO_2 emissions in 2002. The square second farthest to the right shows per capita emissions in 2001. The third farthest to the right shows 2000 and so on. Note that per capita emissions have been roughly constant over the last 30 years (squares corresponding to per capita GDP greater than approximately $16,000).

Figure 3-3. Historical U.S. GDP divided among the manufacturing, services and agricultural sectors. *Source*: Mitchell (1998) and WRI (2005).

Figure 3-4. Historical U.S. carbon emissions divided among the residential, commercial, industrial, and transportation sectors. *Source*: EIA (2005).

Carbon Sinks (see Tables 3-1and 3-2 for citations and data)

Approximately 30% of North American fossil fuel emissions are offset by a natural sink of 592 Mt C yr^{-1} caused by a variety of factors, including forest regrowth, fire suppression, and agricultural soil conservation. The sink currently absorbs 506 Mt C yr^{-1} in the United States and 134 Mt C yr^{-1} in Canada. Mexican ecosystems create a net source of 48 Mt C yr^{-1}. Rivers and international trade also export a net of 161 Mt C yr^{-1} that was captured from the atmosphere by the continent's ecosystems, and so North America absorbs 753 Mt C yr^{-1} of atmospheric CO_2 (753 = 592 + 161). Because most of these net exports will return to the atmosphere elsewhere within 1 year (e.g. carbon in exported grain will be eaten, metabolized, and exhaled as CO_2), the net North American sink is rightly thought of as 592 Mt C yr^{-1} even though the continent absorbs a net of 753 Mt C yr^{-1}. Moreover, coastal waters may be small net emitters to the atmosphere at the continental scale (19 Mt C yr^{-1}), but this flux is highly uncertain (see Chapter 15). The portion of the coastal flux caused by human activity is thought to be close to zero, and so coastal sea-air exchanges should also be excluded from the continental carbon sink.

As reported in Chapter 2, the United States is responsible for 27% of the global carbon sink and 86% of the North American sink. The reason for the disproportionate importance of U.S. sinks is probably the unique land use history of the country (summary in Appendix 3A). During European settlement, large amounts of carbon were released from the harvest of virgin forests and the plowing of virgin soils to create agricultural lands. The abandonment of many of the formerly agricultural lands in the east and the regrowth of forest is a unique event globally and is responsible for about one-half of the U.S. sink (Houghton *et al*., 2000). Most of the U.S. sink thus represents a one-time recapture of some of the carbon that was released to the atmosphere during settlement. In contrast, Mexican ecosystems, like those of many tropical nations, are still a net carbon source because of ongoing deforestation (Masera *et al*., 1997).

Table 3-1. Annual net carbon emissions (source = positive) or uptake (land sink = negative) of carbon in millions of tons.

Table 3-2. Annual net horizontal transfers of carbon in millions of tons.

Table 3-3. Carbon stocks in North America in billions of tons.

The non-fossil fluxes in Tables 3-1 and 3-2, are derived exclusively from inventory methods in which the total amount of carbon in a pool (i.e., living forest trees plus forest soils) is measured on two occasions. The difference between the two measurements shows if the pool is gaining (sink) or losing (source) carbon. Carbon inventories are straightforward in principle, but of uneven quality in practice. For example, we know the carbon in living trees in the United States relatively accurately because the U.S. Forest Service Forest Inventory program measures trees systematically in more than 200,000 locations. However, we must extrapolate from a few measurements of forest soils with models because there is no national inventory of carbon in forest soils.

Although the fluxes in Tables 3-1 and 3-2 represent the most recent published estimates, with most less than five years old, a few are older than ten years (see the citations at the bottom of each Table). Also, the time interval between inventories varies among the elements of the Tables, with most covering a five to ten year period. We report uncertainties using six categories: ***** = 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%, * = uncertainty > 100%.

In addition to inventory methods, it is also possible to estimate carbon sources and sinks by measuring carbon dioxide in the atmosphere. For example, if air exits the border of a continent with more CO_2 than it contained when it entered, then there must be a net source of CO_2 somewhere inside the continent. We do not include estimates obtained in this way because they are still highly uncertain at continental scales. Pacala *et al*. (2001) found that atmosphere- and inventory-based methods gave consistent estimates of U.S. ecosystem sources and sinks but that the range of uncertainty from the former was considerably larger than the range from the latter. For example, by far the largest published estimate for the North American carbon sink was produced by an analysis of atmospheric data by Fan *et al*. (1998) (-1700 Mt C yr^{-1}). The appropriate inventory-based estimate to compare this to is our –753 Mt C yr^{-1} of net absorption (atmospheric estimates include net horizontal exports by rivers and trade), and this number is well within the wide uncertainty limits in Fan *et al*. (1998). The allure of estimates from atmospheric data is that they do not risk missing critical uninventoried carbon pools. But, in practice, they are still far less accurate at continental scales than a careful inventory (Pacala *et al*., 2000). Using today's technology, it should be possible to complete a comprehensive inventory of the sink at national scales, with the same accuracy as the U.S. forest inventory currently achieves for above-ground carbon in forests (25%, Smith and Heath, 2005). Moreover, this inventory would provide disaggregated information about the sink's causes and geographic distribution. In contrast, estimates from atmospheric methods rely on the accuracy of atmospheric models, and estimates obtained from different models vary by 100% or more at the scale of the United States, Canada, or Mexico (Gurney *et al*., 2004). Nonetheless, extensions of the atmospheric sampling network should improve the accuracy of atmospheric methods and might allow them to achieve the accuracy of inventories at regional and whole-country scales. In addition, atmospheric methods will continue to provide an independent check on inventories to make sure that no large flux is missed, and atmospheric methods will remain the only viable method to assess inter-annual variation the continental flux of carbon.

The magnitude of the North American sink documented in Tables 3-1 and 3-2 offers the possibility that significant carbon mitigation could be accomplished by managing forests, rangelands, and croplands to increase the carbon stored in them. However, many of the estimates in Tables 3-1 and 3-2 are highly uncertain; for some the range of uncertainty is larger than the value reported. The largest contributors to the uncertainty in the U.S. sink are the amount of carbon stored on rangelands because of the encroachment of woody vegetation and the lack of comprehensive and continuous inventory of Alaskan lands. A carbon inventory of these lands would do more to constrain the size of the U.S. sink than would any other measurement program of similar cost. Also we still lack comprehensive U.S. inventories of carbon in soils, woody debris, wetlands, rivers, and reservoirs. Finally, we lack estimates of any kind for five significant components of the carbon budget in Canada and six in Mexico (see Table 3-1 and 3-2).

The cause and future of the North American carbon sink is also highly uncertain. Although we can document the accumulation of carbon in ecosystems and wood products, we do not know how much of the sink is due to fertilization of the ecosystems by the nitrogen in air pollution and by the added CO_2 in the atmosphere, we do not fully understand the impact of tropospheric ozone, nor do we understand precisely how the sink will change as the climate changes. Research is mixed about the importance of nitrogen and CO_2 fertilization (Casperson *et al*., 2000; Oren *et al*., 2001; Hungate *et al*., 2003; Luo 2006**;** Körner *et al*., 2005). If these factors are weak, then, all else equal, we expect the North American sink to decline over time as ecosystems complete their recovery from past exploitation (Hurtt *et al*., 2002). However, if these factors are strong, then the sink could grow in the future. Similarly, global warming is expected to lengthen the growing season in most parts of North America, which should increase the sink (but see Goetz et al. 2005). But warming is also expected to increase the rate of decomposition of dead organic matter, which should decrease the sink. The relative strength of these two factors is still difficult to predict. Experimental manipulations of climate, atmospheric CO_2, tropospheric ozone, and nitrogen, at the largest possible scale, will be required to reduce uncertainty about the future of the carbon sink.

In what follows, we provide additional detail about the elements in Tables 3-1 and 3-2.

Forests

Based on U.S. Forest Service inventories, forest ecosystem carbon stocks in the United States, excluding soil carbon, have increased since 1953. The rate of increase has recently slowed because of increasing harvest and declining growth in some areas with maturing forests. The current average annual increase in carbon in trees is 146 Mt C yr^{-1} (Smith and Heath, 2005, uncertainty ****) plus 23 Mt C yr^{-1} from urban and suburban trees (the midpoint of the range in Chapter 14, uncertainty ***). The total estimate of the carbon sink in forested ecosystems is –259 Mt C yr^{-1} and includes a sink of 90 Mt C yr^{-1} (uncertainty **) from the accumulation of nonliving carbon in the soil (-90-146-23 = –259) (Pacala *et al*., 2001; Goodale *et al*., 2002). Although the magnitude of the forest soil sink has always been uncertain, it is now possible to measure the total above-and below-ground sink in a few square kilometers by monitoring the atmospheric carbon dioxide that flows into and out of the site over the course of a year. Note that these spatially intensive methods appropriate for monitoring the sink over a few square kilometers are unrelated to the spatially extensive methods described above, which attempt to constrain the sink at continental scales. As described in Appendix 3B, these studies are producing data that so far confirm the estimates of inventories and show that most of the forest sink is above ground.

According to Canada's Greenhouse Gas Inventory (Environment Canada, 2005), managed forests in Canada (comprising 53% of the total forest area) sequestered 101 Mt C aboveground in 1990 (uncertainty ***). Since then, carbon sequestration has decreased gradually to 69 Mt C in 2003, as managed forests have recovered from past disturbances (Kurz and Apps, 1999, uncertainty ***). In addition, Goodale *et al*. (2002) estimate the sink of nonliving carbon belowground to be –30 Mt C yr^{-1} for the period 1990–1994 (uncertainty **).

The two published carbon inventories for Mexican forests (Masera *et al*., 1997 and Cairns *et al*., 2000) both report substantial losses of forest carbon, primarily because of deforestation in the tropical south. However, both of these studies rely on calculations of carbon loss from remote imagery, rather than direct measurements, and both report results for a period that ended more than 10 years ago. Thus, in addition to being highly uncertain, the estimates for Mexican forests in Table 3-1 are not recent.

Wood Products

Wood products create a carbon sink because they accumulate both in use (e.g., furniture, house frames, etc.) and in landfills. The wood products sink is estimated at –57 Mt C yr^{-1} in the United States (Skog and Nicholson, 1998) and –10 Mt C yr^{-1} in Canada (Goodale *et al*., 2002). We know of no estimates for Mexico.

Woody Encroachment

Woody encroachment is the invasion of woody plants into grasslands or the invasion of trees into shrublands. It is caused by a combination of fire suppression and grazing. Fire inside the United States has been reduced by more than 95% from the pre-settlement level of approximately 80 million hectares burned per year, and this favors shrubs and trees in competition with grasses (Houghton *et al*., 2000). Field studies show that woody encroachment both increases the amount of living plant carbon and decreases the amount of dead carbon in the soil (Guo and Gifford, 2002; Jackson *et al*., 2002). Although the gains and losses are of similar magnitude (Jackson *et al*., 2002), the losses occur within approximately a decade after the woody plants invade (Guo and Gifford, 2002), while the gains occur over a period of up to a century or more. Thus, the net source or sink depends on the distribution of times since woody plants invaded, and this is not known. Estimates for the size of the current U.S. woody encroachment sink (Kulshreshtha *et al*., 2000; Houghton and Hackler, 1999; and Hurtt *et al*., 2002) all rely on methods that do not account for the initial rapid loss of carbon from soil when grasslands were converted to shrublands or forest. The estimate of –120 Mt C yr^{-1} in Table 3-1 is from Kulshreshtha *et al*. (2000) but is similar to the estimates from the other two studies (–120 and –130 Mt C yr^{-1}). No estimates are currently available for Canada or Mexico. Note the error estimate of more than 100% in Table 3-1. A comprehensive set of measurements of woody encroachment would reduce the error in the national and continental carbon budgets more than any other inventory.

Agricultural Lands

Soils in croplands and grazing lands have been historically depleted of carbon by humans and their animals, especially if the land was converted from forest to non-forest use. Harvest or consumption by animals reduces the input of organic matter to the soil, while tillage and manure inputs increase the rate of decomposition. Changes in cropland management, such as the adoption of no-till agriculture (see Chapter 10), have reversed the losses of carbon on some croplands, but the losses continue on the remaining lands. The net is an approximate carbon balance for agricultural soils in Canada and estimates for the United States ranging from a small source of 2Mt C yr^{-1} to small sink of -6 Mt C yr^{-1}.

Wetlands

Peatlands are wetlands that have accumulated deep soil carbon deposits because plant productivity has exceeded decomposition over thousands of years. Thus, wetlands form the largest carbon pool of any North American ecosystem (Table 3-3). If drained for development, this soil carbon pool is rapidly lost. Canada's extensive frozen and unfrozen wetlands create a net sink of between –19 and –20 Mt C yr^{-1} (see Chapters 12 and 13), but drainage of U.S. peatlands have created a net source of 5 Mt C yr^{-1}. The very large pool of peat in northern wetlands is vulnerable to climate change and could add more than 100 ppm to the atmosphere (1 ppm ≈ 2.1 Gt C) during this century if released because of global warming (see the model result in Cox *et al*., 2000 for an example).

The carbon sink due to sedimentation in wetlands is between 0 and –21 Mt C yr^{-1} in Canada and between 0 and –112 Mt C yr^{-1} in the United States (see Chapter 13). Another important priority for research is to better constrain carbon sequestration due to sedimentation in wetlands, lakes, reservoirs, and rivers.

The focus on this chapter is on carbon dioxide; we do not include estimates for other greenhouse gases. However, wetlands are naturally an important source of methane (CH_4). Methane emissions effectively cancel out the positive benefits of any carbon storage as peat in Canada and make U.S. wetlands a source of warming on a decadal time scale (Chapter 13). Moreover, if wetlands become warmer and remain wet with future climate change, they have the potential to emit large amounts of methane. This is probably the single most important consideration, and unknown, in the role of wetlands and future climate change.

Rivers and Reservoirs

Organic sediments accumulate in artificial lakes and in alluvium (deposited by streams and rivers), and colluvium (deposited by wind or gravity) and represent a carbon sink. Pacala *et al*. (2001) extended an analysis of reservoir sedimentation (Stallard, 1998) to an inventory of the 68,000 reservoirs in the United States and also estimated net carbon burial in alluvium and colluvium. Table 3-1 includes the midpoint of their estimated range of 10 to 40 Mt C yr^{-1} in the coterminous United States. This analysis has also recently been repeated and produced an estimate of 17 Mt C yr^{-1} (E. Sundquist, personal communication). We know of no similar analysis for Canada or Mexico.

Exports Minus Imports of Wood and Agricultural Products

The United States imports 14 Mt C yr^{-1} more wood products than it exports and exports 30–50 Mt C yr^{-1} more agricultural products than it imports (Pacala *et al*., 2001). The large imbalance in agricultural products is primarily because of exported grains and oil seeds. Canada and Mexico are net wood exporters, with Canada at –74 Mt C yr^{-1} (Environment Canada, 2005) and Mexico at –1 Mt C yr^{-1} (Masera *et al*., 1997). We know of no analysis of the Canadian or Mexican export-import balance for agricultural products.

River Export

Rivers in the coterminous United States were estimated to export 30–40 Mt C yr^{-1} to the oceans in the form of dissolved and particulate organic carbon and inorganic carbon derived from the atmosphere (Pacala *et al*., 2001). An additional 12–20 Mt C yr^{-1} of inorganic carbon is also exported by rivers but is derived from carbonate minerals. We know of no corresponding estimates for Alaska, Canada, or Mexico.

Coastal Waters

Chapter 15 summarizes the complexity and large uncertainty of the sea-air flux of CO_2 in North American coastal waters. It is important to understand that the source in Mexican coastal waters is not caused by humans and would have been present in pre-industrial times. It is simply the result of the purely physical upwelling of carbon-rich deep waters and is a natural part of the oceanic carbon cycle. It is not yet known how much of the absorption of carbon by U.S. and Canadian coastal waters is natural and how much is caused by nutrient additions to the coastal zone by humans. Accordingly, it is essentially impossible to currently assess the potential or costs for carbon management in coastal waters of North America.

CONCLUDING SUMMARY

Fossil fuel emissions currently dominate the net carbon balance in the United States, Canada, and Mexico (Fig. 3-1, Tables 3-1, 3-2). U.S. fossil fuel consumption currently emits 1582 Mt C yr^{-1} to the atmosphere. This is partially balanced by a flow of 506 Mt C yr^{-1} from the atmosphere to land caused by net ecosystem sinks in the United States. Canadian fossil consumption transfers 164 Mt C yr^{-1} to the atmosphere, but net ecological sinks capture 134 Mt C yr^{-1}. Mexican fossil emissions of 110 Mt C yr^{-1} are supplemented by a net ecosystem source of 48 Mt C yr^{-1} from tropical deforestation. Each of the three countries has always been a net source of carbon dioxide emissions to the atmosphere for the past three centuries (Houghton *et al*., 1999, 2000; Houghton and Hackler, 2000; Hurtt *et al*., 2002).

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Table 3-1. Annual net emissions (source = positive) or uptake (land sink = negative) of carbon in millions of tons

Source (positive) or Sink (negative)	United States	Canada	Mexico	North America
Fossil source (positive)				
Fossil fuel [a] (oil, gas, coal)	1582[****] (681, 328, 573)	164[****] (75, 48, 40)	110[****] (71, 29, 11)	1857[****] (828, 405, 624)
Nonfossil carbon sink (negative) or source (positive)				
Forest	–259[b,***]	–99[c,***]	+52[d,**]	–306[***]
Wood products	–57[e,***]	–10 [f,***]	ND	–67,***
Woody encroachment	–120[g,*]	ND	ND	–120[*]
Agricultural soils	–4[h,*]	–0[h]	–0[h]	–4[*]
Wetlands	–41[i,*]	–25[i,*]	–4[i,*]	–70[*]
Rivers and reservoirs	–25 [j,**]	ND	ND	–25[*]
Total carbon source or sink	–506[***]	–134[**]	48[*]	–592[***]

Uncertainty:

*****(95% confidence within 10%)
****(95% confidence within 25%)
***(95% confidence within 50%)
**(95% confidence within 100%)
*(95% confidence bounds >100%)
ND = No data available
[a]http://www.eia.doe.gov/env/inlenv.htm
[b]Smith and Heath (2005) for above ground carbon, but including 23 Mt C/yr^{-1} for U.S. urban and suburban forests from Chapter 14, and Pacala *et al*. (2001) for below ground carbon.
[c]Environment Canada (2005)
[d]Masera *et al.* (1997)
[e]Skog *et al*. (2004), Skog and Nicholson (1998)
[f]Goodale *et al.* (2002)
[g]Kulshreshtha *et al*. (2000), Hurtt *et al*. (2002), Houghton and Hackler (1999).
[h]Chapter 10; Highly uncertain; Could range from -5 Mt C yr^{-1} to 5 Mt C yr^{-1}.
[i]Chapter 13
[j]Stallard, 1998; Pacala *et al*. (2001)

Table 3-2. Annual net horizontal transfers of carbon in millions of tons.

Net horizontal transfer: imports exceed exports = positive; exports exceed imports = negative	United States	Canada	Mexico	North America
Wood products	14[c,****]	–74[a,****]	–1[b,*]	–61[****]
Agriculture products	–65[d,***]	ND	ND	–65[***]
Rivers to ocean	–35[d,**]	ND	ND	–35[*]
Total net absorption (Total carbon source or sink in Table 3-1 plus exports)	–592[***]	–208[**]	47[*]	–753[**]
Net absorption (negative) or emission (positive) by coastal waters	ND	ND	ND	19[e,*]

Uncertainty:

*****(95% confidence within 10%)
****(95% confidence within 25%)
***(95% confidence within 50%)
**(95% confidence within 100%)
*(95% confidence bounds >100%)
ND = No data available
[a]Environment Canada (2005)
[b]Masera *et al.* (1997)
[c]Skog *et al*. (2004), Skog and Nicholson (1998)
[d]Pacala et al. (2001)
[e]Chapter 15

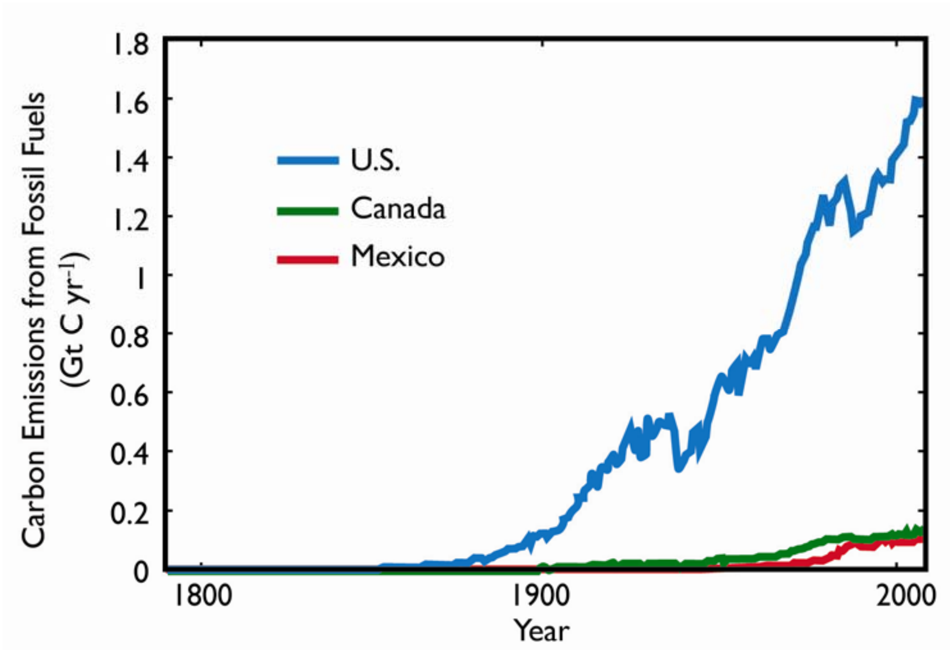
Table 3-3. Carbon stocks in North America in billions of tons

	United States	Canada	Mexico	North America
Forest	53[a,***]	85[a,***]	9[d,**]	147[***]
Cropland	14[b,****]	4[b,****]	1[b,**]	19[****]
Pasture	33[b,***]	12[b,***]	10[b,***]	55[***]
Wetlands	42[c,***]	152[c,***]	2[c,*]	196[***]
Total	142[***]	253[***]	22[**]	417[***]

Uncertainty:

*****(95% confidence within 10%)
****(95% confidence within 25%)
***(95% confidence within 50%)
**(95% confidence within 100%)
*(95% confidence bounds >100%)

[a]Goodale *et al*. (2002)
[b]Chapter 10
[c]Chapter 13
[d]Masera *et al*. (1997)



2 **Fig. 3-1. Historical carbon emissions from fossil fuel in the United States, Canada, and Mexico.** Data from
3 the U.S. Energy Information Administration (EIA 2005).

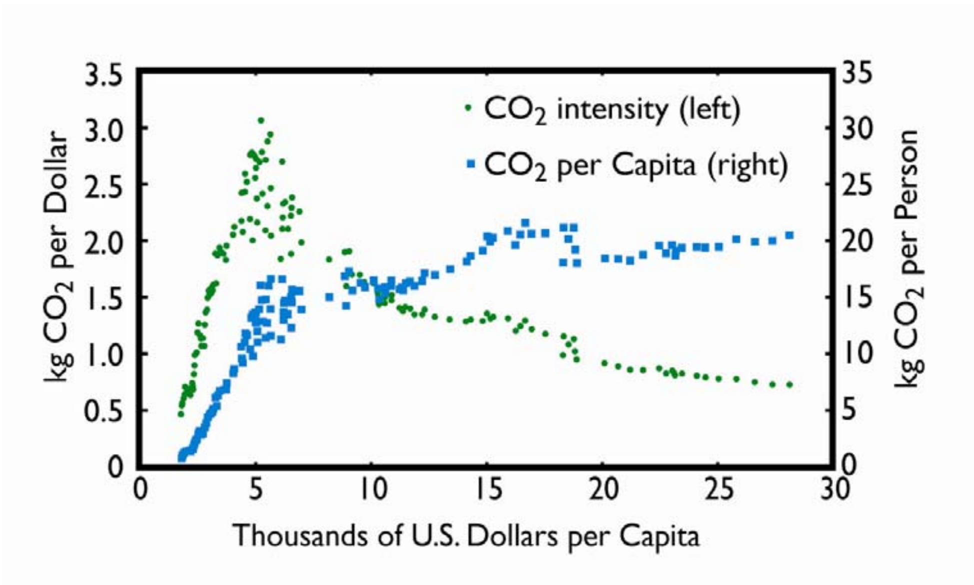


Fig. 3-2. The historical relationship between U.S. per capita GDP and U.S. carbon intensity (green symbols, kg CO_2 emitted per 1995 dollar of GDP) and per capita carbon emissions (blue symbols, kg CO_2 per person). Each symbol shows a different year and each of the two time series progresses roughly chronologically from left (early) to right (late) and ends in 2002. *Source*: Maddison (2003), Marland *et al*. (2005). Thus, the red square farthest to the right shows U.S. per capita CO_2 emissions in 2002. The square second farthest to the right shows per capita emissions in 2001. The third farthest to the right shows 2000, and so on. Note that per capita emissions have been roughly constant over the last 30 years (squares corresponding to per capita GDP greater than approximately $16,000).

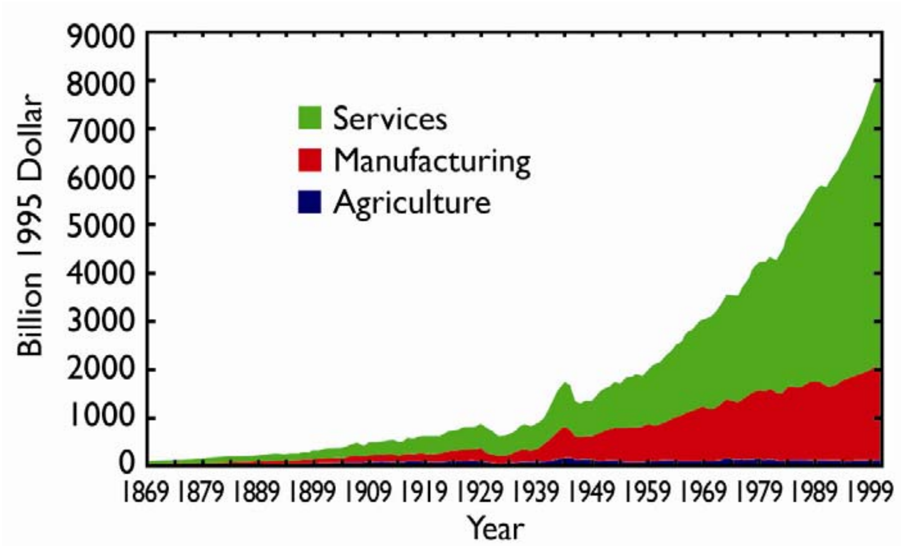


Figure 3-3. Historical U.S. GDP divided among the manufacturing, services, and agricultural sectors.

Source: Mitchell (1998), WRI (2005).

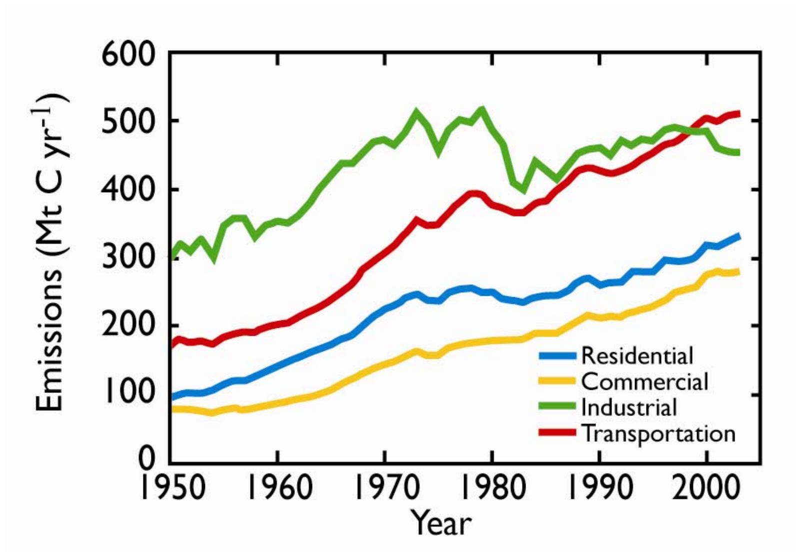


Figure 3-4. Historical U.S. carbon emissions divided among the residential, services, manufacturing, and transportation sectors. *Source*: EIA (2005).

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Appendix 3A

Historical Overview of the Development of U.S., Canadian, and Mexican Ecosystem Sources and Sinks for Atmospheric Carbon

Although the lands of the New World were inhabited before the arrival of Europeans, the changes since arrival have been enormous, especially during the last two centuries. Peak U.S. emissions from land-use change occurred late in the 19th century, and the last few decades have experienced a carbon sink (Houghton *et al*., 1999; Hurtt *et al*., 2002). In Canada, peak emissions occurred nearly a century later than in the United States, and current data show that land-use change causes a net carbon sink (Environment Canada, 2005). In Mexico, the emissions of carbon continue to increase from net deforestation. All three countries may be in different stages of the same development pattern (see Fig. 3-2).

The largest changes in land use and the largest emissions of carbon came from the expansion of croplands. In addition to the carbon lost from trees, soils lose 25–30% of their initial carbon content (to a depth of 1 m) when cultivated. In the United States, croplands increased from about 0.25 million ha in 1700 to 236 million ha in 1990 (Houghton *et al*., 1999; Houghton and Hackler, 2000). The most rapid expansion (and the largest emissions) occurred between 1800 and 1900, and since 1920 there has been little net change in cropland area. Pastures expanded nearly as much, from 0.01 million to 231 million ha, most of the increase taking place between 1850 and 1950. As most pastures were derived from grasslands, the associated changes in carbon stocks were modest.

The total area of forests and woodlands in the United States declined as a result of agricultural expansion by 160 million ha (38%), but this net change obscures the dynamics of forest loss and recovery, especially in the eastern part of the United States. After 1920, forest areas increased by 14 million ha nationwide as farmlands continued to be abandoned in the northeast, southeast, and north central regions. Nevertheless, another 4 million ha of forest were lost in other regions, and the net recovery of 10 million ha offset only 6% of the net loss (Houghton and Hackler, 2000).

Between 1938 and 2002, the total area of forest land in the conterminous United States decreased slightly, by 3 million ha (Smith *et al*., 2004). This small change is the net result of much larger shifts among land-use classes (Birdsey and Lewis, 2003). Gains of forest land, primarily from cropland and pasture, were about 50 million ha for this period. Losses of forest land to cropland, pasture, and developed use were about 53 million ha for the same period. Gains of forest land were primarily in the

Eastern United States, whereas losses to cropland and pasture were predominantly in the South, and losses to developed use were spread around all regions of the United States.

In the United States, harvest of industrial wood (timber) generally followed the periods of major agricultural clearing in each region. In the last few decades, total volume harvested increased until a recent leveling took place (Smith *et al*., 2004). The volume harvested in the Pacific Coast and Rocky Mountain regions has declined sharply, whereas harvest in the South increased and in the North, stayed level. Fuel wood harvest peaked between 1860 and 1880, after which fossil fuels became the dominant type of fuel (Houghton and Hackler, 2000).

The arrival of Europeans reduced the area annually burned, but a federal program of fire protection was not established until early in the 20th century. Fire exclusion had begun earlier in California and in parts of the central, mountain and Pacific regions. However, neither the extent nor the timing of early fire exclusion is well known. After about 1920, the Cooperative Fire Protection Program gradually reduced the areas annually burned by wildfires (Houghton *et al*., 1999, 2000). The reduction in wildfires led to an increase in carbon storage in forests. How long this “recovery” will last is unclear. There is some evidence that fires are becoming more widespread, again, especially in Canada and the western United States. Fire exclusion and suppression are also thought to have led to woody encroachment, especially in the southwestern and western United States. The extent and rate of this process is poorly documented, however, and estimates of a carbon sink are very uncertain. Gains in carbon aboveground may be offset by losses belowground in some systems, and the spread of exotic annual grasses into semiarid deserts and shrublands may be converting the recent sink to a source (Bradley *et al*., in preparation).

The consequence of this land-use history is that U.S. forests, at present, are recovering from agricultural abandonment, fire suppression, and reduced logging (in some regions), and, as a result, are accumulating carbon (Birdsey and Heath, 1995; Houghton *et al*., 1999; Caspersen *et al*., 2000; Pacala *et al*., 2001). The magnitude of the sink is uncertain, and whether any of it has been enhanced by environmental change (CO_2 fertilization, nitrogen deposition, and changes in climate) is unclear. Understanding the mechanisms responsible for the current sink is important for predicting its future behavior (Hurtt *et al*., 2002).

In the mid-1980s, Mexico lost approximately 668,000 ha of closed forests annually, about 75% of them tropical forests (Masera *et al*., 1997). Most deforestation was for pastures. Another 136,000 ha of forest suffered major perturbations, and the net flux of carbon from deforestation, logging, fires, degradation, and the establishment of plantations was 52.3 Mt C yr^{-1}, about 40% of the country’s estimated annual emissions of carbon. A later study found the deforestation rate for tropical Mexico to be about 12% higher (1.9% per year) (Cairns *et al*., 2000).

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Appendix 3B

Eddy-Covariance Measurements Now Confirm Estimates of Carbon Sinks from Forest Inventories

Long-term, tower-based, eddy-covariance measurements (e.g., Wofsy *et al*., 1993) represent an independent approach to measuring ecosystem-atmosphere CO_2 exchange. The method describes fluxes over areas of approximately 1 km^2 (Horst and Weil, 1994), measures hour-by-hour ecosystem carbon fluxes, and can be integrated over time scales of years. A network of more than 200 sites now exists globally (Baldocchi *et al*., 2001); more than 50 of these are in North America. None of these sites existed in 1990, so these represent a relatively new source of information about the terrestrial carbon cycle. An increasing number of these measurement sites include concurrent carbon inventory measurements.

Where eddy-covariance and inventory measurements are concurrent, the rates of accumulation or loss of biomass are often consistent to within several tens of g C m^{-2} yr^{-1} for a one-year sample (10 g C yr^{-1} is 5% of a typical net sink of 2 metric tons of carbon per hectare per year for an Eastern deciduous successional forest) . Published intercomparisons in North America exist for western coniferous forests (Law *et al*., 2001), agricultural sites (Verma *et al*., 2005), and eastern deciduous forests (Barford *et al*., 2001; Cook *et al*., 2004; Curtis *et al*., 2002; Ehmann *et al*., 2002; Gough *et al*., in review). Multiyear studies at two sites (Barford *et al*., 2001; Gough *et al*., in review) show that 5- to 10-year averages converge toward inventory measurements. Table 3B-1 from Barford *et al*. (2001) shows the results of nearly a decade of concurrent measurements in an eastern deciduous forest.

This concurrence between eddy-covariance flux measurements and ecosystem carbon inventories is relevant because it provides independent validation of the inventory measurements used to estimate long-term trends in carbon stocks. The eddy-covariance data are also valuable because the assembly of global eddy-covariance data provides independent support for net storage of carbon by many terrestrial ecosystems and the substantial year-to-year variability in this net sink. The existence of the eddy-covariance data also makes the sites suitable for co-locating mechanistic studies of inter-annual and shorter, time-scale processes governing the terrestrial carbon cycle. Chronosequences show trends consistent with inventory assessments of forest growth, and comparisons across space and plant functional types are beginning to show broad consistency. These results show a consistency across a mixture of observational methods with complementary characteristics, which should facilitate the development of an increasingly complete understanding of continental carbon dynamics (Canadell *et al*., 2000).

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Table 3B-1. Carbon budget for Harvard Forest from forest inventory and eddy-covariance flux measurements, 1993–2001. *Source*: Barford *et al.* (2001), Table 1. Numbers in parentheses give the ranges of the 95% confidence intervals.

Component	Change in carbon stock or flux (g C m^{-2} yr^{-1})	Totals
Change in live biomass		
A. Aboveground		
1. Growth	1.4 (±0.2)	
2. Mortality	–0.6 (±0.6)	
B. Belowground (estimated)		
1. Growth	0.3	
2. Mortality	–0.1	
Subtotal		1.0 (±0.2)
Change in dead wood		
A. Mortality		
1. Aboveground	0.6 (±0.6)	
2. Belowground	0.1	
B. Respiration	–0.3 (±0.3)	
Subtotal		0.4 (±0.3)
Change in soil carbon (net)		0.2 (±0.1)
Sum of carbon budget figures		1.6 (±0.4)
Sum of eddy-covariance flux measurements		2.0 (±0.4)

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Chapter 4. What Are the Options and Measures That Could Significantly Affect the Carbon Cycle?

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KEY FINDINGS

- Options to reduce energy-related CO_2 emissions include improved efficiency, fuel switching (among fossil fuels and non-carbon fuels), and CO_2 capture and storage.
- Most energy use, and hence energy-related CO_2 emissions, involves equipment or facilities with a relatively long life—5 to 50 years. Many options for reducing these CO_2 emissions are most cost-effective, and sometimes only feasible, in new equipment or facilities. This means that cost-effective reduction of energy-related CO_2 emissions may best be achieved as existing equipment and facilities are replaced. It also means that technological change will have a significant impact on the cost because emission reductions will be implemented over a long time.
- Options to increase carbon sinks include forest growth and agricultural soil sequestration. The amount of carbon that can be captured by these options is significant, but small relative to the excess carbon in the atmosphere. These options can be implemented in the short-term, but the amount of carbon sequestered typically is low initially then rising for a number of years before tapering off again as the total potential is achieved. There is also a significant risk that the carbon sequestered may be released again by natural phenomena or human activities.
- A number of policy options can help reduce carbon emissions and increase carbon sinks. The effectiveness of a policy depends on the technical feasibility and cost-effectiveness of the portfolio of measures it seeks to promote, on its suitability given the institutional context, and on its interaction with policies implemented to achieve other objectives.
- Policies to reduce atmospheric CO_2 concentrations cost effectively in the short- and long-term would: (1) encourage adoption of cost-effective emission reduction and sink enhancement measures through an emissions trading program or an emissions tax; (2) stimulate development of technologies that

lower the cost of emissions reduction, geological storage and sink enhancement; (3) adopt appropriate regulations to complement the emissions trading program or emission tax for sources or actions subject to market imperfections, such as energy efficiency measures and co-generation; (4) Revise existing policies with other objectives that lead to higher CO_2 or CH_4 emissions so that the objectives, if still relevant, are achieved with lower emissions.

- Implementation of such policies is best achieved by national governments with international cooperation. This provides maximum coverage of CO_2 emissions and carbon sinks and so enables implementation of the most cost-effective options. It also allows better allocation of resources for technology research and development. National policies may need to be coordinated with state/provincial governments, or state/provincial governments may implement coordinated policies without the national government.

INTRODUCTION

This chapter provides an overview of measures that can reduce carbon dioxide (CO_2) and methane (CH_4) emissions and those that can enhance carbon sinks, and it attempts to compare them. Finally, it discusses policies to encourage implementation of source reduction and sink enhancement measures.

SOURCE REDUCTION OPTIONS

Energy-Related CO_2 Emissions

Combustion of fossil fuels is the main source of CO_2 emissions, although some CO_2 is also released in non-combustion and natural processes. Most energy use, and hence energy-related CO_2 emissions, involves equipment or facilities with a relatively long life—5 to 50 years. Many options for reducing these CO_2 emissions are most cost-effective, and sometimes only feasible, in new equipment or facilities (Chapters 6 through 9).

To stabilize the atmospheric concentration of CO_2 “would require global anthropogenic CO_2 emissions to drop below 1990 levels . . . and to steadily decrease thereafter” (IPCC, 2001a).[1] That entails a transition to an energy system where the major energy carriers are electricity and hydrogen produced by non-fossil sources or from fossil fuels with capture and geological storage of the CO_2 generated. The transition to such an energy system, while meeting growing energy needs, will take at least several decades. Thus, shorter term (2015–2025) and longer term (post-2050) options are differentiated.

[1] The later the date at which global anthropogenic CO_2 emissions drop below 1990 levels, the higher the level at which the CO_2 concentration is stabilized.

Options to reduce energy-related CO_2 emissions can be grouped into a few categories:

- efficiency improvement,
- fuel switching to fossil fuels with lower carbon content per unit of energy produced and to non-carbon fuels, and
- switching to electricity and hydrogen produced from fossil fuels in processes with CO_2 capture and geological storage.

Efficiency Improvement

Energy is used to provide services such as heat, light, and motive power. Any measure that delivers the desired service with less energy is an efficiency improvement.[2] Efficiency improvements reduce CO_2 emissions whenever they reduce the use of fossil fuels at any point between production of the fuel and delivery of the desired service.[3] Energy use can be reduced by improving the efficiency of individual devices (such as refrigerators, industrial boilers, and motors), by improving the efficiency of systems (using the correct motor size for the task), and by using energy that is not currently utilized, such as waste heat.[4] Opportunities for efficiency improvements are available in all sectors.

It is useful to distinguish two levels of energy efficiency improvement: (1) the amount consistent with efficient utilization of resources (the economic definition) and (2) the maximum attainable (the engineering definition). Energy efficiency improvement thus covers a broad range, from measures that provide a cost saving to measures that are too expensive to warrant implementation. Market imperfections inhibit adoption of some cost-effective efficiency improvements (NCEP, 2005).[5]

Energy efficiency improvements tend to occur gradually, but steadily, across the economy in response to technological developments, replacement of equipment and buildings, changes in energy prices, and other factors.[6] In the short term, the potential improvement depends largely on greater deployment and use of available efficient equipment and technology. In the long term, it depends largely on technological developments.

[2] In the transportation sector, for example, energy efficiency can be increased by improving the fuel performance of vehicles, shifting to less emissions-intensive modes of transport, and adopting measures that reduce transportation demand, such as telecommuting and designing communities so that people live closer to shopping and places of work.

[3] Increasing the fuel economy of vehicles or the efficiency of coal-fired generating units reduces fossil fuel use directly. Increasing the efficiency of refrigerators or electricity transmission reduces electricity use and hence the fossil fuel used to generate electricity.

[4] For example, 40 to 70% of the energy in the fuel used to generate electricity is wasted. Cogeneration or combined heat and power systems generate electricity and produce steam or hot water. Cogeneration requires a nearby customer for the steam or heat.

[5] Examples include limited foresight, externalities, capital market barriers, and principal/agent split incentive problems.

[6] The rate of efficiency improvement varies widely across different types of equipment such as lighting, refrigerators, electric motors, and motor vehicles.

Fuel Switching

Energy-related CO_2 emissions are primarily due to combustion of fossil fuels. Thus, CO_2 emissions can be reduced by switching to a less carbon-intensive fossil fuel or to a non-carbon fuel.

The CO_2 emissions per unit of energy for fossil fuels (carbon intensity) differ significantly, with coal being the highest, oil and related petroleum products about 25% lower, and natural gas over 40% lower than coal. Oil and/or natural gas can be substituted for coal in all energy uses, mainly electricity generation. However, natural gas is not available everywhere in North America and is much less abundant than coal, limiting the large-scale long-term replacement of coal with natural gas. Technically, natural gas can replace oil in all energy uses but to substitute for gasoline and diesel fuel, by far the largest uses of oil would require conversion of millions of vehicles and development of a refueling infrastructure.

Non-carbon fuels include

- biomass and fuels, such as ethanol and biodiesel, produced from biomass; and
- electricity and hydrogen produced from carbon-free sources.

Biomass can be used directly as a fuel in some situations. Pulp and paper plants and sawmills, for example, can use wood waste and sawdust as fuel. Ethanol, currently produced mainly from corn, is blended with gasoline and biodiesel is produced from vegetable oils and animal fats. Wood residuals and cellulose materials, such as switch grass, can be utilized both for energy and the production of syngases, which can be used to produce biopetroleum (AF&PA, 2006). The CO_2 emission reduction achieved depends on whether the biomass used is replaced, on the emissions associated with production of the biomass fuel, and the carbon content of the fuel displaced.[7]

Carbon-free energy sources include hydro, wind, solar, biomass, geothermal, and nuclear fission.[8] Sometimes they are used to provide energy services directly, such as solar water heating and wind mills for pumping water. But they are mainly used to generate electricity, about 35% of the electricity in North America. Currently, generating electricity using any of the carbon free energy sources is usually more costly than using fossil fuels.

Most of the fuel switching options are currently available, and so are viable short-term options in many situations.

[7]The CO_2 reductions achieved depend on many factors including the inputs used to produce the biomass (fertilizer, irrigation water), whether the land is existing cropland or converted from forests or grasslands, and the management practices used (no-till, conventional till).

Electricity and Hydrogen from Fossil Fuels with CO_2 Capture and Geological Storage

About 65% of the electricity in North America is generated from fossil fuels, mainly coal but with a rising share for natural gas (EIA, 2003). The CO_2 emissions from fossil-fired generating units can be captured and injected into a suitable geological formation for long-term storage.

Hydrogen (H_2) is an energy carrier that emits no CO_2 when burned, but may give rise to CO_2 emissions when it is produced (National Academies, 2004). Currently, most hydrogen is produced from fossil fuels in a process that generates CO_2. The CO_2 from this process can be captured and stored in geological formations. Alternatively, hydrogen can be produced from water using electricity, in which case the CO_2 emissions depend on how the electricity is generated. Hydrogen could substitute for natural gas in most energy uses and be used by fuel cell vehicles.

Carbon dioxide can be captured from the emissions of large sources, such as power plants, and pumped into geologic formations for long-term storage, thus permitting continued use of fossil fuels while avoiding CO_2 emissions to the atmosphere.[9] Many variations on this basic theme have been proposed; for example, pre-combustion vs. post-combustion capture, production of hydrogen from fossil fuels, and the use of different chemical approaches and potential storage reservoirs. While most of the basic technology exists, much work remains too safely and cost effectively integrates CO_2 capture and storage into our energy system, so this is mainly a long-term option (IPCC, 2005).

Industrial Processes

The processes used to make cement, lime, and ammonia release CO_2. Because the quantity of CO_2 released is determined by chemical reactions, the process emissions are determined by the output. But, the CO_2 could be captured and stored in geological formations. CO_2 also is released when iron ore and coke are heated in a blast furnace to produce molten iron, but alternative steel-making technologies with lower CO_2 emissions are commercially available. Consumption of the carbon anodes during aluminum smelting leads to CO_2 emissions, but good management practices can reduce the emissions. Raw natural gas contains CO_2 that is removed at gas processing plants and could be captured and stored in geological formations.

Methane Emissions

Methane (CH_4) is produced as organic matter decomposes in low-oxygen conditions and is emitted by landfills, wastewater treatment plants, and livestock manure. In many cases, the methane can be collected

[8]Reservoirs for hydroelectric generation produce CO_2 and methane emissions, and production of fuel for nuclear reactors generates CO_2 emissions, so such sources are not totally carbon free.

and used as an energy source. Methane emissions also occur during production of coal, oil, and natural gas. Such emissions usually can be flared or collected for use as an energy source.[10] Ruminant animals produce CH_4 while digesting their food. Emissions by ruminant farm animals can be reduced by measures that improve animal productivity. All of these emission reductions are currently available.

TERRESTRIAL SEQUESTRATION OPTIONS

Trees and other plants sequester carbon as biological growth captures carbon from the atmosphere and sequesters it in the plant cells (IPCC, 2000b). Currently, very large volumes of carbon are sequestered in the plant cells of the earth's forests. Increasing the stock of forest through afforestation[11], reforestation, or forest management draws carbon from the atmosphere and increases the carbon sequestered in the forest and the soil of the forested area. Sequestered carbon is released by fire, insects, disease, decay, wood harvesting, conversion of land from its natural state, and disturbance of the soil.

Agricultural practices can increase the carbon sequestered by the soil. Some crops build soil organic matter, which is largely carbon, better than others. Some research shows that crop-fallow systems result in lower soil carbon content than continuous cropping systems. No-till and low-till cultivation builds soil organic matter.

Conversion of agricultural land to forestry can increase carbon sequestration in soil and tree biomass, but the rate of sequestration depends on environmental factors (such as type of trees planted, soil type, climate, and topography) and management practices (such as thinning, fertilization, and pest control). Conversion of agricultural land to other uses can result in positive or negative net carbon emissions depending upon the land use.

Although forest growth and soil sequestration cannot capture all of the excess carbon in the atmosphere, they do have the potential to capture a significant portion.[12] These options can be implemented in the short-term, but the amount of carbon sequestered typically is low initially then rising for a number of years before tapering off again as the total potential is achieved.

[9]Since combustion of biomass releases carbon previously removed from the atmosphere, capture and storage of these emissions results in negative emissions.

[10]Flaring or combustion of methane as an energy source produces CO_2 emissions.

[11]Afforestation is the establishment of forest on land that has been unforested for a long time.

[12]The IPCC (2001b) estimated that biological growth including soils has the potential of capturing up to 20% of the globe's releases of excess atmospheric carbon over the next 50 years (Chapter 4). Nabuurs *et al.* (2000) estimate potential annual forest sequestration in the United States at 6% to 11% of 1990 emissions and 125% to 185% of 1990 emissions for Canada. For the two countries together, the figure is 17% to 27%.

INTEGRATED COMPARISON OF OPTIONS

As is clear from the previous sections, there are many options to reduce emissions of or to sequester CO_2. To help them decide which options to implement, policy makers need to know the magnitude of the potential emission reduction at various costs for each option so they can select the options that are the most cost-effective—have the lowest cost per metric ton of CO_2 reduced or sequestered.

This involves an integrated comparison of options, which can be surprisingly complex in practice. It is most useful and accurate for short-term options where the cost and performance of the option can be forecast with a high degree of confidence. The performance of many options is interrelated; for example, the emission reductions that can be achieved by blending ethanol in gasoline depend, in addition to the factors previously cited, on other measures, such as telecommuting to reduce travel demand, the success of modal shift initiatives, and the efficiency of motor vehicles. The prices of fossil fuels affect the cost-effectiveness of many options. Finally, the policy selected to implement an option, incentives vs. a regulation for example, can affect its potential.

The emission reduction potential and cost-effectiveness of options also vary by location. Energy sources and sequestration options differ by location; for example, natural gas may not be available, the wind and solar regime vary, hydro potential may be small or large, land suitable for afforestation/reforestation is limited, the agricultural crops may or may not be well suited to low-till cropping. Climate, lifestyles, and consumption patterns also affect the potential of many options; for example, more potential for heating options in a cold climate, more for air conditioning options in a hot climate. The mix of single-family and multi-residential buildings affects the potential for options focused on those building types, and the scope for public transit options tends to increase with city size. Institutional factors affect the potential of many options as well; for example, the prevalence of rented housing affects the potential to implement residential emission reduction measures, the authority to specify minimum efficiency standards for vehicles, appliances, and equipment may rest with the state/provincial government or the national government, and the ownership and regulatory structure for gas and electric utilities can affect their willingness to offer energy efficiency programs.

TEXT BOX on “Emission Reduction Supply Curve” goes here

The estimated cost and emission reduction potential for the principal short-term CO_2 emission reduction and sequestration options are summarized in Table 4-1. All estimates are expressed in 2004

U.S. dollars per metric ton of carbon.[13] The limitations of emission reduction supply curves noted in the text box apply equally to the cost estimates in Table 4-1.

Table 4-1. Standardized cost estimates for short-term CO_2 emission reduction and sequestration options [annualized cost in 2004 constant U.S. dollars per metric ton of carbon (t C)].

Most options have a range of costs. The range is due to four factors. First, the cost per unit of emissions reduced varies by location even for a very simple measure. For example, the emission reduction achieved by installing a more efficient light bulb depends on the hours of use and the generation mix that supplies the electricity. Second, the cost and performance of any option in the future is uncertain. Different assumptions about future costs and performance contribute to the range. Third, most mitigation and sequestration options are subject to diminishing returns, that is, cost rises at an increasing rate with greater use, as in the power generation, agriculture, and forestry cost estimates.[14] So the estimated scale of adoption contributes to range. Finally, some categories include multiple options, notably those for the U.S. economy as a whole, each with its own marginal cost. For example, the "All Industry" category is an aggregation of seven subcategories discussed in Chapter 8. The result again is a range of cost estimates.

The cost estimates in Table 4-1 are the direct costs of the options. A few options, such as the first estimate for power generation in Table 4-1, have a negative annualized cost. This implies that the option is likely to yield cost savings for reasons such as improved combustion efficiency. Some options have ancillary benefits (e.g., reductions in ordinary pollutants, reduced dependence on imported oil, expansion of wildlife habitat associated with afforestation) that reduce their cost from a societal perspective. Indirect (multiplier, general equilibrium, macroeconomic) effects in the economy tend to increase the direct costs (as when the increased cost of energy use raises the price of products that use energy or energy-intensive inputs). Examples of these complicating effects are presented in Chapters 6 through 11, along with some estimates of their impacts on costs.

As indicated in several segments of Table 4-1, costs are sensitive to the policy instrument used to implement the option. In general, the less restrictive the policy, the lower the cost. That is why the cost estimates for the Feebate are lower than the cost estimate for the CAFÉ standard. In a similar vein, costs are lowered by expanding the number of participants in an emissions trading arrangement, especially those with a prevalence of low-cost options, such as developing countries. That is why the global trading costs are lower than the industrialized country trading case for the U.S. economy.

[13]A metric ton (sometimes written as "tonne") is 1000 kg, which is 2205 lb or 1.1025 tons.

[14]For example, increasing the scale of tree planting to sequester carbon requires more land. Typically the value of the extra land used rises, so the additional sequestration becomes increasingly costly.

The task of choosing the "best" combination of options may seem daunting given the numerous options, their associated cost ranges and ancillary impacts. This combination will depend on several factors including the emission target, the emitters covered, the compliance period, and the ancillary benefits and costs of the options. The best combination will change over time as cheap options become more costly with additional installations, and technological change lowers the costs of more expensive options. It is unlikely that policy-makers can identify the least-cost combination of options to achieve a given emission target. They can adopt policies, such as emissions trading or emissions taxes, that cover a large number of emitters and allow them to use their first-hand knowledge to choose the lowest cost reduction options.[15]

POLICY OPTIONS

Overview

No single technology or approach can achieve a sufficiently large CO_2 emission reduction or sequestration to stabilize the carbon cycle (Hoffert *et al*., 1998, 2002). Policies will need to stimulate implementation of a portfolio of options to reduce emissions and increase sequestration in the short-term, taking into account constraints on and implications of the mitigation strategies. The portfolio of short-term options will include greater efficiency in the production and use of energy; expanded use of non-carbon and low-carbon energy technologies; and various changes in forestry, agricultural, and land use practices. Policies will also need to encourage research and development of technologies that can reduce emissions even further in the long term, such as technologies for removing carbon from fossil fuels and sequestering it in geological formations and possibly other approaches, some of which are currently very controversial, such as certain types of "geoengineering."

Because CO_2 has a long atmospheric residence time,[16] immediate action to reduce emissions and increase sequestration allows its atmospheric concentration to be stabilized at a lower level.[17] Policy instruments to promote cost-effective implementation of a portfolio of options covering virtually all emissions sources and sequestration options are available for the short term. Such policy instruments are discussed below.

The effectiveness of the policies is determined by the technical feasibility and cost-effectiveness of the portfolio of measures they seek to promote, their interaction with other policies that have unintended

[15] Swift (2001) finds that emissions trading programs yield greater environmental and economic benefits than regulations. Several other studies of actual policies (e.g., Ellerman *et al*., 2000) and proposed policies (e.g., Rose and Oladosu, 2002) have indicated relative cost savings of these incentive-based instruments.

[16] CO_2 has an atmospheric lifetime of 5 to 200 years. A single lifetime can not be defined for CO_2 because of different rates of uptake by different removal processes. (IPCC, 2001a, Table 1, p. 38)

[17] IPCC, 2001a, p. 187.

impacts on CO_2 emissions and by their suitability given the institutional and socioeconomic context (Raupach *et al*., 2004). This means that the effectiveness of the portfolio can be limited by factors such as

- The institutional and timing aspects of technology transfer. The patenting system for instance does not allow all countries and sectors to get the best available technology.
- Demographic and social dynamics. Factors such as land tenure, population growth, and migration may pose an obstacle to afforestation/reforestation strategies.
- Institutional settings. The effectiveness of taxes, subsidies, and regulations to induce the deployment of certain technology may be limited by factors such as corruption or existence of vested interests.
- Environmental considerations. The portfolio of measures may incur environmental costs such as waste disposal or biodiversity reduction.

General Considerations

Policies to encourage reduction and sequestration of CO_2 emissions could include information programs, voluntary programs, conventional regulation, emissions trading, and emissions taxes (Tietenberg, 2000). Voluntary agreements between industry and governments and information campaigns are politically attractive, raise awareness among stakeholders, and have played a role in the evolution of many national policies, but to date have generally yielded only modest results.[18] While some programs and agreements have reduced emissions, it appears that the majority of voluntary agreements have achieved limited emissions reductions beyond business as usual. (OECD, 2003b).

Reducing emissions will require the use of policy instruments such as regulations, emissions trading, and emissions taxes. Regulations can require designated sources to keep their emissions below a specified limit, either a quantity per unit of output or an absolute amount per day or year. Regulations can also stipulate minimum levels of energy efficiency of appliances, buildings, equipment, and vehicles.

An emissions trading program establishes a cap on the annual emissions of a set of sources. Allowances equal to the cap are issued and can be traded. Each source must monitor its actual emissions and remit allowances equal to its actual emissions to the regulator. An emission trading program creates an incentive for sources with low-cost options to reduce their emissions and sell their excess allowances. Sources with high-cost options find it less expensive to buy allowances at the market price than to reduce their own emissions enough to achieve compliance.

An emissions tax requires designated sources to pay a specified levy for each unit of its actual emissions. In a manner analogous to emissions trading, emitters will mitigate emissions up to the point

[18]Information and voluntary programs may have some impact on behavior through an appeal to patriotism or an environmental ethic; publishing information that may reveal negative actions, as in a pollutant registry; and providing public recognition, as in green labeling or DOE's Energy Star Program (Tietenberg and Wheeler, 2001).

where mitigation costs are lower than the tax, but once mitigation costs exceed the tax, they will opt to pay it.

The framework for choosing a policy instrument needs to consider technical, institutional and socioeconomic constraints that affect its implementation, such the ability of sources to monitor their actual emissions, the constitutional authority of national and/or provincial/state governments to impose emissions taxes, regulate emissions and/or regulate efficiency standards. It is also important to consider potential conflicts between carbon reduction policies and policies with other objectives, such as keeping energy costs to consumers as low as possible.

Practically every policy (except cost-saving conservation and other “no regrets” options), no matter what instrument is used to implement it, has a cost in terms of utilization of resources and ensuing price increases that leads to reductions in output, income, employment, or other measures of economic well-being. The total cost is usually higher than the direct cost due to interactions with other segments of the economy (“general equilibrium” effects) and with existing policies. Regardless of where the compliance obligation is imposed, the cost ultimately is borne by the general public as consumers, shareholders, employees, taxpayers, and recipients of government services.[19] The cost can have competitiveness impacts if some emitters in other jurisdictions are not subject to similar policies. But societal benefits, such as improved public health and reduced environmental damage, may offset the cost of implementing the policy.

To achieve a given emission reduction target, regulations that require each affected source to meet a specified emissions limit or implement specified controls are almost always more costly than emissions trading or emissions taxes because they require each affected source to meet the regulation regardless of cost rather than allowing emission reductions to be implemented where the cost is lowest (Bohm and Russell, 1986).[20] The cost saving available through trading or an emissions tax generally increases with the diversity of sources and share of total emissions covered by the policy (see, e.g., Rose and Oladosu, 2002).[21] A policy that raises revenue (an emissions tax or auctioned allowances) has a lower cost to the

[19]The source with the compliance obligation passes on the cost through some combination of higher prices for its products, negotiating lower prices with suppliers, layoffs, and/or lower wages for employees, and lower profits that lead to lower tax payments and lower share prices. Other firms that buy the products or supply the inputs make similar adjustments. Governments raise taxes or reduce services to compensate for the loss of tax revenue. Ultimately all of the costs are borne by the general public.

[20]As well, regulation is generally inferior to emissions trading or taxes in inducing technological change.

[21]These policies encourage implementation of the lowest cost emission reductions available to the affected sources. They establish a price (the emissions tax or the market price for an allowance) for a unit of emissions and then allow affected sources to respond to the price signal. In principle, these two instruments are equivalent in terms of achievement of the efficient allocation of resources, but they may differ in terms of equity because of how the emission permits are initially distributed and whether a tax or subsidy is used. It is easier to coordinate emissions trading programs than emissions taxes across jurisdictions.

economy than a policy that does not, if the revenue is used to reduce existing distortionary taxes[22] such as sales or income taxes (see, e.g., Parry *et al*., 1999).

Source Reduction Policies

Historically CO_2 emissions have not been regulated directly. Some energy-related CO_2 emissions have been regulated indirectly through energy policies, such as promotion of renewable energy, and efficiency standards and ratings for equipment, vehicles, and some buildings. Methane emissions from oil and gas production, underground coal mines, and landfills have been regulated, usually for safety reasons.

Policies with other objectives can have a significant impact on CO_2 emissions. Policies to encourage production or use of fossil fuels, such as favorable tax treatment for fossil fuel production, increase CO_2 emissions. Similarly, urban plans and infrastructure that facilitate automobile use rather than public transit increase CO_2 emissions. In contrast, a tax on vehicle fuels reduces CO_2 emissions.[23]

Carbon dioxide emissions are well suited to emissions trading and emissions taxes. These policies allow considerable flexibility in the location and, to a lesser extent, the timing of the emission reductions. The environmental impacts of CO_2 depend on its atmospheric concentration, which is not sensitive to the location or timing of the emissions. Apart from ground-level safety concerns, the same is true of CH_4 emissions. In addition, the large number and diverse nature of the CO_2 and CH_4 sources means that use of such policies can yield significant cost savings but may also be difficult to implement.

Despite the advantages of emissions trading and taxes, there are situations where regulations setting maximum emissions on individual sources or efficiency standards for appliances and equipment are preferred. Such regulations may be desirable where monitoring actual emissions is costly or where firms or individuals do not respond well to price signals due to lack of information or other barriers. Energy efficiency standards for appliances, buildings, equipment and vehicles tend to fall into this category (OECD, 2003a).[24] In some cases, such as refrigerators, standards have been used successfully to drive technology development.

Terrestrial Sequestration Policies

Currently there are few, if any, policies whose primary purpose is to increase carbon uptake by forests or agricultural soils. But policies designed to achieve other objectives, such as afforestation of marginal lands, green payments, conservation compliance, Conservation Reserve Program, and CSP increase

[22]A distortionary tax is one that changes the relative prices of goods or services. For example, income taxes change the relative returns from work, leisure and savings.

[23]Initially the reduction may be small because demand for gasoline is not very sensitive to price, but over time the tax causes people to adjust their travel patterns and the vehicles they drive thus yielding larger reductions.

carbon uptake. Policies that affect crop choice (support payments, crop insurance, disaster relief) and farmland preservation (conservation easements, use value taxation, agricultural zoning) may increase or reduce the carbon stock of agricultural soils. And policies that encourage higher agricultural output (support payments) can reduce the carbon stored by agricultural soils.

Policies to increase carbon uptake by forests and agricultural soils could take the form of

- Regulations, such as requirements to reforest areas that have been logged, implement specified forest management practices, and establish land conservation reserves;
- Incentive-based policies, such as subsidies for adoption of specified forest management or agricultural practices, or issuance of tradable credits for increases in specified carbon stocks.[25] Since the carbon is easily released from these sinks, for example by a forest fire or tilling the soil, ensuring the permanence of the carbon sequestered is a major challenge for such policies. (Feng *et al.*, 2003);[26]
- Voluntary actions, such as "best practices" that enhance carbon sequestration in soils and forests while realizing other benefits (e.g., managing forests for both timber and carbon storage), establishment of plantation forests for carbon sequestration, and increased production of wood products (Sedjo, 2001; Sedjo and Swallow, 2002).

The carbon cycle impacts of such programs would not be large, compared with emission levels; and in nearly every case they face serious challenges in verifying and monitoring the net carbon uptake, especially over relatively long periods (e.g., Marland *et al.*, 2001).

Research and Development Policy

Policies to stimulate research and development of lower emissions technologies for the long term are also needed. Policies to reduce CO_2 emissions influence the rate and direction of technological change (OECD, 2003a). By stimulating additional technological change, such policies can reduce the cost of meeting a given reduction target (Goulder, 2004; Grubb *et al.*, 2006). Such induced technological change justifies earlier and more stringent emission reduction targets.

Two types of policies are needed to achieve a given cumulative CO_2 reduction or concentration target at least cost. Policies to reduce emissions and increase sequestration are needed to create a market for less

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[24]The efficiency of standards sometimes can be improved by allowing manufacturers that exceed the standard to earn credits that can be sold to manufacturers that do not meet the standard.

[25]There needs to be a buyer for the credits, such as sources subject to CO2 emissions trading program or an offset requirement. Determination of the quantity of credits earned requires resolution of many issues, including the baseline, leakage, and additionally. Projects to increase forest sequestration are envisaged in the Kyoto Protocol through Articles 3.3 and 3.4 and through the use of the Clean Development Mechanism (CDM).

[26]Agriculture and forestry credits could be temporary. Temporary credits could be valuable additions to a carbon reduction portfolio.

emission-intensive technologies. But direct support for research and development is also important; the combination of “research push” and “market pull” policies is more effective than either strategy on its own (Goulder, 2004). Policies should encourage research and development for all promising technologies because there is considerable ambiguity about which ones will ultimately prove most useful, socially acceptable, and cost-effective.[27]

CONCLUSIONS

Policies to reduce projected CO_2 and CH_4 concentrations in the atmosphere must recognize the following:

- Emissions are produced by millions of diverse sources, most of which (e.g., power plants, factories, building heating and cooling systems, and large appliances) have lifetimes of 5 to 50 years, and so can adjust only slowly at reasonable cost;
- Potential uptake by agricultural soils and forests is significant but small relative to emissions and can be reversed easily at any given location by natural phenomena or human activities;
- Technological change will have a significant impact on the cost because emission reductions will be implemented over a long time, and new technologies should lower the cost of future reductions; and
- Many policies implemented to achieve other objectives by different national, state/provincial, and municipal jurisdictions increase or reduce CO_2/CH_4 emissions.

Under a wide range of assumptions, cost-effective policies to reduce atmospheric CO_2 and CH_4 concentrations cost-effectively in the short and long term would

- Encourage adoption of cost-effective emission reduction and sink enhancement measures. An emissions trading program or emissions tax that covers as many sources and sinks as possible, combined with regulations where appropriate, could achieve this. National policies can improve cost-effectiveness by providing broader coverage of sources and sinks while reducing adverse competitiveness effects. Use of revenue from auctioned allowances and emissions taxes to reduce existing distortionary taxes can reduce the economic cost of emission reduction policies.
- Stimulate development of technologies that lower the cost of emissions reduction, geological storage, and sink enhancement. Policies that encourage research, development, and dissemination of a portfolio of technologies combined with policies to reduce emissions and enhance sinks to create a “market pull” tend to be more effective than either type of policy alone.

[27] In other words, research and development is required for a portfolio of technologies. Because technologies have global markets, international cooperation to stimulate the research and development is appropriate.

- Adopt appropriate regulations to complement the emissions trading program or emissions tax for sources or actions subject to market imperfections, such as energy-efficiency measures and co-generation. In some situations, credit trading can improve the efficiency of efficiency regulations.
- Revise existing policies at the national, state/provincial, and local level with other objectives that lead to higher CO_2 or CH_4 emissions so that the objectives, if still relevant, are achieved with lower emissions.

Implementation of such policies is best achieved by national governments with international cooperation. This provides maximum coverage of CO_2 and CH_4 emissions and carbon sinks. It also allows better allocation of resources for technology research and development. However, constitutional jurisdiction over emissions sources or carbon sinks may reside with state/provincial governments. In that case national policies may need to be coordinated with state/provincial governments, or state/provincial governments may implement coordinated policies without the national government.

CHAPTER 4 REFERENCES

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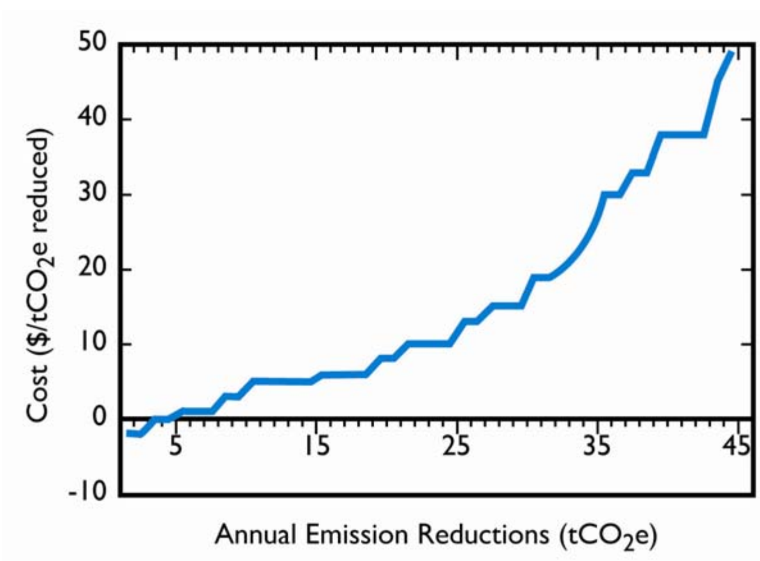
[START OF TEXT BOX]

Emission Reduction Supply Curve

A tool commonly used to compare emission reduction and sequestration options is an emission reduction supply curve, such as that shown in the figure. It compiles the emission reduction and sequestration options available for a given jurisdiction at a given time. If the analysis is for a future date, a detailed scenario of future conditions is needed. The estimated emission reduction potential of each option is based on local circumstances at the specified time, taking into account the interaction among options. The options are combined into a curve starting with the most cost-effective and ending with the least cost-effective. For each option, the curve shows the cost per metric ton of CO_2 reduced on the vertical axis and the potential emission reduction, tons of CO_2 per year, on the horizontal axis. The curve can be used to identify the lowest cost options to meet a given emission reduction target, the associated marginal cost (the cost per metric ton of the last measure included), and total cost (the area under the curve).

An emission reduction supply curve is an excellent tool for assessing alternative emission reduction targets. The best options and cost are easy to identify. The effect on the cost of dropping some options is easy to calculate. And the cost impact of having to implement additional measures due to underperformance by some measures is simple to estimate. The drawbacks are that constructing the curve is a complex analytical process and that the curve is out of date almost immediately because fuel prices and the cost or performance of some options change.



The curve shows the estimated unit cost ($/t CO_2 equivalent) and annual emission reduction (t CO_2 equivalent) for emission reduction and sequestration options for a given region and date arranged in order of increasing unit cost.

When constructed for a future date, such as 2010 or 2020, the precision suggested by the curve is misleading because the future will differ from the assumed scenario. A useful approach in such cases is to group options into cost ranges, such as less than $5 per metric ton of CO_2, $5 to $15 per metric ton of CO_2, etc., ignoring some interaction effects and the impacts of the policy used to implement the option. This still identifies the most cost-effective options. Comparing the emissions reduction target with the emission reduction potential of the options in each group indicates the most economic strategy.

[END OF TEXT BOX]

Table 4.1. Standardized cost estimates for short-term CO_2 emission reduction and sequestration options [annualized cost in 2004 constant U.S. dollars per metric ton of carbon (t C)]

Option/applicable date(s)	Annualized average cost (in $2004 U.S.)	Potential range (Mt C yr^{-1}) or % reduction	Source
Power generation	–$206 to 1067/t C	N.A.	DOE/EIA (2000)
Transportation/2010 (U.S. permit trading)	$76/t C	N.A.	DOE/EIA (2003)
Transportation/2025 (U.S. permit trading)	$214/t C	90	DOE/EIA (2003)
Transportation/2017 (CAFÉ standard)	$74/t C	43	US CBO (2003)
Transportation/2030 (Feebate)	$44/t C	74	Greene *et al*. (2005)
Afforestation/2010–2110	$54 to 109/t C	41 to 247	Lewandrowski (2004),
Forest management/2010–2110	$4 to 109/t C	8 to 94	Stavins and Richards (2005),
Biofuels/2010–2110	$109 to181/t C	123 to 169	EPA (2005)
Agricultural soil carbon sequestration/2010–2110	$4 to 109/t C	19 to 49	EPA (2005)
All industry			
Reduction of fugitives	$92 to 180/t C	3%	Hertzog (1999);
Energy efficiency	$0 to 180/t C	12% to 20%	Martin *et al*. (2001);
Process change	$92 to 180/t C	20%	Jaccard *et al*. (2002, 2003a, 2003b);
Fuel substitution	$0 to 92/t C	10%	Worrel *et al*. (2004);
CO_2 capture and storage	$180 to 367/t C	30%	DOE (2006)
Waste management			
Reduction of fugitives	$0 to 180/t C	90%	Hertzog (1999),
CO_2 capture and storage	>$367/t C	30%	Jaccard *et al*. (2002)
Entire U.S. economy			
No trading	$102 to 548/t C [a]	Not specified	EMF (2000)
Industrialized country trading	$19 to 299/t C [a]	Not specified	EMF (2000)
Global trading	$7 to 164/t C [a]	Not specified	EMF (2000)

Sources: Chapters 6–10 of this report.
[a]Annualized marginal cost (cost at upper limit of application, and therefore typically higher than average cost).

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Chapter 5. How can we improve the usefulness of carbon science for decision-making?

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KEY FINDINGS

- Decision-makers are beginning to seek Information on the carbon cycle and on carbon management options across scales and sectors. Carbon management is a relatively new concept not only for decision-makers and members of the public, but also for the science community.
- Improving the usefulness of carbon science in North America will require stronger commitments to generating high quality science that is also decision-relevant.
- Research on the production of policy-relevant scientific information suggests a several ways to improve the usefulness of carbon science for decision-making, including co-production of knowledge, development of applied modeling tools for decision support, and "boundary organizations" that can help carbon scientists and decision-makers communicate and collaborate.
- A number of initiatives to improve understanding of decision support needs and options related to the carbon cycle are under way, some as a part of the Climate Change Science Program (CCSP).
- Additional pilot projects should be considered aimed at enhancing interactions between climate change scientists and parties involved in carbon management activities and decisions.

INTRODUCTION: THE CHALLENGE OF "USABLE" CARBON SCIENCE

This chapter answers two questions:

- How well is the carbon cycle science community doing in "decision support" of carbon cycle management, i.e., in responding to decision-makers' demands for carbon cycle management information?
- How can the carbon cycle science community improve such decision support?

Chapters in Parts 2 and 3 of this report identify many research priorities, including assessing the potential for geological storage of carbon dioxide, quantifying expansion of the North American carbon sink, and identifying the economic impact of carbon tax systems. This chapter focuses on improving communication and collaboration between scientific researchers and carbon managers, to help researchers be more responsive to decision-making, and carbon managers be better informed in making policy, investment and advocacy decisions.

Humans have been inadvertently altering the Earth's carbon cycle since the dawn of agriculture, and more rapidly since the industrial revolution. These influences have become large enough to cause significant climate change (IPCC, 2001). In response, environmental advocates, business executives, and policy-makers have increasingly recognized the need to deliberately manage the carbon cycle. Effective carbon management requires that the variety of people whose decisions affect carbon emissions and sinks have relevant, appropriate science. Yet, carbon cycle science is rarely organized or conducted to support decision-making on managing carbon emissions, sequestration, and impacts. This reflects that, until recently, scientists have approached carbon cycle science as basic science and non-scientist decision-makers have not demanded carbon cycle information. Consequently, emerging efforts to manage carbon are less informed by carbon cycle science than they could be (Dilling *et al*., 2003). Applying carbon science to carbon management requires making carbon cycle science more useful to public and private decision-makers. In particular, scientists and decision-makers will need to identify the information most needed in specific sectors for carbon management, to adjust research priorities, and to develop mechanisms that enhance the credibility of the information generated and the responsiveness of the information-generating process to stakeholder's views (Mitchell *et al*., 2006; Cash *et al*., 2003). Combining some "applied" or "solutions-oriented" research with a basic science portfolio would make carbon science more directly relevant to decision-making.

TAKING STOCK: WHERE ARE WE NOW IN PROVIDING DECISION SUPPORT TO IMPROVE CAPACITIES FOR CARBON MANAGEMENT?

How effective is the scientific community at providing decision support for carbon management? The Climate Change Science Program (CCSP) Strategic Plan defines decision support as: "the set of analyses and assessments, interdisciplinary research, analytical methods, model and data product development, communication, and operational services that provide timely and useful information to address questions confronting policymakers, resource managers and other stakeholders" (U.S. Climate Change Science Program, 2003).

Who are the potential stakeholders for information related to the carbon cycle and options and measures for altering human influences on that cycle? Most people constantly but unconsciously make decisions that affect the carbon cycle, through their use of energy, transportation, living spaces, and natural resources. Increasing attention to climate change has led some policy makers, businesses, advocacy groups and consumers to begin making choices that consciously limit carbon emissions.[1] Whether carbon emission reductions are driven by political pressures or legal requirements, by economic opportunities or consumer pressures, or by moral or ethical commitments to averting climate change, people and organizations are seeking information that can help them achieve their specific carbon-related or climate-related goals.[2] Even in countries and economic sectors that lack a consensus on the need to manage carbon, some people and organizations have begun to experiment with carbon-limiting practices and investments in anticipation of a carbon-constrained future.

In designing and producing this report, we engaged individuals from a wide range of sectors and activities, including forestry, agriculture, utilities, fuel companies, carbon brokers, transportation, non-profits, and local and federal governments. Although we did not conduct new research on the informational or decision support needs of stakeholders, a preliminary review suggests that many stakeholders may be interested in carbon-related information (see Text Box 1).

CURRENT APPROACHES AND TRENDS

As we enter an era of deliberate carbon management, decision-makers from the local to the national level are increasingly open to or actively seeking carbon science information as a direct input to policy and investment decisions (Apps *et al.*, 2003). The government of Canada, having ratified the Kyoto Protocol, has been exploring emission reduction opportunities and offsets and has identified specific needs for applied research (Government of Canada, 2005). For example, Canada's national government

[1]For examples, see Text Box 1

[2]For example, carbon science was presented at recent meetings of the West Coast Governors' Global Warming Initiative and the Climate Action Registry [http://www.climateregistry.org/EVENTS/PastConferences/; http://www.climatechange.ca.gov/events/2005_conference/presentations/]

recently entered a research partnership with the province of Alberta, to assess geological sequestration of carbon dioxide, to develop fuel cell technologies using hydrogen, and to expand the use of biomass and biowaste for energy production (Government of Canada 2006).

Some stakeholders in the U.S. are actively using carbon science to move forward with voluntary emissions offset programs. For example, the Chicago Climate Exchange brokers agricultural carbon credits in partnership with the Iowa Farm Bureau.[3] Many cities and several states have established commitments to manage carbon emissions, including regional partnerships on the east and west coasts, and non-governmental organizations and utilities have begun to experiment with pilot sequestration projects (Text Box 1). The eventual extent of interest in carbon information may well depend on whether and how mandatory and incentive-based policies related to carbon management evolve. In Europe, for example, mandatory carbon emissions policies have resulted in intense interest in carbon science by those directly affected by such policies (Schröter *et al*., 2005).

In the U.S., federal carbon science has very few mechanisms to assess demand for carbon information across scales and sectors. Thus far, federally-funded carbon science has focused on basic research to clarify fundamental uncertainties in the global carbon cycle and local and regional processes affecting the exchange of carbon (Dilling, in press). Most federal efforts are organized under the Climate Change Science Program (CCSP). The National Aeronautics and Space Administration (NASA) and the National Science Foundation (NSF) manage almost two-thirds of this effort, and their missions are limited to basic research, not decision support (U.S. Climate Change Science Program, 2006; Dilling, in press). There are relatively smaller investment research efforts at the Department of Energy (DOE) and the Department of Agriculture (USDA) under the CCSP[4] as well as significant technology efforts under the Climate Change Technology Program (CCTP), a sister program to the CCSP focused on technology development. Increasing linkages among these programs may increase the usefulness of CCSP carbon-related research to decision-makers. For over a decade, the National Oceanic and Atmospheric Administration (NOAA) Climate Program Office has invested in research and institutions intended to improve the usability of climate science, although that investment is small relative to the investment in climate science itself and has focused on the usability of climate, rather than carbon cycle, science.

Until recently, the concept of "carbon management" has not been widely recognized—even now, most members of the public do not understand the term "carbon sequestration" or its potential implications (Shackley *et al*., 2005; Curry *et al*., 2004). However, the carbon cycle science community is

[3]http://www.iowafarmbureau.com/special/carbon/default.aspx

[4]For example, The Consortium for Agricultural Soil Mitigation of Greenhouse Gases (CASMGS) was recently funded by the USDA to provide information and technology necessary to develop, analyze and implement carbon sequestration strategies.

beginning to recognize that it may have information relevant to policy and decision-making. Thus, prominent carbon scientists have called for "coordinated rigorous, interdisciplinary research that is strategically prioritized to address societal needs" (Sarmiento and Wofsy, 1999) and the North American Carbon Program's (NACP) "Implementation Plan" lists decision support as one of four organizing questions (Denning *et al*., 2005).

That same plan, however, states that the scientific community knows relatively little about the likely users of information that the NACP will produce. Indeed, the National Academy of Sciences' review of the CCSP stated that "as the decision support elements of the program are implemented, the CCSP will need to do a better job of identifying stakeholders and the types of decisions they need to make" (National Research Council, 2004). Moreover, they state that "managing risks and opportunities requires stakeholder support on a range of scales and across multiple sectors, which in turn implies an understanding of the decision context for stakeholders" (National Research Council, 2004). Successful decision support, i.e., science that improves societal outcomes, requires knowledge of what decision-makers might use the information being generated, and what information would be most relevant to their decisions. Without such knowledge, information runs the risk of being "left on the loading-dock" and not used (Cash et al. 2006).

Two programs within CCSP may shed light on how to link carbon science to user needs. NASA has an Applied Sciences program that seeks to find uses for its data and modeling products using "benchmarking systems," and USDA and DOE have invested significant resources in science that might inform carbon sequestration efforts and carbon accounting in agriculture and forests. However, these programs have not been integrated into a broader framework self-consciously aimed at making carbon cycle science more useful to decision-makers.

Improving the usefulness of carbon science in North America will require more explicit commitments by funding agencies, scientists, policy makers, and private sector managers to generate decision-relevant carbon cycle information. The participatory methods and boundary spanning institutions identified in the next section help both refine research agendas and accelerate the application of research results to carbon management and societal decision-making.

OPTIONS FOR IMPROVING THE APPLICABILITY OF SCIENTIFIC INFORMATION TO CARBON MANAGEMENT AND DECISION-MAKING

Studies of the creation and use of knowledge for decision-making have found that information must be perceived not only as *credible*, but also as *relevant* to high priority decisions and as stemming from a process that decision-makers view as *responsive* to their concerns (Mitchell *et al*., 2006; Cash *et al*., 2003). Even technically and intellectually rigorous science lacks influence with decision-makers if decision-makers perceive it as not addressing the decisions they face, as being biased, or as having ignored their views and interests.

Research on the production of policy-relevant scientific information suggests several strategies that can maintain the integrity of the research endeavor while increasing its policy relevance. Although communicating results more effectively is clearly important, generating science that is more applicable to decision-making may require deeper changes in the way scientific information is produced. Carbon cycle scientists and carbon decision-makers will need to develop methods for interaction that work best in the specific arenas in which they work. At their core, strategies will be effective to the extent that they promote interaction among scientists and stakeholders in the development of research questions, selection of research methods, and review, interpretation and dissemination of results (Adler *et al*., 1999; Ehrmann and Stinson, 1999; National Research Council, 1999; National Research Council, 2005; Farrell and Jaeger, 2005; Mitchell *et al*., 2006). Such processes work best when they enhance the usability of the research while preserving the credibility of both scientists and stakeholders. Transparency and expanded participation are important for guarding against politicization and enhancing usability.

Examples of joint scientist-stakeholder development of policy relevant scientific information include:

- *Co-production of research knowledge (e.g., Regional Integrated Sciences and Assessments)*: In regional partnerships across the U.S., university researchers work closely with local operational agencies and others that might incorporate climate information in decision-making. New research is developed through ongoing, iterative consultations with all partners (Lemos and Morehouse, 2005).
- *Institutional experimentation and adaptive behavior* (*e.g., adaptive management)*: Adaptive management acknowledges our inherent uncertainty about how natural systems respond to human management, and periodically assesses the outcomes of management decisions and adjusts those decisions accordingly, a form of deliberate "learning by doing" (c.f. Holling 1978). Adaptive management principles have been applied to several resources where multiple stakeholders are involved, including management of river systems and forests (Holling 1995; Pulwarty and Redmond, 1997; Mitchell *et al*., 2004; Lemos and Morehouse, 2005).
- *Assessments as policy component* (*e.g., recovering the stratospheric ozone layer)*: Assessments that were credible, relevant, and responsive played a significant role in the Montreal Protocol's success in phasing out the use of ozone-depleting substances. A highly credible scientific and technical assessment process with diverse academic and industry participation is considered crucial in the Protocol's success (Parson, 2003).
- *Mediated modeling*: Shared tools can facilitate scientist-user interactions, help diverse groups develop common knowledge and understanding of a problem, and clarify common assumptions and differences. In mediated modeling, participants from a wide variety of perspectives jointly construct a

computer model to solve complex environmental problems or envision a shared future. The process has been used for watershed management, endangered species management, and other difficult environmental issues (Van den Belt, 2004).

- *Carbon modeling tools as decision support*: Although the U.S. government has not yet adopted a carbon management policy, some federal agencies have begun to develop online decision support tools, with customizable user interfaces, to estimate carbon sequestration in various ecosystems and under various land use scenarios (see the NASA Ames Carbon Query and Evaluation Support Tools, http://geo.arc.nasa.gov/website/cquestwebsite/; the U.S. Forest Service Carbon Online Estimator, http://ncasi.uml.edu/COLE/; and Colorado State's CarbOn Management Evaluation Tool, http://www.cometvr.colostate.edu/).

Over time, well-structured scientist-stakeholder interaction can help both scientists and decision-makers (Moser, 2005). Scientists learn to identify research questions that are both scientifically interesting and relevant to decisions, and to present their answers in ways that audiences are more likely to find compelling. Non-scientists learn what questions science can and cannot answer. Such interactions clarify the boundary between empirical questions that scientists can answer (e.g., the sequestration potential of a particular technology) and issues that require political resolution (e.g., the appropriate allocation of carbon reduction targets across firms). Institutional arrangements can convert ad hoc successes in scientist-stakeholder interaction into systematic and ongoing networks of scientists, stakeholders, and managers. Such "co-production of knowledge," can enhance both the scientific basis of policy and management and the research agenda for applied science (Lemos and Morehouse, 2005; Gibbons *et al*., 1994; Patt *et al*., 2005a).

That said, such interactive approaches have limitations, risks, and costs. Scientists may be reluctant to involve non-scientists who "should" be interested in a given issue, but who can add little scientific value to the research, and whose involvement requires time and effort. Involving private sector firms may require scientists accustomed to working in an open informational environment to navigate in a world of proprietary information. Scientists may also avoid applied, participatory research if they do not see it producing the "cutting edge" (and career enhancing) science most valued by other scientists (Lemos and Morehouse, 2005).

Some stakeholders may lack the financial resources, expertise, time, or other capacities necessary to meaningful participation. Some will distrust scientists in general and government-sponsored science in particular for cultural, institutional, historical, or other reasons. Some may reject the idea of interacting with those with whom they disagree politically or compete economically. Stakeholders may try to manipulate research questions and findings to serve their political or economic interests. And, stakeholders often show little interest in diverting their time from other activities to what they perceive as the slow and too-often fruitless pursuit of scientific knowledge (Patt *et al*., 2005b).

Where direct stakeholder participation proves too difficult, costly, unmanageable, or unproductive, scientists and research managers need other methods to identify the needs of potential users. Science on the one hand and policy, management, and decision-making on the other often exist as separate social and professional realms, with different traditions, norms, codes of behavior, and reward systems. The boundaries between such realms serve many useful functions but can inhibit the transfer of useful knowledge across those boundaries. A boundary organization is an institution that "straddles the shifting divide" between politics and science (Guston, 2001). Boundary organizations are accountable to both sides of the boundary and involve professionals from each. Boundary spanning individuals and organizations facilitate the uptake of science by translating scientific findings so that stakeholders find them more useful and by stimulating adjustments in research agendas and approach. Boundary organizations can exist at a variety of scales and for a variety of purposes. For example, cooperative agricultural extension services and non-governmental organizations (NGOs) successfully convert large-scale scientific understandings of weather, aquifers, or pesticides into locally-tuned guidance to farmers (Cash, 2001). The International Research Institute for Climate Prediction focuses on seasonal-to-interannual scale climate research and modeling to make their research results useful to farmers, fishermen, and public health officials (e.g., Agrawala *et al*., 2001). The Subsidiary Body for Scientific and Technological Advice of the United Nations Framework Convention on Climate Change serves as an international boundary organization that links information and assessments from expert sources (such as the IPCC) to the Conference of the Parties, which focuses on setting policy.[5] The University of California Berkeley Digital Library Project Calflora project has explicitly designed their database on plants to support environmental planning (Van House *et al*., 2003).

Of course, other significant challenges exist to the use of knowledge. People fail to integrate new research and information in their decisions for many reasons. People often are not motivated to use information that supports policies they dislike; that conflicts with pre-existing preferences, interests, or beliefs; or that conflicts with cognitive, organizational, sociological, or cultural norms (e.g., Douglas and Wildavsky, 1984; Lahsen, 1998; Yaniv, 2004; Lahsen, forthcoming). These tendencies are important components of a healthy democratic process. Developing processes to make carbon science more useful to decision-makers will not guarantee its use but will make its use more likely.

[5] http://unfccc.int/2860.php

RESEARCH NEEDS TO ENHANCE DECISION SUPPORT FOR CARBON MANAGEMENT

The demand for detailed analysis of carbon management issues and options across major economic sectors, nations and levels of government in North America is likely to grow substantially in the near future. This will be especially true in jurisdictions that place policy constraints on carbon budgets, such as Canada, the U.S. states comprising the Regional Greenhouse Gas Initiative, or the U.S. State of California. Although new efforts are underway in some federal agencies, carbon cycle science in the U.S. could be organized and carried out to better and more systematically meet this potential demand. Effective implementation of the goals of the Climate Change Science Program "requires focused research to develop decision support resources and methods" (National Research Council, 2004).

Creating information for decision support should differ significantly from doing basic science. In such "use-inspired research," societal need is as important as scientific curiosity (Stokes, 1997). Scientists and carbon managers need to improve their joint understanding of the top priority questions facing carbon-related decision-making. They need to collaborate more effectively in undertaking research and interpreting results in order to answer those questions.

A first step might involve developing a formal process "for gathering requirements and understanding the problems for which research can inform decision-makers outside the scientific community," including forming a decision support working group (Denning *et al*., 2005). The NRC has recommended that the CCSP's decision support components could be improved by organizing various deliberative activities, including workshops, focus groups, working panels, and citizen advisory groups to: "1) expand the range of decision support options being developed by the program; 2) to match decision support approaches to the decisions, decision-makers, and user needs; and 3) to capitalize on the practical knowledge of practitioners, managers and laypersons" (National Research Council, 2004).

SUMMARY AND CONCLUSIONS

The carbon cycle is influenced through both deliberate and inadvertent decisions by diverse and spatially dispersed people and organizations, working in many different sectors and at different scales. To make carbon cycle science more useful to decision-makers, we suggest that leaders in the scientific and program level carbon science community initiate the following steps:

- Identify categories of decision-makers for whom carbon cycle science is a relevant concern, focusing on policy makers and private sector managers in carbon-intensive sectors (energy, transport, manufacturing, agriculture and forestry)

- Evaluate existing information about carbon impacts of actions in these arenas, and assess the need and demand for additional information. In some cases, demand may need to be fostered through an interactive process.
- Encourage scientists and research programs to experiment with incremental and major departures from existing practice with the goal of making carbon cycle science more credible, relevant, and responsive to carbon managers.
- Involve experts in the social sciences and communication as well as experts in physical, biological, and other natural science disciplines in efforts to produce usable science.
- Consider initiating participatory pilot research projects and identifying existing boundary organizations (or establishing new ones) to bridge carbon management and carbon science.

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Sectors Expressing Interest and/or Participating in the SAP 2.2 Process. This list of sectors is neither exhaustive nor is it based on a statistically rigorous assessment, but is meant to demonstrate the wide variety of stakeholders with a potential interest in carbon-related information.

Agriculture: Tillage and other farming practices significantly influence carbon storage in agricultural soils. Managing these practices presents opportunities both to slow carbon loss and to restore carbon in soils. Farmers have been quite interested in carbon management as a means to stimulate rural economic activity. Since much of the agricultural land in the United States is privately owned, both economic forces and governmental policies will be critical factors in the participation of this sector in carbon management. (Chapter 10).

Forestry: Forests accumulate carbon in above-ground biomass as well as soils. The carbon impact of planting, conserving, and managing forests has been an area of intense interest in international negotiations on climate change (IPCC, 2000). Whether seeking to take advantage of international carbon credits, to offset other emissions, or to simply identify environmental co-benefits of forest actions taken for other reasons, governments, corporations, land-owners, and conservation groups may need more information on and insight into the carbon implications of forestry decisions ranging from species selection to silviculture, harvesting methods, and the uses of harvested wood. (Chapter 11).

Utilities and Industries: In the US, over 85% of energy produced comes from fossil fuels with relatively high carbon intensity. The capital investment and fuel source decisions of utilities and energy-intensive industries thus have major carbon impacts. A small but growing number of companies have made public commitments to reducing carbon emissions, developed business models that demonstrate sensitivity to climate change, and begun exploring carbon capture and storage opportunities. For example, Cinergy, a large Midwestern utility, has experimented with carbon offset programs in partnership with The Nature Conservancy. (Chapter 6 and 8).

Transportation: Transportation accounts for approximately 37% of carbon emissions in the U.S., and about 22% worldwide. In transportation, governmental infrastructure investments, automobile manufacturers' decisions about materials, technologies and fuels, and individual choices regarding auto purchases, travel modes, and distances all have significant impacts on carbon emissions. (Chapter 7)

Government: In the US, national policies currently rely primarily on voluntary measures and incentive structures (U.S. Department of State, 2004; Richards, 2004). Canada, having ratified the Kyoto Protocol, has direct and relatively immediate needs for information that can help it meet its binding targets as cost-effectively as possible (Government of Canada, 2005). The Mexican government appears to be particularly interested in locally-relevant research on natural and anthropogenic influences on the carbon cycle, likely impacts across various regions, and the costs, benefits, and viability of various management options (Martinez and Fernandez-Bremauntz, 2004). Below the national level, more and more states and local governments are taking steps, including setting mandatory policies, to reduce carbon emissions, and may need new carbon cycle science scaled to the state and local level to manage effectively [for example, nine New England and mid-Atlantic states have formed a regional partnership, also observed by Eastern Canadian provinces, to reduce carbon emissions through a cap and trade program combined with a market-based emissions trading system (Regional Greenhouse Gas Initiative—RGGI—www.rggi.org] (see Chapters 4 and 14).

Non-Profits and Non-Governmental Organizations (NGOs): Many environmental and business-oriented organizations have an interest in carbon management decision making. Such organizations rely on science to support their positions and to undercut the arguments of opposing advocates. There has been substantial criticism of "advocacy science" in the science-for-policy literature, and new strategies will need to be developed to promote constructive use of carbon cycle science by advocates (Ehrmann and Stinson, 1999; Adler *et al*., 2001).

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PART II OVERVIEW

Energy, Industry, and Waste Management Activities: An Introduction to CO_2 Emissions from Fossil Fuels

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THE CONTEXT

Fossil fuels (coal, oil, and natural gas) are used primarily for their concentration of chemical energy, energy that is released as heat when the fuels are burned. Fossil fuels are composed primarily of compounds of hydrogen and carbon (C), and when the fuels are burned the hydrogen and carbon oxidize to water and CO_2, and heat is released. If the water and CO_2 are released to the atmosphere, the water will soon fall out as rain or snow. The CO_2, however, will increase the concentration of CO_2 in the atmosphere and join the active cycling of carbon that takes place among the atmosphere, biosphere, and hydrosphere. Since humans began taking advantage of fossil-fuel resources for energy, we have been releasing to the atmosphere, over a very short period of time, carbon that was stored deep in the Earth over millions of years. We have been introducing a large perturbation to the active cycling of carbon.

Estimates of fossil-fuel use globally show that there have been significant emissions of CO_2 dating back at least to 1750, and from North America back at least to 1785. However, this human perturbation of the active carbon cycle is largely a recent process, with the magnitude of the perturbation growing as population grows and demand for energy grows. Over half of the CO_2 released from fossil-fuel burning globally has occurred since 1980 (Fig. 1).

Figure 1. Cumulative global emissions of CO_2 from fossil-fuel combustion and cement manufacture from 1751 to 2002 (data from Marland *et al.*, 2006).

Some CO_2 is also released to the atmosphere during the manufacture of cement. Limestone ($CaCO_3$) is heated to release CO_2 and produce the calcium oxide (CaO) used to manufacture cement. In North America, cement manufacture now releases less than 1% of the mass of CO_2 released by fossil-fuel combustion. However, cement manufacture is the third largest anthropogenic source of CO_2 (after fossil-fuel use and the clearing and oxidation of forests and soils; see Part III of this report). The CO_2 emissions from cement manufacture are often included with the accounting of anthropogenic CO_2 emissions from fossil fuels.

Part II of this report addresses the magnitude and pattern of CO_2 emissions from fossil-fuel consumption and cement manufacture in North America. This introductory section addresses some general issues associated with CO_2 emissions and the annual and cumulative magnitude of total emissions. It looks at the temporal and spatial distribution of emissions and some other data likely to be of interest. The following four chapters delve into the sectoral details of emissions so that we can understand the forces that have driven the growth in emissions to date and the possibilities for the magnitude and pattern of emissions in the future. These chapters reveal, for example, that 38% of CO_2 emissions from North America come from enterprises whose primary business is to provide electricity and heat and another 31% come from the transport of passengers and freight. This introduction focuses on the total emissions from the use of fossil fuels and the subsequent chapters provide insight into how these fuels are used and the economic and human factors motivating their use.

Estimating CO_2 Emissions

It is relatively straightforward to estimate the amount of CO_2 released to the atmosphere when fossil fuels are consumed. Because CO_2 is the equilibrium product of oxidizing the carbon in fossil fuels, we need to know only the amount of fuel used and its carbon content. For greater accuracy, we adjust this estimate to take into consideration the small amount of carbon that is left as ash or soot and is not actually oxidized. We also consider the fraction of fossil fuels that is used for things like asphalt, lubricants, waxes, solvents, and plastics and may not be soon converted to CO_2. Some of these long-lived, carbon-containing products will release their contained carbon to the atmosphere as CO_2 during use or during processing of waste. Other products will hold the carbon in use or in landfills for decades or longer. One of the differences among the various estimates of CO_2 emissions is the way they deal with the carbon in these products.

Fossil-fuel consumption is often measured in mass or volume units and, in these terms, the carbon content of fossil fuels is quite variable. However, when we measure the amount of fuel consumed in terms of its energy content, we find that for each of the primary fuel types (coal, oil, and natural gas) there is a strong correlation between the energy content and the carbon content. The rate of CO_2 emitted per unit of useful energy released depends on the ratio of hydrogen to carbon and on the details of the organic compounds in the fuels; but, roughly speaking, the numerical conversion from energy released to carbon released as CO_2 is about 25 kg C per 10^9 joules for coal, 20 kg C per 10^9 joules for petroleum, and 15 kg C per 10^9 joules for natural gas. Figure 2 shows details of the correlation between energy content and carbon content for more than 1000 coal samples. Detailed analysis of the data suggests that hard coal contains 25.16 ± 2.09% kg C per 10^9 joules of coal (measured on a net heating value basis[1]). The value is slightly higher for lignite and brown coal (26.23 kg C ± 2.33% per 10^9 joules (also shown in Fig. 2). Similar correlations exist for all fuels and Table 1 shows some of the coefficients reported by the Intergovernmental Panel on Climate Change (IPCC) for estimating CO_2 emissions. The differences between the values in Table 1 and those in Fig. 1 are small, but they begin to explain how different data compilations can end up with different estimates of CO_2 emissions.

Figure 2. The carbon content of coal varies with the heat content, shown here as the net heating value.

Table 1. A sample of the coefficients used for estimating CO_2 emissions from the amount of fuel burned (from IPCC, 1997).

Data on fossil-fuel production, trade, consumption, etc. are generally collected at the level of some political entity, such as a country, and over some time interval, typically a year. Estimates of national, annual fuel consumption can be based on estimates of fuel production and trade, estimates of actual final consumption, data for fuel sales or some other activity that is clearly related to fuel use, or on estimates and models of the activities that consume fuel (such as vehicle miles driven). In the discussion that follows, some estimates of national, annual CO_2 emissions are based on "apparent consumption" (defined as production + imports – exports +/– changes in stocks) while others are based on more direct estimates of fuel consumption. All of the emissions estimates in this chapter are as the mass of carbon released[2].

The uncertainty in estimates of CO_2 emissions will thus depend on the variability in the chemistry of the fuels, the quality of the data or models of fuel consumption, and on uncertainties in the amount of carbon that is used for non-fuel purposes (such as asphalt and plastics) or is otherwise not burned. For

[1]Net heating value (NHV) is the heat release measured when fuel is burned at constant pressure so that the H_2O is released as H_2O vapor. This is distinguished from the gross heating value (GHV), the heat release measured when the fuel is burned at constant volume so that the H_2O is released as liquid H_2O. The difference is essentially the heat of vaporization of the H_2O and is related to the H content of the fuel.

[2]The C is actually released to the atmosphere as CO_2 and it is accurate to report (as is often done) either the amount of CO_2 emitted or the amount of C in the CO_2. The numbers can be easily converted back and forth using the ratio of the molecular masses, i.e. (mass of C) x (44/12) = (mass of CO_2).

countries like the United States—with good data on fuel production, trade, and consumption—the uncertainty in national emissions of CO_2 is on the order of ± 5% or less. In fact, the US Environmental Protection Agency (USEPA, 2005) suggests that their estimates of CO_2 emissions from energy use in the United States are accurate, at the 95% confidence level, within –1 to +6 % and Environment Canada (2005) suggests that their estimates for Canada are within –4 to 0 %. The Mexican National Report (Mexico, 2001) does not provide estimates of uncertainty, but our analyses with the Mexican data suggest that uncertainty is larger than for the United States and Canada. Emissions estimates for these same three countries, as reported by the Carbon Dioxide Information Analysis Center (CDIAC) and the International Energy Agency (IEA) (see the following section), will have larger uncertainty because these groups are making estimates for all countries. Because they work with data from all countries, they use global average values for things like the emissions coefficients, whereas agencies within the individual countries use values that are more specific to the particular country. When national emissions are calculated by consistent methods it is likely that year-to-year changes can be estimated more accurately than would be suggested by the uncertainties of the individual annual values.

The Magnitude of National and Regional CO_2 Emissions

Figure 3 shows that from the beginning of the fossil-fuel era (1751 in these graphs) to the end of 2002, there were 93.5 Gt C released as CO_2 from fossil-fuel consumption (and cement manufacture) in North America: 84.4 Gt C from the United States, 6.0 from Canada, and 3.1 from Mexico. All three countries of North America are major users of fossil fuels and this 93.5 Gt C was 31.5 % of the global total. Among all countries, the United States, Canada, and Mexico ranked as the first, eighth, and eleventh largest emitters of CO_2 from fossil-fuel consumption, respectively (for 2002) (Marland *et al*., 2006). Figure 4 shows, for each of these countries and for the sum of the three, the annual total of emissions and the contributions from the different fossil fuels.

Figure 3. The cumulative total of CO_2 emissions from fossil-fuel consumption and cement manufacture, as a function of time, for the three countries of North America and for the sum of the three (from Marland *et al*., 2006).

Figure 4. Annual emissions of CO_2 from fossil-fuel use by fuel type.

The long time series of emissions estimates in Figs. 1, 3, and 4 are from the CDIAC (Marland *et al*., 2006). These estimates are derived from the "apparent consumption" of fuels and are based on data from the UN Statistics Office back to 1950 and on data from a mixture of sources for the earlier years (Andres *et al*., 1999). There are other published estimates (with shorter time series) of national, annual CO_2 emissions. Most notably the IEA (2005) has reported estimates of emissions for many countries for all years back to 1971, and most countries have now provided some estimates of their own emissions as part of their national obligations under the United Nations Framework Convention on Climate Change (UNFCCC, see http://unfccc.int). These latter two sets of estimates are based on data on actual fuel consumption and thus are able to provide details as to the sector of the economy where fuel use is taking place[3].

Comparing the data from multiple sources can give us some insight into the reliability of the estimates generally. These different estimates of CO_2 emissions are not, of course, truly independent because they all rely ultimately on national data on fuel use; but they do represent different manipulations of this primary data and in many countries there are multiple potential sources of energy data. Many developing countries do not collect or do not report all of the data necessary to precisely estimate CO_2 emissions and in these cases differences can be introduced by how the various agencies derive the basic data on fuel production and use. Because of the way data are collected, there are statistical differences between "consumption" and "apparent consumption" as defined above.

To make comparisons of different estimates of CO_2 emissions we would like to be sure that we are indeed comparing estimates of the same thing. For example emissions from cement manufacture are not available from all of the sources, so they are not included in the comparisons in Table 2. All of the estimates in Table 2, except those from the IEA, include emissions from flaring natural gas at oil production facilities. It is not easy to identify the exact reason the estimates differ, but the differences are generally small. The differences have mostly to do with the statistical difference between consumption and apparent consumption, the way in which correction is made for non-fuel usage of fossil-fuel resources, the conversion from mass or volume to energy units, and/or the way in which estimates of carbon content are derived. Because the national estimates from CDIAC do not include emissions from the non-fuel uses of petroleum products, we expect them to be slightly smaller than the other estimates shown here, all of which do include these emissions[4]. The comparisons in Table 2 reveal one number for which there is a notable relative difference among the multiple sources, emissions from Mexico in 1990. Losey (2004) has suggested, based on other criteria, that there is a problem in the United Nations energy data set with the Mexican natural gas data for the 3 years 1990-1992, and these kinds of analyses result in re-examination of some of the fundamental data.

[3]The International Energy Agency provides estimates based on both the reference approach (estimates of apparent consumption) and the sectoral approach (estimates of actual consumption) as described by the IPCC (IPCC, 1997). In the comparison here we use the numbers that they believe to be the most accurate, those based on the sectoral approach.

[4]The CDIAC estimate of global total emissions does include estimates of emissions from oxidation from non-fuel use of hydrocarbons.

Table 2. Different estimates (in Mt C) of CO_2 emissions from fossil-fuel consumption for the United States, Canada, and Mexico.

The IEA (2005, p. 1.4) has systematically compared their estimates with those reported to the UNFCCC by the different countries and they find that the differences for most developed countries are within 5%. The IEA attributes most of the differences to the following:

- use of the IPCC Tier 1 method that does not take into account different technologies,
- use of energy data that may have come from different “official” sources within a country,
- use of average values for net heating value of secondary oil products,
- use of average emissions values,
- use of incomplete data on non-fuel uses,
- different treatment of military emissions, and
- a different split between what is identified as emissions from energy and emissions from industrial processes.

Emissions by Month and/or State

With increasing interest in the details of the global carbon cycle there is increasing interest in knowing emissions at spatial and temporal scales finer than countries and years. For the United States, energy data have been collected for many years at the level of states and months and thus estimates of CO_2 emissions can be made by state or by month. Figure 5 shows the variation in U.S. emissions by month and preliminary analyses by Gurney *et al*. (2005) reveal that proper recognition of this variability can be very important in some exercises to model the details of the global carbon cycle.

Figure 5. Emissions of CO_2 from fossil-fuel consumption in the United States, by month.

Because of differences in the way energy data are collected and aggregated, it is not obvious that an estimate of emissions from the United States will be identical to the sum of estimates for the 50 U.S. states. Figure 6 shows that estimates of total annual CO_2 emissions are slightly different if we use data directly from the U.S. Department of Energy (DOE) and sum the estimates for the 50 states or if we sum the estimates for the 12 months of a given year, or if we take U.S. energy data as aggregated by the UN Statistics Office and calculate the annual total of CO_2 emissions directly. Again, the state and monthly emissions data are based on estimates of fuel consumption while the national emissions estimates calculated using UN data result from estimates of "apparent consumption." There is a difference between annual values for consumption and annual values of "apparent consumption" (the IEA calls this difference simply "statistical difference") that is related to the way statistics are collected and aggregated. There are also differences in the way values for fuel chemistry and non-fuel usage are averaged at different spatial and temporal scales, but the differences in CO_2 estimates are seen to be within the error bounds generally expected.

Figure 6. A comparison of three different estimates of national annual emissions of CO_2 from fossil-fuel consumption in the United States.

Data from DOE permit us to estimate emissions by state or by month (Blasing *et al*., 2005a and 2005b), but they do not permit us to estimate CO_2 emissions for each state by month directly from the published energy data. Nor do we have sufficiently complete data to estimate emissions from Canada and Mexico by month or province. Andres *et al*. (2005), Gregg (2005), and Losey (2004) have shown that we can disaggregate national total emissions by month or by some national subdivision (such as states or provinces) if we have data on some large fraction of fuel use. Because this approach relies on determining the fractional distribution of an otherwise-determined total, it can be done with incomplete data on fuel use. The estimates will, of course, improve as the fraction of the total fuel use is increased. Figure 7 is based on sales data for most fossil fuel commodities and the CDIAC estimates of total national emissions, and shows how the CO_2 emissions from North America vary at a monthly time scale.

Figure 7. CO_2 emissions from fossil-fuel consumption in North America, by month.

Emissions by Economic Sector

To understand how CO_2 emissions from fossil-fuel use interact in the global and regional cycling of carbon, it is necessary to know the masses of emissions and their spatial and temporal patterns. We have tried to summarize this information here. To understand the trends and the driving forces behind the growth in fossil-fuel emissions, and the opportunities for controlling emissions, it is necessary to look in detail at how the fuels are used. This is the goal of the next four chapters of this report.

Before looking at the details of how energy is used and where CO_2 emissions occur in the economies of North America, however, there are two indices of CO_2 emissions at the national level that provide perspective on the scale and distribution of emissions. These two indices are emissions per capita and emissions per unit of economic activity, the latter generally represented by CO_2 per unit of gross domestic product (GDP). Figure 8 shows the 1950–2002 record of CO_2 emissions per capita for the three countries of North America and, for perspective, includes the same data for the Earth as a whole. Similarly, Table 3 shows CO_2 emissions per unit of GDP for the three countries of North America and for the world total. These are, of course, very complex indices and though they provide some insight they say nothing about the details and the distributions within the means. The data on CO_2 per capita for the 50 U.S. states (Fig. 9) show that values range over a full order of magnitude, differing in complex ways with the structure of the economies and probably with factors like climate, population density, and access to resources (Blasing *et al*., 2005b; Neumayer, 2004).

Figure 8. Per capita emissions of CO_2 from fossil-fuel consumption (and cement manufacture) in the United States, Canada, and Mexico and for the global total of emissions (from Marland *et al.*, 2005).

Table 3. Emissions of CO_2 from fossil-fuel consumption (cement manufacture and gas flaring are not included) per unit of GDP for the United States, Canada, and Mexico and for the global total.

Figure 9. Per capita emissions of CO_2 from fossil-fuel consumption for the 50 U.S. states in 2000.

Chapters 6 through 9 of this report discuss the patterns and trends of CO_2 emissions by sector and the driving forces behind the trends that are observed. Estimating emissions by sector brings special challenges in defining sectors and assembling the requisite data. Readers will find that there is consistency and coherence within each of the following chapters but will encounter difficulty in aggregating or summing numbers across chapters. Different experts use different sector boundaries, different data sources, different conversion factors, etc. Different analysts will find data for different base years and may treat electricity and biomass fuels differently. Despite these differences in accounting procedures, the four chapters accurately characterize the patterns of emissions and the opportunities for controlling the growth in emissions. They reveal that there are major differences between the countries of North America where, for example, the United States derives 51% of its electricity from coal, Mexico gets 68% from petroleum and natural gas, and Canada gets 58% from hydroelectric stations. Partially as a reflection of this difference, 40% of U.S. CO_2 emissions are from enterprises whose primary business is to generate electricity and heat, while this number is only 31% in Mexico and 23% in Canada (for 2003; from IEA, 2005). Chapter 8 reveals that the sectors are not independent as, for example, a change from fuel burning to electricity in an industrial process will decrease emissions from the industrial sector but increase emissions in the electric power sector. The database of the IEA allows us to summarize CO_2 emissions for the three countries according to sectors that closely correspond to the sectoral division of chapters 6 through 9 (Table 4).

Table 4. Percent of CO_2 emissions by sector for 2003.

CONCLUSION

There are a variety of reasons that we want to know the emissions of CO_2 from fossil fuels, there are a variety of ways of coming up with the desired estimates, and there are a variety of ways of using the estimates. By the nature of the process of fossil-fuel combustion, and because of its economic importance, there are reasonably good data over long time intervals that we can use to make reasonably accurate estimates of CO_2 emissions to the atmosphere. In fact, it is the economic importance of fossil-fuel burning that has assured us of both good data on emissions and great challenges in altering the rate of emissions.

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Table 1. A sample of the coefficients used for estimating CO_2 emissions from the amount of fuel burned (from IPCC, 1997)

Fuel	Emissions coefficient (kg C/10^9 J net heating value)
Lignite	27.6
Anthracite	26.8
Bituminous coal	25.8
Crude oil	20.0
Residual fuel oil	21.1
Diesel oil	20.2
Jet kerosene	19.5
Gasoline	18.9
Natural gas	15.3

Table 2. Different estimates (in Mt C) of CO_2 emissions from fossil-fuel consumption for the United States, Canada, and Mexico

Country	1990		1998		2002	
United States	CDIAC	1305	CDIAC	1501	CDIAC	1580
	IEA	1320	IEA	1497	IEA	1545
	USEPA	1316	USEPA	1478	USEPA	1534
Canada	CDIAC	112	CDIAC	119	CDIAC	139
	IEA	117	IEA	136	IEA	145
	Canada	117	Canada	133	Canada	144
Mexico	CDIAC	99	CDIAC	96	CDIAC	100
	IEA	80	IEA	96	IEA	100
	Mexico	81	Mexico	96	Mexico	NA

Notes:

Many of these data were published in terms of the mass of CO_2, and these data have been multiplied by 12/44 to get the mass of carbon for the comparison here.

Values are from CDIAC (Marland *et al.*, 2005), IEA (2005), USEPA (2005), Canada (Environment Canada, 2005), and Mexico (2001).

All data except CDIAC include oxidation of non-fuel hydrocarbons.

All data except IEA include flaring of gas at oil and gas processing facilities.

Table 3. Emissions of CO_2 from fossil-fuel consumption (cement manufacture and gas flaring are not included) per unit of GDP for the United States, Canada, and Mexico for the global total

Country	CO_2 emissions per unit of GDP[a]		
	Year		
	1990	1998	2002
United States	0.19	0.17	0.15
Canada	0.18	0.18	0.16
Mexico	0.13	0.12	0.11
Global total	0.17	0.15	0.14

[a]CO_2 is measured in kg carbon and GDP is reported in 2000 US$ purchasing power parity (from IEA, 2005).

Table 4. Percentage of CO_2 emissions by sector for 2003

Sector	United States	Canada	Mexico	North America
Energy extraction and conversion[a]	46.2	36.2	47.7	45.4
Transportation[b]	31.3	27.7	30.3	31.0
Industry[c]	11.2	16.8	13.6	11.8
Buildings[d]	11.3	19.3	8.4	11.8

[a]The sum of three IEA categories, “public electricity and heat production,” “unallocated autoproducers,” and “other energy industries.” (IEA, 2005).
[b]IEA category “transport.” (IEA, 2005).
[c]IEA category “manufacturing industries and construction.” (IEA, 2005).
[d]IEA category “other sectors.” (IEA, 2005).

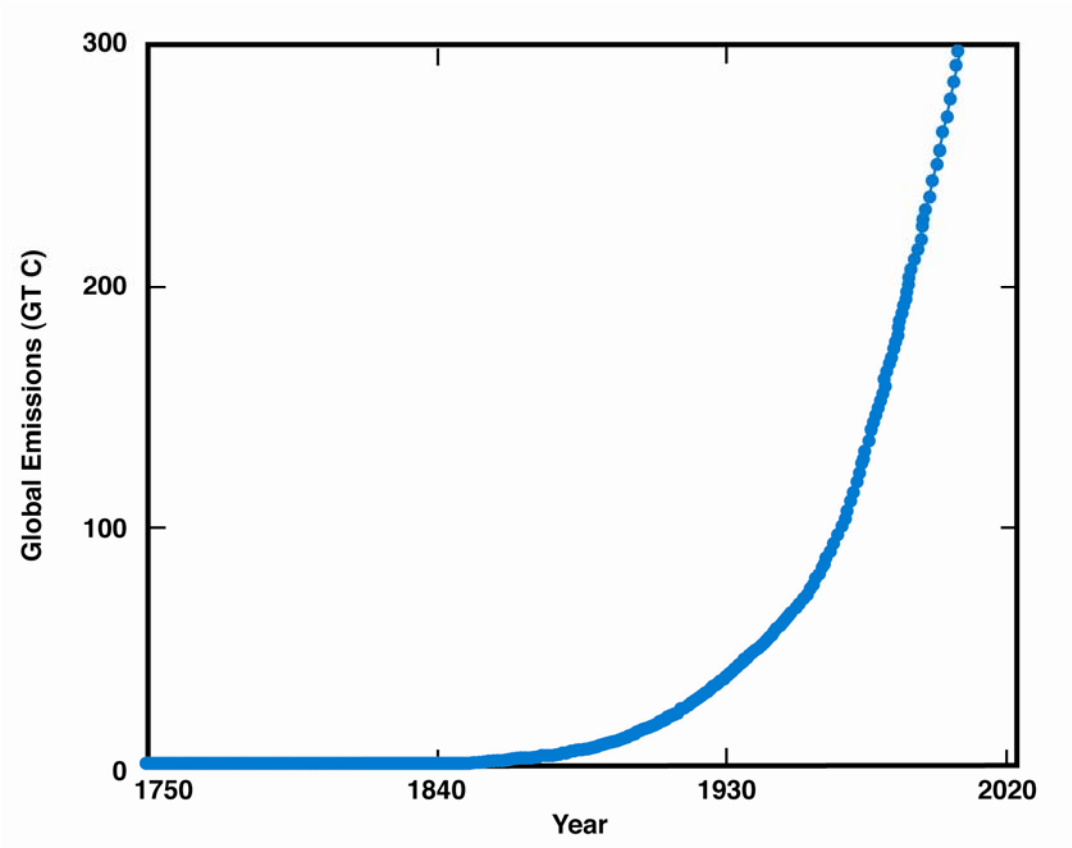


Fig. 1. Cumulative global emissions of CO_2 from fossil-fuel combustion and cement manufacture from 1751 to 2002 (data from Marland *et al.*, 2006).

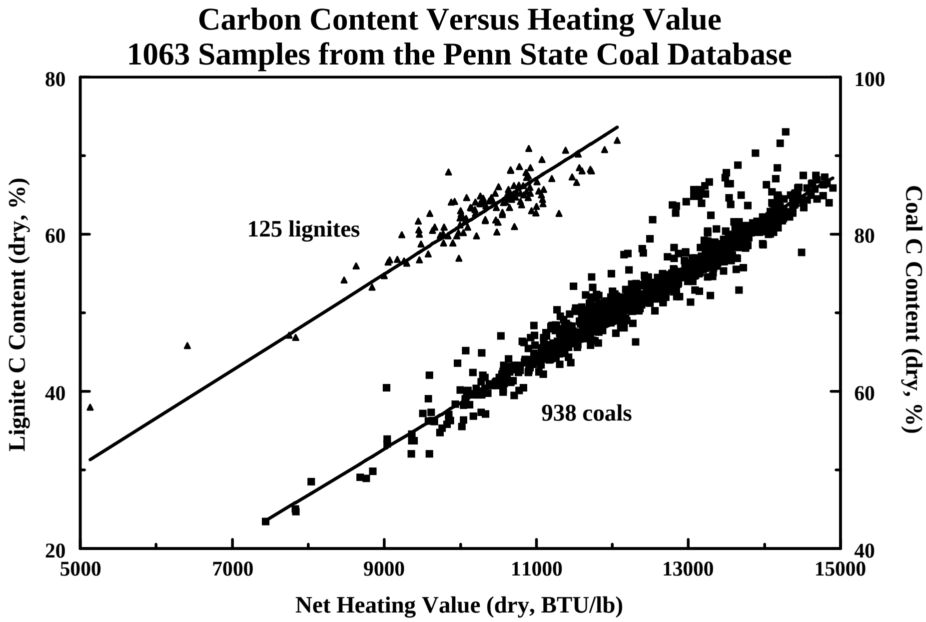


Fig. 2. The carbon content of coal varies with the heat content, shown here as the net heating value. To make them easier to distinguish, data for lignites and brown coals are shown on the left axis, while data for hard coals are offset by 20% and shown on the right axis. Heating value is plotted in the units at which it was originally reported, Btu/lb, where 1 Btu/lb = 2324 J/kg (from Marland *et al*., 1995).

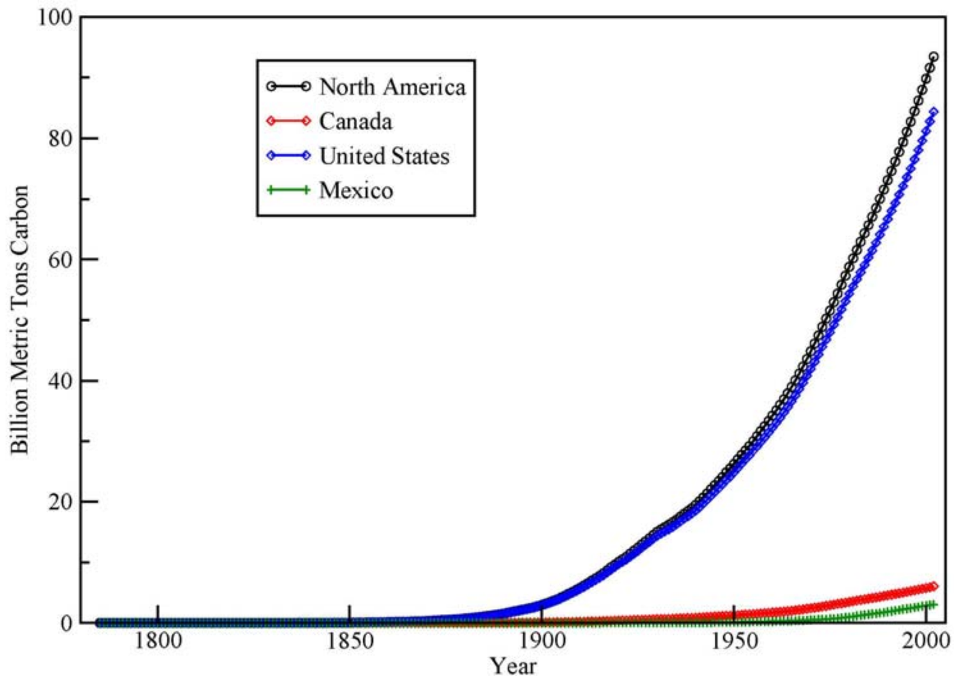
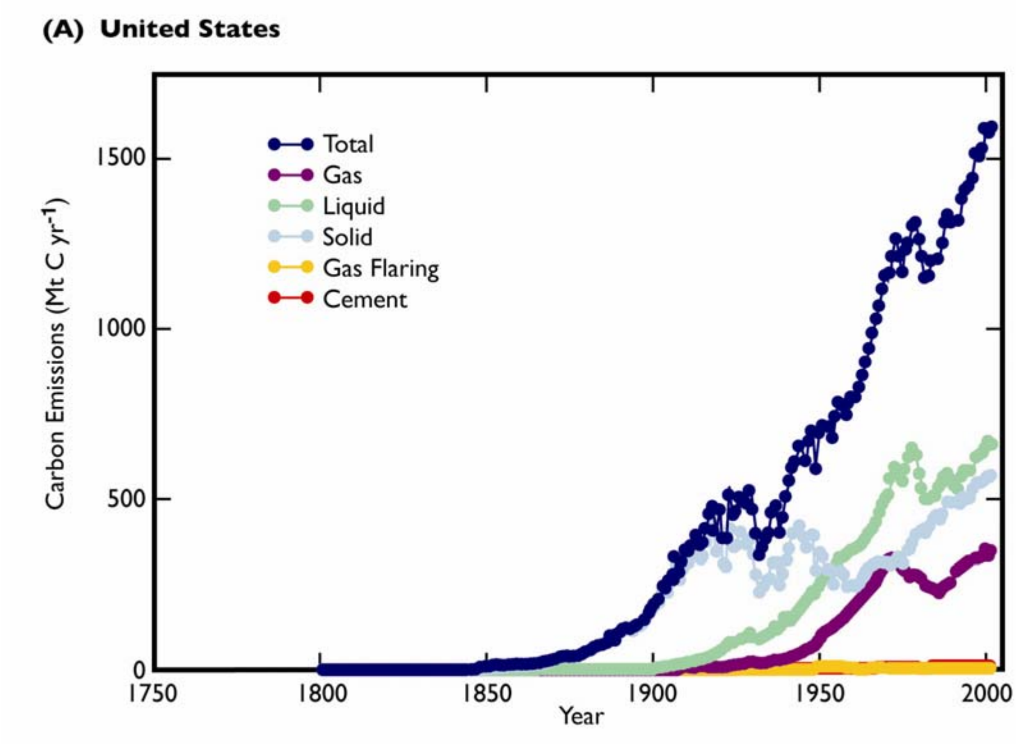


Fig. 3. The cumulative total of CO_2 emissions from fossil-fuel consumption and cement manufacture, as a function of time, for the three countries of North America and for the sum of the three (from Marland *et al.*, 2006).



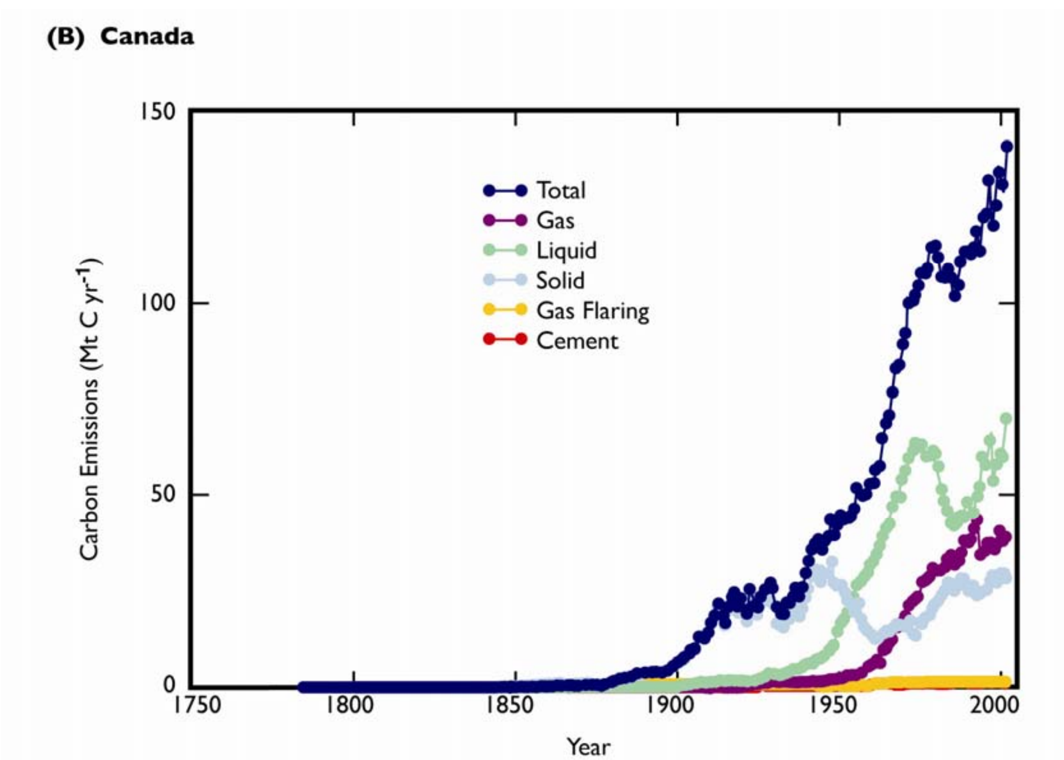
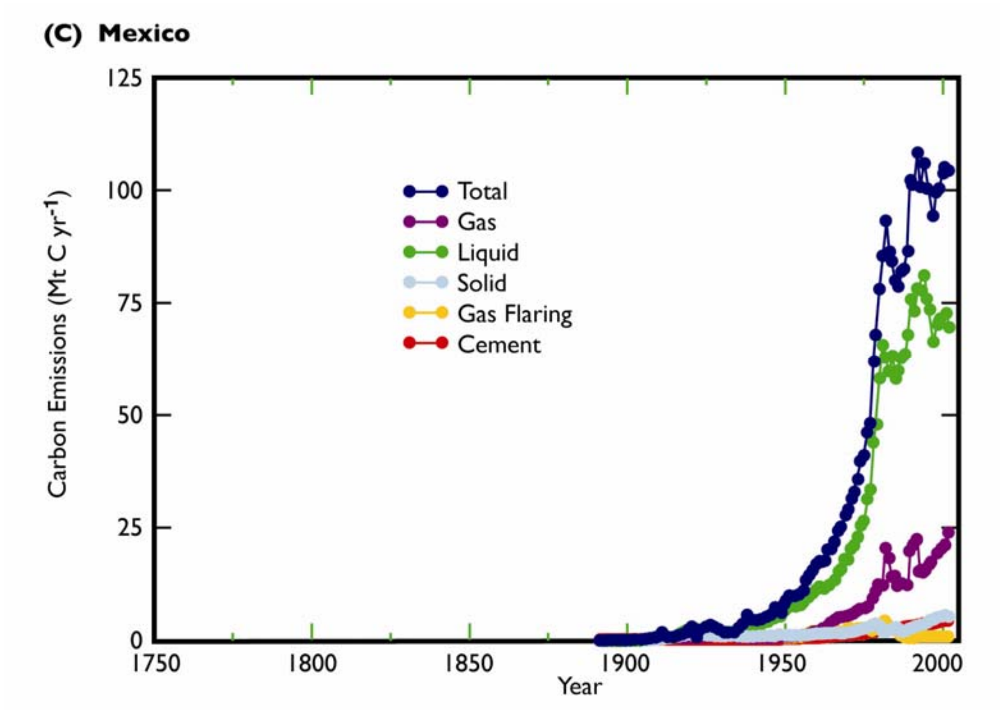


Fig. 4A and 4B. Annual emissions of CO_2 from fossil-fuel use by fuel type. Figure 4A is for the United States, Figure 4B is for Canada, Figure 4C is for Mexico, and Figure 4D is for the sum of the three. Note that in order to illustrate the contributions of the different fuels, the four plots are not to the same vertical scale (from Marland *et al*., 2006).



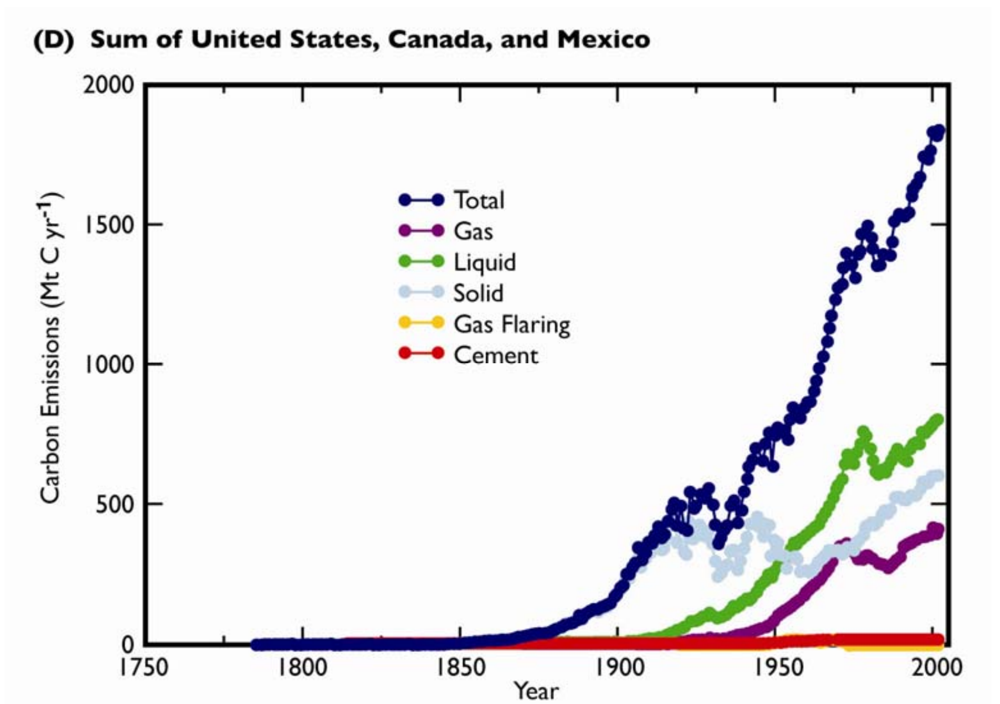


Fig. 4C and 4D. Annual emissions of CO_2 from fossil-fuel use by fuel type.
Figure 4A is for the United States, Figure 4B is for Canada, Figure 4C is for Mexico, and Figure 4D is for the sum of the three. Note that in order to illustrate the contributions of the different fuels, the four plots are not to the same vertical scale (from Marland *et al*., 2006).

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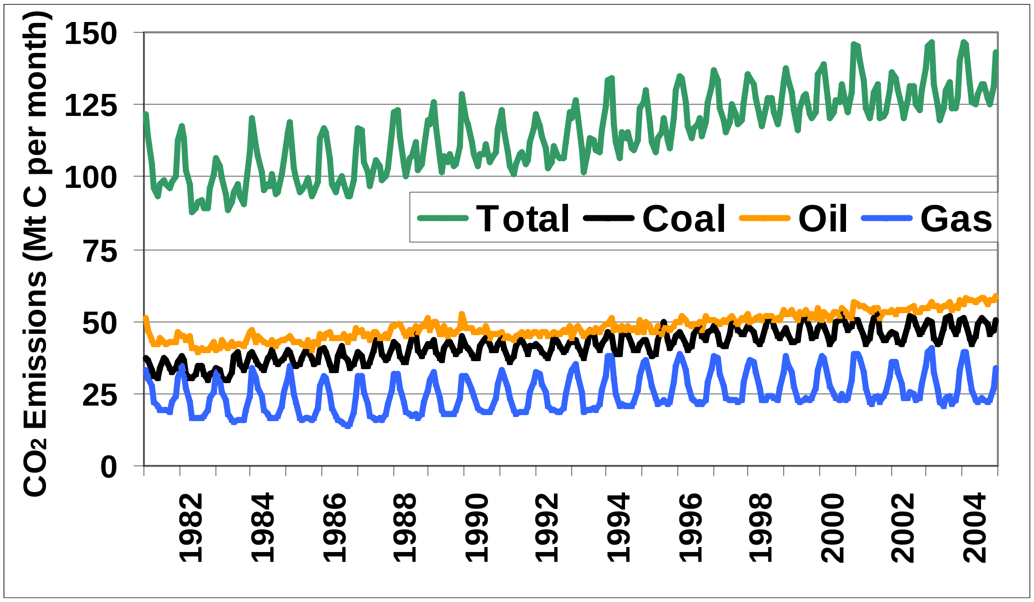

Fig. 5. Emissions of CO_2 from fossil-fuel consumption in the United States, by month. Emissions from cement manufacturing are not included (from Blasing *et al*., 2005a).

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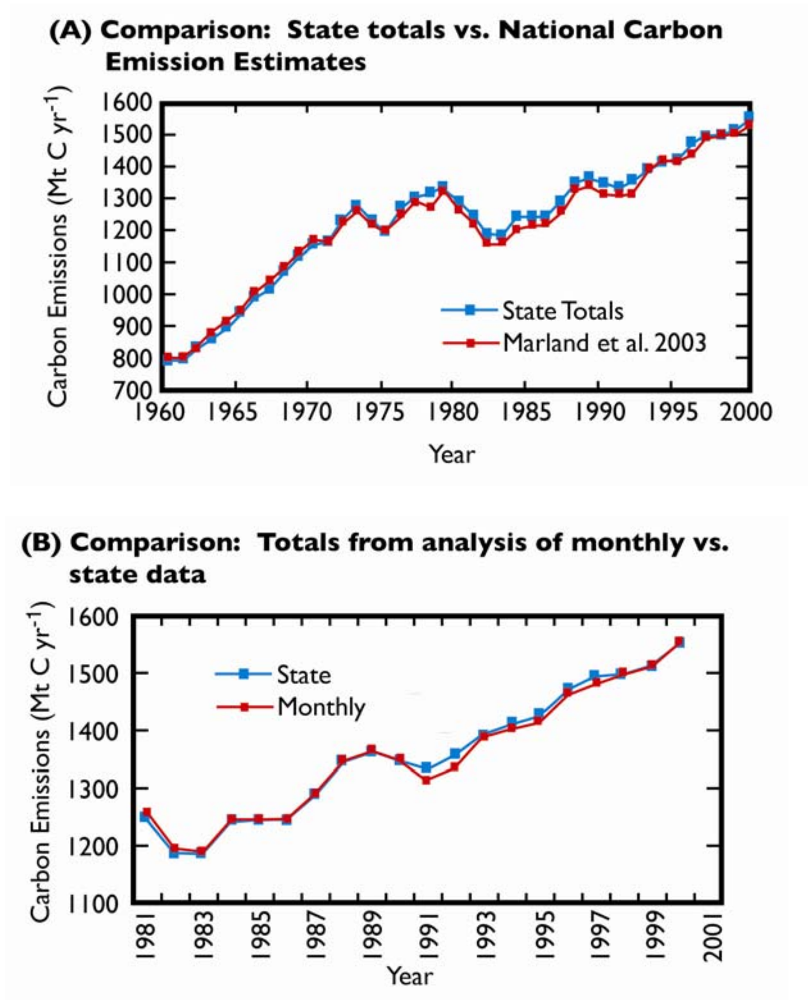

Fig. 6. A comparison of three different estimates of national annual emissions of CO_2 from fossil-fuel consumption in the United States. (6A) Estimates from U.S. Department of Energy data on fuel consumption by state (blue squares) vs. estimates based on UN Statistics Office data on apparent fuel consumption for the full United States (red squares) from Marland *et al*. (2003). (6B) Estimates based on DOE data on fuel consumption in the 50 U.S. states (blue squares) vs. estimates based on national fuel consumption for each of the 12 months (red squares). The state and monthly data include estimates of oxidation of non-fuel hydrocarbon products; the UN-based estimates do not (from Blasing *et al.*, 2005b).

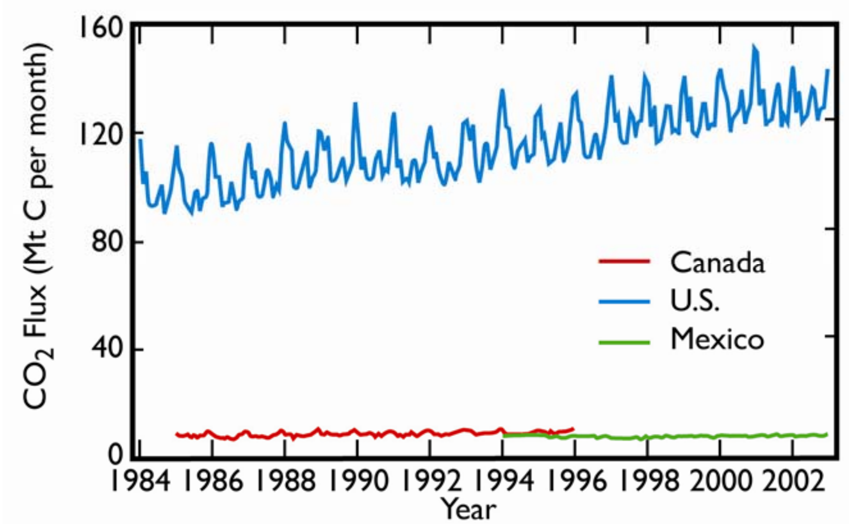

Fig. 7. CO_2 emissions from fossil-fuel consumption in North America, by month. Monthly values are shown where estimates are justified by the availability of monthly data on fuel consumption or sales (from Andres *et al*., 2005).

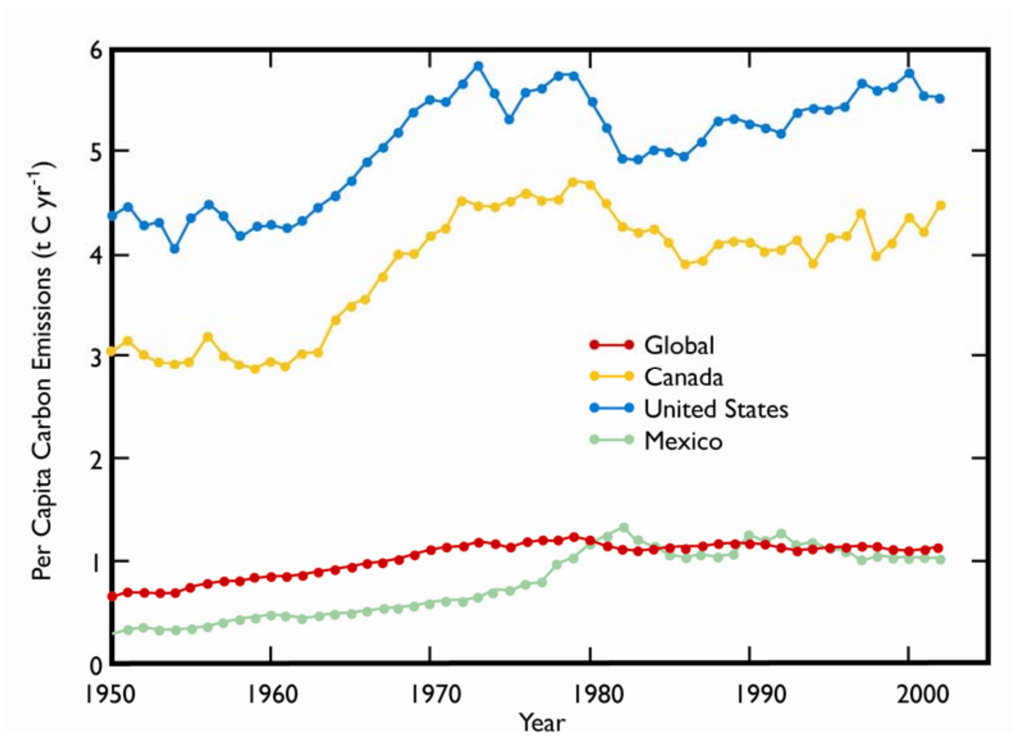

Fig. 8. Per capita emissions of CO_2 from fossil-fuel consumption (and cement manufacture) in the United States, Canada, and Mexico and for the global total of emissions (from Marland *et al*., 2005).

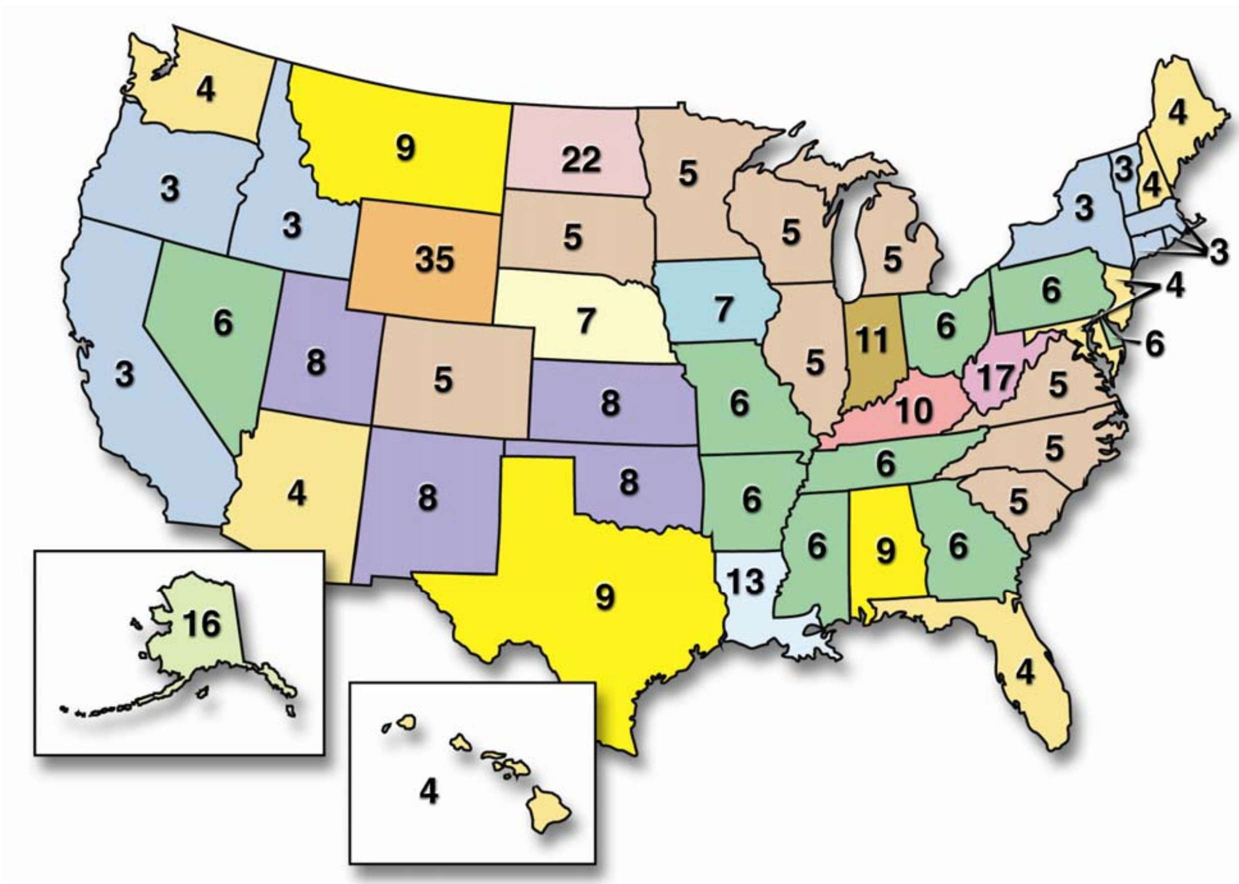

Fig. 9. Per capita emissions of CO_2 from fossil-fuel consumption for the 50 U.S. states in 2000. To demonstrate the range, values have been rounded to whole numbers of metric tons carbon per capita. A large portion of the range for extreme values is related to the occurrence of coal resources and inter-state transfers of electricity (from Blasing *et al.*, 2005b).

Chapter 6. Energy Extraction and Conversion

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KEY FINDINGS

- In recent years, the extraction of primary energy sources and their conversion into energy commodities in North America released on the order of 2700 Mt CO_2 per year to the atmosphere, approximately 40% of total North American emissions in 2003 and 10% of total global emissions. Electricity generation is responsible for a very large share of North America's energy extraction and conversion emissions.
- Carbon dioxide emissions from energy supply systems in North America are currently rising.
- The principal drivers behind carbon emissions from energy supply systems are (1) the growing appetite for energy services, closely related to economic and social progress, and (2) the market competitiveness of fossil energy compared with alternatives.
- Emissions from energy supply systems in North America are projected to increase in the future. Projections vary among the countries, but increases approaching 50% or more in coming decades appear likely. Projections for the United States., for example, indicate that CO_2 emissions from electricity generation alone will rise to above 3300 Mt CO_2 by 2030, an increase of about 45% over emissions in 2004, with three-quarters of the increase associated with greater coal use in electric power plants.
- The prospects for major reductions in CO_2 emissions from energy supply systems in North America appear dependent upon (a) the extent, direction, and pace of technological innovation and (b) whether policy conditions favoring carbon emissions reduction that do not now exist will emerge (Fig. 6-1). In these regards, the prospects are brighter in the long term (e.g., more than several decades in the future) than in the near term.
- Research and development priorities for managing carbon emissions from energy supply systems include, on the technology side, clarifying and realizing potentials for carbon capture and sequestration, and, on the policy side, understanding the public acceptability of policy incentives for reducing dependence on carbon-intensive energy sources.

Figure 6-1. Prospects for carbon emissions from energy extraction and conversion in North America, assuming substantial improvement in energy efficiency.

INTRODUCTION

The energy supply system in North America is a significant part of the North American carbon cycle, because so many of its primary energy resources are fossil fuels, associated with extraction and conversion activities that emit greenhouse gases. This chapter summarizes the knowledge bases related to emissions from energy extraction, energy conversion, and other energy supply activities such as energy movement and energy storage, along with options and measures for managing emissions.

Clearly, this topic overlaps the subject matter of other chapters. For instance, the dividing line between energy conversion and other types of industry is sometimes indistinct. One prominent case is emissions associated with electricity and process heat supply for petroleum refining and other fossil fuel processing – a large share of their total emissions, included in industrial sector emission totals; another example is industrial co-generation as an energy-efficiency strategy. Also, biomass energy extraction/conversion is directly related to agriculture and forestry. Moreover, emission-related policy alternatives for energy supply systems are often directed at both supply and demand responses, involving not only emission reductions but also potential payoffs from efficiency improvements in buildings, industry, and transportation, especially where they reduce the consumption of fossil fuels.

CARBON EMISSIONS INVENTORY

Carbon Emissions from Energy Extraction and Conversion

Carbon emissions from energy resource extraction, conversion into energy commodities, and transmission are one of the "big three" sectors accounting for most of the total emissions from human systems in North America, along with industry and transportation. The largest share of total emissions from energy supply (not including energy end use) is from coal and other fossil fuel use in producing electricity; fossil fuel conversion activities such as oil refining and natural gas transmission and distribution also contribute to this total, but in much smaller amounts. Other emission sources are less well-defined but generally small, such as emissions from oil production and methane from reservoirs established partly to support hydropower production (Tremblay *et al.*, 2004), or from materials production (e.g., metals production) associated with other renewable or nuclear energy technologies. Generally, data on emissions have a relatively low level of uncertainty, although the source materials do not include quantitative estimates of uncertainty.

Data on emissions from energy supply systems are unevenly available for the countries of North America. Most emission data sets are organized by fuel consumed rather than by consuming sector, and countries differ in sectors identified and the units of measurement. As a result, inventories are reported in this chapter by country in whatever forms are available rather than constructing a North American inventory that could not be consistent across all three major countries. It is worth noting that Canada and Mexico export energy supplies to the United States; therefore, some emissions from energy *supply* systems in these countries are associated with energy *uses* in the United States.

Canada

Canada is the world's fifth-largest energy producing country, a significant exporter of both natural gas and electricity to the United States. In Alberta, which produces nearly two-thirds of Canada's energy, energy accounts for about one-quarter of the province's economic activity; its oil sands are estimated to have more potential energy value than the remaining oil reserves of Saudi Arabia (DOE, 2004). Although Canada has steadily reduced its energy and carbon intensities since the early 1970s, its overall energy intensity remains high—in part due to its prominence as an energy producer—and total greenhouse gas emissions have grown by 9% since 1990. As of 2003, greenhouse gas emissions in Mt CO_2 equivalents were 134 for electricity and heat generation and 71 for petroleum refining and upgrading and other fossil fuel production (Environment Canada, 2003). Although the mix of CO_2 and CH_4 in these figures is unclear, the carbon emission equivalent is probably in a 60-80 Mt C range.

Mexico

Mexico is one of the largest sources of energy-related greenhouse gas emissions in Latin America, although its per capita emissions are well below the per capita average of industrialized countries. The first large oil-producing nation to ratify the Kyoto Protocol, it has promoted shifts to natural gas use to reduce greenhouse gas emissions. The most recent emission figures are from the country's Second National Communication to the UN United Nations Framework Convention on Climate Change in 2001, which included relatively comprehensive data from 1996 and some data from 1998. In 1998, total CO_2 emissions from "energy industries" were 47.3 Mt CO_2 (13 Mt C); from electricity generation they totaled 101.3 Mt CO_2 (27.6 Mt C), and "fugitive" emissions from oil and gas production and distribution were between 1.9 and 2.6 Mt of CH_4 (1.4 – 2 Mt C), depending on the estimated "emission factor" (Government of Mexico, 2001).

United States

The United States is the largest national emitter of greenhouse gases in the world, and CO_2 emissions associated with electricity generation in 2004 account for 2299 Mt of CO_2 (627 Mt C), or 39% of a national total of 5890 (EIA, 2006a). Greenhouse gases are also emitted from oil refining, natural gas transmission, and other fossil energy supply activities, but apart from energy consumption figures included in industry sector calculations these emissions are relatively small compared with electric power plant emissions. For instance, emissions from petroleum consumed in refining processes in the U.S. are about 40 Mt C per year (EIA, 2004: Ch. 2), while fugitive emissions from gas transmission and distribution pipelines in the U.S. are about 2.2 Mt C yr^{-1} (ORNL estimate). On the other hand, a study of greenhouse gas emissions from a six-county area in southwestern Kansas found that compressor stations for natural gas pipeline systems are a significant source of emissions at that local scale (AAG, 2003).

Carbon Sinks Associated with Energy Extraction and Conversion

Generally, energy supply in North America is based heavily on mining hydrocarbons from carbon sinks accumulated over millions of years; but current carbon sequestration occurs in plant growth, including the cultivation of feedstocks for bioenergy production. Limited strictly to energy sector applications, the total contribution of these *sinks* to the North American carbon cycle is relatively small, while other aspects of bioenergy development are associated with carbon *emissions*.

TRENDS AND DRIVERS

Three principal drivers are behind carbon emissions from energy extraction and conversion.

(1) The growing global and national appetite for energy services such as comfort, convenience, mobility, and labor productivity, so closely related to progress with economic and social development and the quality of life (Wilbanks, 1992). Globally, the challenge is to increase total energy *services* (not necessarily *supplies*) over the next half-century by a factor of at least three or four—more rapidly than overall economic growth—while reducing environmental impacts from the associated supply systems (NAS, 1999). Mexico shares this need, while increases in Canada and the United States are likely to be more or less proportional to rates of economic growth.

(2) The market competitiveness of fossil energy sources compared with supply- and demand-side alternatives. Production costs of electricity from coal, oil, or natural gas at relatively large scales are currently lower than other sources besides large-scale hydropower, and production costs of liquid and gas fuels are currently far lower than other sources, though rising. This is mainly due to the fact that the energy density and portability of fossil fuels is as yet unmatched by other energy sources, and in some cases policy conditions reinforce fossil fuel use. These conditions appear likely to continue for some years. In many cases, the most cost-competitive alternative to fossil fuel production and use is not alternative supply sources but efficiency improvement.

(3) Enhanced future markets for alternative energy supply sources. In the longer run, however, emissions from energy supply systems may—and in fact are likely to—begin to decline as alternative technology options are developed and/or improved. Other possible driving forces for attention to alternatives to fossil fuels, at least in the mid to longer term, include the possibility of shrinking oil and/or gas reserves and changes in attitudes toward energy policy interventions.

Given the power of the first two of these drivers, total carbon emissions from energy extraction and conversion in North America are currently rising (e.g., Fig. 6-2). National trends and drivers are as follows. As is always the case, projections of the future involve higher levels of uncertainty than measurements of the present, but source materials do not include quantitative estimates of uncertainties associated with projections of future emissions.

Figure 6-2. U.S. carbon dioxide emissions from electricity generation, 1990–2004.

Canada

Canada has ratified the Kyoto Protocol, and it is seeking to meet the Kyoto target of CO_2 emission reduction to 6% below 1990 levels. Of these reductions, 25% are to be through domestic actions and 75% through market mechanisms such as purchases of carbon credits (Government of Canada, 2005). Domestic actions will include a significant reduction in coal consumption. Available projections, however, indicate a total national increase of emissions in CO_2 equivalent of 36.1% by 2020 from 1990 levels (Environment Canada, 2005). Emissions from electricity generation could increase 2000–2020 by as much as two-thirds, while emissions from fossil fuel production would remain relatively stable (although substantial expansion of oil sands production could be a factor).

Mexico

It has been estimated that total Mexican CO_2 emissions will grow 69% by 2010, although mitigation measures could reduce this rate of growth by nearly half (Pew Center, 2002). Generally, energy sector emissions in Mexico vary in proportion to economic growth (e.g., declining somewhat with a recession in 2001), but such factors as a pressing need for additional electricity supplies, calling for more than doubling production capacity between 1999 and 2008, could increase net emissions while a national strategy to promote greater use of natural gas (along with other policies related in part to concerns about emissions associated with urban air pollution) could reduce emissions compared with a reference case (EIA, 2005).

United States

The Energy Information Administration (EIA, 2006a) projects that CO_2 emissions from electricity generation in the United States will rise between 2004 and 2030 from about 2299 (627 Mt C) to more than 3300 Mt (900 Mt C), an increase of about 45%, with three-quarters of the increase associated with greater coal use in electric power plants. EIA projects that technology advances could lower emissions by as much as 9%. Projections of other emissions from energy supply systems appear to be unavailable, but emissions could be expected to rise at a rate just below the rate of change in product consumption in the U.S. economy.

OPTIONS FOR MANAGEMENT OF EMISSIONS FROM ENERGY EXTRACTION AND CONVERSION

Few aspects of the carbon cycle have received more attention in the past several decades than emissions from fossil energy extraction and conversion. As a result, there is a wide array of technology and policy options, many of which have been examined in considerable detail, although there is not a strong consensus on courses of action.

Technology Options

Technology options for reducing energy-supply-related emissions (other than reduced requirements due to end-use efficiency improvements) consist of

- reducing emissions from fossil energy extraction, production, and movement (e.g., for electricity generation, improving the efficiency of existing power plants or moving toward the use of lower-emission technologies such as coal gasification–combined cycle generation facilities) and
- shifting from fossil energy sources to other energy sources [e.g., energy from the sun (renewable energy) or from the atom (nuclear energy)].

The most comprehensive description of emission-reducing and fuel switching technologies and their potentials is the U.S. Climate Change Technology Program (CCTP) draft *Strategic Plan* (CCTP, 2005), especially Chapters 5 (energy supply) and 6 (capturing and sequestering CO_2)—see also National Laboratory Directors (1997). The CCTP report focuses on five energy supply technology areas: low-emission fossil-based fuels and power, hydrogen as an energy carrier, renewable energy and fuels, nuclear fission, and fusion energy.

There is a widespread consensus that no one of these options, nor one family of options, is a good prospect to stabilize greenhouse gas emissions from energy supply systems, nationally or globally, because each faces daunting constraints (Hoffert *et al.*, 2002). An example is possible physical and/or technological limits to effective global "decarbonization" (i.e., reducing the use of carbon-based energy sources as a proportion of total energy supplies), including renewable or other non-fossil sources of energy use at scales that would dramatically change the global carbon balance between now and 2050. One conclusion is that "the disparity between what is needed and what can be done without great compromise may become more acute."

Instead, progress with technologies likely to be available in the coming decades may depend on adding together smaller "wedges" of contributions by a variety of resource/technology combinations (Pacala and Socolow, 2004), each of which may be feasible if the demands upon it are moderate. If many such contributions can be combined, the total effect could approach requirements for even relatively ambitious carbon stabilization goals, at least in the first half of the century, although each contribution would need to be economically competitive with current types of fossil energy sources.

A fundamental question is whether prospects for significant decarbonization depend on the emergence of new technologies, in many cases requiring advances in science. For instance, efforts are being made to develop economically affordable and socially acceptable options for large-scale capture of carbon from fossil fuel streams—with the remaining hydrogen offering a clean energy source—and sequestration of the carbon in the ground or the oceans. This approach is known to be technologically feasible (and is being practiced commercially in the North Sea), and recent assessments suggest that it may have considerable promise (e.g., IPCC, 2006). If so, there is at least some chance that fossil energy sources may be used to provide energy services in North America and the world in large quantities in the mid to longer terms without contributing to a carbon cycle imbalance.

What can be expected from technology options over the next quarter to half a century is a matter of debate, partly because the pace of technology development and use depends heavily on policy conditions. Chapter 3 in the CCTP draft *Strategic Plan* (2005) shows three advanced technology scenarios drawn from work by the Pacific Northwest National Laboratory, varying according to carbon constraints. Potential contributions to global emission reduction by energy supply technology initiatives between 2000 and 2100 range from about 25 Gt C equivalent to nearly 350 Gt, which illustrates uncertainties related to both science and policy issues. Carbon capture and storage, along with terrestrial sequestration, could add reductions between about 100 and 325 Gt C. It has been suggested, however, that significantly decarbonizing energy systems by 2050 could require massive efforts on a par with the Manhattan project or the Apollo space program (Hoffert *et al.*, 2002).

Estimated costs of potential technology alternatives for reducing greenhouse gas emissions from energy supply systems are summarized after the following discussion of policy options, because cost estimates are generally based on assumptions about policy interventions.

Policy Options

Policy options for carbon emission reduction from energy supply systems revolve around either *incentives* or regulatory *requirements* for such reductions. Generally, interventions may be aimed at (a) shaping technology choice and use or (b) shaping technology development and supply. Many of the policy options are aimed at encouraging end-use efficiency improvement as well as supply-side emission reduction.

Options for intervening to change the relative attractiveness of available energy supply technology alternatives include appealing to voluntary action (e.g., improved consumer information, "green power"), a variety of regulatory actions (e.g., mandated purchase policies such as energy portfolio standards), carbon emission rights trading (where emission reduction would have market value), technology/product standards, production tax credits for non-fossil energy production, tax credits for alternative energy use, and carbon emission taxation or ceilings. Options for changing the relative attractiveness of investing in carbon-emission-reducing technology development and dissemination include tax credits for certain kinds of energy R&D, public-private sector R&D cost sharing, and electric utility restructuring. For a more comprehensive listing and discussion, see Chapter 6 in IPCC (2002, Chapter 6).

In some cases, perceptions that policies and market conditions of the future will be more favorable to emission reduction than at present are motivating private industry to consider investments in technologies whose market competitiveness would grow in such a future. Examples include the CO_2 Capture Project and industry-supported projects at MIT, Princeton, and Stanford.

Most estimates of the impacts of energy policy options on greenhouse gas emissions do not differentiate the contributions from energy supply systems from the rest of the energy economy [e.g., Interlaboratory Working Group (IWG), 1997; IWG, 2000; IPCC, 2001; National Commission on Energy Policy, 2004; also see OTA, 1991, and NAS, 1992]. For instance the IWG (1997) considered effects of $25 and $50 per ton carbon emission permits on both energy supply and use, while IWG considered fifty policy/technology options (IWG, 2000; also see IPCC, 2001), most of which would affect both energy supply and energy use decisions.

Estimated Costs of Implementation

Estimating the costs of emission reduction associated with the implementation of various technology and policy options for energy supply and conversion systems is complicated by several realities. First, many estimates are aggregated for the United States or the world as a whole, without separate estimates for the energy extraction and conversion sector. Second, estimates differ in the scenarios considered, the modeling approaches adopted, and the units of measure that are used.

More specifically, estimates of costs of emission reduction vary widely according to assumptions about such issues as how welfare is measured, ancillary benefits, and effects in stimulating technological innovation; and therefore any particular set of cost estimate includes considerable uncertainty. According to IWG (2000), benefits of emission reduction would be comparable to costs, and the National Commission on Energy Policy (2004) estimates that their recommended policy initiatives would be, on the whole, revenue-neutral with respect to the federal budget. Other participants in energy policymaking, however, are convinced that truly significant carbon emission reductions would have substantial economic impacts (GAO, 2004).

Globally, IPCC (2001) projected that total CO_2 emissions from energy supply and conversion could be reduced in 2020 by 350 to 700 Mt C equivalents per year, based on options that could be adopted through the use of generally accepted policies, generally at a positive direct cost of less than U.S.$100 per t C equivalents. Based on DOE/EIA analyses in 2000, this study includes estimates of the cost of a range of specific emission-reducing technologies for power generation, compared with coal-fired power, although the degree of uncertainty is not clear. Within the United States, the report estimated that the cost of emission reduction per metric ton of carbon emissions reduced would range from –$170 to +$880, depending on the technology used. Marginal abatement costs for the total United States economy, in 1990 U.S. dollars per metric ton carbon, were estimated by a variety of models compared by the Energy Modeling Forum at $76 to $410 with no emission trading, $14 to $224 with Annex I trading, and $5 to $123 with global trading.

Similarly, the National Commission on Energy Policy (2004) considered costs associated with a tradable emission permit system that would reduce United States national greenhouse gas emission growth from 44% to 33% from 2002 to 2025, a reduction of 760 Mt CO_2 (207 Mt C) in 2025 compared with a reference case. The cost would be a roughly 5% increase in total end-use expenditures compared with the reference case. Electricity prices would rise by 5.4% for residential users, 6.2% for commercial users, and 7.6% for industrial users.

The IWG (2000) estimated that a domestic carbon trading system with a $25/t C permit price would reduce emissions by 13% compared with a reference case, or 230 Mt CO_2 (63 Mt C), while a $50 price would reduce emissions by 17 to 19%, or 306 to 332 Mt CO_2 (83-91 Mt C). Both cases assume a doubling of United States government appropriations for cost-shared clean energy research, design, and development.

For carbon capture and sequestration, IPCC (2006) concluded that this option could contribute 15 to 55% to global mitigation between now and 2100 if technologies develop as projected in relatively optimistic scenarios and very large-scale geological carbon sequestration is publicly acceptable. Under these assumptions, the cost is projected at \$30 to \$70/t CO_2. With less optimistic assumptions, the cost could rise to above \$200/t.

Net costs to the consumer, however, are balanced in some analyses by benefits from advanced technologies which are developed and deployed on an accelerated schedule due to policy interventions and changing public preferences. The U.S. Climate Change Technology Program (2005: pp. 3–19) illustrates how costs of achieving different stabilization levels can conceivably be reduced substantially by the use of advanced technologies, and IWG (2000) estimates that net end-user costs of energy can actually be reduced by a domestic carbon trading system if it accelerates the market penetration of more energy-efficient technologies.

In many cases, however, discussions of the promise of technology options are not associated with cost estimates. Economic costs of energy are not one of the drivers of the IPCC SRES scenarios, and such references as Hoffert *et al.* (2002) and Pacala and Socolow (2004) are concerned with technological potentials and constraints as a limiting condition on market behavior rather than with comparative costs and benefits of particular technology options at the margin.

Summary

In terms of prospects for major emission reductions from energy extraction and conversion in North America, the key issues appear to be the extent, direction, and pace of technological innovation and the likelihood that policy conditions favoring carbon emissions reduction that do not now exist will emerge if concerns about carbon cycle imbalances grow. In these regards, the prospects are brighter in the long term (e.g., more than several decades in the future) than in the near term. History suggests that technology solutions are usually easier to implement than policy solutions, but it is possible that observed impacts of carbon cycle imbalances might change the political calculus for policy interventions in the future.

RESEARCH AND DEVELOPMENT NEEDS

If it is possible that truly effective management of carbon emissions from energy supply and conversion systems cannot be realized with the current portfolio of technology alternatives under current policy conditions, then research and development needs and opportunities deserve expanded attention and support (e.g., National Commission on Energy Policy, 2004). If so, the priorities include:

Technology. Several objectives seem to be especially relevant to carbon management potentials:

- clarifying and realizing potentials for carbon capture and sequestration;
- clarifying and realizing potentials of affordable renewable energy systems at a relatively large scale;

- addressing social concerns about the nuclear energy fuel cycle, especially in an era of concern about terrorism;
- improving estimates of economic costs and emission reduction benefits of a range of energy; technologies across a range of economic, technological, and policy scenarios; and
- "Blue Sky" research to develop new technology options and families, such as innovative approaches for energy from the sun and from biomass, including possible applications of nanoscience (Caldeira *et al*., 2005; Lewis, 2005).

Policy. Research and development can also be applied to policy options in order to enlarge their knowledge bases and explore their implications. For instance, research priorities might include learning more about:

- the public acceptability of policy incentives for reducing dependence on energy sources associated with carbon emissions,
- possible effects of incentives for the energy industry to increase its support for pathways not limited to fossil fuels,
- approaches toward a more distributed electric power supply enterprise in which certain renewable (and hydrogen) energy options might be more attractive, and
- transitions from one energy system/infrastructure to another.

In these ways, technology and policy advances might be combined with multiple technologies to transform the capacity to manage carbon emissions from energy supply systems, if that is a high priority for North America.

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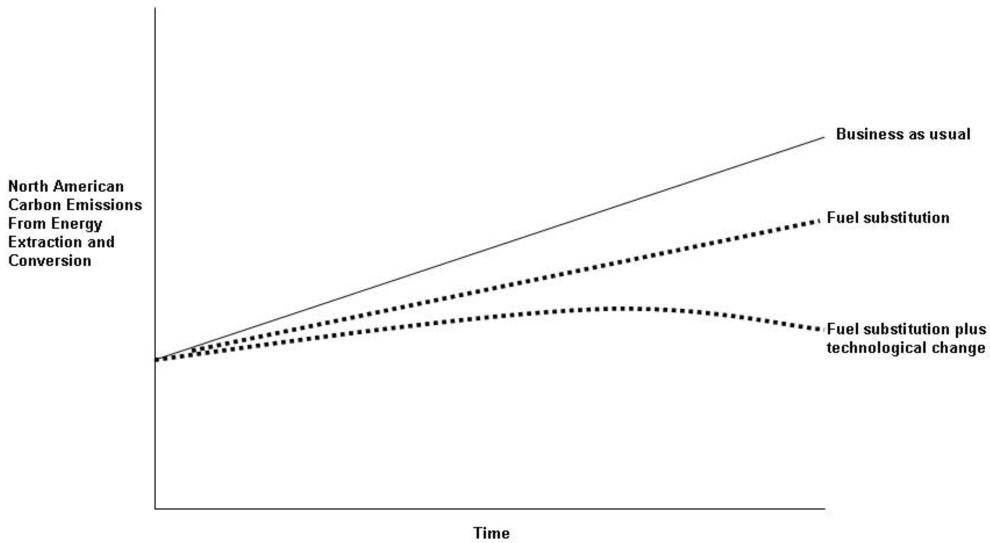

Fig. 6-1. Prospects for carbon emissions from energy extraction and conversion in North America, assuming substantial improvements in energy efficiency.

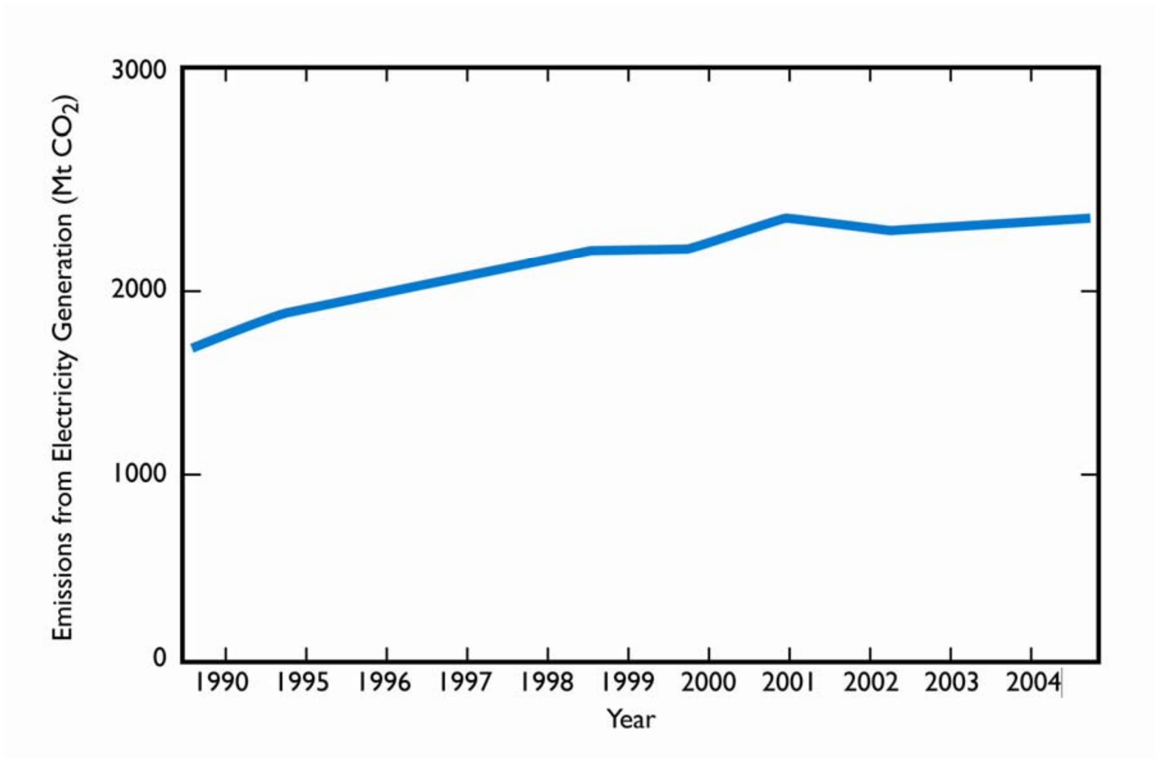

Fig. 6-2. U.S. carbon dioxide emissions from electricity generation, 1990–2004.

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Chapter 7. Transportation

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KEY FINDINGS

- The transportation sector of North America released 587 Mt of C into the atmosphere in 2003, nearly all in the form of carbon dioxide from combustion of fossil fuels. This comprises 37% of the total CO_2 emissions from worldwide transportation activity which, in turn, accounts for about 22% of total global CO_2 emissions.
- Transportation energy use in North America and the associated C emissions have grown substantially and relatively steadily over the past 40 years. Growth has been most rapid in Mexico, the country most dependent upon road transport.
- Carbon emissions by transport are determined by the levels of passenger and freight activity, the shares of transport modes, the energy intensity of passenger and freight movements, and the carbon intensity of transportation fuels. The growth of passenger and freight activity is driven by population, per capita income, and economic output.
- Chiefly as a result of economic growth, energy use by North American transportation is expected to increase by 46% from 2003 to 2025. If the mix of fuels is assumed to remain the same, carbon dioxide emissions would increase from 587 Mt C in 2003 to 859 Mt C in 2025. Canada, the only one of the three countries in North America to have committed to specific GHG reduction goals, is expected to show the lowest rate of growth in C emissions.
- The most widely proposed options for reducing the carbon emissions of the North American transportation sector are increased vehicle fuel economy, increased prices for carbon-based fuels, liquid fuels derived from biomass, and in the longer term, hydrogen produced from renewables, nuclear energy, or from fossil fuels with carbon sequestration. Biomass fuels appear to be a promising near- and long-term option, while hydrogen could become an important energy carrier after 2025.
- After the development of advanced energy efficient vehicle technologies and low-carbon fuels, the most pressing research need in the transportation sector is for comprehensive, consistent, and rigorous assessments of carbon emissions mitigation potentials and costs for North America.

Transportation is the largest source of carbon emissions among North American energy end uses. This fact reflects the vast scale of passenger and freight movements in a region that comprises one-fourth of the global economy, as well as the dominance of relatively energy-intensive road transport and the near total dependence of North American transportation systems on petroleum as a source of energy. If present trends continue, carbon emissions from North American transportation are expected to increase by more than one-half by 2050. Options for mitigating carbon emissions from the transportation sector like increased vehicle fuel economy and biofuels could offset the expected growth in transportation activity. However, at present only Canada has committed to achieving a specific reduction in future greenhouse gas emissions: 6% below 1990 levels by 2012 (Government of Canada, 2005).

INVENTORY OF CARBON EMISSIONS

Worldwide, transportation produced about 22% (1.5 Gt C) of total global carbon dioxide emissions from the combustion of fossil fuels (6.6 Gt C) in 2000 (page 3-1 in U.S. EPA, 2005; Marland, Boden and Andres, 2005). Home to 6.7% of the world's 6.45 billion people and source of 24.8% of the world's $55.5 trillion gross world product (CIA, 2005), North America produces 37% of the total carbon emissions from worldwide transportation activity (Fulton and Eads, 2004).

Transportation activity is driven chiefly by population, economic wealth, and geography. Of the approximately 435 million residents of North America, 68.0% reside in the United States, 24.5% in Mexico, and 7.5% in Canada. The differences in the sizes of the three countries' economies are far greater. The United States is the world's largest economy, with an estimated gross domestic product (GDP) of $11.75 trillion in 2004. Although Mexico has approximately three times the population of Canada, its GDP is roughly the same, $1.006 trillion compared to $1.023 trillion (measured in 2004 purchasing power parity dollars). With the largest population and largest economy, the United States has by far the largest transportation system. The United States accounted for 87% of the energy used for transportation in North America in 2003, Canada for 8%, and Mexico 5% (Fig. 7-1) (see Table 4-1 in NATS, 2005). These differences in energy use are directly reflected in carbon emissions from the three countries' transportation sectors (Table 7-1).

Figure 7-1. Transportation energy use in North America, 1990–2003.

Table 7-1. Carbon emissions from transportation in North America in 2003.

Transportation is defined as private and public vehicles that move people and commodities (U.S. EPA, 2005, p. 296). This includes automobiles, trucks, buses, motorcycles, railroads and railways (including streetcars and subways), aircraft, ships, barges, and natural gas pipelines. This definition excludes petroleum, coal slurry, and water pipelines, as well as the transmission of electricity, although many countries consider all pipelines part of the transport sector. It also generally excludes mobile sources not engaged in transporting people or goods, such as construction equipment, and on-farm agricultural equipment. In addition, carbon emissions from international bunker fuel use in aviation and waterborne transport, though considered part of transport emissions, are generally accounted for separately from a nation's domestic greenhouse gas inventory. In this chapter, however, they are included as are carbon emissions from military transport operations because they are real inputs to the carbon cycle. Upstream, or well-to-tank, carbon emissions are not included with transportation end-use, nor are end-of-life emissions produced in the disposal or recycling of materials used in transportation vehicles or infrastructure because these carbon flows are in the domain of other chapters. These two categories of emissions typically comprise 20–30% of total life cycle emissions for transport vehicles (see Table 5.4 in Weiss *et al*., 2000). In the future, it is likely that upstream carbon emissions will be of greater importance in determining the total emissions due to transportation activities.

In addition to carbon dioxide, the combustion of fossil fuels by transportation produces other greenhouse gases including methane (CH_4), nitrous oxide (N_2O), carbon monoxide (CO), nitrogen oxides (NO_x), and non-methane volatile organic compounds (VOCs). Those containing carbon are generally oxidized in the atmosphere to ultimately produce CO_2. However, the quantities of non-CO_2 gases produced by transportation vehicles are very minor sources of carbon in comparison to the volume of CO_2 emissions. For example, North American emissions of CH_4 by transportation accounted for only 0.03% of total transportation carbon emissions in 2003. This chapter will therefore address primarily the carbon dioxide emissions from transportation activities (methane emissions are included in the totals presented in Table 7-1, but they are not included in any other estimates presented in this chapter).

Four main sources of information on carbon emissions are used in this chapter. The estimates shown in Table 7-1 were obtained from the greenhouse gas inventory reports of the three countries, estimated by environmental agencies in accordance with IPCC guidelines. As Annex 1 countries, Canada and the United States are obliged to compile annual inventories under IPCC guidelines. As a non-Annex 1 country, Mexico is not. These inventories are the most authoritative sources for estimates of carbon emissions. The inventory reports, however, do not generally provide estimates of associated energy use and the most recent inventory data available for Mexico are for 2001. Estimates of energy use and carbon emissions produced by the countries' energy agencies are also used in this chapter to illustrate the relationship between energy use and carbon emissions and its historical trends. There are some minor differences between the carbon emissions estimates from the two sources. Finally, future projections of carbon emissions for North America to 2025 were taken from the U.S. Energy Information's Annual Energy Outlook 2005, and projections to 2050 were taken from the World Business Council on Sustainable Development's Sustainable Mobility Project (WBCSD, 2004).

Fuels Used in Transportation

Virtually all of the energy used by the transport sector in North America is derived from petroleum, and most of the remainder comes from natural gas (Table 7-2). In the United States, 96.3% of total transportation energy is obtained by combustion of petroleum fuels (U.S. DOE/EIA, 2005a). Most of the non-petroleum energy is natural gas used to power natural gas pipelines (2.5%, 744 PJ). During the past two decades, ethanol use as a blending component for gasoline has increased from a negligible amount to 1.1% of transportation energy use (312 PJ). Electricity, mostly for passenger rail transport, comprises only 0.1% of U.S. transport energy use. This pattern of energy use has persisted for more than half a century.

Table 7-2. Summary of North American transport energy use and carbon dioxide emissions in 2003 by fuel type.

The pattern of energy sources is only a little different in Mexico where 96.2% of transportation energy use is gasoline, diesel, or jet fuel: 3.4% is liquefied petroleum gas (LPG), and less than 0.2% is electricity (Rodríguez, 2005). In Canada, natural gas use for natural gas pipelines accounts for 7.5% of transport energy use, 91.8% is petroleum, 0.5% is propane (LPG) and only 0.1% is electricity (see Table 1 in NRCan, 2006).

Mode of Transportation

Mode of transportation refers to how people and freight are moved about, whether by road, rail, or air, in light or heavy vehicles. Carbon dioxide emissions from the North American transportation sector are summarized by mode in Table 7-3, and the distribution of emissions by mode for North America in 2003 is illustrated in Fig. 7-2.

Table 7-3. Summary of North American transport energy use and carbon dioxide emissions in 2003 by fuel type.

Figure 7-2. North American carbon emissions from transportation by mode; U.S.A and Canada 2003, Mexico 2001.

Freight Transport

Movement of freight is a major component of the transportation sector in North America. Total freight activity in the United States, measured in metric ton-km, is 20 times that in Mexico and more than 10 times the levels observed in Canada (Figs. 7-3A, 7-3B, 7-3C).

Figure 7-3A. Freight activity by mode in Canada.

Figure 7-3B. Freight activity by mode in Mexico.

Figure 7-3C. Freight activity by mode in the United States.

In Mexico, trucking is the mode of choice for freight movements. Four-fifths of Mexican metric ton-km are produced by trucks. Moreover, trucking's modal share has been increasing over time.

In Canada, rail transport accounts for the majority of freight movement (65%). Rail transport is well suited to the approximately linear distribution of Canada's population in close proximity to the U.S. border, the long-distances from east to west, and the large volumes of raw material flows typical of Canadian freight traffic (see Table 5-2 in NATS, 2005).

In the United States, road freight plays a greater role than in Canada, and rail is less dominant, although rail still carries the largest share of metric ton-km (40%). In none of the countries does air freight account for a significant share of metric ton-km.

Passenger Transport

In all three countries, passenger transport is predominantly by road, followed in distant second by air travel. The rate of growth in air travel in North America is more than double that of road transport, so that air transport's share of carbon emissions will increase in the future. Nearly complete data are available for passenger-kilometers-traveled (pkt) by mode in the United States and Canada in 2001. Of the more than 8 trillion pkt accounted for by the United States, 86% was by light-duty personal vehicles, most by passenger car but a growing share by light trucks (Fig. 7-4A) (motorcycle pkt, about 0.2% of the total, is included with passenger car). Air travel claims 10%; other modes are minor.

Figure 7-4A. Distribution of passenger travel in the United States by mode.

Canadian passenger travel exhibits a very similar modal structure, but with a smaller role played by light trucks and air and a larger share for buses (Fig. 7-4B) (transit numbers for Canada were not available at the time these figures were compiled).

Figure 7-4B. Distribution of passenger travel by mode in Canada.

TRENDS AND DRIVERS

Driven by economic and population growth, transportation energy use has increased substantially in all three countries since 1990. Figures 7-5A and 7-5B illustrate the evolution of transport energy use by mode for Mexico and the United States. Energy use has grown most rapidly in Mexico, the country most dependent on road transport. In the United States, the steady growth of transportation oil use was interrupted by oil price shocks in 1973–74, 1979–80, and to a much lesser degree in 1991. The impact of the attack on the World Trade Center in 2001 and subsequent changes in air travel procedures had a visible effect on energy use for air travel.

Figure 7-5A. Evolution of transport energy use in Mexico.

Figure 7-5B. Evolution of transport energy use in the United States.

The evolution of transport carbon emissions has closely followed the evolution of energy use. Carbon dioxide emissions by mode are shown for the United States and Canada for the period 1990–2003 in Figs. 7-6A and 7-6B. The Canadian data include light-duty commercial vehicles in road freight transport, while all light trucks are included in the light-duty vehicle category in the U.S. data. These data illustrate the relatively faster growth of freight transport energy use. Fuel economy standards in both countries restrained the growth of passenger car and light-truck energy use (NAS, 2002). From 1990 to 2003 passenger kilometers traveled by road in Canada increased by 23%, while energy use increased by only 15%. In 2003, freight activity accounted for more than 40% of Canada's transport energy use. And while passenger transport energy use increased by 15% from 1990 to 2003, freight energy use increased by 40%. The Canadian transport energy statistics do not include natural gas pipelines as a transport mode.

Figure 7-6A. Transport CO_2 emissions in Canada.

Figure 7-6B. Transport CO_2 emissions in the United States.

Carbon emissions by transport are determined by the levels of passenger and freight activity, the shares of transport modes, the energy intensity of passenger and freight movements, and the carbon intensity of transportation fuels. In North America, petroleum fuels supply over 95% of transportation's energy requirements and account for 98% of the sector's GHG emissions. Among modes, road vehicles are predominant, producing almost 80% of sectoral GHG emissions. As a consequence, the driving forces for transportation GHG emissions have been changes in activity and energy intensity. The principal driving forces of the growth of passenger transportation are population and per capita income (WBCSD, 2004). Increased vehicle ownership follows rising per capita income, as do vehicle use, fuel consumption, and emissions. In general, energy forecasters expect the greatest growth in vehicle ownership and fossil fuel use in transportation over the next 25–50 years to occur in the developing economies (U.S. DOE/EIA, 2005b; IEA, 2004; WBCSD, 2004; Nakićenović, Grűbler, McDonald, 1998). The chief driving forces for freight activity are economic growth and the integration of economic activities at both regional and global scales (WBCSD, 2004).

Projections of North American transportation energy use and carbon emissions to 2030 have been published by the U.S. Energy Information Administration (U.S. DOE/EIA, 2005b) and the International Energy Agency (2005a). Historical population growth rates are similar in the three countries, 0.92% per year in the United States, 1.17% per year in Mexico, and 0.90% per year in Canada. Recent annual GDP growth rates are 4.4% for the United States, 4.1% for Mexico, and 2.4% for Canada (CIA, 2005). The U.S. Energy Information Administration's Reference Case projection assumes annual GDP growth rates of 3.1% for the United States, 2.4% for Canada, and 3.9% for Mexico (see Table A3 in U.S. DOE/EIA, 2005b). Assumed population growth rates are United States: 0.9%; Canada: 0.6%; Mexico: 1.0% (see Table A14 in U.S. DOE/EIA, 2005b). Chiefly as a result of economic growth, energy use by North American transportation is expected to increase by 46% from 2003 to 2025 (U.S. DOE/EIA, 2005b). If the mix of fuels is assumed to remain the same, as it nearly does in the IEO 2005 Reference Case projection, carbon dioxide emissions would increase from 587 Mt C in 2003 to 859 Mt C in 2025 (Fig. 7-7). Canada, the only one of the three countries to have committed to specific GHG reduction goals, is expected to show the lowest rate of growth in CO_2 emissions.

Figure 7-7. Projected carbon dioxide emissions from the North American transport sector in 2025, based on EIA IEO 2005 reference case.

The World Business Council for Sustainable Development (WBCSD), in collaboration with the International Energy Agency developed a model for projecting world transport energy use and greenhouse gas emissions to 2050 (Table 7-4). The WBCSD's reference case projection foresees the most rapid growth in carbon emissions from transportation occurring in Asia and Latin America (Fig. 7-8). Still, in 2050 North America accounts for 26.4% of global carbon dioxide emissions from transport vehicles (down from a 37.2% share in 2000).

Table 7-4. Global carbon emissions from transportation vehicles to 2050 by regions, WBCSD reference case projection (Mt C).

Figure 7-8. WBCSD projections of world transportation vehicle CO_2 emissions to 2050.

OPTIONS FOR MANAGEMENT

Dozens of policies and measures for reducing petroleum consumption and mitigating carbon emissions from transportation in North America have been identified and assessed (e.g., U.S. DOT, 1998; IEA, 2001; Greene and Schafer, 2003; Greene *et al*., 2005; CBO, 2003; Harrington and McConnell, 2003; NRTEE, 2005). However, there is no consensus about how much transportation GHG emissions can be reduced and at what cost. In general, top-down models estimating the mitigation impacts of economy-wide carbon taxes or cap-and-trade systems find the cost of mitigation high and the potential modest. On the other hand, bottom-up studies evaluating a wide array of policy options tend to reach the opposite conclusion. Part of the explanation of this paradox may lie in the predominant roles that governments play in constructing, maintaining, and operating the majority of transportation infrastructure and in the strong interrelationship between land use planning and transportation demand. In addition, top down models typically assume that all markets are efficient, whereas there is evidence of real-world transportation energy market failures, especially with respect to the determination of light-duty vehicle fuel economy (e.g., Turrentine and Kurani, 2004; NAS, 2002, Ch. 5). Estimates of the costs and benefits of mitigation policies also vary widely and depend critically on premises concerning (1) the efficiency of transportation energy markets, (2) the values consumers attach to vehicle attributes such as acceleration performance and vehicle weight, and (3) the current and future status of carbon-related technology.

A U.S. Energy Information Administration evaluation of a greenhouse gas cap and trade system, expected to result in carbon permit prices of $79/t C in 2010 and $221/t C in 2025, was estimated to reduce 2025 transportation energy use by 4.3 PJ and to cut transportation's carbon emissions by 10% from 225 Mt C in the reference case to 203 Mt C under this policy (U.S. DOE/EIA, 2003). The average fuel economy of new light-duty vehicles was estimated to increase from 26.4 mpg (8.9 L per 100 km) to 29.0 mpg (8.1 L per 100 km) in the policy case, an improvement of only 10%. A 2002 study by the U.S. National Academy of Sciences (NAS, 2002) estimated that "cost-efficient" fuel economy improvements for U.S. light-duty vehicles using proven technologies ranged from 12% for subcompact cars to 27% for large cars, and from 25% for small SUVs to 42% for large SUVs. The NAS study did not include the potential impacts of diesel or hybrid vehicle technologies and assumed that vehicle size and horsepower would remain constant.

The U.S. Congressional Budget Office (CBO, 2003) estimated that achieving a 10% reduction in U.S. gasoline use would create total economic costs of approximately $3.6 billion per year if accomplished by means of Corporate Average Fuel Economy (CAFE) standards, $3.0 billion if the same standards allowed trading of fuel economy credits among manufacturers, and $2.9 billion if accomplished via a tax on gasoline. This partial equilibrium analysis assumed that it would take about 14 years for the policies to have their full impact. If one assumes that the United States would consume 22,600 PJ of gasoline in 2017, resulting in 387 Mt of CO_2 emissions, then a 10% reduction amounts to 39 Mt C. At a total cost of $3 billion per year, and attributing the full cost to carbon reduction (vs. other objectives such as reducing petroleum dependence) produces an upper-bound mitigation cost estimate of $77/t C.

Systems of progressive vehicle taxes on purchases of less efficient new vehicles and subsidies for more efficient new vehicles ("feebates") are yet another alternative for increasing vehicle fuel economy. A study of the U.S. market (Greene *et al*., 2005) examined a variety of feebate structures under two alternative assumptions: (1) consumers consider only the first three years of fuel savings when making new vehicle purchase decisions, and (2) consumers consider the full discounted present value of lifetime fuel savings. The study found that if consumers consider only the first three years of fuel savings, then a feebate of $1000 per 0.01 gal/mile (3.5 L per 100 km), designed to produce no net revenue to the government, would produce net benefits to society in terms of fuel savings and would reduce carbon emissions by 139 Mt C in 2030. If consumers fully valued lifetime fuel savings, the same feebate system would cause a $3 billion loss in consumers' surplus (a technical measure of the change in economic well-being closely approximating income loss) and reduce carbon emissions by only 67 Mt C, or an implied cost of $44/Mt CO_2.

The most widely proposed options for reducing the carbon content of transportation fuels are liquid fuels derived from biomass and hydrogen produced from renewables, nuclear energy, or from fossil fuels with carbon sequestration. Biomass fuels, such as ethanol from cellulosic feedstocks or liquid hydrocarbon fuels produced via biomass gasification and synthesis, appear to be a promising mid- to long-term option, while hydrogen could become an important energy carrier but not before 2025 (WBCSD, 2004). The carbon emission reduction potential of biomass fuels for transportation is strongly dependent on the feedstock and conversion processes. Advanced methods of producing of ethanol from grain, the predominant feedstock in the United States can reduce carbon emissions by 10% to 30% (Wang, 2005; p. 16 in IEA, 2004). Production of ethanol from sugar cane, as is the current practice in

Brazil, or by not-yet-commercialized methods of cellulosic conversion can achieve up to a 90% net reduction over the fuel cycle. Conversion of biomass to liquid hydrocarbon fuels via gasification and synthesis may have a similar potential (Williams, 2005). The technical potential for liquid fuels production from biomass is very large and very uncertain; recent estimates of the global potential range from 10 to 400 exajoules per year (see Table 6.8 in IEA, 2004). The U.S. Departments of Energy and Agriculture have estimated that 30% of U.S. petroleum use could be replaced by biofuels by 2030 (Perlack *et al*., 2005). The economic potential will depend on competition for land with other uses, the development of a global market for biofuels, and advances in conversion technologies.

Hydrogen must be considered a long-term option because of the present high cost of fuel cells, technical challenges in hydrogen storage, and the need to construct a new infrastructure for hydrogen production and distribution (NAS, 2004; U.S. DOE, 2005; IEA, 2005b). Hydrogen's potential to mitigate carbon emissions from transport will depend most strongly on how hydrogen is produced. If produced from coal gasification without sequestration of CO_2 emissions in production, it is conceivable that carbon emissions could increase. If produced from fossil fuels with sequestration, or from renewable or nuclear energy, carbon emissions from road and rail vehicles could be virtually eliminated (General Motors *et al*., 2001).

In a comprehensive assessment of opportunities to reduce GHG emissions from the U.S. transportation sector, a study published by the Pew Center on Global Climate Change (Greene and Schafer, 2003) estimated that sector-wide reductions in the vicinity of 20% could be achieved by 2015 and 50% by 2030 (Table 7-5). The study's premises assumed no change in the year 2000 distribution of energy use by mode. A wide range of strategies was considered, including research and development, efficiency standards, use of biofuels and hydrogen, pricing policies to encourage efficiency and reduce travel demand, land-use transportation planning options, and public education (Table 7-5). Other key premises of the analysis were that (1) for efficiency improvements the value of fuel saved to the consumer must be greater than or equal to the cost of the improvement, (2) there is no change in vehicle size or performance, (3) pricing policies shift the incidence but do not increase the overall cost of transportation, and (4) there is a carbon cap and trade system in effect equivalent to a charge of approximately $50/t C. Similar premises underlie the 2030 estimates, except that technological progress is assumed to have expanded the potential for efficiency improvement and lowered the cost of biofuels.

Table 7-5. Potential impacts of transportation GHG reduction policies in the United States by 2015 and 2030 based on the 2000 distribution of emissions by mode and fuel.

The Pew Center study notes that if transportation demand continues to grow as the IEO 2005 and WBCSD projections anticipate, the potential reductions shown in Table 7.4 would be just large enough to hold U.S. transportation CO_2 emissions in 2030 to 2000 levels.

A study for the U.S. Department of Energy (ILWG, 2000) produced estimates of carbon mitigation potential for the entire U.S. economy using a variety of policies generally consistent with carbon taxes of $25–$50/t C. In the study's business as usual case, transportation CO_2 emissions increased from 478 Mt C in 1997 to 700 Mt C in 2020. A combination of technological advances, greater use of biofuel, fuel economy standards, paying for a portion of automobile insurance as a surcharge on gasoline, and others, were estimated to reduce 2020 transportation CO_2 emissions by 155 Mt C to 545 Mt CO_2. The study did not produce cost estimates and did not consider impacts on global energy markets.

A joint study of the U.S. Department of Energy and Natural Resources Canada (Patterson *et al.*, 2003) considered alternative scenarios of highway energy use in the two countries to 2050. The study did not produce estimates of cost-effectiveness for greenhouse gas reduction strategies but rather focused on the potential impacts of differing social, economic, and technological trends. Two of the scenarios describe paths that lead to essentially constant greenhouse gas emissions from highway vehicles through 2050 through greatly increased efficiency and biofuel and hydrogen use and, in one scenario, reduced demand for vehicle travel.

INCONSISTENCIES AND UNCERTAINTIES

There are some inconsistencies in the way the three North American countries report transportation carbon emissions. The principal source for Mexican emissions data breaks out transportation into four modes (road, air, rail and waterborne), does not report emissions for pipelines but does report emissions from use of international bunker fuels. The U.S. and Canada report transport emissions in much greater modal detail, by vehicle type and fuel type within modes. The U.S. and Mexico report emissions from international bunker fuels in their national inventory reports while Canada does not. Estimates of international bunker fuel emissions for Canada presented in this chapter were derived by subtracting Air and Waterborne emissions reported by Environment Canada (2005) which exclude international bunker fuels from total air and waterborne emissions as reported by Natural Resources Canada (2006) which include them. Environment Canada reports off-road emissions from mobile sources separately; in the tables and figures in this chapter Canadian off-road emissions have been added to road emissions. Both Canada and the U.S. include emissions from military transport operations in their inventories. It is not clear whether these are included in the estimates for Mexico.

All three countries' greenhouse gas inventories discuss uncertainties in estimated emissions. In general, the uncertainties were estimated in accordance with IPCC guidelines. The U.S. EPA provides only an estimate of a 95% confidence interval for all carbon dioxide emissions from the combustion of fossil fuels (-1% to 6%) which can be inferred to apply to transportation. Mexico's INE estimates a total uncertainty for transportation greenhouse gas emissions on the order of +/- 10%. For carbon dioxide emissions from road transport, the uncertainty is put at +/- 9% (INE, 2003, Appendix B). The Canadian Greenhouse Gas Inventory provides by far the most extensive and detailed estimates of uncertainty. Given the similarity in methods, the Canadian uncertainty estimates are probably also approximately correct for the United States, and therefore may be considered indicative of the uncertainty of North American carbon emission estimates (Table 7-6). Most significant is the apparent overestimation of carbon emissions from on-road vehicles, offset to a degree by the underestimation of off-road mobile source emissions. Still, total mobile source carbon emissions are estimated to have a 95% confidence interval of (-4% to 0%).

Table 7-6. Uncertainty in estimates of carbon dioxide emissions from energy use in transport: Canada 2003.

RESEARCH AND DEVELOPMENT NEEDS

Research needs with respect to the transport sector as a part of the carbon cycle fall into three categories: (1) improved data, (2) comprehensive assessments of mitigation potential, and (3) advances in key mitigation technologies and policies for transportation. The available data are adequate to describe carbon inputs by fuel type and carbon emissions by very broad modal breakdowns by country. Environment Canada (2005) and the U.S. Environmental Protection Agency (2005) annually publish estimates of transportation's carbon emissions that closely follow IPCC guidelines with respect to methods, data sources and quantification of uncertainties (GAO, 2003). The Mexican Instituto Nacional de Ecología has published estimates for 2001 that are also based on IPCC methods. However, that report also notes deficiencies in the data available for Mexico's transport sector and recommends establishing an information system for estimating Mexico's transportation's greenhouse gas emissions on a continuing basis (INE, 2003, p. 21). Knowledge of the magnitudes of GHG emissions by type of activity and fuel and of trends is essential if policies are to be focused on the most important GHG sources.

The most pressing research need is for comprehensive, consistent, and rigorous assessments of the carbon emissions mitigation potential for North American transportation. The lack of such studies for North America parallels a similar dearth of consistent and comprehensive global analyses noted by the Intergovernmental Panel on Climate Change (Moomaw and Moreira, 2001). Existing studies focus almost exclusively on a single country, with premises and assumptions varying widely from country to country. Even the best single country studies omit the impacts of carbon reduction policies on global energy markets. Knowledge of how much contribution the transport sector can make to GHG mitigation at what cost and what options and measures are capable of achieving those potentials is crucial to the global GHG policy discussion.

Continued research and development of vehicle technologies and fuels that can cost-effectively increase energy efficiency and displace carbon-based fuels is essential to achieving major reductions in transportation carbon emissions. Highly promising technologies for reducing transportation GHG emissions include hybrid vehicles, which are available today, and in the future, plug-in hybrid vehicles capable of accepting electrical energy from the grid, and eventually fuel cell vehicles powered by hydrogen. While hybrids are already in the market and fuel cell vehicles are still years away, all three technologies would benefit from cost reduction. Hydrogen fuel cell vehicles also face significant technological challenges with respect to hydrogen storage and fuel cell durability. Technologies exist that could greatly reduce greenhouse gas emissions from other transport modes. For example, blended wing-body aircraft designs could reduce fuel burn rates by one-third. Biofuels in the near term and hydrogen in the longer term appear to be the most promising low-carbon fuel options. To achieve the greatest greenhouse gas reduction benefits, biofuels must be made from plants' lingo-cellulosic components either by conversion to alcohol or by gasification and synthesis of liquid hydrocarbon fuels. Cost reductions in both feedstock production and fuel conversion are needed.

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Table 7-1. Carbon emissions from transportation in North America in 2003

North American Carbon Emissions by Country and Mode, 2003/2001 (Mt C)				
	U.S.A. 2003	**Canada 2003**	**Mexico 2001**	**North America 2003/2001**
Road	399.4	36.7	26.0	462.0
Domestic Air	46.7	1.9	1.8	50.4
Rail	11.7	1.4	0.4	13.5
Domestic Water	15.7	1.6	0.9	18.1
Pipeline	9.5	2.4		11.9
International Bunker	23.0	3.0	0.5	26.4
Off-Road		4.6		4.6
Total	505.9	51.7	29.4	587.0

Sources: U.S. EPA, 2005; Environment Canada, 2005; INE, 2003.
Note: Data for Mexico is 2001, U.S.A. and Canada are 2003.

Table 7-2. Summary of North American transport energy use and carbon dioxide emissions in 2003 by energy source or fuel type

North America energy source	Energy input (Petajoules)	Carbon input (Mt C)
Gasoline	20,923	358.3
Diesel/distillate	7,344	129.5
Jet fuel/kerosene	2,298	68.5
Residual	681	14.5
Other fuels	124	1.3
Natural gas	926	9.7
Electricity	36	0.0
Unalloc./error	466	-
Total	32,798	581.8
United States		
Gasoline	18,520	312.5
Diesel/distillate	6,193	107.1
Jet fuel/kerosene	1,986	62.3
Residual	612	13.1
Other fuels	50	0.2
Natural gas	748	9.7
Electricity	20	0.0
Unalloc./error	466.2	-
Total	28,595.2	504.9

Sources: U.S. EPA, 2005, Tables 3-7 and 2-17; Davis and Diegel, 2004, Tables 2.6 and 2.7.

Canada		
Gasoline	1,355	26.2
Diesel/distillate	698	13.9
Jet fuel/kerosene	223	4.3
Residual	67	1.3
Other fuels	17	0.2
Natural gas	2	0.0
Electricity	3	0.0
Unalloc./error	0	
Total	2,363	45.9

NRCan, 2006, Tables 1 and 8.

Mexico		
Gasoline	1,066	19.5
Diesel/distillate	447	8.5
Jet fuel/kerosene	106	1.9
Residual	4	0.1
Other fuels	57	0.9
Natural gas	1	0.0
Electricity	4	0.0
Unalloc./error		
Total	1,685	31.0

Sources: Transportation energy use by fuel and mode from Rodriguez, 2005.

Source: Fulton and Eads, 2004, spreadsheet model, output worksheet.

Data sources differ somewhat by country with respect to modal, fuel, and greenhouse gas definitions so that the numbers are not precisely comparable. Canadian carbon emissions data include all greenhouse gases produced by transportation in CO_2 equivalents, while the U.S. data are CO_2 emissions only. Carbon dioxide emissions for Mexico were estimated by applying U.S. EPA emissions factors to the Mexican energy use data. For Mexico, it is assumed that no transportation carbon emissions result from electricity use.

Table 7-3. Summary of North American transport energy use and carbon dioxide emissions in 2003 by mode of transportation

North America transport mode	Energy use (Petajoules)	Carbon emissions (Mt C)
Road	25,830	463.5
Air	2,667	53.0
Rail	751	13.7
Waterborne	1,386	18.4
Pipeline	990	12.3
	0	23.0
Total	31,624	583.9
United States		
Road		
Light vehicles	17,083	303.8
Heavy vehicles	5,505	95.5
Air	2,335	46.7
Rail	655	11.7
Waterborne	1,250	15.7
Pipeline/other	986	9.5
Internatl./Bunker		23.0
Total	27,814	505.8
Source: U.S. EPA, 2005, Tables 3-7 and 2-17; Davis and Diegel, 2004, Tables 2-6 and 2-7.		
Canada		
Road		
Light vehicles	1,233	23.8
Heavy vehicles	491	12.4
Air	226	4.3
Rail	74	1.6
Waterborne	103	2.1
Pipeline/other		1.8
Total	2,126	46.1
Source: NRCan, 2006; Tables 1 and 8.		
Mexico		
Road	1,518	27.9
Light vehicles		
Heavy vehicles		
Air	107	2.0
Rail	22	0.5
Waterborne	33	0.6
Electric	4	-
Total	1,684	32.0
Source: Rodriguez, 2005.		

Data sources differ somewhat by country with respect to modal, fuel, and greenhouse gas definitions so that the numbers are not precisely comparable. Canadian carbon emissions data include all greenhouse gases produced by transportation in CO_2 equivalents, while the U.S. data are CO_2 emissions only. Carbon dioxide emissions for Mexico were estimated by applying U.S. EPA emissions factors to the Mexican energy use data. Electricity is assumed to produce no carbon emissions in end use.

Table 7-4. Global carbon emissions from transportation vehicles to 2050 by regions, WBCSD reference case projection (Mt C)

	2000	2010	2020	2030	2040	2050
OECD North America	544	623	708	768	824	882
OECD Europe	313	359	392	412	420	428
OECD Pacific	133	142	153	161	169	179
FSU	48	64	88	109	132	153
Eastern Europe	23	28	36	42	52	66
China	69	108	163	225	308	417
Other Asia	98	131	174	220	283	368
India	38	54	80	108	146	203
Middle East	59	71	88	106	122	138
Latin America	95	127	172	216	275	352
Africa	43	58	80	103	127	158
TOTAL - All Regions	1463	1766	2134	2470	2858	3343

Source: Fulton and Eads, 2004.

Table 7-5. Potential impacts of transportation GHG reduction policies in the United States by 2015 and 2030[a] based on the 2000 distribution of emissions by mode and fuel (Greene and Schafer, 2003)

		Reduction potential per mode/fuel (%)		Transportation sector reduction potential (%)	
Management option	**Carbon emission (Mt C) 2000**	**2015**	**2030**	**2015**	**2030**
Research, development and demonstration					
Light-duty vehicles (LDVs)	289	11[b]	38[b]	7[b]	23[b]
Heavy trucks	80	11[b]	24[b]	2[b]	4[b]
Commercial aircraft	53	11[b]	27[b]	1[b]	3[b]
Efficiency standards					
Light-duty vehicles	289	9	31	6	18
Heavy trucks	80	9	20	2	3
Commercial aircraft	53	9	22	1	2
Replacement and alternative fuels					
Low-carbon replacement fuels (~10% of LDV fuel)	27	30	100	2	7
Hydrogen fuel (All LDV fuel)	289	1	6	1	4
Pricing policies					
Low-carbon replacement fuels (~10% of LDV fuel)	27	30	100	2	6
Carbon pricing (All transportation fuel)	489	3	6	3	6
Variabilization (All highway vehicle fuel)	370	8	12	6	9
Behavioral					
Land use and infrastructure (2/3 of highway fuel)	246	5	10	3	5
System efficiency (25% LDV fuel)	72	2	5	0	1
Climate change education (All transportation fuel)	489	1	2	1	2
Fuel economy information (All LDV fuel)	289	1	2	1	1
Total	**489**			**22**	**48**

Notes:

[a]Carbon emissions for the year 2000 are used to weight percent reductions for the respective emissions source and example policy category in calculating total percent reduction potential. The elasticity of vehicle travel with respect to fuel price is –0.15 for all modes. Price elasticity of energy efficiency with respect to fuel price is –0.4.

[b]R&D efficiency improvements have no direct effect on total. Their influence is seen through efficiency standards impacts.

Policies affecting the same target emissions, such as passenger car efficiency, low carbon fuels, and land use policies are multiplicative, to avoid double counting [e.g. (1–0.1)*(1.0–0.2) = 1–0.28, a 28% rather than a 30% reduction.]

1 **Table 7-6. Uncertainty in estimates of carbon dioxide emissions from energy use in transport: Canada 2003**

Mode	% Below (2.5th Percentile)	% Above (97.5th Percentile)
Total Mobile Sources excluding pipeline	-4	0
Road Transportation	-8	-3
On-Road Gasoline Vehicles	-7	-3
On-Road Diesel Vehicles	-13	-1
Railways	-5	3
Navigation	-3	3
Off-Road Mobile Sources	4	45
Pipeline	-3	3

Source: Environment Canada, 2005, table A7-9.

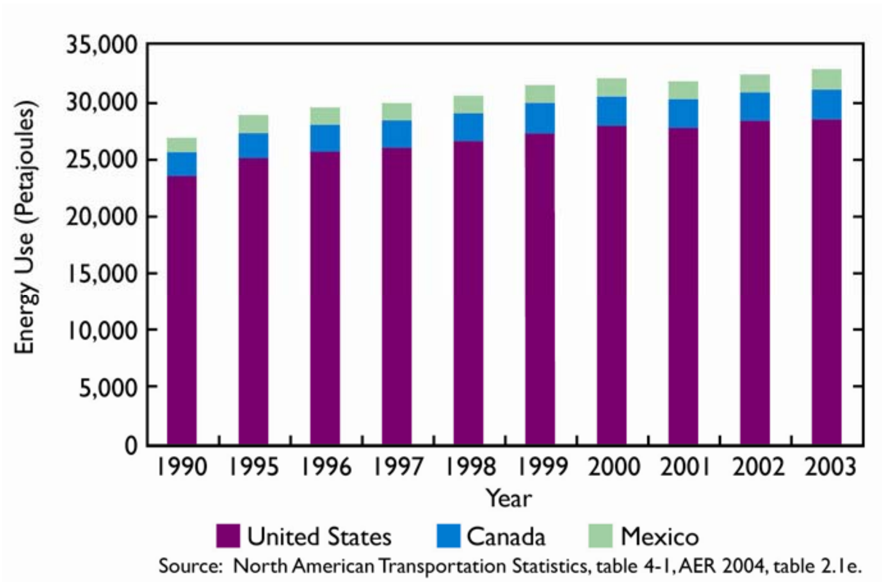


Fig. 7-1. Transportation energy use in North America, 1990–2003.

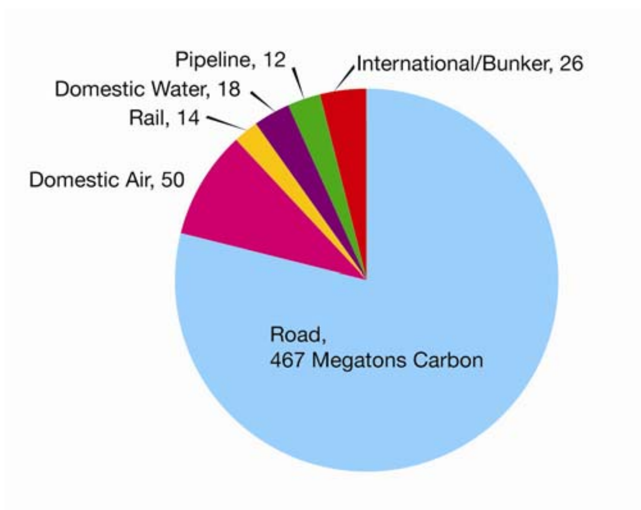


Fig. 7-2. North American carbon emissions from transportation by mode; U.S.A and Canada 2003, Mexico 2001. *Sources*: U.S. EPA, 2005; Environment Canada, 2005; INE, 2003.

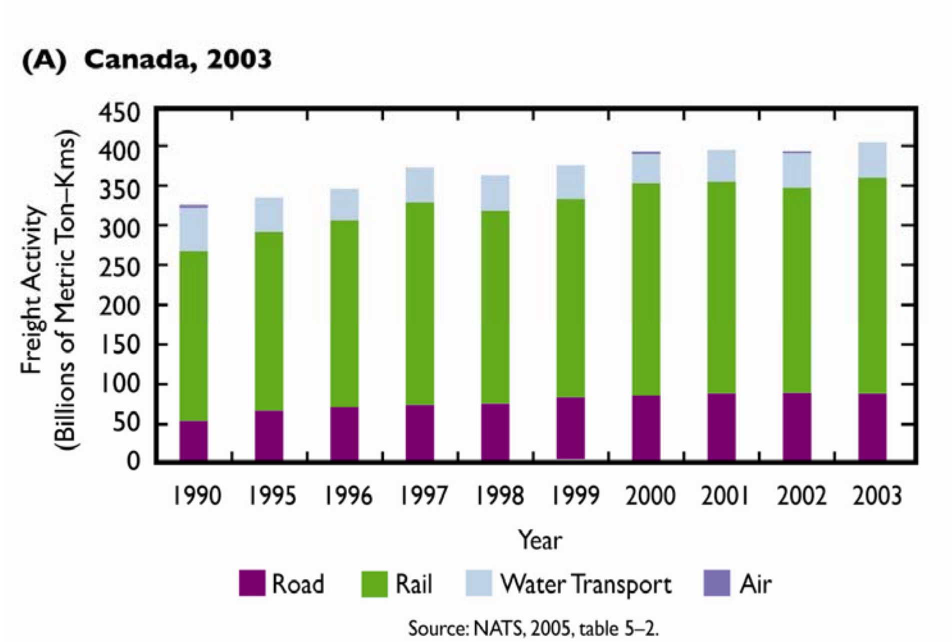


Fig. 7-3A. Freight activity by mode in Canada.

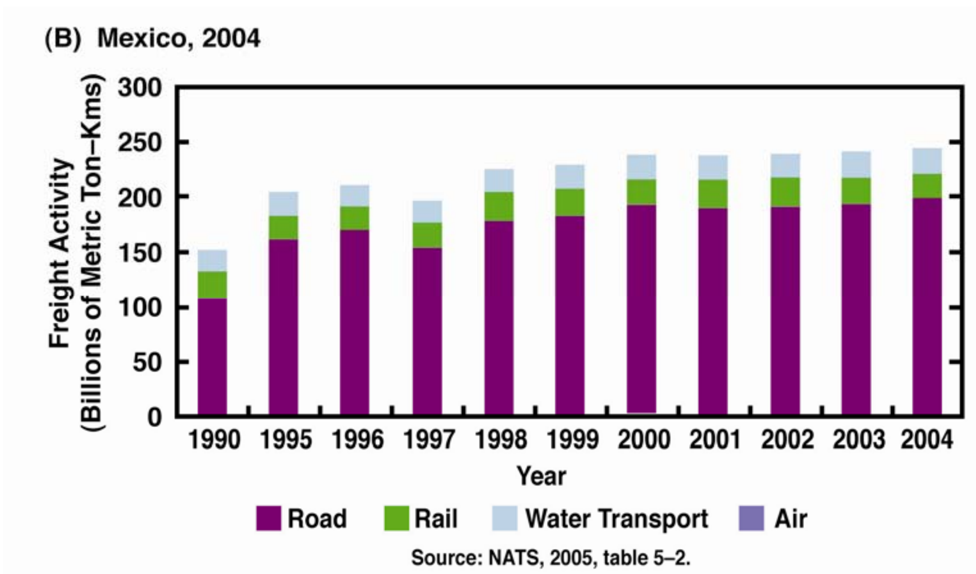
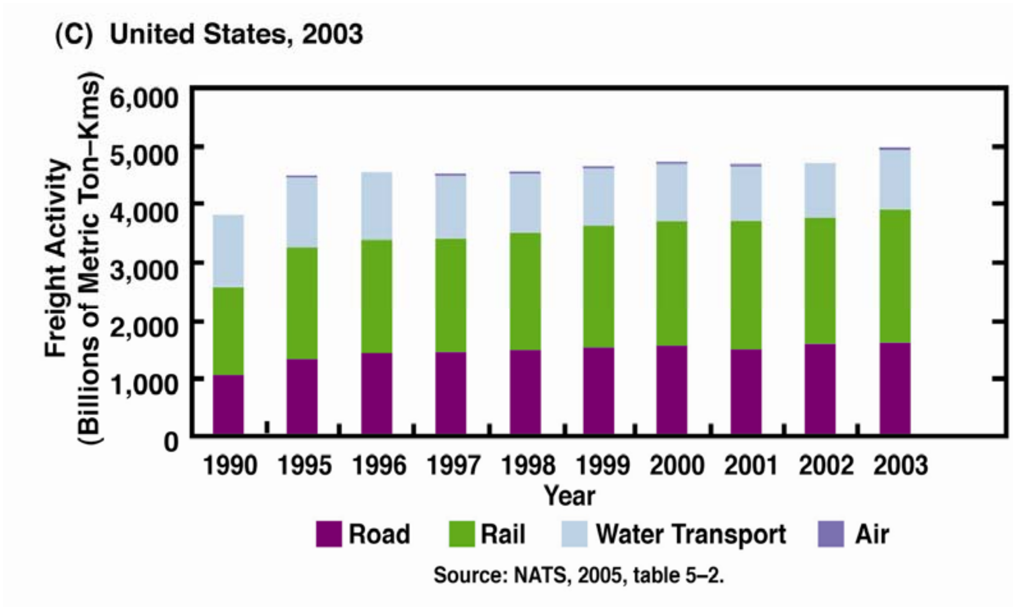


Fig. 7-3B. Freight activity by mode in Mexico.

1


2 **Fig. 7-3C. Freight activity by mode in the United States.**

3

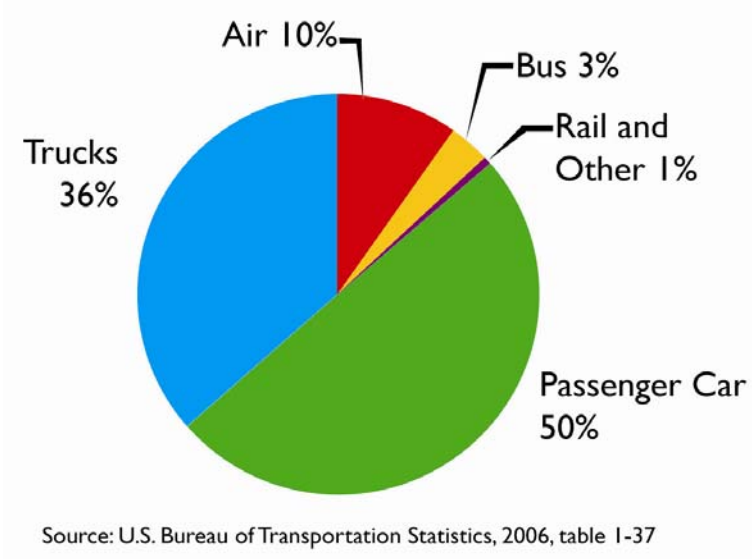


Fig. 7-4A. Distribution of passenger travel in the United States by mode.

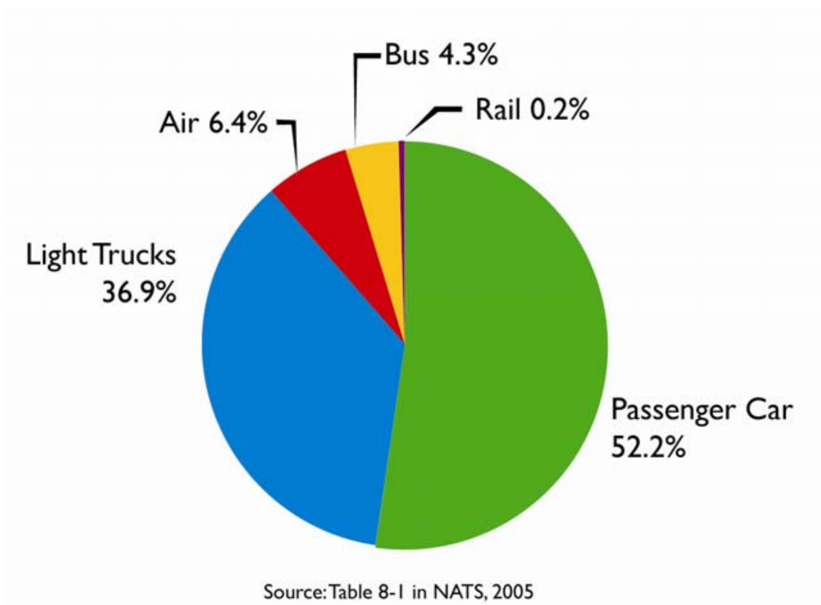


Fig. 7-4B. Distribution of passenger travel by mode in Canada. *Source*: Table 8-1 in NATS, 2005.

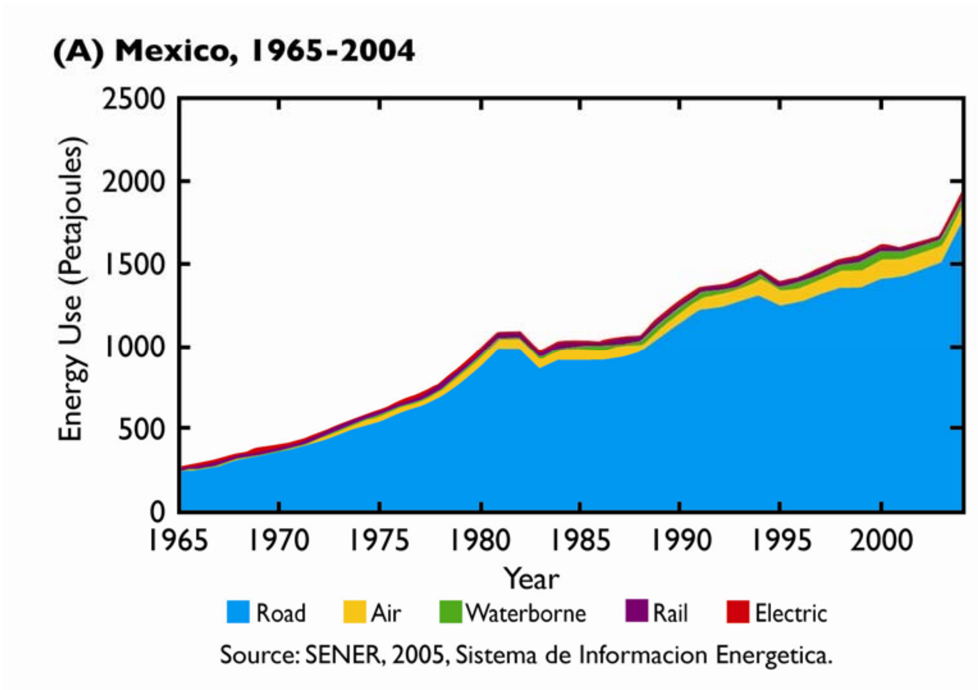


Fig. 7-5A. Evolution of transport energy use in Mexico.

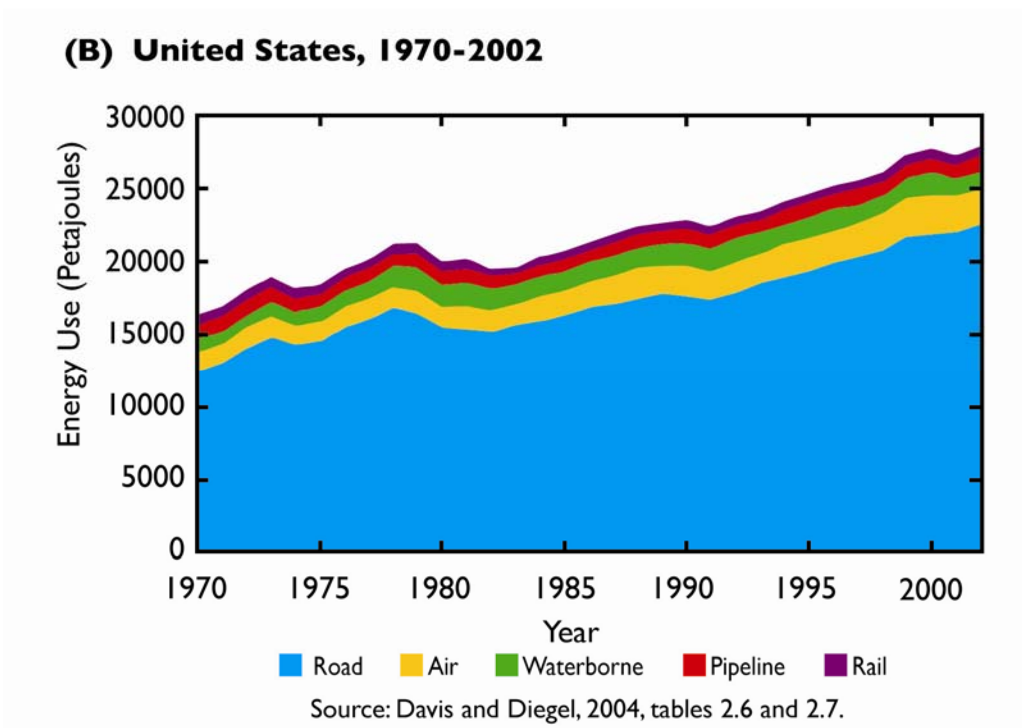


Fig. 7-5B. Evolution of transport energy use in the United States.

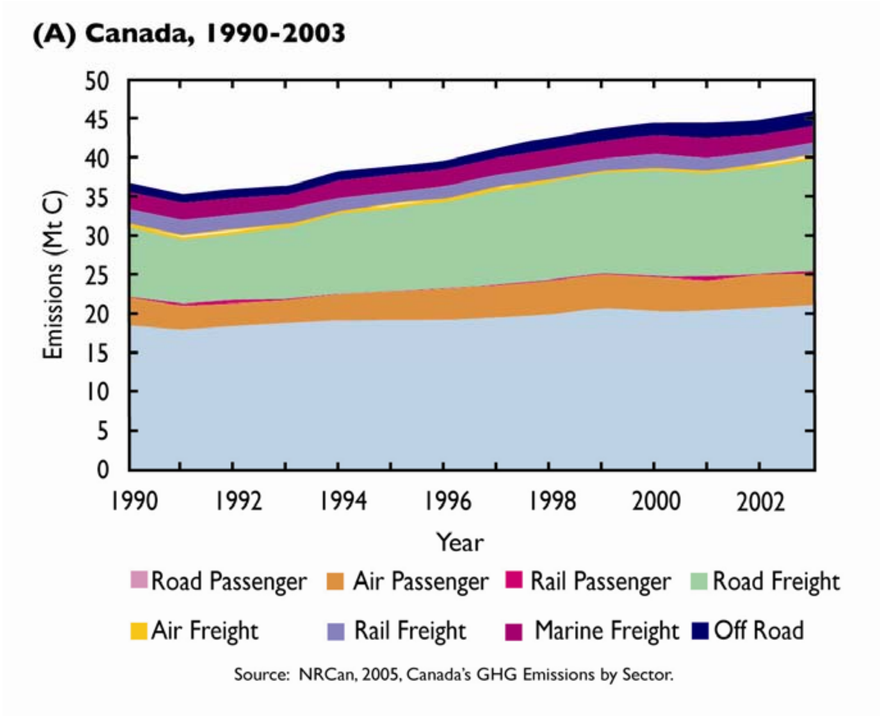


Fig. 7-6A. Transport CO_2 emissions in Canada.

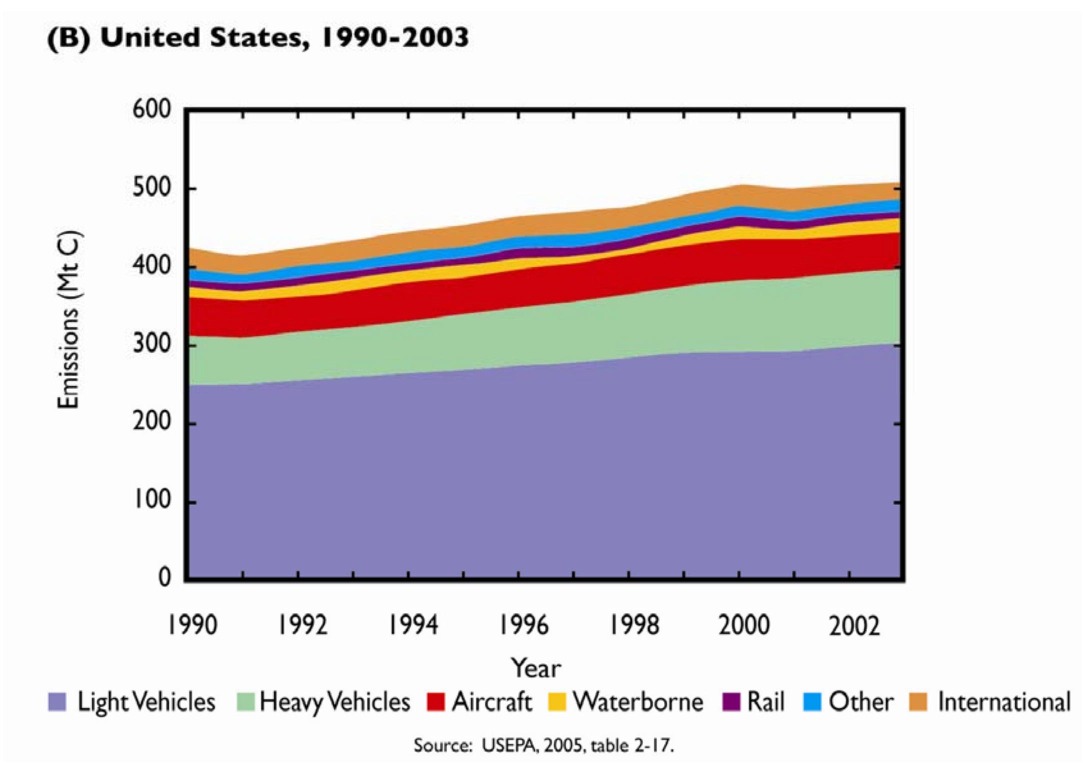


Fig. 7-6B. Transport CO_2 emissions in the United States.

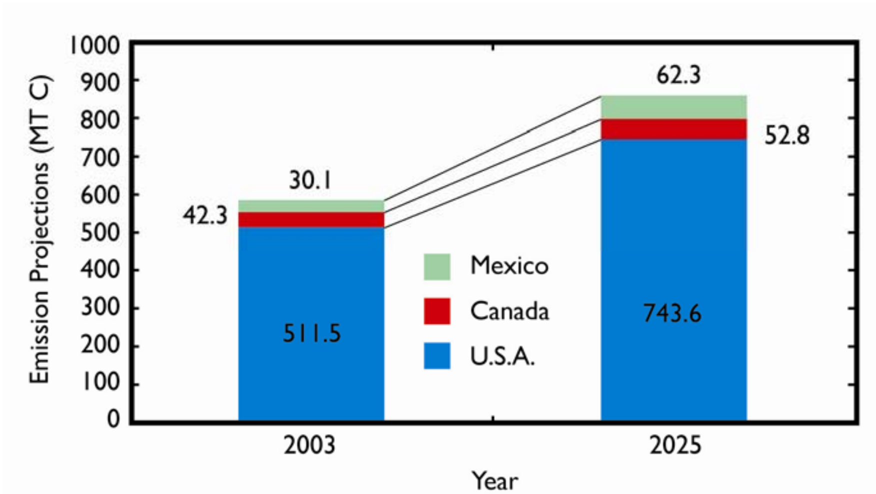


Fig. 7-7. Projected carbon dioxide emissions from the North American transport sector in 2025, based on EIA IEO 2005 reference case. *Source*: U.S. DOE Energy Information Administration, 2005b.

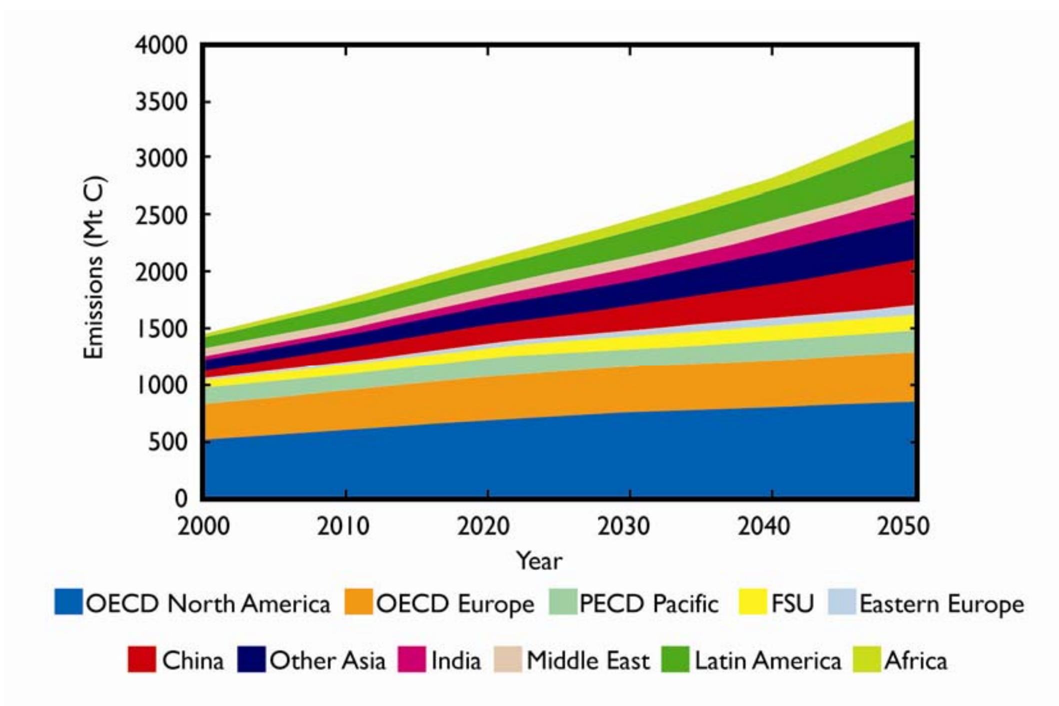


Fig. 7-8. WBCSD projections of world transportation vehicle CO_2 emissions to 2050. *Source*: Fulton and Eads, 2004.

Chapter 8. Industry and Waste Management

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KEY FINDINGS

- In 2002, North America's industry (not including fossil fuel mining and processing or electricity generation) contributed 826 Mt CO_2, 16% of the world's CO_2 emissions to the atmosphere from industry. Waste treatment plants and landfill sites in North America accounted for 13.4 Mt of CH_4 (282 Mt CO_2e), roughly 20% of global totals.
- Industrial CO_2 emissions from North America decreased nearly 11% between 1990 and 2002, while energy consumption in the United States and Canada increased 8% to 10% during that period. In both countries, a shift in production activity toward less energy-intensive industries and dissemination of more energy efficient equipment kept the rate of energy demand growth lower than industrial GDP growth.
- Changes in industrial CO_2 emissions are a consequence of changes in industrial energy demand and changes in the mix of fossil fuels used by industry to supply that demand. Changes in industrial energy demand are themselves a consequence of changes in total industrial output, shifts in the relative shares of industrial sectors, and increases in energy efficiency. Shifts from coal and refined petroleum products to natural gas and electricity contributed to a decline in total industrial CO_2 emissions since 1997 in both Canada and the United States.
- An increase in CO_2 emissions from North American industry is likely to accompany the forecasted increase in industrial activity (2.3% yr^{-1} until 2025 for the United States). Emissions per unit of industrial activity will likely decline as non-energy intensive industries grow faster than energy intensive industries and with increased penetration of energy efficient equipment. However, continuation of the trend toward less carbon-intensive fuels is uncertain given the rise in natural gas prices relative to coal in recent years.
- Options and measures for reducing CO_2 emissions from North American industry can be broadly classified as methods to: (1) reduce process/fugitive emissions or converting currently released emissions; (2) increase energy efficiency, including combined heat and power; (3) change industrial processes (materials efficiency, recycling, substitution between materials or between materials and energy); (4) substitute less carbon intense fuels; and (5) capture and store carbon dioxide.

- Further work on materials substitution holds promise for industrial emissions reduction, such as the replacement of petrochemical feedstocks by biomass feedstocks, of steel by aluminium in the transport sector, and of concrete by wood in the buildings sector. The prospects for greater energy efficiency technologies, including efficient Hall-Heroult cell retrofits in aluminium production, black liquor gasification in kraft pulp production, and shape casting in iron and steel industries are equally substantial.

INTRODUCTION

This chapter assesses carbon flows through industry (manufacturing, construction, including industry process emissions, but excludes fossil fuel mining and processing)[1] and municipal waste disposal.

In 2002, industry was responsible for 5220.6 Mt of CO_2, 21% of anthropogenic CO_2 emissions to the atmosphere (4322.9 Mt from fuel combustion and 897.7 Mt from industrial processes). North America's industry contributed 758.7 Mt of combustion-sourced emissions and 66.8 Mt of process emissions for a total of 826 Mt, 16% of global totals. The manufacturing industry contributed 12% of total North American greenhouse gas (GHG) emissions, lower than in many other parts of the world. But with North America's population at 6.8% of the world's total, industry contributed a proportionally larger share of total industrial emissions per capita than the rest of the world (see Fig. 8-1A).[2]

Figure 8-1A. CO_2 emissions by sector in 2002.

Industrial CO_2 emissions decreased nearly 11% between 1990 and 2002 while energy consumption in the United States and Canada increased 8% to 10% (EIA, 2005; CIEEDAC, 2005). In both countries, a shift in production activity toward less energy-intensive industries and dissemination of more energy efficient equipment kept the rate of growth in energy demand lower than industrial GDP growth (IEA, 2004).[3] This slower demand growth, in concert with a shift toward less carbon-intensive fuels, explains the decrease in industrial CO_2 emissions.

The municipal waste stream excludes agricultural and forestry wastes but includes wastewater. CO_2, generated from aerobic metabolism in waste removal and storage processes, arises from biological material and is considered GHG neutral. Methane (CH_4), released from anaerobic activity at waste

[1]This includes direct flows only. Indirect carbon flows (e.g., due to electricity generation) are associated with power generation.

[2]North America, including Mexico, was responsible for about 27% of global CO_2 emissions in 2002.

[3]Decomposition analyses can assess changes in energy consumption due to, for example, increases in industry activity, changes in relative productivity to or from more intense industry subsectors, or changes in material or energy efficiency in processes.

treatment plants and landfill sites, forms a substantial portion of carbon emissions to the atmosphere. Given its high global warming potential, methane plays an important role in the evaluation of possible climate change impacts (see Fig. 8-1B).[4] Globally, CH_4 emissions from waste amount to 66 Mt, or 1386 Mt CO_2 equivalent. North American activity accounts for 13.4 Mt of CH_4 (282 Mt CO_2 equivalent), roughly 20% of global totals.

Figure 8-1B. GHG emissions by sector in 2000, CO_2, CH_4, N_2O, PFCs, HFCs, and SF_6.

Substantial sequestration of carbon occurs in landfills.[5] Data on carbon buried there are poor. The Environmental Protection Agency (EPA), using data from Barlaz (1990, 1994), estimated that 30% of carbon in food waste and up to 80% of carbon in newsprint, leaves, and branches remain in the landfill. Plastics show no deterioration. In all, 80% of the carbon entering a landfill site may be sequestered, depending on moisture, aeration, and site conditions. Bogner and Spokas (1993) estimate that "more than 75% of the carbon deposited in landfills remains in sedimentary storage."

INDUSTRY CARBON CYCLE

Carbon may enter industry as a fuel or as a feedstock where the carbon becomes entrained in the industry's final product. Carbon in the waste stream can be distinguished as atmospheric and non-atmospheric, the former being comprised of process and combustion-related emissions. Process CO_2 emissions, a non-combustive source, are the result of the transformation of the material inputs to the production process. For example, cement production involves the calcination of lime, which chemically alters limestone to form calcium oxide and releases CO_2. Of course, combustion-related CO_2 emissions occur when carbon-based fuels provide thermal energy to drive industrial processes.

Overview of Carbon Inputs and Outputs

Industry generates about one-third as much emitted carbon as the production of electricity and other fuel supply in North America and only about 55% as much as is generated by the transportation sector.

Carbon In

Carbon-based raw materials typically enter industrial sites as biomass (primarily wood), limestone, soda ash, oil products, coal/coke, natural gas and natural gas liquids. These inputs are converted to

[4] While not carbon-based, N_2O from sewage treatment is shown in Fig. 2 to show its relative GHG importance.
[5] IPCC guidelines currently do not address landfill sequestration. Such guidelines will be in the 2006 publication.

dimension lumber and other wood products, paper and paperboard, cement and lime, glass, and a host of chemical products, plastics, and fertilizers.

While the bulk of the input carbon leaves the industrial site as a product, some leaves as process CO_2 and some is converted to combustible fuel. Waste wood (or hog fuel) and black liquor, generated in the production of chemical pulps, are burned to provide process heat/steam for digesting wood chips or for drying paper or wood products, in some cases providing electricity through cogeneration. Chemical processes utilizing natural gas often generate off-gases that, mixed with conventional fuels, provide process heat. Finally, some of the carbon that enters as a feedstock leaves as solid or liquid waste.

In some industries, carbon is used to remove oxygen from other input materials through "reduction." In most of the literature, such carbon is considered an input to the process and is released as "process" CO_2, even though it acts as a fuel (i.e., it unites with oxygen to form CO_2 and releases heat). For example, in metal smelting and refining processes, a carbon-based reductant separates oxygen from the metal atoms. Coke, from the destructive distillation of coal, enters a blast furnace with iron ore to strip off the oxygen associated with the iron. Carbon anodes in electric arc furnaces in steel mills and specialized electrolytic "Hall-Heroult" cells oxidize to CO_2 as they melt recycled steel or reduce alumina to aluminum.

Carbon Out

Carbon leaves industry as part of the intended commodity or product, as a waste product or as a gas, usually CO_2.

Process emissions are CO_2 emissions that occur as a result of the process itself—the calcining of limestone releases about 0.5 tons CO_2 per ton of clinker (unground cement) or about 0.8 tons per ton of lime.[6,7] The oxidation of carbon anodes generates about 1.5 tons CO_2 to produce a ton of aluminum. Striping hydrogen from methane to make ammonia releases about 1.6 tons CO_2 per ton of ammonia.

Combustion of carbon-based fuels results in the emission of CO_2. In many cases, the combustion process is not complete and other carbon-based compounds may also be released (carbon monoxide, methane, volatile organic compounds). These often decompose into CO_2, but their life spans in the atmosphere vary.

Carbon Flow

Figure 8-2 illustrates the flows of carbon in and out of industries in North America. Comparable diagrams for individual countries are presented in Appendix 8A. On the left side of Fig. 8-2, all carbon-

[6] In these industries, more CO_2 is generated from processing limestone than from the fossils fuels combusted.
[7] The calcination of limestone also takes place in steel, pulp and paper, glass and sugar industries.

based material by industry sector is accounted for, whether in fuel or in feedstock. On the right, the exiting arrows portray how much of the carbon leaves as part of the final products from that industry. The carbon in the fossil fuel and feedstock materials leave in the waste stream as emissions from fuel combustion (including biomass), as process emissions, or as other products and waste. Carbon capture and storage potentials are assessed in the industry subsections below.

Figure 8-2. Carbon flows for Canada, the United States, and Mexico combined.

Sectoral Trends in the Industrial Carbon Cycle

Figure 8-2 shows that energy-intensive industries differ significantly in their carbon cycle dynamics.

Pulp and Paper

While pulp and paper products are quite energy-intensive, much of the energy is obtained from biomass. By using hog fuel and black liquor, some types of pulp mills are energy self-sufficient. Biomass fuels are considered carbon neutral because return of the biomass carbon to the atmosphere completes a cycle that began with carbon uptake from the atmosphere by vegetation.[8] Fuel handling difficulties and air quality concerns can arise from the use of biomass as a fuel.

Cement, Lime, and Other Nonmetallic Minerals

Cement and lime production require the calcination of limestone, which releases CO_2; about 0.78 tons of CO_2 per ton of lime calcined.

$CaCO_3$	→	CaO	+	CO_2
calcium carbonate		calcium oxide		carbon dioxide

Outside of the combustion of fossil fuels, lime calcining is the single largest anthropogenic source of CO_2 emissions. Annual growth in cement production is forecast at 2.4% in the United States for at least the next decade. This industry could potentially utilize sequestration technologies to capture and store CO_2 generated.

The production of soda ash (sodium carbonate) from sodium bicarbonate in the Solvay Process releases CO_2 and, as in glass production, in its utilization. Soda ash is used to produce pulp and paper, detergents and soft water.

[8] This is also reflected in the United Nations Framework Convention on Climate Change IPCC guidelines to estimate CO_2 emissions.

$2NaHCO_3$	→	Na_2CO_3	+	CO_2	+	H_2O
sodium bicarbonate		sodium carbonate		carbon dioxide		water

Nonferrous Metal Smelting and Iron and Steel Smelting

Often metal smelting requires the reduction of metal oxides to obtain pure metal through the use of a "reductant", usually coke. Because reduction processes generate relatively pure streams of CO_2, the potential for capture and storage is good.

In electric arc furnaces, carbon anodes decompose to CO_2 as they melt the scrap iron and steel feed in "mini-mills". In Hall-Heroult cells, a carbon anode oxidizes when an electric current forces oxygen from aluminium oxide (alumina) in the production of aluminum. [9]

Metal and Nonmetal Mining

Mining involves the extraction of ore and its transformation into a concentrated form. This involves transportation from mine site, milling and separating mineral-bearing material from the ore. Some transportation depends on truck activity but the grinding process is driven by electric motors (i.e., indirect release of CO_2). Some processes, like the sintering or agglomeration of iron ore and the liquid extraction of potash, use a considerable amount of fossil fuels directly.

Chemical Products

This diverse group of industries includes energy-intensive electrolytic processes as well as the consumption of large quantities of natural gas as a feedstock to produce commodities like ammonia, methanol, and hydrogen. Ethylene and propylene monomers from natural gas liquids are used in plastics production. Some chemical processes generate fairly pure streams of CO_2 suitable for capture and storage.

Forest Products

This industry uses biomass waste to dry commercial products such as lumber, plywood and other products. The industry also includes silviculture, the practice of replanting and managing forests.

Other Manufacturing

Most of the remaining industries, while economically important, individually play a relatively minor role in the carbon cycle because they are not energy intensive and use little biomass.[10] In aggregate, however, these various industries contribute significantly to total industrial CO_2 emissions. Industries in

[9] Ceramic anodes may soon be available to aluminum producers and significantly reduce process CO_2 emissions.
[10] Except, of course, the food, beverage and some textile industries.

this group include the automotive industry, electronic products, leather and allied products, fabricated metals, furniture and related products, and plastics and rubber products.

Changing Role of Industry in the Carbon Cycle

Energy consumption per unit GDP has declined in Canada and the United States by more than 30% since the mid-1970s. In manufacturing, the decline was even greater—more than 50% in the United States since 1974.

The National Energy Modelling System operated by the United States' Energy Information Administration applies growth forecasts from the Global Insight macroeconomic model. While the United States economy is forecast to grow at an average rate of 3.1% per year to 2025, industrial growth is forecast at 2.3% per year—an amalgam of manufacturing growth of 2.6% per year and non-manufacturing of 1.5% per year. Manufacturing is further disaggregated into energy-intensive industries, growing at 1.5% per year, and non-energy intensive industries at 2.9% per year. The slower growth in the energy-intensive industries is reflected in the expected decline in industrial energy intensity of 1.6% per year over the EIA (2005) forecast.

The International Energy Agency reviewed energy consumption and emissions during the last 30 years to identify and project underlying trends in carbon intensity.[11] The review's decomposition analysis (Fig. 8-3) attributes changes in industrial energy demand to changes in total industrial output (activity), shifts in the relative shares of industrial sectors (structure), and increases in energy efficiency (intensity).

Figure 8-3. Decomposition of energy use, manufacturing section, 1990–1998.

Changes in carbon emissions result from these three factors, but also from changes in fuel shares—substitution away from or toward more carbon-intensive fuels. The shift from coal and refined petroleum products to natural gas and electricity[12] contributed to a decline in total industrial CO_2 emissions since 1997 in both Canada and the United States. The continuation of this trend is uncertain given the rise in natural gas prices relative to coal in recent years.

[11] Most of the information in this section is obtained from this report (IEA, 2004a).

[12] As noted earlier, emissions associated with electricity are allocated to the electricity supply sector. Thus, a shift to electricity reduces the GHG intensity of the industry using it. If electricity is made in coal-fired plants, however, total CO_2 emissions may actually increase.

Actions and Policies for Carbon Management in Industry

Industry managers can reduce carbon flows through industry by altering the material or energy intensity and character of production (IPCC, 2001). Greater materials efficiency typically reduces energy demands in processing because of reduced materials handling. For example, recycling materials often reduces energy consumption per unit of output by 26 to 95% (Table 8-1). Further work on materials substitution also holds promise for reduced energy consumption and emissions reduction.[13]

Table 8-1. Energy reductions in recycling

The prospects for greater energy efficiency are equally substantial. Martin *et al.* (2001) characterized more than 50 key emerging energy efficient technologies, including efficient Hall-Heroult cell retrofits, black liquor gasification in pulp production, and shape casting in steel industries. Worrell *et al.* (2004) covers many of the same technologies and notes that significant potential exists in utilizing efficient motor systems and advanced cogeneration technologies.

At the same time, energy is a valuable production input that, along with capital, can substitute for labor as a means of increasing productivity. Thus, overall productivity gains in industry can be both energy-saving and energy-augmenting, and the net impact depends on the nature of technological innovation and the expected long-run cost of energy relative to other inputs. This suggests that, if policies to manage carbon emissions from industry are to be effective, they would need to provide a significant signal to technology innovators and adopters to reflect the negative value that society places on carbon emissions. This in turn suggests the application of regulations or financial instruments, examples being energy efficiency regulations, carbon management regulations, and fees on carbon emissions.

WASTE MANAGEMENT CARBON CYCLE

The carbon cycle associated with human wastes includes industrial, commercial, construction, demolition, and residential waste. Municipal solid waste contains significant amounts of carbon. Paper, plastics, yard trimmings, food scraps, wood, rubber, and textiles made up more than 80% of the 236 Mt of municipal solid waste generated in the United States in 2003 (EPA, 2005) and the 25 Mt generated in Canada (Statistics Canada, 2004), as shown in Table 8-2. In Mexico, as much as 20% of wastes are not systematically collected; no disaggregated data are available (EPA, 2005).

Table 8-2. Waste materials flows by region in North America, 2003

[13] For example, substitute petrochemical feedstocks by biomass or concrete by wood in home foundations.

A portion of municipal solid waste is recycled: 31% in the United States, 27% in Canada. Up to 14% of the remaining waste is incinerated in the United States, slightly less in Canada. Incineration can reduce the waste stream by up to 80%, but this ensures that more of the carbon reaches the atmosphere as opposed to being sequestered (or subsequently released as methane) in a landfill. Incineration, however, can be used to cogenerate electricity and useful heat, which may reduce carbon emissions from stand-alone facilities.

Once in a landfill, carbon in wastes may be acted upon biologically, releasing roughly equal amounts of CO_2 and methane (CH_4) by volume[14] depending on ambient conditions, as well as a trace amount of carbon monoxide and volatile organic compounds. While no direct data on the quantity of CO_2 released from landfills exists, one can estimate the CO_2 released by using this ratio; the estimated amount of CO_2 released from landfills in Canada and the United States (no data from Mexico) would be approximately 38 Mt,[15] a relatively small amount compared to total other (sub)sectors in this chapter. Also recall that these emissions are from biomass and, in the context of IPCC assessment guidelines, are considered GHG-neutral.

Depending on the degree to which aerobic or anaerobic metabolism takes place, a considerable amount of carbon remains unaltered and more or less permanently stored in the landfill (75%–80%; see Barlaz, 1990, 1994; and Bogner and Spokas, 1993). Because data on the proportions of carboniferous material entering landfills can be estimated, approximate carbon contents of these materials can be determined and the degree to which these materials can decompose, it would be possible to estimate the amount of carbon sequestered in a landfill site (see EPIC, 2001; Mohareb *et al.*, 2003; EPA, 2003; EPA, 2005). While EPA (2005) provides an estimate of carbon sequestered in US landfills (see Table 8-2), no data are available for other regions.

Anaerobic digestion generates methane gases that can be captured and used in cogenerators. Many of the 1,800 municipal solid waste sites in 2003 in the United States captured and combusted landfill-generated methane; about half of all the methane produced was combusted or oxidized in some way (EPA, 2005). In Canada, about 23% of the methane emissions were captured and utilized to make energy in 2002 (Mohareb *et al.*, 2003). The resultant CO_2 released from such combustion is considered biological in origin. Thus, only methane emissions, at 21 times the CO_2 warming potential, are included as part of GHG inventories. Their combustion greatly alleviates the net contribution to GHG emissions and, if used in cogeneration, may offset the combustion of fossil fuels elsewhere.

[14]Based on gas volumes, this means that roughly equivalent amounts of carbon are released as CO_2 as CH_4.

[15]14 Mt of CH_4 (see Table 8-3) are equivalent, volume wise at standard temperature and pressure, to 38 Mt of CO_2. This derived estimate is highly uncertain and not of the same caliber as other emissions data provided here.

COSTS RELATED TO CONTROLLING ANTHROPOGENIC IMPACTS ON THE CARBON CYCLE

Defining costs associated with reducing anthropogenic impacts on the carbon cycle is a highly contentious issue. Different approaches to cost assessments (top-down, bottom-up, applicable discount rates, social costing, cost effectiveness, no regrets), different understandings of what costs include (risk, welfare, intangibles, capital investment cycles), different values associated with energy demand in different countries (accessibility, availability, infrastructure, resource type and size), actions and technologies included in the analysis, and the perspective on technology development all have an impact on evaluating costs. Should analysts consider only historical responses to energy prices, production and demand elasticities or income changes? Does one consider only technology options and their strict financial costs or see historic technology investments as sunk costs? Should one include producers' or consumers' welfare? Are there local, national, international issues?

Cost variation within industries is significant. Costs associated with various methods to reduce emissions also vary. Reduction methods can be classified as:

- reducing or altering process/fugitive emissions,
- energy efficiency, including combined heat and power,
- process changes,
- fuel substitution,
- carbon capture and storage.

One can attribute potential reductions over a set time period under a range of costs. We suggest the cost-range categories ("A" through "D") shown in Table 8-3. The table contains estimates of the percentage reduction by industry under these cost categories. Costs are not drawn from a single source but are the authors' estimates based on a long history of costs reported in various documents.[16] Some studies focus on technical potential and don't provide the cost of achieving the reductions. As such, achievable reductions are likely overestimated. Others describe optimization models that provide normative costs and likely overestimate potentials and underestimate costs. Still others use top-down approaches where historic data sets are used to determine relationships between emissions and factors of production; costs are often high and emissions reductions underestimated.

Table 8-3. Approximate costs and reductions potential

[16]Studies vary widely in how they define system boundaries, baseline and time periods, which sectors or subsectors are included, economic assumptions, and many other factors. See *Some Explanatory Notes* below Table 8-3 for a list.

When looking at cost numbers like this, one should remember that, for each $10 cost increment per t CO_2 (or about $37 per t C), gasoline prices would increase about 2.4¢/L (9¢/U.S. gal). Diesel fuel cost would be nearly 2.7¢/L (10¢/U.S. gal). Costs per GJ[17] vary by fuel: coal rises about 90¢/GJ, depending on type, HFO by 73¢, and natural gas by 50¢. At 35% efficiency, coal-fired electricity generation would be about 0.8¢/kWh higher, about 0.65¢/kWh for HFO, and about 0.45¢/kWh for natural gas.

Of course, as the cost of carbon increases, one moves up the carbon supply curve for industrial sectors. But reductions become marginal or insignificant and so are not included in Table 8-3. If a cell in Table 8-3 shows two cost categories (e.g., A/B) and two reduction levels ($\%Q_{red}$ is 15/20), the value associated with the second portrays the *additional* reduction at that increased expenditure level. Thus, spending up to $50/t CO_2 to improving efficiency in metal smelting implies a potential reduction of 35% (see Table 8-3). Reductions in each category are *not* additive for an industry type because categories are not independent.

Because not all reduction methods are applicable to all industries, as one aggregates to an "all industry" level (top line, Table 8-3), the total overall emissions reduction level may be less than any of the individual industries sited.

Some Explanatory Notes

Data come from a variety of sources and do not delineate costs as per the categories describe here. Data sources can be notionally categorized into the following groups (with some references listed twice):[18]

- *General overviews*: Grubb *et al*., 1993; Weyant *et al*., 1999;[19] Grubb *et al*., 2002; Löschel, 2002.
- *Top-down analyses*: McKitrick, 1996; Herzog, 1999; Sands, 2002; McFarland *et al*., 2004; Schäfer and Jacoby, 2005; Matysek, *et al*., 2006.
- *Bottom up analyses*: Martin *et al.*, 2001; Humphreys and Mahasenan, 2002; Worrell *et al.*, 2004; Kim and Worrell, 2002; Morris *et al.*, 2002; Jaccard *et al.*, 2003; DOE, 2006; IEA, 2006.
- *Hybrid model analyses*: Böhringer, 1998; Jacobsen, 1998; Edmonds *et al.*, 2000; Koopmans and te Velde, 2001; Jaccard, 2002; Frei *et al.*, 2003; Jaccard *et al.*, 2003; Jaccard, Nyboer, *et al.*, 2003; Edenhofer *et al.*, 2006.
- *Others*: Newell *et al.*, 1999; Sutherland, 2000; Jaffe *et al.*, 2002.

[17] A GJ is slightly smaller than 1 MMBtu (1 GJ = 0.948 MMBtu)

[18] Two authors are currently involved with IPCC's upcoming fourth assessment report where estimated costs of reduction are provided. Preliminary reviews of the cost data presented there do not differ substantially from those in table 8-3.

[19] John Weyant, Stanford, is currently editing another similar analysis to this listed publication to be released some time in 2006. ***DETAILS FORTHCOMING…***

Process and Fugitives: Process and fugitive reductions are only available in certain industries. For example, because wood-products industries burn biomass, fugitives are higher than in other industries and reduction potentials exist.

In the waste sector, the reductions potentials are very large; we have simply estimated possible reductions if we were to trap and burn all landfill methane. The costs for this are quite low. EPA (2003a) estimates of between 40% and 60% of methane available for capture may generate net economic benefits.

Energy Efficiency: The potential for emissions reductions from efficiency improvements is strongly linked with both process change and fuel switching. For example, moving to Cermet-based processes in electric arc furnaces in steel and aluminum smelting industries can significantly improve efficiencies and lower both combustion and process GHG emissions.

A "bottom up" technical analyses tends to show higher potentials and lower costs than when one uses a hybrid or a "top-down" approach to assess reduction potentials due to efficiency improvements; Table 8-3 portrays the outcome of the more conservative hybrid (mix of top-down and bottom-up) approach and provides what some may consider conservative estimates of reduction potential (see particularly Martin *et al.*, 2001; Jaccard *et al.*, 2002; Jaccard *et al.*, 2003; Jaccard, Nyboer, *et al.*, 2003; Worrell *et al.*, 2004).

Process Change: Reductions from process change requires not only an understanding of the industry and its potential for change but also an understanding of the market demand for industry products that may change over time. In pulp production, for example, one could move from higher quality kraft pulp to mechanical pulp and increase production ratios (the kraft process only converts one-half the input wood into pulp), but will market acceptability for the end product be unaffected? Numerous substitution possibilities exist in the rather diverse Other Manufacturing industries (carpet recycling, alternative uses for plastics, etc.).

Fuel Substitution: It is difficult to isolate fuel substitution and efficiency improvement because fuels display inherent qualities that affect efficiency. Fuel substitution can reduce carbon flow but efficiency may become worse. In wood products industries, shifts to biomass reduces emissions but increases energy use. In terms of higher heating values, shifts from coal or oil to natural gas may worsen efficiencies while reducing emissions.[20]

Carbon Capture and Storage (CC&S): In one sense, all industries and landfills could reduce emissions through CC&S but the range of appropriate technologies has not been fully defined and/or the costs are very high. For example, one could combust fuels in a pure oxygen environment such that the exhaust steam is CO_2-rich and suitable for capture and storage. Even so, some industries, like cement

[20] As the ratio of hydrogen to carbon rises in a fossil fuel, more of the total heat released upon combustion is caught up in the latent heat of vaporization of water and is typically lost to process. This loss is equivalent the difference between a fuel's higher heating value and its lower heating value.

production, are reasonable candidates for capture, but cost of transport of the CO_2 to storage may prohibit implementation (see particularly Herzog, 1999; DOE, 2006).

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Table 8-1. Energy reductions in recycling

Recycled material	Energy saved	Recycled material	Energy saved
Aluminum	95%	Glass	31%
Tissue paper	54%	Newsprint	45%
Printing/writing paper	35%	Corrugated cardboard	26%
Plastics	57%–75%	Steel	61%

Source: Hershkowitz, 1997.

Table 8-2. Waste materials flows by region in North America, 2003

	United States	Canada	Mexico
Total waste (Mt yr^{-1})	236.0	24.8	29.2
Recycled	72.0	6.6	–
Carbon-based waste	197.1	19.6	–
Carbon-based waste recycled	47.3*	4.3	–
Carbon sequestered (CO_2 equivalents)	10.1	–	–
Methane (kt yr^{-1})			
Generated	12,486	1,452	–
Captured, oxidized	6,239	336	–
Emitted	6,247	1,117	–
Emitted (CO_2 equivalents)	131,187	23,453	–

* Calculated estimate

Source: EPA, 2003b, 2005; Statistics Canada, 2004; Mohareb, 2003 for Canada methane data; California Environmental Protection Agency, 2003 for Mexico data point.

Table 8-3. Approximate costs and reductions potential

	Reduction of fugitives		Energy efficiency		Process change		Fuel substitution		Carbon Capture and Storage	
Sector	Cost category	$\%Q_{red}$	Cost category*	$\%Q_{red}$*	Cost category	$\%Q_{red}$	Cost category	$\%Q_{red}$	Cost category	$\%Q_{red}$
All industry	B	3	A/B	12/8	B	20	A	10	C	30
P&P	B	5	A/B	10/5	B	40	A	40	D	?
Nonmetal min			A	10	A	40	A	40	C	80
Metal smelt			A/B	15/20	B	10	A	15	C	40
Mining			A	5						
Chemicals	B	10	A/B	10/5	B	25	A	5	C/D	40/20
Forest products	B	5	A	5						
Other man			A	15	A	20	A	5	D	?
Waste	A	90							D	30

*If two letters appear, two percent quantities reduced are shown. Each shows the quantity reduced at that cost. That is, if all lesser and higher costs were made, emissions reduction would be the sum of the two values.

Note: The reductions across categories are NOT additive. For example, if “Carbon Capture and Storage” is employed, then fuel switching would have little bearing on the emissions reduction possible. Also, it is difficult to isolate process switching and efficiency improvements.

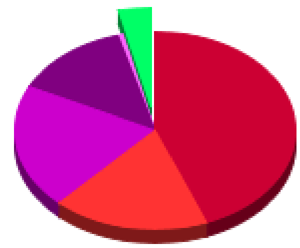
The “Cost Categories” are as follows:

CO_2-Based: **A:** \$0–\$25/t CO_2; **B:** \$25–\$50/t CO_2; **C:** \$50–\$100/t CO_2; **D:** >\$100/t CO_2

Carbon-Based: **A:** \$0–\$92/t C; **B:** \$92–\$180/t C; **C:** \$180–\$367/t C; **D:** >\$367/t C

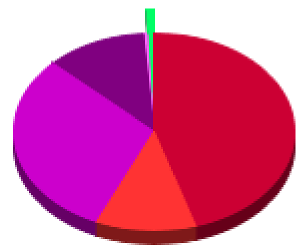
World

Sector	Mt CO_2	%
Energy	23,432.1	96.3
Electricity & Heat	*10,731.8*	*44.1*
Manufacturing & Construction	*4,322.9*	*17.8*
Transportation	*4,964.5*	*20.4*
Other Fuel Combustion	*3,265.3*	*13.4*
Fugitive Emissions	*147.6*	*0.6*
Industrial Processes	897.7	3.7
Total	**24,329.8**	



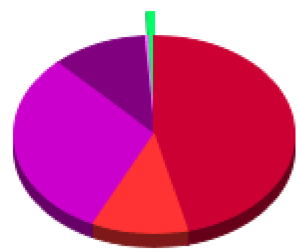
North America (w/ Mexico)

Sector	Mt CO_2	%
Energy	6,576.5	98.9
Electricity & Heat	*3,017.0*	*45.3*
Manufacturing & Construction	*758.7*	*11.3*
Transportation	*2,016.6*	*30.5*
Other Fuel Combustion	*757.1*	*11.6*
Fugitive Emissions	*27.2*	*0.4*
Industrial Processes	66.8	1.0
Total	**6,643.3**	



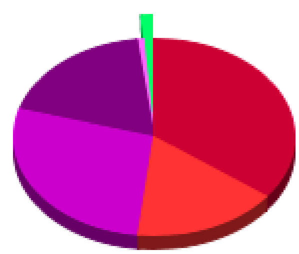
United States of America

Sector	Mt CO_2	%
Energy	5,675.4	99.2
Electricity & Heat	*2,645.0*	*46.2*
Manufacturing & Construction	*621.4*	*10.9*
Transportation	*1,761.4*	*30.8*
Other Fuel Combustion	*624.5*	*10.9*
Fugitive Emissions	*23.1*	*0.4*
Industrial Processes	44.7	0.8
Total	**5,720.1**	



Canada

Sector	Mt CO_2	%
Energy	535.9	98.8
Electricity & Heat	*191.7*	*35.3*
Manufacturing & Construction	*89.2*	*16.4*
Transportation	*150.5*	*27.7*
Other Fuel Combustion	*100.5*	*18.5*
Fugitive Emissions	*4.1*	*0.7*
Industrial Processes	6.6	1.2
Total	**542.5**	



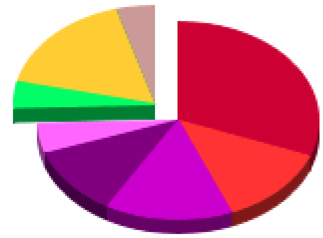
Mexico

Sector	Mt CO_2	%
Energy	365.2	95.9
Electricity & Heat	*180.3*	*47.4*
Manufacturing & Construction	*48.1*	*12.6*
Transportation	*104.7*	*27.5*
Other Fuel Combustion	*32.1*	*8.4*
Fugitive Emissions	*--*	*--*
Industrial Processes	15.5	4.1
Total	**380.6**	

Fig. 8-1A. CO_2 emissions by sector in 2002. *Source*: Climate Analysis Indicators Tool (CAIT) Version 3.0 (Washington, D.C.: World Resources Institute, 2005).

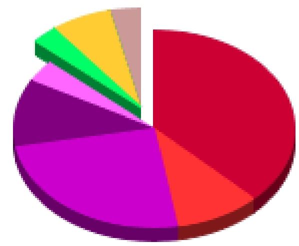
World

Sector	Mt CO_2	%
Energy	24,789.9	74.5
Electricity & Heat	*10,269.4*	*30.9*
Manufacturing & Construction	*4,327.9*	*13.0*
Transportation	*4,809.7*	*14.5*
Other Fuel Combustion	*3,742.4*	*11.2*
Fugitive Emissions	*1,640.5*	*4.9*
Industrial Processes	1,366.8	4.1
Agriculture	5,631.5	16.9
Waste	1,483.6	4.5
Total	**33,271.8**	



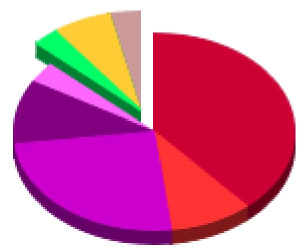
North America (w/ Mexico)

Sector	Mt CO_2	%
Energy	7,004.8	86.2
Electricity & Heat	*3,027.6*	*37.3*
Manufacturing & Construction	*809.6*	*10.0*
Transportation	*1,971.1*	*24.3*
Other Fuel Combustion	*877.2*	*10.8*
Fugitive Emissions	*319.2*	*3.9*
Industrial Processes	239.0	2.9
Agriculture	580.9	7.1
Waste	300.0	3.7
Total	**7,610.9**	



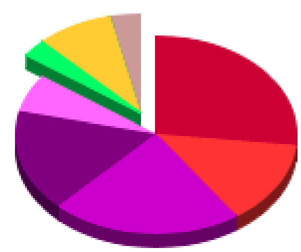
United States of America

Sector	Mt CO_2	%
Energy	6,005.5	86.8
Electricity & Heat	*2,670.6*	*38.6*
Manufacturing & Construction	*657.9*	*9.5*
Transportation	*1,719.9*	*24.9*
Other Fuel Combustion	*723.6*	*10.5*
Fugitive Emissions	*233.5*	*3.4*
Industrial Processes	198.4	2.9
Agriculture	469.9	6.8
Waste	243.3	3.5
Total	**6,917.1**	



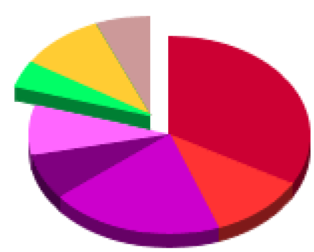
Canada

Sector	Mt CO_2	%
Energy	589.5	85.0
Electricity & Heat	*185.9*	*26.8*
Manufacturing & Construction	*94.6*	*13.6*
Transportation	*150.0*	*21.6*
Other Fuel Combustion	*115.3*	*16.6*
Fugitive Emissions [1]	*43.6*	*6.3*
Industrial Processes [2]	19.3	2.8
Agriculture	60.8	8.8
Waste	24.2	3.5
Total	**693.8**	



Mexico

Sector	Mt CO_2	%
Energy	409.8	79.8
Electricity & Heat	*171.1*	*33.3*
Manufacturing & Construction	*57.1*	*11.1*
Transportation	*101.2*	*19.7*
Other Fuel Combustion	*38.3*	*7.5*
Fugitive Emissions [1]	*42.1*	*8.2*
Industrial Processes [2]	21.3	4.2
Agriculture	50.2	9.8
Waste	32.5	6.3
Total	**513.8**	



[1] N_2O data not available. [2] CH_4 data not available.

Fig. 8-1B. GHG emissions by sector in 2000, CO_2, CH_4, N_2O, PFCs, HFCs, and SF_6. *Source*: Climate Analysis Indicators Tool (CAIT) Version 3.0 (Washington, D.C.: World Resources Institute, 2005).

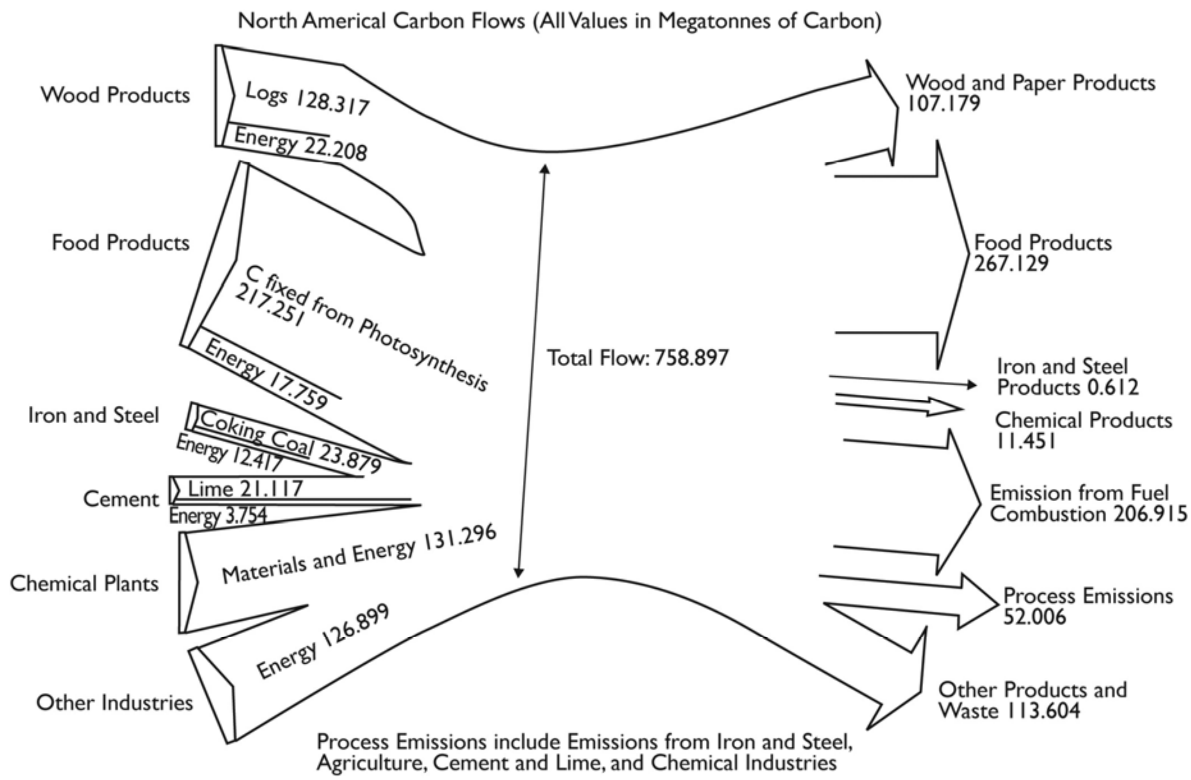


Fig. 8-2. Carbon flows for Canada, the United States and Mexico combined. Values in kilotons carbon can be converted to kilotons CO_2 equivalents by multiplying by 44/12, the ratio of carbon dioxide mass to carbon mass. Comparable diagrams for the individual countries are in Appendix 8A**.** *Source*: Energy data from Statistics Canada Industrial Consumption of Energy survey, Conversion coefficients, IEA Oil Information 2004, IEA Coal Information 2005, IEA Natural Gas Information 2004. Process emissions from Environment Canada, *Canada GHG Inventory, 2002,* EPA, U.S. Emissions Inventory. Production data from Statistics Canada, CANSIM Table 002-0010, Tables 303-0010, -0014 to -0021, -0024, -0060, Pub. Cat. Nos.: 21-020, 26-002, 45-002, Canadian Pulp and Paper Association on forestry products. Production of forestry products: USDA Database; FO-2471000, -2472010, -2482000, -2483040, -6342000, -6342040, U.S. Timber Production, Trade, Consumption, and Price Statistics 1965–2005. Production of organic products (e.g., food): USDA PS&D Official Statistical Results. Steel: International Iron and Steel institute, World steel in figures 2003. Minerals production: USGS mineral publications.

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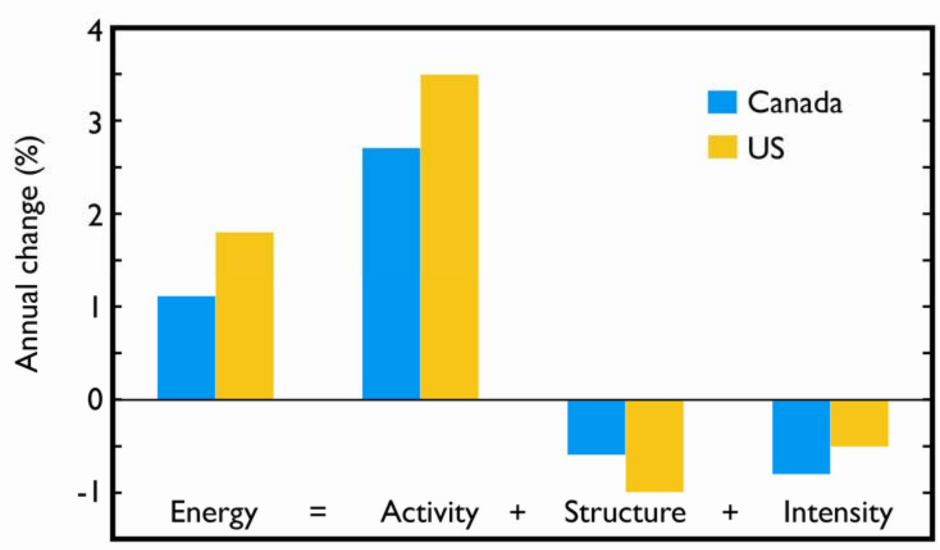
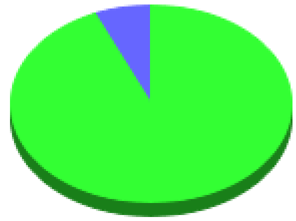


Fig. 8-3. Decomposition of energy use, manufacturing sector, 1990–1998. *Source*: IEA, 2004.

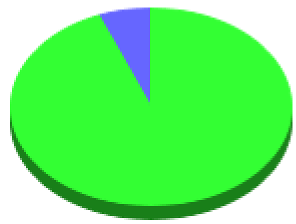
World

Gas	Mt CO_2	%
CH_4	1,386.4	93.5
N_2O	97.2	6.5
Total	**1,483.6**	



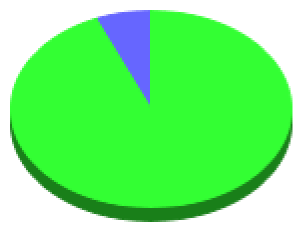
North America (w/ Mexico)

Gas	Mt CO_2	%
CH_4	281.8	93.9
N_2O	18.2	6.1
Total	**300.0**	



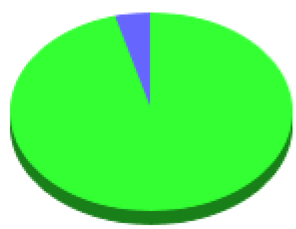
United States of America

Gas	Mt CO_2	%
CH_4	227.7	93.6
N_2O	15.6	6.4
Total	**243.3**	



Canada

Gas	Mt CO_2	%
CH_4	23.2	95.8
N_2O	1.0	4.2
Total	**24.2**	



Mexico

Gas	Mt CO_2	%
CH_4	31.0	95.2
N_2O	1.6	4.8
Total	**32.5**	

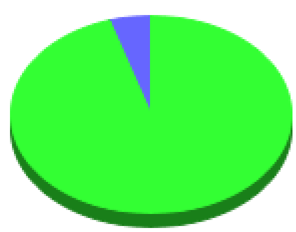


Fig. 8-4. GHG emissions by gas from waste in 2000. *Source*: Climate Analysis Indicators Tool (CAIT) Version 3.0 (Washington, D.C.: World Resources Institute, 2005).

Appendix 8A

Industry and Waste Management – Supplemental Material

This appendix presents diagrams of the carbon flows in Canada, the United States, and Mexico, respectively (Figs. 8A-1 through 8A-3). The numerical data in these figures are shown in thousands of metric tons of carbon, which can be converted into thousands of metric tons of CO_2 equivalents by multiplying the carbon values by 44/12 (i.e., the ratio of carbon dioxide mass to carbon mass). The combined carbon flows for all three nations are presented in Fig. 8-2 in Chapter 8 of this report.

Figure 8A-1. Carbon flows, Canada.

Figure 8A-2. Carbon flows, United States.

Figure 8A-3. Carbon flows, Mexico.

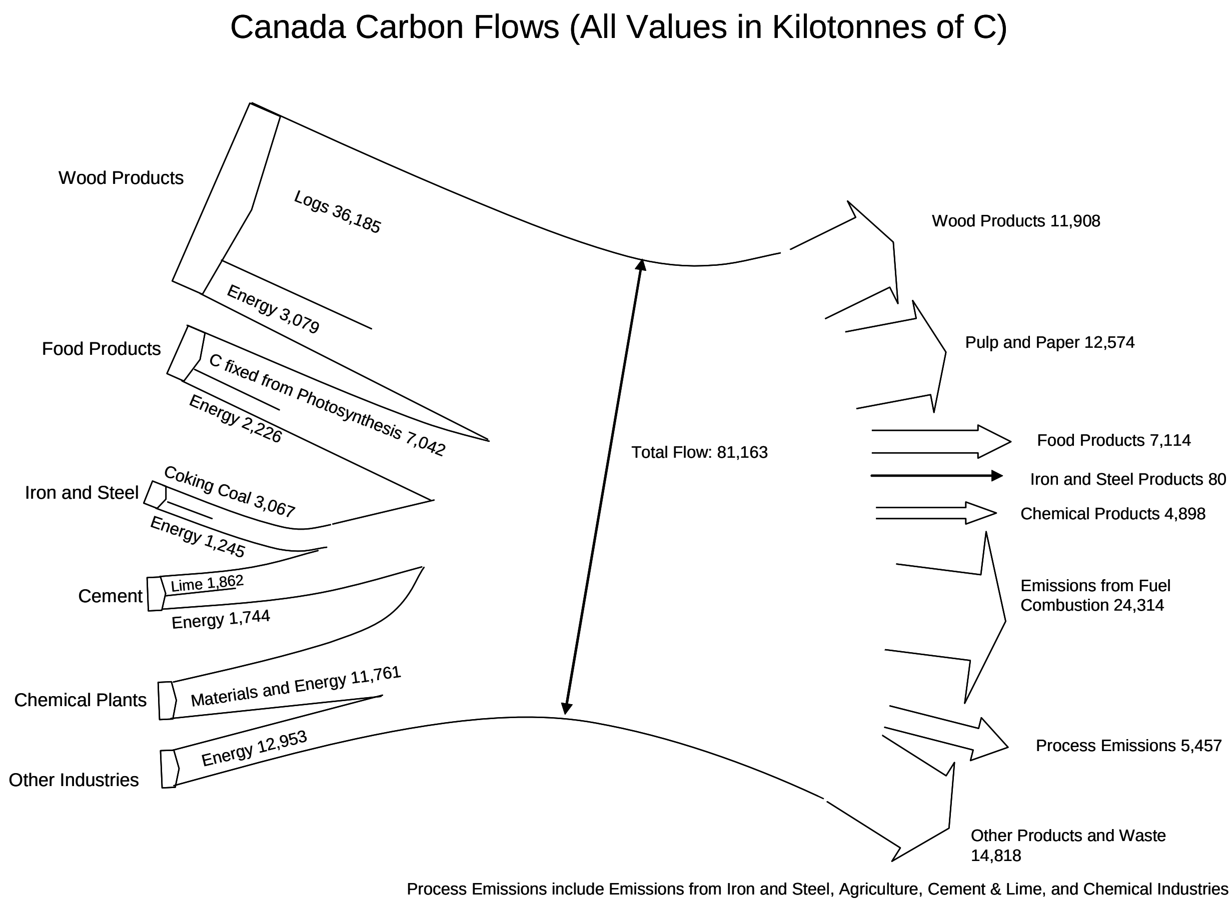


Fig. 8A-1. Carbon flows, Canada. *Source*: Energy data from Statistics Canada Industrial Consumption of Energy survey, conversion coefficients and process emissions from Environment Canada, *Canada GHG Inventory, 2002*. Production data from Statistics Canada, CANSIM Table 002-0010, Tables 303-0010, -0014 to -0021, -0024, -0060, Pub. Cat. Nos.: 21-020, 26-002, 45-002, Canadian Pulp and Paper Association on forestry products.

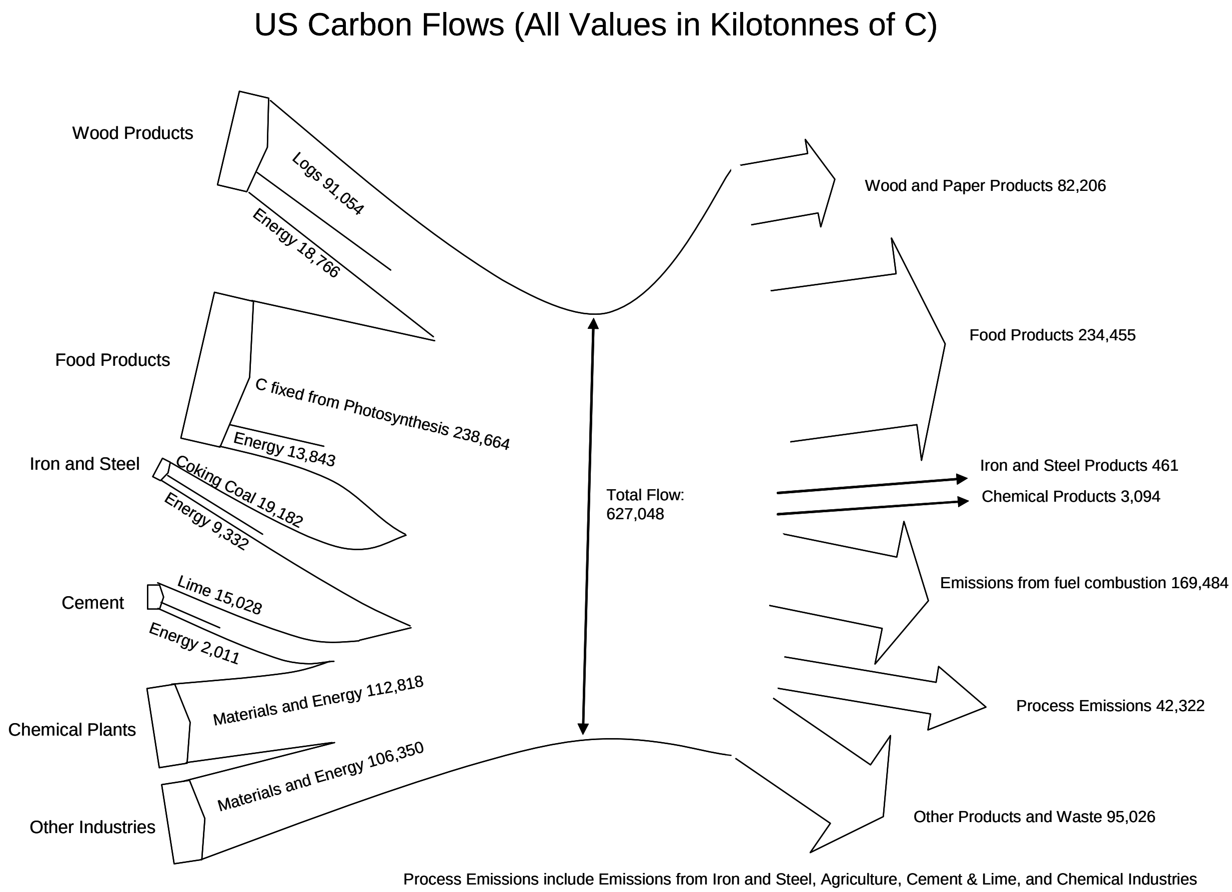


Fig. 8A-2. Carbon flows, United States. *Source*: Energy data from IEA Oil Information 2004, IEA Coal Information 2005, IEA Natural Gas Information 2004. Process emissions: EPA, U.S. Emissions Inventory. Production of forestry products: USDA Database; FO-2471000 and -2472010, U.S. Timber Production, Trade, Consumption, and Price Statistics 1965–2005, Production of organic products (e.g., food): USDA PS&D Official Statistical Results, Steel: International Iron and Steel institute, World steel in figures 2003, Minerals production: USGS mineral publications.

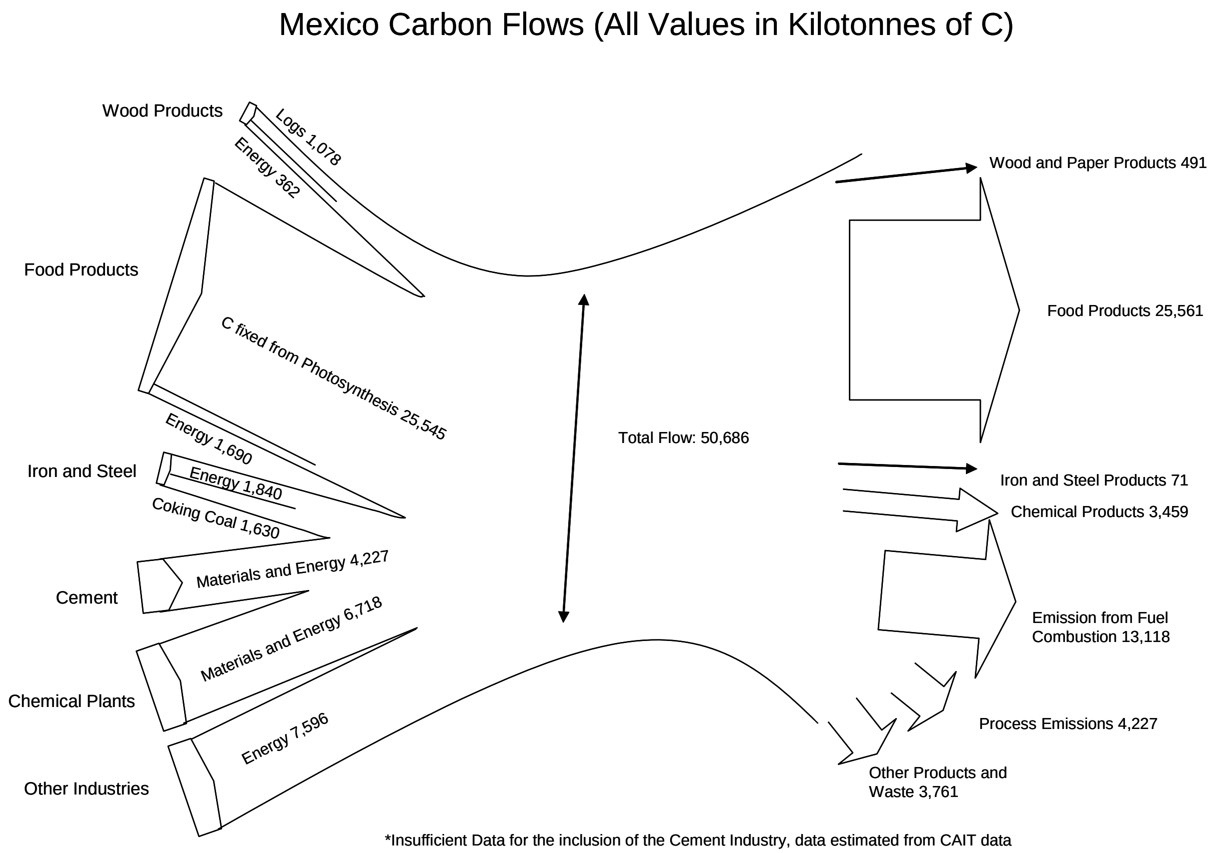


Fig. 8A-3. Carbon flows, Mexico. *Source*: Energy data from IEA Oil Information 2004, IEA Coal Information 2005, IEA Natural Gas Information 2004. Process emissions: EPA, U.S. Emissions Inventory. Production of forestry products: USDA Database; FO-2471000, -2472010, -2482000, -2483040, -6342000, -6342040. Production of organic products (e.g., food): USDA PS&D Official Statistical Results. Steel: International Iron and Steel institute, World steel in figures 2003.

Chapter 9. Buildings

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KEY FINDINGS

- The buildings sector of North America was responsible for annual carbon dioxide (CO_2) emissions of 671 Mt C in 2003, which is 37% of total North American CO_2 emissions and 10% of global emissions. U.S. buildings alone are responsible for more CO_2 emissions than total CO_2 emissions of any other country in the world, except China.
- Carbon dioxide emissions from energy use in buildings in the United States and Canada increased by 30% from 1990 to 2003, an annual growth rate of 2.1% per year.
- Carbon dioxide emissions from buildings have grown with energy consumption, which in turn is increasing with population and income. Rising incomes have led to larger residential buildings and increased household appliance ownership.
- These trends are likely to continue in the future, with increased energy efficiency of building materials and equipment and slowing population growth, especially in Mexico, only partially offsetting the general growth in population and income.
- Options for reducing the CO_2 emissions of new and existing buildings include increasing the efficiency of equipment and implementing insulation and passive design measures to provide thermal comfort and lighting with reduced energy. Current best practices can reduce emissions from buildings by at least 60% for offices and 70% for homes. Technology options need to be supported by a portfolio of policy options that take advantage of synergies, avoid unduly burdening certain sectors and are cost effective.
- Because reducing CO_2 emissions from buildings is currently secondary to reducing building costs, continued improvement of energy efficiency in buildings and reduced CO_2 emissions from the building sector will require a better understanding of the total societal cost of CO_2 emissions as an externality of building costs, including the costs of mitigation compared to the costs of continued emissions.

In 2003, buildings were responsible for 615 Mt C[1] in the United States (DOE-EIA, 2005), 40 Mt C in Canada (Natural Resources Canada, 2005) and 17 Mt C in Mexico (SENER México, 2005), for a total of 671 Mt C in North America. According to the International Energy Agency, total energy-related emissions in North America in this year were 1815 Mt (IEA, 2005). Therefore, buildings were responsible for 37% of energy-related emissions in North America. North American buildings accounted for 10% of global energy emissions, which totaled 6814 Mt C. U.S. buildings alone are responsible for more CO_2 emissions than total CO_2 emissions of any other country in the world except China (Kinsey *et al.*, 2002). Significant carbon emissions are due to energy consumption during the operation of the buildings; other emissions, not well quantified, may occur from water use in and around the buildings and from land-use impacts related to buildings. Buildings are responsible for 72% of U.S. electricity consumption and 54% of natural gas consumption (DOE/EERE, 2005).[2] The discussions in this chapter include an accounting of CO_2 emissions from electricity consumed in the buildings sector; however, this represents a potential double-counting of the CO_2 emissions from fossil fuels that are used to generate that electricity (see Chapter 6). This chapter provides a description of how energy, including electrical energy, is used within the buildings sector. Following the discussion of such end uses of energy, this chapter then describes the opportunities and potential for reducing energy consumption within the sector.

Many options are available for reducing the carbon impacts of new and existing buildings, including increasing equipment efficiency and implementing alternative design, construction, and operational measures to provide thermal comfort and lighting with reduced energy. Current best practices can reduce carbon emissions for buildings by at least 60% for offices[3] and up to 70% for homes.[4] Residential and commercial buildings in the United States and Canada occupy 27 billion m^2 (2.7 million hectares) of floor space, providing a large area available for siting non-carbon-emitting on-site energy supplies (e.g., photovoltaic panels on roofs)[5]. With the most cutting-edge technology, at the least, emissions can be dramatically reduced, and, at best, buildings can produce electricity without carbon emissions by means of on-site renewable electricity generation.

Carbon Fluxes

Carbon fluxes from energy emissions in buildings are well understood, since primary energy inputs from the source of production are tracked, their emissions rates are known, and the total end user consumption data are gathered and reported by energy utilities, typically monthly. The quantity of energy

[1]Carbon dioxide emissions only.
[2]See Tables 1.1.6 and 1.1.7 in DOE/EERE (2005).
[3]Leadership in Energy and Environment Design (LEED) Gold Certification (USGBC, 2005).
[4]U.S. DOE Building America Program (DOE/EERE, 2006).
[5]A recent study estimates a potential of 711 GW generation capacity from rooftop installation of photovoltaic systems (Chaudhari *et al*., 2004).

consumed by each particular end use is slightly less well known because attribution requires detailed data on use patterns in a wide variety of contexts. The governments of North America have invested in detailed energy consumption surveys, which allow researchers to identify opportunities for reducing energy use.

The largest contribution to carbon emissions from buildings is through the operation of energy-using equipment. The energy consumed in the average home accounts for 2.9 metric tons[6] of carbon per year in the United States, 1.7 metric tons[7] per year in Canada, and 0.6 metric tons[8] in Mexico (DOE/EIA, 2005; Natural Resources Canada, 2005; SENER México, 2004). Energy consumption in a 500-m^2 commercial, government, or public-use building in the United States produces 1.9 metric tons of carbon (DOE/EIA, 2005).[9] Energy consumption includes electricity as well as the direct combustion of fossil fuels (natural gas, bottled gas and petroleum distillates) and the burning of wood. Because most electricity in North America is produced from fossil fuels, each kilowatt-hour consumed in a building contributed about 180 g of carbon to the atmosphere in 2003 (DOE/EIA, 2005).[10] The equivalent amount of energy from natural gas or other fuels contributed about 52 g of carbon (DOE/EIA, 2005).[11] Renewable energy accounted for 9% of electricity production in 2003, down from 12% in 1990. Renewable site energy use in buildings also decreased in that time, from 4% to 2%, mostly due to decreasing use of wood as a household fuel (DOE/EERE, 2005).[12]

Buildings-sector CO_2 emissions and the relative contribution of each end use are shown in Fig. 9-1. In the United States, five end uses account for 87% of primary energy consumption in buildings: space conditioning (including space heating, cooling and ventilation), 40.9%; lighting, 19.8%; water heating, 10.5%; refrigeration, 7.9%; and electronics (including televisions, computers, and office equipment), 7.7% (DOE/EERE, 2005).[13] Space heating and cooling are the largest single uses for residences, commercial, and public-sector buildings, accounting for 46% and 35% of primary energy, respectively, in the United States (DOE/EERE, 2005).[14] Water heating is the second-highest energy consumer in the United States and Canada, while lighting is the second-highest source of carbon dioxide emissions, due to the higher emissions per unit of electricity compared to natural gas.

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[6]U.S. residential sector emissions of 334 Mt CO_2 divided by 114 million households in 2004; the numerical value given for "tons of carbon" is for carbon dioxide emissions only.

[7]Canada residential sector emissions of 20.6 Mt CO_2 divided by 12.2 million households in 2003.

[8]Mexico residential sector emissions of 13.2 Mt CO_2 divided by 23.8 million households in 2004.

[9]U.S. commercial sector emissions per m^2 in 2003 times 500 m^2.

[10]U.S. emissions from electricity divided by delivered energy.

[11]U.S. emissions from electricity divided by delivered energy.

[12]See Table 1.5.4 and Summary Table 2 in DOE/EERE (2005).

[13]Does not include adjustment EIA uses to relieve differences between data sources.

[14]Table 1.2.3 and Table 1.3.3 in DOE/EERE (2005); available at http://buildingsdatabook.eere.energy.gov (2003 data).

Fig. 9-1. U.S. carbon emissions by sector and—for commercial and residential buildings—by end use.

Heating and cooling loads are highly climate dependent; colder regions use heating during much of the year (primarily with natural gas), while warm regions seldom use heating. The majority of U.S. households own an air conditioner; and, although air-conditioner ownership has been historically low Mexico,[15] sales of this equipment are now growing significantly, 14% per year over the past 10 years.[16] Space-conditioning energy end use depends significantly on building construction (e.g., insulation, air infiltration) and operation (thermostat settings). Water heating is a major consumer of energy in the United States and Canada, where storage-tank systems are common.

Aside from heating and cooling, lighting, and water heating, energy is consumed by a variety of appliances, mostly electrical. Most homes in the United States and Canada own all of the major appliances, including refrigerators, freezers, clothes washers, clothes dryers, dishwashers, and at least one color television. The remainder of household energy consumption comes from small appliances (blenders and microwaves, for example) and increasingly from electronic devices, such as entertainment equipment and personal computers. In Mexico, 96.6% of households used electricity in 2005, and recent years have shown a marked growth in appliance ownership: ownership rates in 2000 were 85.9% for televisions, 68.5% for refrigerators, 52% for washing machines, and only 9.3% for computers. By the end of 2005 ownership rates had grown to 91% for televisions, 79% for refrigerators, 62.7% for washing machines, and 19.6% for computers (INEGI, 2005).

Many end uses—such as water heating, and space heating, cooling, and ventilation—occur in most commercial sector buildings. Factors such as climate and building construction influence the carbon emissions by these buildings. In addition, commercial buildings contain specialized equipment, such as large-scale refrigeration units in supermarkets; cooking equipment in food preparation businesses; and computers, printers, and copiers in office buildings. Office equipment is the largest component of electricity use aside from cooling and lighting. Due to heat from internal loads, many commercial buildings use air-conditioning year round in most climates in North America.

Residential and commercial buildings in the United States are responsible for 38% of CO_2 emissions from energy nationally and 33% of emissions from energy in North America as a whole. Total emissions from buildings in the United States are ten times as high as in the other two countries combined, due to a large population compared to Canada, and high per capita consumption compared to Mexico. On a per capita basis, building energy consumption in the United States is comparable with that of Canada, about

[15]Air conditioners have typically been used only in the northern and coastal areas of Mexico.
[16]Air conditioner sales 1995–2004 from Asociacion Nacional de Fabricantes de Aparatos Domesticos (ANFAD).

40 GJ equivalent per person per year. This is about six times higher than in Mexico, where 7 GJ is consumed per person per year.

In general, contributions from the residential sector are roughly equal to that of the commercial sector, except in Mexico, where the commercial sector contributes less. Electricity contributes twice as many emissions as all other fuels combined in the United States and Mexico (2.2 and 2.1 times as much, respectively). In Canada, natural gas is on par with electricity (1.03 times as many emissions), due to high heating loads resulting from the cold climate. Fuel oil represents most of Canada's "other fuels" for the commercial sector. Firewood (*leña*) remains an important fuel for many Mexican households for heating, water heating, and cooking. Table 9-1 summarizes CO_2 emissions by country, sector, and fuel type.

Table 9-1. Carbon dioxide emissions from energy consumed in buildings.

The energy consumed during building operation is the most important input to the carbon cycle from buildings; but it is not the only one. The construction, renovation, and demolition of buildings also generate a significant flux of wood and other materials. Construction of a typical 204-m^2 (2200-ft^2) house requires about 20 metric tons of wood and creates 2 to 7 metric tons of construction waste (DOE/EERE, 2005).[17] Building lifetimes are many decades and, especially for commercial buildings, may include several cycles of remodeling and renovation. In the United States as a whole, water supplied to residential and commercial customers accounts for about 6% of total national fresh water consumption. This water consumption also impacts the carbon cycle because water supply, treatment, and waste disposal require energy.

Trends and Drivers

Several factors influence trends in carbon emissions in the buildings sector. Some driver variables tend to increase emissions, while others decrease emissions. Emissions from energy use in buildings in the United States and Canada increased 30% from 1990 to 2003 (DOE/EERE, 2005; Natural Resources Canada, 2005),[18] corresponding to an annual growth rate of 2.1%.

Carbon emissions from buildings have grown with energy consumption, which in turn is increasing with population and income. Demographic shifts therefore have a direct influence on residential energy consumption. Rising incomes have led to larger residential buildings—the amount of living area per capita is increasing in all three countries in North America. On one hand, total population growth is

[17]Construction data from Table 2.1.7 in DOE/EERE (2005); wood content estimated from lumber content. Construction waste from Table 3.4.1 in DOE/EERE (2005).

[18]Data from Table 3.1.1 in DOE/EERE (2005).

slowing, especially in Mexico, as families are having fewer children than in the past. Annual population growth during the 1990s was 1.1% in the United States, 1.0% in Canada, and 1.7% in Mexico. In the period from 1970 to 1990 it was 1.0%, 1.2%, and 2.5%, respectively.[19] By 2005, annual population growth in Mexico declined to 1% (INEGI, 2005). On the other hand, a shift from large, extended-family households to nuclear-family and single-occupant households means an increase in the number of households per unit population[20]—each with its own heating and cooling systems and appliances.

The consumption of energy on a per capita basis or per unit economic activity [gross domestic product (GDP)] is also not constant but depends on several underlying factors. Economic development is a primary driver of overall per capita energy consumption and influences the mix of fuels used.[21] Per capita energy consumption generally grows with economic development, since wealthier people live in larger dwellings and use more energy.[22] Recently, computers, printers, and other office equipment have become commonplace in nearly all businesses and in most homes. These end uses now constitute 7% of primary household energy consumption. As a result of these growing electricity uses, the ratio of electricity to total household primary energy has increased. This is significant to emissions because of the large emissions associated with the combustion of fossil fuels in power plants. Electricity can be generated from renewable sources, such as solar or wind, but their full potential has yet to be realized.

In the United Stages, the major drivers of energy consumption growth are growth in commercial floor space and an increase in the size of the average home. The size of an average U.S. single-family home has grown from 160 m^2 (1720 ft^2) for a house built in 1980 to 216 m^2 (2320 ft^2) in 2003. In the same time, commercial floor space per capita has increased from 20 to 22.6 m^2 (215 to 240 ft^2) (DOE/EERE, 2005).[23] Certain end uses once considered luxuries have now become commonplace. Only 56% of U.S. homes in 1978 used mechanical space-cooling equipment (DOE/EIA, 2005). By 2001, ownership grew to 83%, driven by near total saturation in warmer climates and a demographic shift in new construction to these regions. Table 9-2 shows emissions trends, as well as the underlying drivers.

Table 9-2. Principal drivers of buildings emissions trends

[SIDEBAR 1 TEXT BOX HERE]

[19]*Source:* UN Department of Economic and Social Affairs.
[20]See household size statistics in Table 9-2.
[21]For example, whether biomass, natural gas or electricity is used for space heating and cooking.
[22]See Table 4.2.6 in DOE/EERE (2005).
[23]See Tables 2.1.6 and 2.2.1 in DOE/EERE (2005). Residential data are from 1981.

Although the general trend has been toward growth in per capita emissions, emissions per unit of GDP have decreased in past decades, due to improvements in efficiency. Efficiency performance of most types of equipment has generally increased, as has the thermal insulation of buildings, due to influences such as technology improvements and voluntary and mandatory efficiency standards and building codes. The energy crisis of the 1970s was followed with a sharp decline in economic energy intensity. Increases in efficiency were driven both by market-related technology improvements and incentives and by the establishment of federal and state/provincial government policies designed to encourage or require energy efficiency.

Options for Management

A variety of alternatives exist for reducing emissions from the buildings sector. Technology- and market-driven improvements in efficiency are expected to continue for most equipment, but this will probably not be sufficient to adequately curtail emissions growth without government intervention. The government has many different ways in which it can manage emissions that have been proven effective in influencing the flow of products from manufacturers to users (Interlaboratory Working Group, 2000). That flow may involve six steps: advancing technologies; product development and manufacturing; supply, distribution, and wholesale purchasing; retail purchasing; system design and installation; and operation and maintenance (Wiel and McMahon, 2005). Options for specific products or packages include government investment in research and development, information and education programs, energy pricing and metering, incentives and financing, establishment of voluntary guidelines, procurement programs, energy audits and retrofits, and mandatory regulation. The most effective approaches will likely include more than one of these options in a policy portfolio that takes advantage of synergies, avoids unduly burdening certain sectors, and is cost effective. Major participants include not only federal agencies, but also state and local governments, energy and water utilities, private research and development firms, equipment manufacturers and importers, energy services companies (ESCOs[24]), nonprofit organizations, building owners and occupants.

- **Technology adoption supported by research and development:** Government has the opportunity to encourage development and adoption of energy-efficient technologies through investment in research and development, which can advance technologies and bring down prices, therefore enabling a larger market. Successful programs have contributed to the development of high-efficiency lighting, heating, cooling, and refrigeration. Research and development has also had an impact on the improvement of insulation, ducting, and windows. Finally, government support of research and

[24]An ESCO is a company that offers to reduce a client's utility costs, often with the cost savings being split with the client through an energy performance contract or a shared savings agreement.

development has been critical in the reduction of costs associated with development of renewable energy.

- **Voluntary Programs:** By now, there are a wide range of efficiency technologies and best practices available, and if the most cost-effective among them were widely utilized, carbon emissions would be reduced. Voluntary measures can be effective in overcoming some market barriers. Government has been active with programs to educate consumers with endorsement labels or ratings [such as the U.S. Environmental Protection Agency's (EPA's) Energy Star Appliances and Homes], public-private partnerships [such as the U.S. Department of Energy's (DOE's) "Building America Program"]. Government is not the only player, however. Energy utilities can offer rebates for efficient appliances, and ESCOs can facilitate best practices at the firm level. Finally, nongovernment organizations and professional societies (such as U.S. Green Building Council and the American Institute of Architects) can play a role in establishing benchmarks and ratings.
- **Regulations:** Governments can dramatically impact energy consumption through well-considered regulations that address market failures with cost-effective measures. Regulations facilitate best practices in two ways: they eliminate the lowest-performing equipment from the market, and they boost the market share of high-efficiency technologies. Widely used examples are mandatory energy efficiency standards for appliances, equipment, and lighting; mandatory labeling programs; and building codes. Most equipment standards are instituted at a national level, whereas most states have their own set of prescriptive building codes (and sometimes energy performance standards for equipment) to guarantee a minimum standard for energy-saving design in homes and businesses.

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Although large strides in efficiency improvement have been made over the past three decades, significant improvements are still possible. They will involve continued improvement in equipment technology, but will increasingly take a whole-building approach that integrates the design of the building and the energy consumption of the equipment inside it. The improvements may also involve alternative ways to provide energy services, such as cogeneration of heat and electricity and thermal energy storage units (Public Technology Inc. and U.S. Green Building Council, 1996).

Whole-building certification standards evaluate a package of efficiency and design options. An example is the Leadership in Energy and Environmental Design (LEED) certification system developed by the U.S. Green Building Council, a non-profit organization. In existence for five years, the LEED program has certified 36 million m^2 (390 million ft^2) of commercial and public-sector buildings and has recently implemented a certification system for homes. The LEED program includes a graduated rating system (Certified, Silver, Gold, or Platinum) for environmentally friendly design, of which energy efficiency is a key component (USGBC, 2005).

On the government side, the EPA's Energy Star Homes program awards certification to new homes that are independently verified to be at least 30% more energy-efficient than homes built to the 1993 national Model Energy Code, or 15% more efficient than state energy code, whichever is more rigorous. Likewise, the DOE's Building America program partners with home builders, providing research and development toward goals to decrease primary energy consumption by 30% for participating projects by 2007, and by 50% by 2015.

Research and Development Needs

Research, development, demonstration, and deployment of technologies and programs to improve energy efficiency in buildings and to produce energy with fewer carbon emissions have involved significant effort over the last 30 years. These efforts have contributed options toward carbon management. Technologies and markets continue to evolve, representing new crops of "low-hanging fruit" available for harvesting. However, in most buildings-related decisions in North America, reducing carbon emissions remains a secondary objective to other goals, such as reducing first costs (DeCanio, 1993 and 1994). The questions for which answers could significantly change the discussion about options for carbon management include the following.

- What is the total societal cost of environmental externalities, including carbon emissions? Energy resources in North America have been abundant and affordable, but externality costs have not been completely accounted for. Most economic decisions are weighted toward the short term and do not consider the complete costs. Total societal costs of carbon emissions are unknown and, because it is a global issue, difficult to allocate. Practical difficulties notwithstanding, this is a key issue, answers to which could influence priorities for research and development as well as policies such as energy pricing, carbon taxes or credits.
- What cost-effective reduced-carbon-emitting equipment and building systems—including energy demand (efficient equipment) and supply (renewable energy)—are available in the short, medium, and long term? Policymakers must have sufficient information to be confident that particular new technology types or programs will be effective and affordable. For consumers to seriously consider a set of options, the technologies must be manifested as products that are widely available and competitive in the marketplace. Therefore, economic and market analyses are necessary before attractive options for managing carbon can be proposed.
- How do the costs of mitigation compare to the costs of continued emissions? The answers to the previous two questions can be compared in order to develop a supply curve of conserved carbon

comprising a series of least-cost options, whether changes to energy demand or to supply, for managing carbon emissions. The supply curve of conserved carbon will need to be updated at regular intervals to account for changes in technologies, production practices, and market acceptance of competing solutions.

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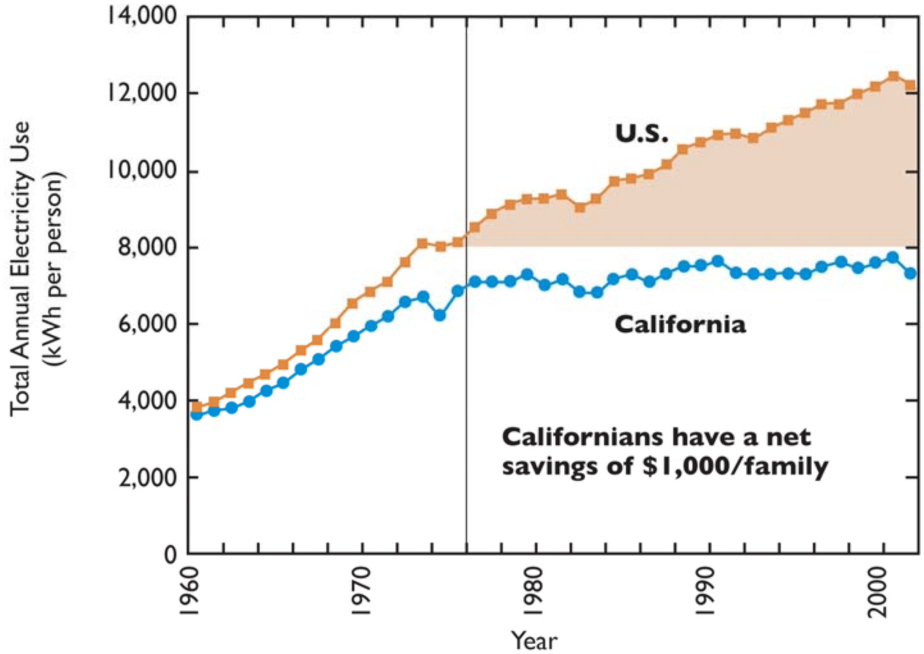
Electricity Consumption in the United States and in California

Since the mid-1970s, the state of California has pursued an aggressive set of efficiency regulations and utility programs. As a result, per capita electricity consumption has stabilized in that state, while it continues to grow in the United States as a whole.



Source: California Energy Commission— Available at http://www.energy.ca.gov/2005publications/CEC-999-2005-007/CEC-999-2005-007.PDF, Slide 5

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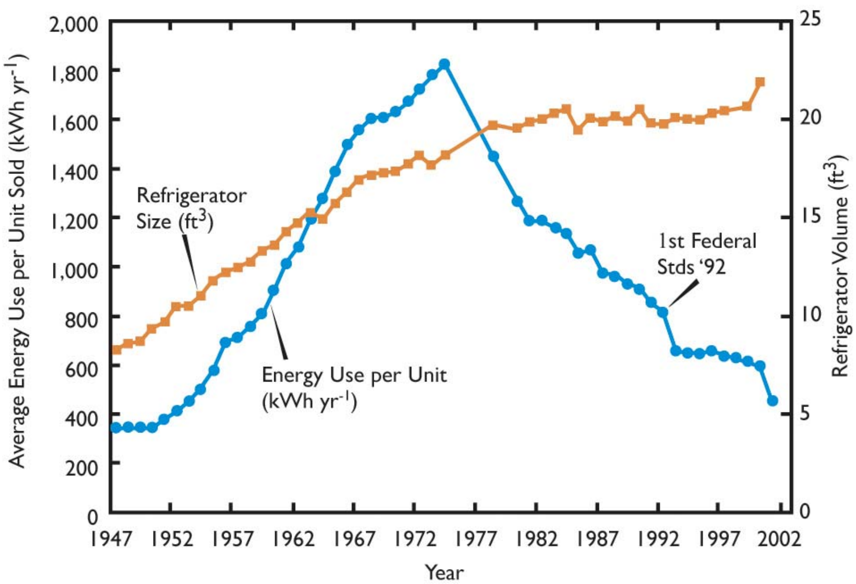
[BEGIN SIDEBAR 2 TEXT BOX]

Impact of Efficiency Improvements

Between 1974 and 2001, the energy consumption of the average refrigerator sold in the United States dropped by 74%, a change driven by market forces and regulations. From 1987 to 2005, the U.S. Congress and DOE promulgated labels or minimum efficiency standards for over 40 residential and commercial product types. Canada and Mexico also have many product labels and efficiency standards, and a program is under way to harmonize standards throughout North America in connection with the North American Free Trade Agreement (NAFTA).



Source: California Energy Commission—Available at http://www.energy.ca.gov/2005publications/CEC-999-2005-007/CEC-999-2005-007.PDF, slide 7

[END SIDEBAR 2 TEXT BOX]

Table 9-1. Carbon dioxide emissions from energy consumed in buildings

	2003 Carbon Dioxide Emissions (Mt C)			
	Electricity	**Natural Gas**	**Other Fuels**	**All Fuels**
United States	**445.8**	**122.1**	**46.5**	**614.5**
Residential	229.2	75.6	29.3	334.1
Commercial	216.6	46.5	17.2	280.4
Canada	**17.7**	**15.8**	**6.1**	**39.5**
Residential	9.4	8.7	2.5	20.6
Commercial	8.2	7.1	3.5	18.9
Mexico	**10.7**	**0.5**	**5.6**	**16.9**
Residential	7.3	0.4	5.5	13.2
Commercial	3.5	0.1	0.1	3.7

Table 9-2. Principal drivers of buildings emissions trends

Driver	**United States**		**Canada**		**Mexico**	
	Total 2000	**Growth Rate 1990-2000**	**Total 2000**	**Growth Rate 1990-2000**	**Total 2000**	**Growth Rate 1990-2000**
Population (Millions)	288	1.1%	31.0	1.0%	100	1.7%
Household Size (persons per household)	2.5	-0.6%	2.6	-0.9%	5.3	-0.1%
Per capita GDP (thousand $US 1995)	31.7	2.0%	23.0	1.8%	3.8	1.8%
Residential Floor space (billion m^2)	15.7	0.0%	1.5	2.4%	0.85	N/A
Commercial Floor space (million m^2)	6.4	0.6%	0.5	1.6%	N/A	N/A
Building Energy Emissions per GDP (g C/$US)	70	-0.5%	59	-0.9%	N/A	N/A

Source: Population - UNDESA; Household Size - UNDP; GDP - World Bank
Source: Floorspace - EIA-EERE (2005), Natural Resources Canada (2005). Mexican residential floor space estimated from Table 1.8 in CONAFOVI (2001)
Source: Emissions - EIA-EERE (2005), Natural Resources Canada (2005)

1

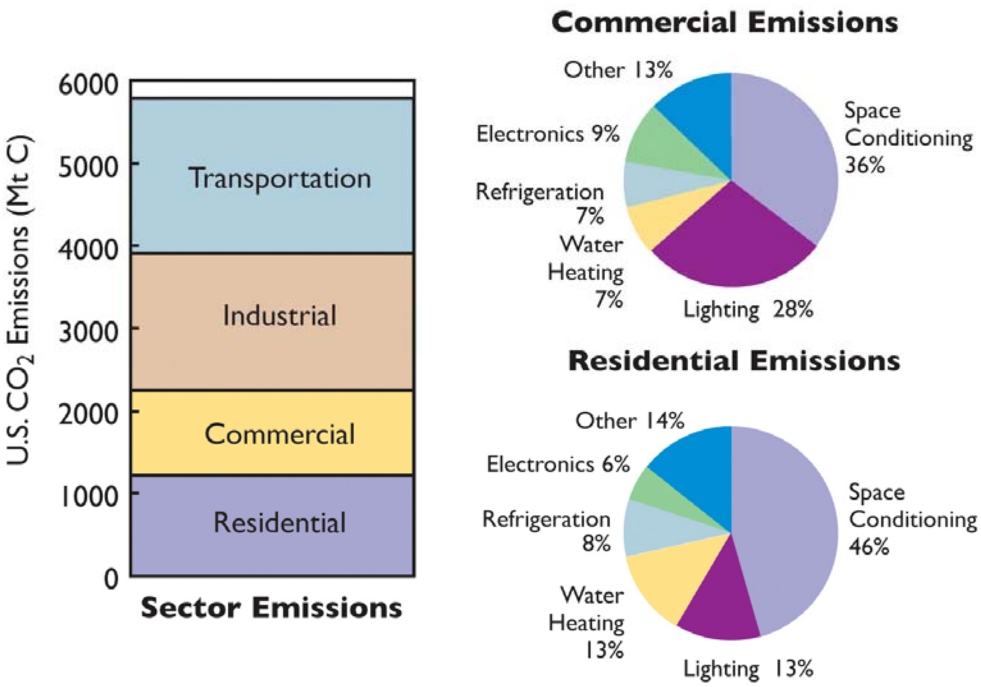


Fig. 9-1. U.S. carbon emissions by sector and—for commercial and residential buildings—by end use.

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PART III OVERVIEW

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 472-592 Mt C yr^{-1}, and offsets only about 25-30% of current fossil fuel emissions from the region (1856 Mt C yr^{-1} in 2003) (see Chapter 3). If managed properly, these systems have the potential to become significantly larger sinks of carbon in the future; they may also become significant net sources of carbon if managed poorly or if the climate warms.

Much of the current North American carbon sink is the result of past changes in land use and management. The large sink in the forests of Canada and the United States, for example, is partly the result of continued forest growth following agricultural abandonment that occurred in the past, partly the result of current and past management practices (e.g., fire suppression), and partly the result of forest responses to a changing environment (climatic change, CO_2 fertilization, and the increased mobilization of nutrients). However, the relative importance of these three 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.*, 2002). 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 19th century 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. Grazing lands, although not directly affected by cultivation, were, nevertheless, 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 the second largest net carbon sink (after forests), 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, 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 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) (Mg C ha^{-1} yr^{-1}).

The largest carbon sink in North America (350 Mt C yr^{-1}) is associated with forests (Chapter 11) (Table 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), including the wetlands overlying permafrost (Chapter 12), 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 nearly two thirds of the carbon in North America. Thus, despite the current net sink in these systems, their potential for future emissions is large.

Table 1. Ecosystems in North America: their areas, net annual fluxes of carbon, and their potential for sources (+) or sinks (–) in the future

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). The carbon lost to the atmosphere from cultivation of organic soils is approximately balanced by the carbon accumulated in mineral soils. In the past, before cultivation, these soils held considerably more carbon than they do today, but about 25% 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). The magnitude (and even sign) of this flux is uncertain, however, in part because some ecosystems lose carbon belowground (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.*, in press).

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 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 ha^{-1} yr^{-1} in Table 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 (see Chapter 3), 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 the Table 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 to potential to become a significant net source of carbon if the permafrost thaws with global warming (Smith *et al*,. 2005, Smith *et al.*, 2001, Osterkamp *et al.*, 1999, 2000). Forests clearly dominate the net sequestration 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 yr^{-1}, for example (about 5% of fossil fuel emissions from North America), requires management over hundreds of millions of hectares (e.g., the area presently in agriculture or forests) (Table 1). Enhancement of this terrestrial carbon sink through management 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 yr^{-1} 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. Somewhat smaller sinks of 10–70 Mt C yr^{-1} might be sequestered in agricultural soils at low to moderate costs (\$3–30/ton CO_2). The maximum amounts of carbon that might be accumulated in forests and agricultural soils are not known, and 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.

Despite the limited nature of carbon sequestration 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).

The effects and cost of managing aquatic systems are less clear. Increasing the area of wetlands, for example, would presumably sequester carbon; but it would also increase emissions of CH_4, countering the desired effect. 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).

A few studies have estimated the potential magnitudes of future carbon sinks as a result of management (Chapters 10, 11). However, the contribution of management, as opposed to the environment, in today's sink is unclear (see Chapter 3), 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). 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 sequestering 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 cycling of carbon is large in comparison to net fluxes. The observation is relevant for carbon management, because the cumulative effects of small managed 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 today's and tomorrow's sources and sinks of carbon? 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). 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*., 2005).
- Several chapters cite studies that have attempted to quantify potential future carbon sinks in countries in North America, but no reference is made to estimates of future sources of carbon. Clearly, there are modeling studies that project large future carbon emissions, although these studies are largely global in scope (e.g., Cox *et al*., 2000; Jones *et al*., 2005). Are there no studies of future carbon sources and sinks for North America? Melting permafrost, in particular, is likely to increase emissions of carbon to the atmosphere, CH_4 as well as CO_2.
- The sum of land areas reported in these chapters is about 330 million ha larger than the area of North America (Table 1). The reason for this double-counting is unclear, but it implies a double counting of carbon stocks and, perhaps, current sinks, as well.

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Table 1. Ecosystems in North America: their areas, net annual fluxes of carbon, and their potential for sources (+) or sinks (–) in the future

Type of ecosystem	Area (10^6 ha)	Current mean flux density (Mg C ha^{-1} yr^{-1})	Current flux (Mt C yr^{-1})	Carbon stocks (Mt C)	Future potential flux (Mt C yr^{-1})
Agriculture	231	0.0	0±15[1]	18,500	–(50 to100) to +??
Grass, shrub and arid	558	–0.01	–6[2]	59,950	–34
Forests	771	–0.45	–350[3]	171,475	–(100 to 200) to+??
Permafrost wetlands	621[4]	–0.02	–14[5]	213,320	
Wetlands	246	–0.28	–70	220,000	
Settled lands	104	–0.31[6]	–32[6]	~1,000[6]	
Coastal waters	384	0.05	19		
Sum	2531[7]	–0.18[8]	–472[9]	684,245	
Total	2126[10.]				

1. Fossil fuel inputs to crop management are not included. Some of the C 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.
2. 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 (Bradley *et al.*, in press) is included in this estimate of flux because the uncertainties are so large.
3. Includes an annual sink of 67 Mt C yr^{-1} in wood products as well as a sink of 283 Mt C yr^{-1} in forested ecosystems.
4. Includes zones with isolated and sporadic permafrost.
5. This estimate is for peatlands (not mineral soils) in permafrost regions. The net flux for mineral soil permafrost areas is unknown. This estimate of flux may be high because it does not include the losses resulting from fires, but it may be low if mineral soils are also accumulating carbon in permafrost regions.
6. Urban trees only (does not include soil carbon).
7. Sum does not include coastal waters. The summed area is too high because an estimated 75×10^6 ha of permafrost peatlands in Canada are treed (and may be included in forest area as well as permafrost area). Nevertheless, another ~330×10^6 ha are double counted (United States forests on non-permafrost wetlands? Other wooded lands that are included as both forests and rangelands? Large areas of grasslands and shublands on non-permafrost lands within areas defined as sporadic or isolated permafrost? Inland waters?).
8. Weighted average; does not include coastal waters.
9. Does not include coastal waters. The total annual sink of 472 Mt C is lower than the estimate of 592 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 yr^{-1}; Table 1, above, presents a net flux of zero (see note 2). Other differences between the two estimates include: (1) an additional sink in Table 1 of 14 Mt C yr^{-1} in permafrost wetlands; (2) an additional sink in Table 1 of 32 Mt C yr^{-1} in settled lands; and (3) a sink of 25 Mt C yr^{-1} in rivers and reservoirs that is included in Table 3-1 but not in Table 1. In addition, there are small differences in the estimates for agricultural lands and grasslands.
10. 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

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 ha (47% of the land area of North America) and contain 78.5±19.5 Gt C (17% of North American terrestrial carbon) in the soil alone.
- The emissions and sequestration 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 CO_2 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 be sequestering currently 6.4-15.9 Mt C yr^{-1}, the cultivation of organic soils releases 5.1-10.1 Mt C yr^{-1}. On-farm fossil fuel use and (30.9 Mt C yr^{-1}) and manufacture of agricultural inputs including fertilizer (6.4 Mt C yr^{-1}) yields a net source from the agricultural sector of 27-41 Mt C yr^{-1}.
- As much as 120 Mt C yr^{-1} 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 biomass carbon accumulation.

- 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/tonne CO_2, would yield mitigation amounts of 168 Mt CO_2 yr^{-1} through agricultural soil C sequestration and 53 Mt CO_2 yr^{-1} from fossil fuel use reduction. At lower prices of $5/tonne CO_2, the corresponding values would be 123 Mt CO_2 yr^{-1} and 32 Mt CO_2 yr^{-1}, 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 sequestration, but may stimulate emissions of CH_4 or N_2O.
- Many of the practices that lead to carbon sequestration and reduced CO_2 and CH_4 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 is needed 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 is needed 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 biomass, and variation in impacts on soil carbon stocks would help reduce the large uncertainty of the carbon impacts of woody encroachment.

INVENTORY

Background

Agricultural and grazing lands (cropland, pasture, rangeland, shrublands, and arid lands[1]) 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) 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

[1] 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.

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 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 CH_4 (27% of United States, 31% of Mexican, and 27% of Canadian CH_4 emissions) (Mosier *et al.*, 1998b; CISCC, 2001; Matin *et al.*, 2004; EPA, 2006); and agricultural and grazing lands are the largest anthropogenic source of N_2O emissions (CAST, 2004; see Text Box 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.

Table 10-1. Carbon pools in agricultural and grazing lands in Canada, Mexico, and the United States; the area (M ha) for each climatic zone are in parentheses.

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 and 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 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 Text Box 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 Text Box 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 aboveground 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 aboveground biomass, have drastically altered carbon cycling within agricultural lands and thus the sources and sinks of CO_2 to the atmosphere.

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 C inputs to the soil, and increased organic matter additions have reversed historic soil C losses in long-term experimental plots (e.g., Buyanovsky and Wagner, 1998). Across Canada and the United States, mineral soils have been sequestering 0.1and 6.5–16 Mt C yr^{-1} (Smith *et al.*, 1997; Smith *et al.*, 2001b; Ogle *et al.*, 2003), respectively, largely through increased production and improved management practices on annual cropland (Fig. 10-1, Table 10-2). Conversion of agricultural land to grassland, like under the Conservation Reserve Program in the United States (6 Mt C yr^{-1} on 14 M ha of land), and afforestation have also sequestered carbon in agricultural and grazing lands. In contrast, cultivation of organic soils (e.g., peat-derived soils) is releasing an estimated 0.1 and 5-10 Mt C yr^{-1} from soils in Canada and the United States (Matin *et al.*, 2004; Ogle *et al.*, 2003). 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.

Figure 10-1. North American agricultural and grazing land CO_2 (left side) and methane (right side), adjusted for global warming potential.

Table 10-2. North American agricultural and grazing land carbon fluxes for the years around 2000

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 yr^{-1} within the United States (Lal *et al.*, 1998) and 4.6 Mt C yr^{-1} in Canada (Sobool and Kulshreshtha, 2005) (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 (U.S.: 4.3 Mt C yr^{-1}; Mexico: 0.4 Mt C yr^{-1}; Canada: 1.7 Mt C yr^{-1}) and phosphoric acid manufacture (U.S.: 0.4 Mt C yr^{-1}; Mexico: 0.2 Mt C yr^{-1}; Canada: not reported) are devoted to agricultural uses.

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 (U.S.: 0.25 Mt CH_4-C yr^{-1}; Mexico: 0.01 Mt CH_4-C yr^{-1}; Canada: negligible, not reported; Table 10-2). Methane is also formed by incomplete biomass combustion of crop residues (U.S.: 0.03 Mt CH_4-C yr^{-1}; Mexico: <0.01 Mt CH_4-C yr^{-1}; 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 methane consumption relative to forests and extensively 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 (Fig. 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).

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 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 (5.8 Mt CH_4-C yr^{-1}, 21% of total U.S. CH_4 emissions), Canada (0.6 Mt CH_4-C yr^{-1}, 22% of total) (Sobool and Kulshreshtha, 2005), and Mexico (3.7 Mt CH_4-C yr^{-1}, 27% of total) with the vast majority of enteric CH_4 emissions are 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).

Methane emissions during manure storage (U.S.: 1.9 Mt CH_4 yr^{-1}; Mexico: 0.06 Mt CH_4 yr^{-1}; Canada: 0.3 Mt CH_4 yr^{-1}) 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).

DRIVERS AND TRENDS

The extent to which agricultural options 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 U.S. 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 cost'). 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, N 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, 1996) would tend to enhance the potential for soil C sequestration (CAST, 2004). Similarly, increased plant growth due to higher concentrations of CO_2 (and N 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 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.

OPTIONS FOR MANAGEMENT

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[2] (Post and Kwon, 2000; Follett *et al.*, 2001b) (Fig. 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;

[2]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.

Paustian *et al.*, 1998; VandenBygaart *et al.*, 2003) (Fig. 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) (Fig. 10-2). Estimates of the technical potential for annual cropland soil carbon sequestration are on the order of 50–100 Mt C yr^{-1} in the United States (Lal *et al.*, 2003; Sperow *et al.*, 2003) and approximately 5 Mt C yr^{-1} in Canada (Boehm *et al.*, 2004).

Figure 10-2. Relative soil carbon following implementation of new agricultural or grassland management practices.

Within grazing lands, historical overgrazing has substantially reduced productive capacity in many areas, leading to loss of soil carbon stocks (Conant and Paustian, 2002) (Fig. 10-2). Conversely, improved grazing management and production inputs—like fertilizer, adding (N-fixing) legumes, organic amendments, and irrigation—can increase productivity, carbon inputs, and soil carbon stocks, potentially storing 0.44 Mt C yr^{-1} in Canada (Lynch *et al.*, 2005) and as much as 33.2 Mt C yr^{-1}in the United States (Follett *et al.*, 2001a). Such improvements will carry a carbon cost, particularly fertilization and irrigation since their production and implementation require the use of fossil fuels.

Fossil Fuel-Derived Emission Reductions

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 (e.g., dairy) facilities. Substitution of crop-derived for fossil fuels could decrease net emissions.

Energy intensity (energy per unit product) for the U.S. 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).

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 as much as 40% (Mosier *et al.*, 1998b).

Biomass burning is uncommon in most Canadian and U.S. 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).

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.

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 NO_3^- leaching and downstream eutrophication.

Economics and Policy Assessment

Policies for agricultural mitigation activities can range from transfer payments (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 North American three countries differs greatly. Canada and the United States are both Annex 1 (developed countries) within the 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).

- Significant amounts (10–70 Mt yr^{-1}) of carbon sequestration in soils can be achieved 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), e.g., conservation tillage, pasture improvement, have a lower cost per ton sequestered carbon compared with practices where mitigation would be a primary income source (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), estimated economic potential for some agricultural mitigation options, assuming constant 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 168 Mt CO_2 per year through agricultural soil carbon sequestration and 53 Mt CO_2 per year from fossil fuel use reduction (compare with estimated U.S. national ecosystem carbon sink of 1760 Mt CO_2 per year). At lower prices of $5/ton CO_2, the corresponding values would be 123 Mt CO_2 per year (for soil sequestration) and 32 Mt CO_2 per year (for fossil fuel reduction), respectively, reflecting the effect of price on the supply of mitigation activities.

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, a given change in management (e.g., tillage reduction, pasture improvement, afforestation) will stimulate carbon storage for a finite duration. For many practices, soil carbon storage 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). Thus, to maintain these higher stocks, the management practices will need to be maintained. Key implications for policy are that the value of sequestered carbon will 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 C 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, production 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—where 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 emission, because N_2O is such a potent greenhouse gas (Robertson and Grace, 2004; Six *et al.*, 2004; Gregorich *et al.*, 2005). (See Text Box 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.*, 2001a; 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 be needed to get a clearer picture of the actual potential of agriculture to contribute to greenhouse gas mitigation efforts.

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 by knowledge 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 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 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 U.S. national agricultural greenhouse gas inventories (Ogle *et al.*, 2003; VandenBygaart *et al.*, 2004). 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.

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[START OF TEXT BOX 1]

Nitrous oxide (N_2O) 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 N 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%).

[END OF TEXT BOX 1]

[START OF TEXT BOX 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 is 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 the United States, about 1 Mt C yr^{-1} is emitted from liming (EPA, 2006).

[END OF TEXT BOX 2]

[START OF TEXT BOX 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 1 Mg C ha^{-1} yr^{-1} (Pacala *et al.*, 2001), with estimated net sequestration of 0.12–0.13 Gt C yr^{-1} in encroaching woody biomass (Houghton *et al.*, 1999; Pacala *et al.*, 2001). In response to woody encroachment, soil carbon stocks can significantly increase or decrease, thus predicting impacts on soil carbon or ecosystem carbon stocks is very difficult (Jackson *et al.*, 2002).

[END OF TEXT BOX 3]

[START OF TEXT BOX 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).

[END OF TEXT BOX 4]

1

Table 10-1. Carbon pools in agricultural and grazing lands in Canada, Mexico, and the United States; the area (M ha) for each climatic zone are 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 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 Fig. 10-2).

Practice	Temperate dry [b,c]	Temperate wet	Tropical dry	Tropical wet	Total
	Gt C				
			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 30º latitude. Tropical zones (<30º 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 1; in wet areas, MAP/PET >1.

1

Table 10-2. North American agricultural and grazing land carbon fluxes for the years around 2000. All units are in Mt C yr^{-1}. 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 U.S. (EPA, 2006) National Inventories and from the second Mexican National Communication (CISCC, 2001). Values are for 2003 for United States and Canada and 1998 for Mexico. A factor of 12/44 was used convert from CO_2 to carbon and a factor of 12/16 to convert CH_4 to carbon.

	Canada	**Mexico**	**United States**	**Total**
			Mt C yr^{-1}	
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	(0.1)	ND	(6.5) – (16)	(6.4) – (15.9)
Organic soil cultivation	0.1	ND	5–10	5.1 – 10.1
Woody encroachment	ND	ND	(120)[c]	(120)
Total	4.6	ND	(98.3) – (83.8)	(93.7) – (79.2)
CH_4				
Rice production	0	0.011	0.25	0.26
Biomass burning	<0.01	<0.01	0.03	0.05
Livestock	0.62	1.48	3.67	5.77
Manure	0.18	0.05	1.28	1.51
Total	0.82	1.54	5.23	7.59

ND = no data reported.
[a]From Sobool and Kulshreshtha (2005).
[b] From Lal *et al.* (1998).
[c]From Houghton *et al.* (1999).

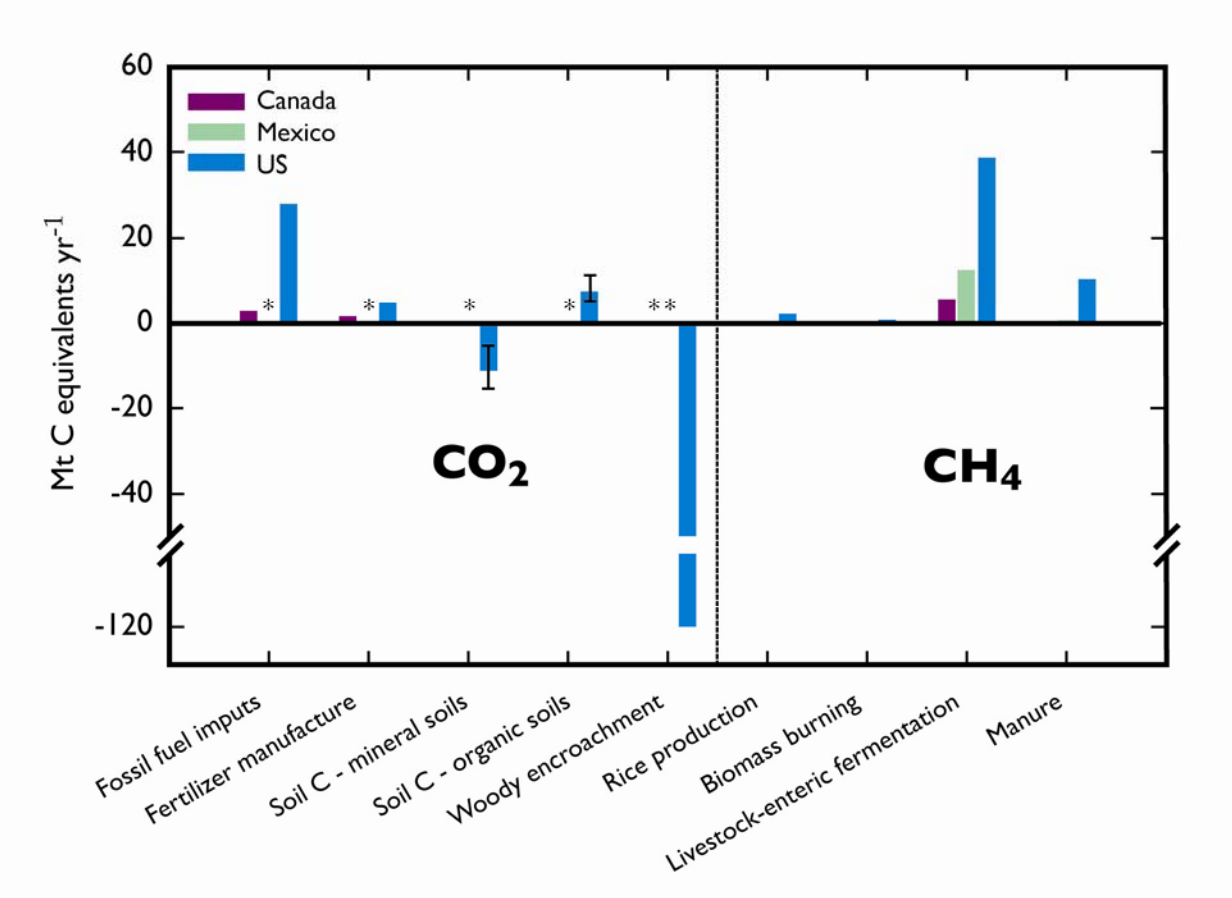

Fig. 10-1. North American agricultural and grazing land CO_2 (left side) and methane (right side), adjusted for global warming potential. All units are in Mt C-equivalent yr^{-1} for years around 2000. Negative values indicate net flux from the atmosphere to soil and biomass carbon pools (i.e., sequestration). 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.

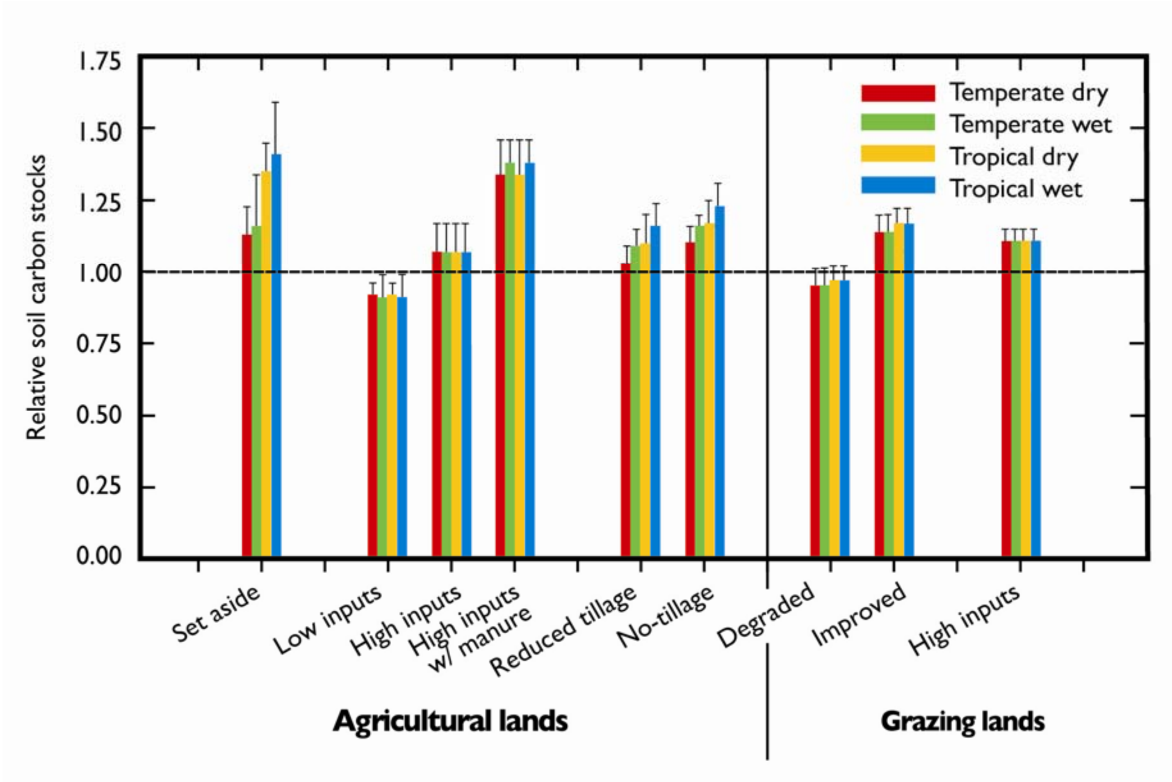

Fig. 10-2. Relative soil carbon following implementation of new agricultural or grassland management practices. Conventionally tilled, medium-input cultivated land and moderately grazed grasslands with moderate inputs are defaults for agricultural and grazing lands, respectively. Default soil carbon stocks (like those in Table 10-1) can be multiplied by one or more emission factors to estimate carbon sequestration rates. The dashed horizontal line indicates default soil carbon stocks (i.e., those under conventional-tillage cropland or undegraded garzingland, with medium inputs). Temperature/precipitation divisions are the same as those described in Table 10-1. Data are from Nabuurs e*t al.* (2004) and Ogle *et al.* (2004).

Chapter 11. North American Forests

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KEY FINDINGS

- North American forests contain more than 170 Gt of carbon, of which 28% is in live biomass and 72% is in dead organic matter.
- North American forests were a net carbon sink of approximately -269 Mt C yr^{-1} over the last 10 to 15 years. This estimate is highly uncertain.
- Deforestation continues in Mexico where forests are a source of CO_2 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 U.S. 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 CO_2, climate change, nitrogen deposition, and tropospheric ozone. There is a lack of consensus about how these different natural and anthropogenic factors contribute to the current sink, and the relative importance of factors varies geographically.
- There have been several continental- to subcontinental-scale assessments of future changes in carbon and vegetation distribution in North America, but the resulting projections of future trends for North American forests are highly uncertain. Some of this is due to uncertainty in future climate, but there is also considerable uncertainty in forest response to climate change and in the interaction of climate with other natural and anthropogenic factors.

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

INTRODUCTION

The forest area of North America totals 771 million hectares, 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). About 45% of this forest area is classified as boreal, mostly in Canada and some in Alaska. Temperate and tropical forests constitute the remainder of the forest area.

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

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

CARBON STOCKS AND FLUXES

Ecosystem Carbon Stocks And Pools

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

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

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 U.S., forests of the Northeast, Upper Midwest, Pacific Coast, and Alaska (with 14,000 Mt C) store the most carbon. In Mexico, temperate forests contain 4,500 Mt C, tropical forests contain 4,100 Mt C, and semiarid forests contain 5,000 Mt C.

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 Mt C yr^{-1} over the last decade or so, with U.S. forests accounting for most of the sink (Table 11-3). This estimate is likely to be within 50% of the true value.

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

Canadian forests were estimated to be a net sink of -17 Mt C yr^{-1} 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 82% of the total area of Canada’s forests. The estimates also include the carbon changes that result from land-use change. Changes in forest soil carbon are not included. High interannual variability is averaged into this estimate—the annual change varied from approximately -50 to +40 between 1990 and 2004. Years with net emissions were generally years with high forest fire activity (Environment Canada, 2006).

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

For Mexico, the most comprehensive available estimate for the forest sector suggests a source of +52 Mt C per year in the 1990s (Masera *et al*., 1997). This estimate does not include changes in the wood products carbon pool. The main cause of the estimated source is deforestation, which is offset to a much lesser degree by restoration and recovery of degraded forestland.

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 11.A.

TRENDS AND DRIVERS

Overview of Trends and Drivers of Change in Carbon Stocks

Many factors that cause changes in carbon stocks of forests and wood products have been identified, but the importance of each is still debated in the scientific literature (Barford *et al*., 2001; Caspersen *et al*., 2000; Goodale *et al*., 2002; Korner, 2000; Schimel *et al*., 2000). Land-use 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 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 yr^{-1} 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 yr^{-1}. 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 form 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).

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 2006).

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

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.

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

Forests and other wooded land in Canada occupy about 402 Mha. Approximately 310 Mha is considered forest of which 255 Mha (83%) are under active forest management (Environment Canada, 2006). Managed forests are considered to be under the direct influence of human activity and not reserved. Less than 1% of the area under active management is harvested annually. Apps *et al*. (1999) used a carbon budget model to simulate carbon in harvested wood products (HWP) for Canada. Approximately 800 Mt C were stored in the Canadian HWP sector in 1989, of which 50 Mt C were in imported wood products, 550 Mt C in exported products, and 200 Mt C in wood products produced and consumed domestically.

Between 1990 and 2000, about 4 Mha yr^{-1} were harvested in the U.S., two-thirds by partial-cut harvest and one-third by clear-cut (Birdsey and Lewis, 2003). Between 1987 and 1997, about 1 Mha yr^{-1} were planted with trees, and about 800,000 ha were treated to improve the quality and/or quantity of timber produced (Birdsey and Lewis, 2003). Harvesting in U.S. forests accounts for substantially more tree mortality than natural causes such as wildfire and insect outbreaks (Smith *et al*., 2004). 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 m^3 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.

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 (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 (Korner *et al*., 2005; Lichter, 2005). Experiments have also shown that the effects of rising CO_2 are significantly moderated by increasing tropospheric ozone (Karnosky *et al*., 2003; Loya *et al*., 2003). When nitrogen 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).

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 Mha yr^{-1} in Canada's forests, with a maximum 7.6 Mha yr^{-1} in 1989. Carbon emissions from forest fires range from less than +1 Mt C yr^{-1} in the interior of British Columbia to more than +10 Mt C yr^{-1} in the western boreal forest. Total emissions from forest fires in Canada averaged approximately +27 Mt C yr^{-1} 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 yr^{-1} in the 1970s to less than +2 Mt C yr^{-1} 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 U.S. forests insects, diseases, and wildfire combined affect more than 30 Mha per decade (Birdsey and Lewis, 2003). Damage from weather events (hurricanes, tornados, ice storms) may exceed 20 Mha per decade (Dale *et al*., 2001). 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.

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

Projections of Future Trends

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

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

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.

Assessments of future changes in carbon and vegetation distribution in the U.S. suggest that under most future climate conditions, 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). Under most climate scenarios the West gets wetter; when coupled with higher CO_2 and longer growing seasons, simulations show woody expansion and increased sequestration of carbon as well as increases in fire (Bachelet *et al*., 2001). However, recent scenarios from the Hadley 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.

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).

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 11.B). 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).

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 yr^{-1} according to several studies (Birdsey *et al*., 2000; Lewandrowski, 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. 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 a 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, 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.

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

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.

DATA GAPS AND INFORMATION NEEDS FOR DECISION SUPPORT

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

Major Data Gaps in Estimates of Carbon Pools and Fluxes

Effective carbon 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.

In the United States, the range of estimates of the size of the land carbon sink is between 0.30 and 0.58 Mt C yr^{-1} (Pacala *et al*., 2001). Significant data gaps among carbon pools include carbon in wood products, soils, woody debris, and water transport (Birdsey, 2004; Pacala *et al*., 2001). Geographic areas that are poorly represented in the available data sets include much of the Intermountain Western United States and Alaska, where forests of low productivity have not been inventoried as intensively as more productive timberlands (Birdsey, 2004). Accurate quantification of the relative magnitude of various causal mechanisms at large spatial scales is not yet possible, 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 forestlands 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 CBM (Kurz *et al*., 2002); however the CBM does not consider climate change impacts (Price *et al*., 1999; Hogg and Bernier, 2005).

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

Major Data Gaps in Knowledge of Forest Management Effects

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 other needs and services of provided by forests (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 utilization as a renewable energy source, 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.

Availability Of Decision-Support Tools

Few decision-support tools for land managers that include complete carbon accounting are available. 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.

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

Ecological zone:	Canada[2]	U.S.[3]	Mexico[4]	Total
Tropical/subtropical	0	115,200	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

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

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

Ecosystem carbon pool:	Canada[2]	U.S.[3]	Mexico[4]	Total
Biomass	14,500	24,900	7,700	47,100
Dead organic matter[5]	71,300	41,700	11,400	124,400
Total	85,800	66,600	19,100	171,500

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

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

Carbon pool:	Canada[1]	U.S.[2]	Mexico[3]	Total
Forest Ecosystem	–17	–236	+52	–201
Wood Products	–11	–57	ND[4]	–68
Total	–28	–293	+52	–269

[1]Data for 1990-2004, taken from Environment Canada (2006), Goodale *et al*. (2002). There is 95% certainty that the actual values are within 100% of those reported for Canada.
[2]From Smith and Heath, 2005 (excluding soils), and Pacala *et al*., 2001 (soils). Estimates do not include urban forests. There is 95% certainty that the actual values are within 50% of those reported for the United States.
[3]From Masera, 1997. There is 95% certainty that the actual values are within 100% of those reported for Mexico.
[4]Estimates are not available.

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

Management class:	Canada	U.S.	Mexico	Total
Protected	19,300	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

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

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

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

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

APPENDIX 11A

ECOSYSTEM CARBON FLUXES

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

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

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

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

APPENDIX 11A REFERENCES

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

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

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APPENDIX 11B

PRINCIPLES OF FOREST MANAGEMENT FOR ENHANCING CARBON SEQUESTRATION

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

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

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

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

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

- Management activities that move live carbon to dead pools (such as CWD or soil C) over short periods of time will often dramatically enhance decomposition (Rh), although considerable carbon can be stored in decomposing pools (Harmon and Marks, 2002). Regimes seeking to reduce the decomposition-related flows from residue following harvest may enhance overall sink capacity of these forests if these materials are used for energy generation or placed into forest products that last longer than the residue.
- Despite the importance of decomposition rates to the overall stand-level forest carbon balance, management of CWD pools is mostly impacted by recruitment of new CWD rather than by changing decomposition rates (Janisch and Harmon, 2002; Pregitzer and Euskirchen, 2004). Decreasing the interval between harvests can significantly decrease the store in this pool.
- Live coarse root biomass accounts for approximately 20–25% of aboveground forest biomass (Jenkins *et al*., 2003), and there is additional biomass in fine roots. Following harvest, this pool of live root biomass is transferred to the dead biomass pool, which can form a significant carbon store. Note that roots of various size classes and existing under varying environmental conditions decompose at different rates.
- Some carbon can be sequestered in wood products from harvested wood, though due to manufacturing losses only about 60% of the carbon harvested is stored in products (Harmon, 1996). Clearly, longer-lived products will sequester carbon for longer periods of time.
- According to international convention, the replacement of fossil fuel by biomass fuel can be counted as an emissions offset if the wood is produced from sustainably managed forests (Schoene and Netto 2005).

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

- Practices aimed at increasing NPP: fertilization; genetically improved trees that grow faster (Peterson *et al*., 1999); any management activity that enhances growth rate without causing a concomitant increase in decomposition (Stanturf *et al*., 2003; Stainback and Alavalapati, 2005).
- Practices aimed at reducing Rh (i.e., minimizing the time forests are a source to the atmosphere following disturbance): low impact harvesting (that does not promote soil respiration); utilization of logging residues (biomass energy and fuels); incorporation of logging residue into soil during site prep (but note that this could also speed up decomposition); thinning to capture mortality; fertilization.

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

APPENDIX 11B REFERENCES

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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 km^2) 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 CO_2 in response to an already detectable polar warming.
- The soils of the permafrost region of North America contain 213 Gt 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 Mt C yr^{-1}.
- The soils of the permafrost region of North America have been slowly accumulating carbon for the last 5-8 thousand 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 Gt) and 41% (40 Gt) of carbon stored in soils of the Subarctic and 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.

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, carbon that is vulnerable to release to the atmosphere as carbon dioxide and methane 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). 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^{o}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).

The permafrost region in North America is divided into four zones on the basis of the percentage of the land area underlain by permafrost (Fig. 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) (Fig. 12-2): the Arctic (north of the arctic tree line), the Subarctic (open canopy coniferous forest), and the Boreal (closed canopy forest, either 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.

Figure 12-2. Arctic, Subarctic, and Boreal ecoclimatic provinces (ecological regions) in North America (Ecoregions Working Group, 1989; Baily and Cushwa, 1981).

Although northern ecosystems (Arctic, Subarctic, and Boreal) in North America cover approximately14% 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. 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.

PROCESSES AFFECTING THE CARBON CYCLE IN A PERMAFROST ENVIRONMENT

Soils of the Permafrost Region

Soils cover approximately 6,211,340 km^2 of the area of the North American permafrost region (Tables 12-1 and 12-2), with approximately 58% of the soil area being occupied by permafrost-affected (perennially frozen) soils (Cryosols/Gelisols) and the remainder by non-permafrost soils. Approximately 17% of this area is associated with organic soils (peatlands), the remainder with mineral soils. 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.

Table 12-1. Areas of mineral soils in the various permafrost zones.

Table 12-2. Areas of peatlands (organic soils) in the various permafrost zones.

Mineral Soils

The schematic diagram in Fig. 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 process of cryoturbation, which moves 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.

Figure 12-3. Carbon cycle in permafrost-affected upland (mineral) soils, showing below-ground organic carbon sinks and 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 produce 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 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 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 carbon dioxide and methane, respectively (Fig. 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.

Peatlands (Organic Soils)

The schematic diagram in Fig. 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 5,000–8,000 years, resulting in peat deposits that are an average of 2–3 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 (Fig. 12-4).

Figure 12-4. Carbon cycle in permafrost peatlands, showing below-ground organic carbon sinks and sources.

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 values for frozen peatlands, which are more widespread, generally range around 13 g C m^{-2} yr^{-1}. 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.

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).

BELOW-GROUND CARBON STOCKS

The carbon content of mineral soils to a 1-m depth is 49–61 kg m^{-2} for permafrost-affected soils and 12–17 kg 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 m^{-2} for permafrost-affected soils and 43–144 kg m^{-2} for unfrozen soils (Tables 12-4 and 12-5) (Tarnocai, 1998 and 2000).

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).

Table 12-5. Average organic carbon content for soils in the various ecological regions (Tarnocai 1998 and 2000).

Soils in the permafrost region of North America contain 213 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).

Table 12-6. Organic carbon mass in mineral soils in the various permafrost zones.

Table 12-7. Organic carbon mass in peatlands (organic soils) in the various permafrost zones.

CARBON FLUXES

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 m^{-2} yr^{-1} (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 1000 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 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 Fig. 12-5 shows the carbon cycle in these soils.

Figure 12-5. Carbon cycle in perennially frozen mineral soils in the permafrost region.

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 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 *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 and 78.5 g m^{-2} yr^{-1}, 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 g m^{-2} yr^{-1}, while in unfrozen fens and bogs the comparable rates were 20.34 and 21.81 g m^{-2} yr^{-1}, 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 of carbon 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 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 Fig. 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 m^{-2} yr^{-1}, or 18 Mt C, is added annually to peatlands in this region of North America. Approximately 1.46 kg C 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 m^{-2} yr^{-1}, and a further 2.0 g C m^{-2} yr^{-1} 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 (Fig. 12-6). Note that these values are based on current measurements. However, rates of peat accumulation have varied during the past 6000–8000 years, with periods during which the rate of peat accumulation was much higher than at present.

Figure 12-6. Carbon cycle in peatlands in the permafrost region.

Total Flux

Based on the limited data available for this vast, and largely inaccessible, area of the continent, approximately 27 Mt C yr^{-1} is deposited on the surface of mineral soils and peatlands (organic soils) in the permafrost region of North America. Approximately 8 Mt yr^{-1} 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 yr^{-1} and as a source for approximately 8 Mt C yr^{-1} and are, therefore, a net carbon sink (Figs. 12-5 and 12-6).

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 methane to the atmosphere.

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 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 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 (Fig. 12-7) (Tarnocai, in press). 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. 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 (Fig. 12-7) (Tarnocai, in press).

Figure 12-7. The organic carbon mass in the various sensitivity classes for the Subarctic and Boreal Ecoclimatic Provinces (ecological regions) (Tarnocai, in press).

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.

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.

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).

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.

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. As a result, carbon is released into the atmosphere as either CO_2 or methane, or it enters the water system as dissolved organic carbon.

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 methane. 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., PCBs, DDT, 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.

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 manmade 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.

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 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.

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Table 12-1. Areas of mineral soils in the various permafrost zones

Permafrost zones	Area ($10^3 \times km^2$)		
	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 ($10^3 \times km^2$)		
	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).

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 yr^{-1}[a]
All Canadian peatlands	–30 Mt yr^{-1}[b]
All mineral and organic soils	–18 mg m^{-2} yr^{-1}[c]
Rich fens	–13.58 g m^{-2} yr^{-1}[d]
Poor fens (unfrozen, Discontinuous Permafrost Zone)	–20.34 g m^{-2} yr^{-1}[d]
Peat plateaus (frozen, Discontinuous Permafrost Zone)	–13.31 g m^{-2} yr^{-1}[d]
Collapse fens	–13.54 g m^{-2} yr^{-1}[d]
Bogs (unfrozen, Discontinuous Permafrost Zone)	–21.81 g m^{-2} yr^{-1}[d]
Dissolved organic carbon (DOC)	+2 g m^{-2} yr^{-1}[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 m^{-2}[g]
Carbon release by fires in all terrain	+27 Mt yr^{-1}[h]
Carbon release by fires in Western Canadian peatlands	+5.9 Mt yr^{-1}[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.

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)

	Peatlands		Mineral soils		
Type	**Perennially frozen**	**Unfrozen**	**Perennially frozen**	**Unfrozen**	**Total**
Current area ($\times 10^3$ 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 m^{-2} yr^{-1})	–5.7[d]	–15.2[e]			
Carbon accumulation (g m^{-2} yr^{-1})	–13.3[f]	–20.3 to –21.8[f]		–60 to –100[g]	
Carbon release by fires (g m^{-2} yr^{-1})[h]	+7.57[i]				
Methane flux (g m^{-2} yr^{-1})		+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 ha^{-1} yr^{-1} (this report).
[e]Using C accumulation rate of 0.194 mg ha^{-1} yr^{-1} (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.

Table 12-5. Average organic carbon content for soils in the various ecological regions (Tarnocai, 1998 and 2000)

Ecological regions	Average carbon content (kg 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).

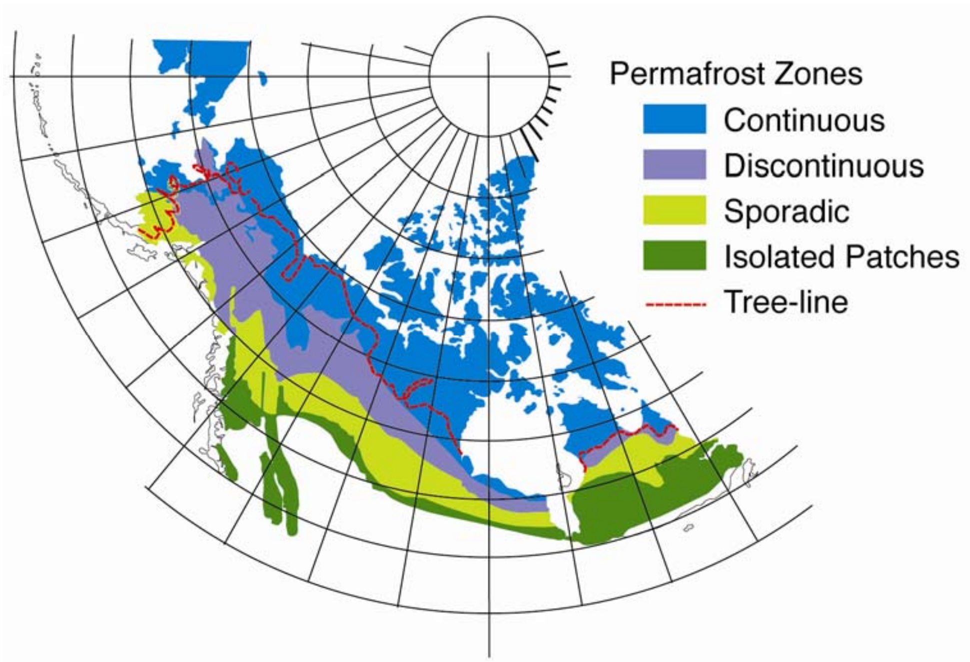


Fig. 12-1. Permafrost zones in North America (Brown *et al*., 1997).

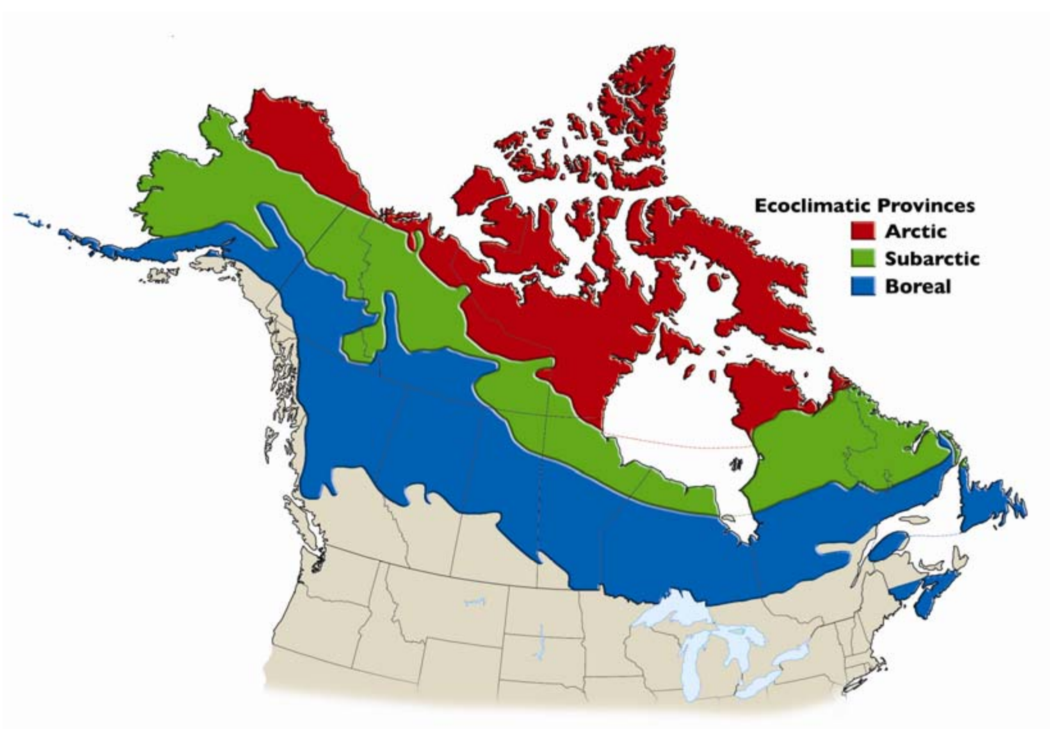


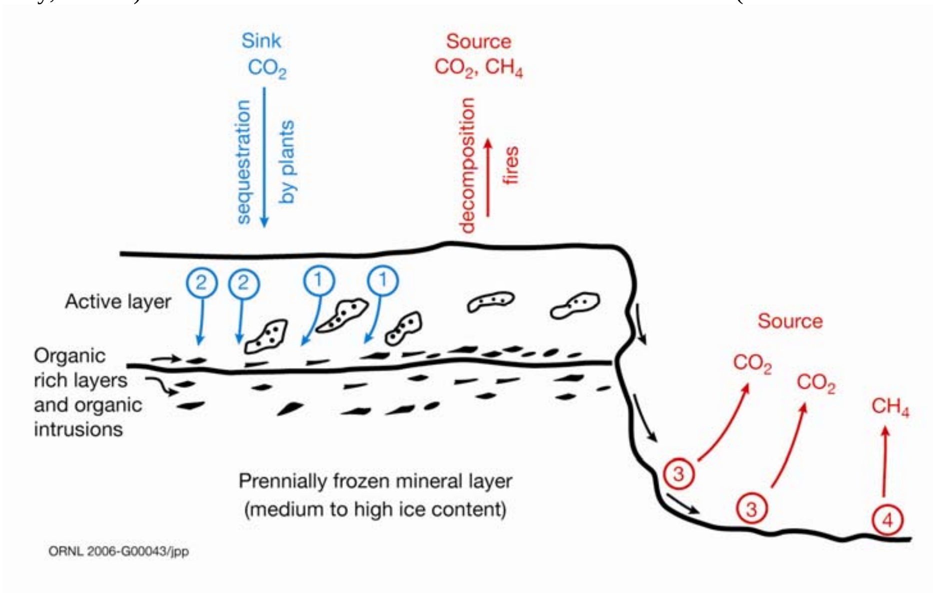
Fig. 12-2. Arctic, Subarctic, and Boreal ecoclimatic provinces (ecological regions) in North America (Ecoregions Working Group, 1989; Baily and Cushwa, 1981).

Carbon sinks

Permafrost-affected soil with a thick surface organic layer, dark-colored organic intrusions in the brown soil layer, and an underlying frozen, high-ice-content layer. The organic intrusions were translocated from the surface by cryoturbation. (Mackenzie Valley, Canada)

Carbon sources

Eroding high-ice-content permafrost soil composed of a dark frozen soil layer with an almost pure ice layer below. The thawing process generated a flow slide in which high-organic- content soil materials slumped into the water-saturated environment. (Mackenzie Delta area, Canada)



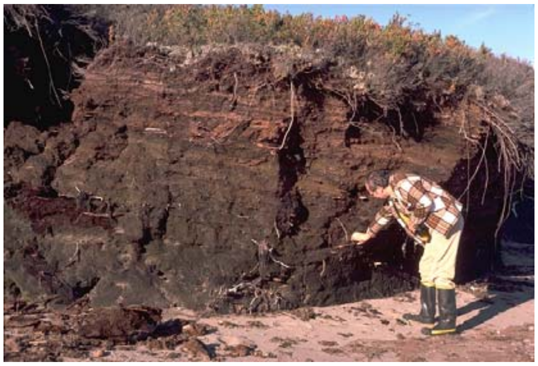
Perennially frozen deposit composed of an active layer that freezes and thaws annually and an underlying perennially frozen layer that has a high ice content.

Organic material deposited annually on the soil surface builds up as an organic soil layer. Some of this surface organic material is translocated into the deeper soil layers by cryoturbation (1). In addition, soluble organic matter is translocated into the deeper soil layers by movement of water to the freezing front and by gravity (2). Because these deeper soil layers have low temperatures (0 to -15°C), the organic material decomposes very slowly. Thus more organic material accumulates as long as the soil is frozen. In this state, the permafrost soil acts as a carbon sink.

Thermal erosion initiated by climate warming, wildfires or human activity causes the high-ice-content mineral soils to thaw, releasing the organic materials locked in the system. In this environment aerobic (3) and anaerobic (4) decomposition occurs releasing carbon dioxide and methane. In this state, the soil is a source of carbon.

Fig. 12-3. Carbon cycle in permafrost-affected upland (mineral) soils, showing below-ground organic carbon sinks and sources.

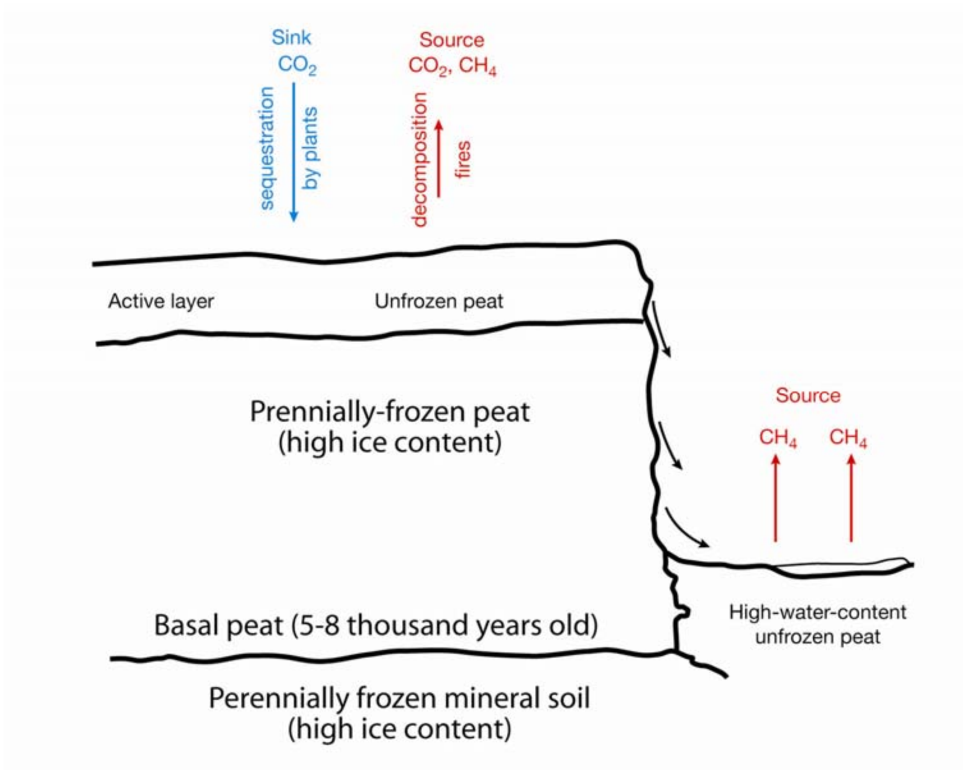
Carbon sinks



Perennially frozen peat deposit with multiple dark-colored peat layers. (Mackenzie River Delta area, Canada)

Carbon sources

Eroding perennially frozen peat deposit, showing the large blocks of peat slumping into the water- saturated collapsed area. (Fort Simpson area, Canada)



Perennially frozen peat deposits consist of an active layer that freezes and thaws annually and an underlying perennially frozen layer composed of ice-rich frozen peat and mineral materials.

Organic material is deposited annually on the peatland surface. Although a large portion (≥90%) of this organic material decomposes, the remainder is added to the peat deposit, producing an annual peat accumulation. The low soil temperatures (0 to –15°C) and the water-saturated and acid conditions cause this added organic carbon to be preserved and stored. This has been occurring for the last 5–8 thousand years. In this state, the peatland is a carbon sink.

Thermal erosion (thawing) of frozen peat deposits occurs as a result of climate change, wildfires, or human disturbances, releasing large amounts of water from the melting ice. This is mixed with the slumped peat material, initiating anaerobic decomposition in the much warmer environment. Anaerobic decomposition produces methane, which is expelled into the atmosphere. In this state, the peatland is a source of carbon.

Fig. 12-4. Carbon cycle in permafrost peatlands, showing below-ground organic carbon sinks and sources.

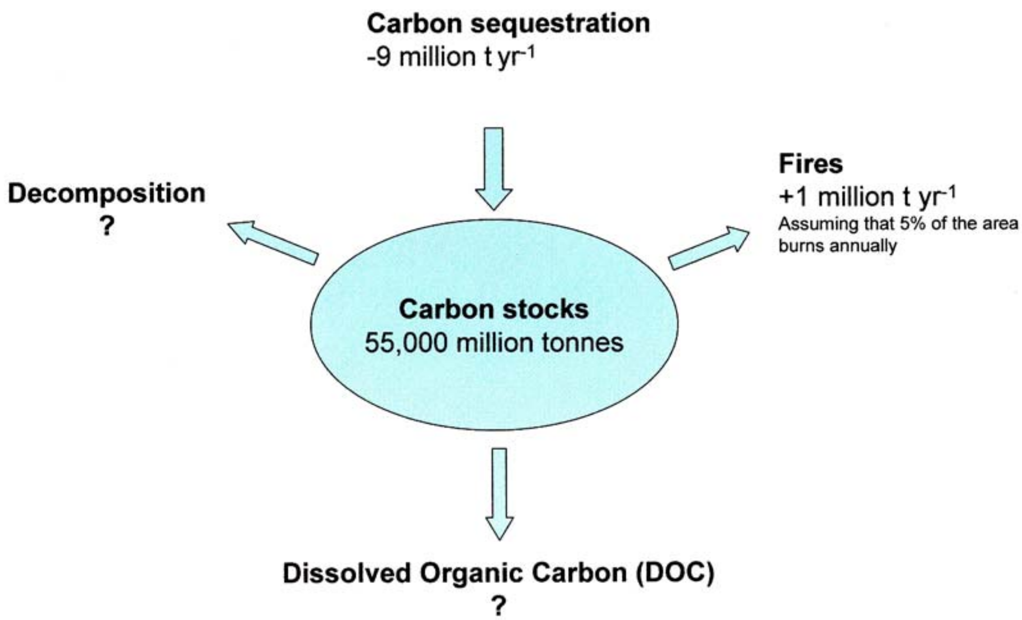


Fig. 12-5. Carbon cycle in perennially frozen mineral soils in the permafrost region.

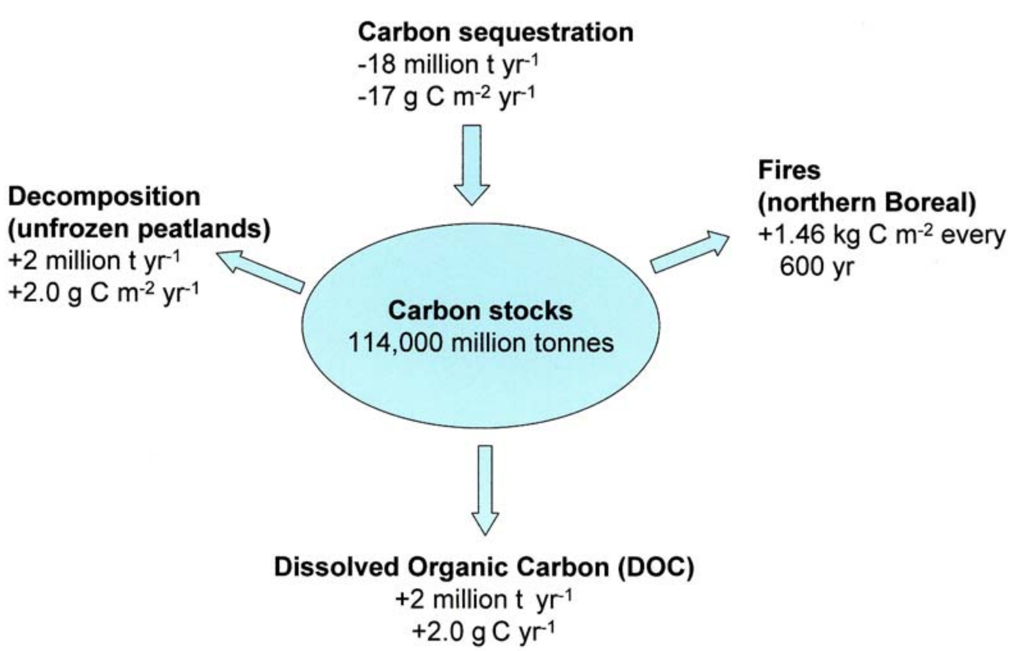


Fig. 12-6. Carbon cycle in peatlands in the permafrost region.

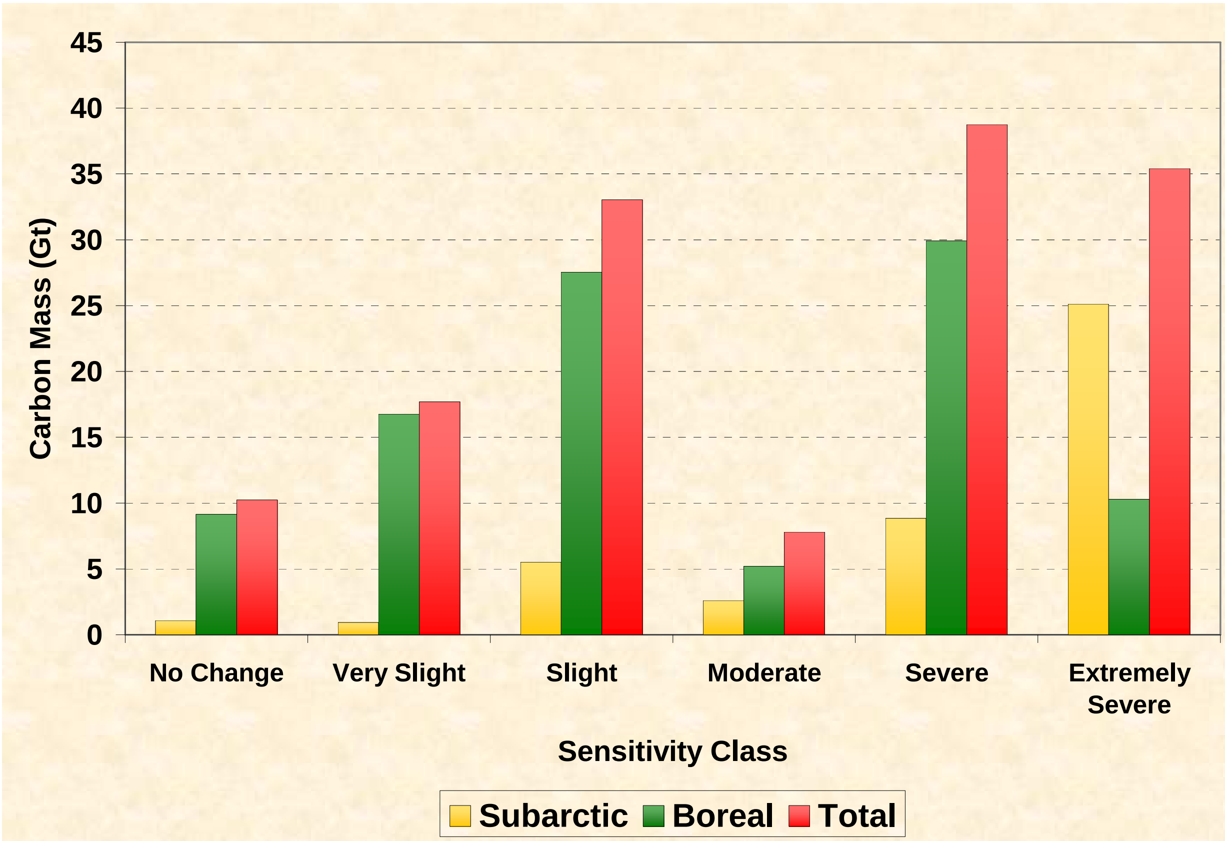


Fig. 12-7. The organic carbon mass in the various sensitivity classes for the Subarctic and Boreal Ecoclimatic Provinces (ecological regions) (Tarnocai, in press).

Chapter 13. Wetlands

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KEY FINDINGS

- North America is home to approximately 41% of the global wetland area, encompassing about 2.5 million km^2 with a carbon pool of approximately 220 Gt, mostly in peatland soils.
- North American wetlands currently are a CO_2 sink of approximately 70 Mt C yr^{-1}, but that estimate has an uncertainty of greater than 100%. North American wetlands are also a source of approximately 26 Mt yr^{-1} 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 43 Mt C yr^{-1}, primarily through the oxidation of carbon in peatland soils as they are drained and a more general reduction in carbon sequestration 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 CO_2 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 biodiversity, 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 sequestration and trace gas fluxes.

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 >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 13A. 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). 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 Mt methane (CH_4) yr^{-1}, which is a large fraction of the total annual global flux of about 600 Mt CH_4 yr^{-1} (Ehhalt *et al.*, 2001). This is important because methane is a potent greenhouse gas, second in importance to only carbon dioxide (Ehhalt *et al.*, 2001). A number of previous studies have examined the role of peatlands in the global carbon balance (reviewed in Mitra *et al.*, 2005). 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.

Given that many undisturbed wetlands are a natural sink for carbon dioxide and a source of methane, a note of caution in interpretation of our data is important. Using the International 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 greenhouse gas 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 methane emissions from wetlands have zero net radiative forcing, but an increase in methane 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 extant soil carbon pool in drained wetlands are both positive radiative forcings. Here we consider changes from a historical baseline of about 1800 A.D. to present and future emissions of greenhouse gas fluxes in North American wetlands.

INVENTORIES

Current Wetland Area and Rates of Loss

The current and historical 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 methane. The strengths and weakness of the wetland inventories of Canada, the United States, and Mexico are discussed in Appendix 13A.

The conterminous United States has 312,000 km^2 of FWMS wetlands, 93,000 km^2 of peatlands, and 23,000 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 13A-1 in Appendix 13A). However, wetland losses in the United States have declined from 1,855 km^2 yr^{-1} in the 1950s–1970s to 237 km^2 yr^{-1} in the 1980s–1990s (Dahl, 2000). Such data mask large differences in loss rates among wetland classes and conversion of wetlands to other classes, with potentially large effects on carbon stocks and fluxes (Dahl, 2000). For example, the majority of wetland losses in the United States have occurred in FWMS wetlands. As of the early 1980s, 84% of U.S. 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.

Table 13-1. The area, carbon pool, net carbon balance, and methane flux from wetlands in North America and the world.

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 13A-1 in Appendix 13A). However, given the lack of a nationwide wetland inventory and a general paucity of data, this number is highly uncertain.

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 historical wetlands have been converted to other uses (Moser *et al.*, 1996).

Carbon Pools

We estimate that North American wetlands have a current soil and plant carbon pool of 220 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 13A).

Soil Carbon Fluxes

North American peatlands currently have a net carbon balance of about -18 Mt C yr^{-1} (Table 13-1), but several large fluxes are incorporated into this estimate. **(Negative numbers indicate net fluxes into the ecosystem, whereas positive numbers indicate next fluxes into the atmosphere.)** Peatlands sequester -34 Mt C yr^{-1} (Table 13A-2 in Appendix 13A), but peatlands in the conterminous United States that have been drained for agriculture and forestry had a net oxidative flux of 18 Mt C yr^{-1} as of the early 1980s (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 2.4 Mt C yr^{-1} (Table 13-1), so the overall soil carbon sink of North American peatlands is about 20 Mt C yr^{-1} 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 methane emissions. Carbon sequestration associated with sediment deposition is a potentially large, but poorly quantified, flux in wetlands (Stallard, 1998). Using a review by Johnston (1991), we calculate a substantial carbon accumulation rate in sedimentation in FWMS wetlands of -129 g C m^{-2} yr^{-1} (see Appendix 13A). However, it is unlikely that the actual sequestration rate is this high. Researchers may have preferentially chosen wetlands with high sedimentation rates to study this process, providing a bias towards greater carbon sequestration. More fundamentally, it is important to distinguish between autochthonous carbon (derived from on-site plant production) and allochthonous carbon (imported from outside the wetland) in soil carbon storage. Almost all of the soil carbon stored in peatlands is of autochthonous origin and represents sequestration of atmospheric carbon dioxide at the landscape scale. In contrast, much of the soil carbon that is stored in FWMS wetlands is likely of allochthonous origin. At a landscape scale, redistribution of sediments from uplands to wetlands does not represent net carbon sequestration if the decomposition rate of carbon is the same in both environments. Carbon exported from upland source areas is likely to be relatively recalcitrant and physically protected from decomposers by association with mineral soil. Thus, despite the anaerobic conditions in wetlands, decomposition rates in deposited sediments may not be substantially lower than in the uplands from which those sediments were eroded. There are no data to our knowledge to evaluate these important caveats. Because of this reasoning, we somewhat arbitrarily assumed that sediment carbon sequestered in FWMS wetlands is of allochthonous origin and decomposed 25% slower than in the uplands from which the sediment was derived. Accordingly, we reduced our calculated rates of *landscape-level* carbon sequestration in FWMS wetlands by 75% to -34 g C m^{-2} yr^{-1} (Table 13A-2 in Appendix 13A). Nevertheless, this still represents a substantial carbon sink. For example, Stallard (1998) estimated that global wetlands are a large sediment sink, with a flux on the order of -1 Gt C yr^{-1}. However, this analysis was based on many assumptions and was acknowledged by the author to be a first guess at best.

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 13A-2 in Appendix 13A). However, due to the historical loss of FWMS wetland area, we estimate that they currently sequester 21 Mt C yr^{-1} 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 FWMS wetlands.

We estimate that estuarine wetlands currently sequester -9.7 Mt C yr^{-1}, with a historical reduction in sequestration capacity of 1.6 Mt C yr^{-1} due to loss of area (Table 13-1). However, the reduction is almost certainly greater because our ‘historical’ area is only from the 1950s. Despite the relatively small area of estuarine wetlands, they currently contribute about 26% of total wetland carbon sequestration in the conterminous United States and about 14% 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 robust, but it is unknown to what extent soil carbon sequestration is due to allochthonous versus autochthonous carbon. As with FWMS wetlands, the contribution of soil carbon sequestration in estuarine wetlands to the North American carbon budget is overestimated to the extent that allochthonous carbon simply represents redistribution of carbon in the landscape. There is also large uncertainty in 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 -70 Mt C yr^{-1} (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 25 Mt C yr^{-1} 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 yr^{-1} from conterminous U.S. peatlands, we estimate that North American wetlands currently sequester 43 Mt C yr^{-1} less than they did historically (Table 13A-2 in Appendix 13A). Furthermore, North American peatlands and FWMS wetlands have lost 2.6 Gt and 4.9 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. The current change in soil carbon flux in peatlands is about 176 to 266 Mt C yr^{-1} (Table 13A-2 in Appendix 13A), 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). Thus, globally peatlands are a moderate atmospheric source of carbon. The cumulative historical shift in soil carbon stocks has been estimated to be 5.5 to 7.1 Gt C (Maltby and Immirzi, 1993).

Methane and Nitrous Oxide Emissions

We estimate that North American wetlands emit 26 Mt CH_4 yr^{-1} (Table 13-1), a value that is substantially higher than the previous estimate by Bartlett and Harriss (1993) (see Appendix 13A). A mechanistic methane model yielded similar rates of 3.8 and 7.1 Mt CH_4 yr^{-1} for Alaska and Canada, respectively (Zhuang *et al.*, 2004). For comparison, a regional inverse atmospheric modeling approach estimated total methane emissions (from all sources) of 16 and 54 Mt CH_4 yr^{-1} for boreal and temperate North America, respectively (Fletcher *et al.*, 2004b).

Methane emissions are currently about 24 Mt CH_4 yr^{-1} less than they were historically in North American wetlands (see Table 13A-4 in Appendix 13A) 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 methane emissions. Similarly, we estimate that global methane emissions from natural wetlands are only about half of what they were historically due to loss of area (Table 13A-4 in Appendix 13A). 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 carbon dioxide and methane 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 carbon dioxide 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 methane 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.

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)[1]. 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 in a range between approximately neutral and a large source of net CO_2-carbon equivalents to the atmosphere (note that we discuss *net radiative forcing* in *Trends and Drivers of Wetland Carbon Fluxes*). It is likely that FWMS wetlands, with their high methane emissions, are a net source of CO_2-carbon equivalents to the atmosphere. In contrast, estuarine wetlands are a net sink for CO_2-carbon equivalents because they support both rapid rates of carbon sequestration and low methane 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 methane 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 carbon dioxide sequestered as peat (Frolking *et al*., 2006).

[1]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 methane flux columns in Table 13-1 can be directly compared by multiplying methane fluxes by the GWPs given here.

Plant Carbon Fluxes

We estimate that wetland forests in the conterminous United States currently sequester -10.3 Mt C yr^{-1} as increased plant biomass (see Table 13A-3 in Appendix 13A). 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.

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 the emission of methane to the atmosphere (a large decrease in radiative forcing) (Table 13A-1 and Appendix 13A). 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 methane 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 methane 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 13A-2 in Appendix 13A) 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 yr^{-1} 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.

Our compilation shows that attention needs to be directed toward understanding climate change impacts to FWMS wetlands, which collectively emit over 3-times more methane than North American peatlands and potentially sequester an equivalent amount of carbon. 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 methane 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 methane 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. 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 methane emissions from wetlands because of their large GWP. Wetlands emit about 107 Mt CH_4 yr^{-1} (Table 4), or 20% of the global total. Increases in atmospheric methane concentrations over the past century have had the second largest radiative forcing (after carbon dioxide) in human-induced climate change (Ehhalt *et al*., 2001). Moreover, methane 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 methane emissions from wetlands (Fletcher *et al*., 2004a; Wang *et al*., 2004; Zhuang *et al*., 2004).

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 methane 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 methane-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^{o}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. A doubling of 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 greenhouse gas (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 (Hoosbeek and VanKessel, 2004; Lichter *et al*., 2005). Elevated CO_2 has the potential to influence the carbon budgets of adjacent aquatic ecosystems by increasing export of DOC (Freeman *et al*., 2004) and DIC (Marsh *et al*., 2005).

Other important anthropogenic forcing factors that will affect future methane 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.

OPTIONS AND MEASURES

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 methane 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.

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 methane emissions from conterminous U.S. 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 methane compared to the carbon they sequester. Restoration of drained peatlands may stop the rapid loss of their soil carbon, which may compensate for increased methane 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 4,157 km^2 of uplands were converted into wetlands in the conterminous United States (Dahl, 2000). Using the soil carbon sequestration rate of 305 g C m^{-2} yr^{-1} found by Euliss *et al.* (2006) for restored prairie pothole wetlands[2], we estimate that wetland restoration in the U.S. would have sequestered 1.3 Tg 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 yr^{-1} through 2045 (Barker *et al.*, 1996). Euliss *et al.* (2006) estimated that if all cropland on former prairie pothole wetlands in the U.S. and Canada (162,244 km^2) were restored that 378 Tg C would be sequestered over 10 years in soils and plants. However, neither study accounted for the GWP of increased methane 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 because of storage as peat and are a small-to-moderate carbon sink (excluding methane effects). The largest unknown in the wetland carbon budget is the amount and significance of sedimentation in FWMS wetlands. With the exception of estuarine wetlands, methane 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 methane 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 to take additional measurements of annual methane 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.

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

[2]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 ($r^2 = 0.04$, $P = 0.241$).

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.

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 carbon dioxide 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.

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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 13A.

	Area[a] (km^2)		**Carbon Pool[b]** (Gt C)		**Net Carbon Balance[c]** ($Mt\ C\ yr^{-1}$)		**Historical Loss in Sequestration Capacity** ($Mt\ C\ yr^{-1}$)		**Methane Flux** ($Mt\ CH_4\ yr^{-1}$)	
Canada										
Peatland	1,135,608	****	149	****	-19	***	0.3	*	3.2	**
Freshwater Mineral	158,720	**	4.9	**	-5.1	*	6.5	*	5.7	*
Estuarine	6,400	***	0.1	***	-1.3	**	0.5	*	0.0	***
Total	1,300,728	****	154	****	-25	**	7.2	*	8.9	*
Alaska										
Peatland	132,196	****	15.9	**	-2.0	**	0.0	****	0.3	*
Freshwater Mineral	555,629	****	27.1	**	-18	*	0.0	****	1.4	*
Estuarine	8,400	****	0.1	***	-1.9	**	0.0	****	0.1	***
Total	696,224	*****	43.2	**	-22	*	0.0	****	1.8	*
Conterminous United States										
Peatland	93,477	****	14.4	***	4	*	2.1	*	3.4	**
Freshwater Mineral	312,193	*****	6.2	***	-18	*	15	*	11.2	**
Estuarine	23,000	*****	0.6	*****	-4.9	**	0.4	*	0.1	***
Total	428,670	*****	21.2	***	-19	*	17	*	14.7	**
U.S. Total	1,124,895	*****	64	**	-41	*	17	*	17	**
Mexico										
Peatland	10,000	*	1.5	*	-1.6	*	ND[d]	*	0.4	*
Freshwater Mineral	20,685	*	0.4	*	-0.7	*	ND	*	0.7	*
Estuarine	5,000	*	0.2	*	-1.6	*	0.5	*	0.0	*
Total	35,685	*	2.1	*	-3.9	*	ND	*	1.1	*

North America										
Peatland	1,371,281	****	180	****	-18	*	2.4	*	7	**
Freshwater Mineral	1,047,227	****	39	***	-42	*	21	*	19	*
Estuarine	42,800	***	1.0	***	-9.7	**	1.4	*	0.2	**
Total	2,461,308		220		-70	*	25	*	26	*
Global										
Peatland	3,443,000	***	460	***	150	**	16	*	37	**
Freshwater Mineral	2,315,000	***	46	***	-75	*	87	*	68	**
Estuarine	203,000	*	5.4	*	-43	*	13.2	*	1.5	**
Total	5,961,000	***	511	***	32	*	116	*	107	**

[a]Estuarine includes salt marsh, mangrove, and mudflat, except for Mexico and global for which no mudflat estimates were available.

[b]Includes soil C and plant C, but overall soil C is 98% of the total pool.

[c]Includes soil C sequestration, plant C sequestration, and loss of C due to drainage of wetlands. Plant C 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 13A).

[d]No data.

The error categories are as follows:

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

Appendix 13A
Wetlands – Supplemental Material

INVENTORIES

Current Wetland Area and Rates of Loss

The ability to estimate soil carbon pools and fluxes in North American wetlands is constrained by the national inventories (or lack thereof) for Canada, the United States, and Mexico (Davidson *et al*., 1999). The National Wetland Inventory (NWI) program of the United States has repeatedly sampled several thousand wetland sites using aerial photographs and more limited field verification. The data are summarized in a series of reports detailing changes in wetland area in the conterminous United States for the periods of the mid-1950s to mid-1970s (Frayer *et al*., 1983), mid-1970s to mid-1980s (Dahl and Johnson, 1991), and 1986 to 1997 (Dahl, 2000). We used these relatively high-quality data sets extensively for estimating wetland area and loss rates in the conterminous United States, including mud flats. However, the usefulness of the NWI inventory reports for carbon budgeting is limited by the level of classification used to define wetland categories within the Cowardin *et al.* (1979) wetland classification system. At the level used in the national status and trend reports, vegetated freshwater wetlands are classified by dominant physiognomic vegetation type, and it is impossible to make the important distinction between wetlands with deep organic soils (i.e., peatlands) and wetlands with mineral soils. The data are not at an adequate spatial resolution to combine with U.S. Department of Agriculture (USDA) National Resources Conservation Service (NRCS) soil maps to discriminate between the two types of wetlands (T. Dahl, personal comm.). Because of these data limitations, we used the NRCS soil inventory of peatlands (i.e., Histosols and Histels, or peatlands with and without permafrost, respectively) to estimate historical peatland area (Bridgham *et al*., 2000) and combined these data with regional estimates of loss (Armentano and Menges, 1986) to estimate current peatland area in the conterminous United States. We calculated the current area of freshwater mineral-soil (FWMS) wetlands in the conterminous United States by subtracting peatland area from total wetland area (Dahl, 2000). This approach was limited by the Armentano and Menges peatland area data being current only up to the early 1980s, although large losses of peatlands since then are unlikely due to the institution of wetland protection laws.

We used a similar approach for Alaskan peatlands: peatland area was determined by the NRCS soil inventory [N. Bliss, query of the NRCS State Soil Geographic (STATSGO) database, February 2006] and overall wetland inventory was determined by standard NWI methods (Hall *et al*., 1994). However, our peatland estimate of 132,000 km^2 (Table 13A-1) is 22% of the often cited value by Kivinen and Pakarinen (1981) of 596,000 km^2.

Table 13A-1. Current and historical area of wetlands in North America and the world ($\times 10^3$ km^2).

Kivinen and Pakarinen also used NRCS soils data (Rieger *et al*., 1979) for their peatland estimates, but they defined a peatland as having a minimum organic layer thickness of 30 cm, whereas the current U.S. and Canadian soil taxonomies require a 40-cm thickness. The original 1979 Alaska soil inventory has been reclassified with current U.S. soil taxonomy (J. Moore, Alaska State Soil Scientist, personal comm.). Using the reclassified soil inventory, Alaska has 417,000 km^2 of wetlands with a histic modifier that are not Histosols or Histels, indicating significant carbon accumulation in the surface horizons of FWMS wetlands. Thus, we conclude that Kivinen and Pakarinen's Alaska peatland area estimate is higher because many Alaskan wetlands have a thin organic horizon that is not deep enough to qualify as a peatland under current soil taxonomy. Our smaller peatland area significantly lowers our estimate of carbon pools and fluxes in Alaskan peatlands compared to earlier studies (see *Carbon Pools* below).

The area of salt marsh in the conterminous U.S. and Alaska were taken from Alexander *et al*. (1986) and Hall (1994), respectively, as reported in Mendelssohn and McKee (2000). Because these estimates include brackish tidal marshes, they cannot be compared directly to the area of Canadian salt marsh. The historical area of tidal wetlands in the conterminous U.S. was based on the NWI (Dahl, 2000), but 'historical' here only refers to the 1950s as we could not find earlier estimates. It is almost certain that historical salt marsh area in the conterminous U.S. was larger than our estimate. We made the reasonable assumption that the historical area of Alaskan tidal wetlands was similar to the current area. The area of freshwater tidal marshes was not included.

A regular national inventory of Canada's wetlands has not been undertaken, although wetland area has been mapped by ecoregion (National Wetlands Working Group, 1988). Extensive recent effort has gone into mapping Canadian peatlands (Tarnocai, 1998; Tarnocai *et al*., 2005). We calculated the current area of mineral-soil wetlands as the difference between total wetland area and peatland area in National Wetland Working Group (1988). Historical FWMS wetland area was obtained from Rubec (1996). Canadian salt marsh estimates were taken from a compilation by Mendelssohn and McKee (2000). The compilation does not include brackish or freshwater tidal marshes, and we were unable to locate other estimates of Canadian brackish marsh area. The historical area of these marshes was estimated from the National Wetland Working Group (1988), but it is highly uncertain. There are no reliable country-wide estimates of mud flat area for Canada, but a highly uncertain extrapolation from a limited number of regional estimates was possible.

No national wetland inventories have been done for Mexico. Current freshwater wetland estimates for Mexico were taken from Davidson *et al*. (1999) and Spiers (1999), who used inventories of discrete wetland regions performed by a variety of organizations. Thus, freshwater wetland area estimates for Mexico are highly unreliable and are possibly a large underestimate. For mangrove area in Mexico, we used the estimates compiled by Mendelssohn and McKee (2000), which are similar to estimates reported in Davidson *et al.* (1999) and Spalding *et al.* (1997). We could find no estimates of tidal marsh or mud flat area for Mexico. Since most vegetated Mexican tidal wetlands are dominated by mangroves (Olmsted, 1993; Mendelssohn and McKee, 2000), the omission of Mexican tidal marshes should not significantly affect our carbon budget. However, there may be large areas of mud flat that would significantly increase our estimate of carbon pools and sequestration in this country. We arbitrarily estimated that 25% of the mangrove area was lost since the late 1800s, which is less than the rough worldwide estimate of 50% wetland loss that is often cited (see Zedler and Kercher, 2005). A lower estimate is reasonable because wetland losses are lower in coastal systems than freshwater systems (Zedler and Kercher, 2005).

CARBON POOLS

Freshwater Mineral-Soil (Gleysol) Carbon Pools

Gleysol is a soil classification used by the Food and Agriculture Organization (FAO) and many countries that denotes mineral soils formed under waterlogged conditions (FAO-UNESCO, 1974). Tarnocai (1998) reported a soil carbon density of 200 Mg C ha^{-1} for Canadian Gleysols but did not indicate to what depth this extended. Batjes (1996) determined soil carbon content globally from the *Soil Map of the World* (FAO, 1991) and a large database of soil pedons. He gave a very similar average value for soil carbon density of 199 Mg C ha^{-1} (CV[3] = 212%, n = 14 pedons) for Gleysols of the world to 2-m depth; to 1-m depth, he reported a soil carbon density of 131 Mg C ha^{-1} (CV = 109%, n =142 pedons).

Gleysols are not part of the U.S. soil taxonomy scheme, and mineral soils with attributes reflecting waterlogged conditions are distributed among numerous soil groups. We used the NRCS State Soil Geographic (STATSGO) soils database to query for soil carbon density in “wet” mineral soils of the conterminous United States (all soils that had a surface texture described as peat, muck, or mucky peat, or appeared on the 1993 list of hydric soils, which were not classified as Histosols) (N. Bliss, query of NRCS STATSGO database, Dec. 2005). We used the average soil carbon densities of 162 Mg C ha^{-1} from this query for FWMS wetlands in the conterminous United States and Mexico.

Some caution is necessary regarding the use of Gleysol or ‘wet’ mineral soil carbon densities because apparently they include large areas of seasonally wet soils that are not considered wetlands by the more conservative definition of wetlands used by the United States and many other countries and organizations.

[3]CV is the “coefficient of variation,” or 100 times the standard deviation divided by the mean.

For example, Eswaran *et al.* (1995) estimated that global wet mineral-soil area was 8,808,000 km^2, which is substantially higher than the commonly accepted mineral-soil wetland area estimated by Matthews and Fung (1987) of 2,289,000 km^2 and Aselmann and Crutzen (1989) of 2,341,000 km^2, even accounting for substantial global wetland loss. In our query of the NRCS STATSGO database for the United States, we found 1,258,000 km^2 of wet soils in the conterminous United States versus our estimate of 312,000 km^2 of FWMS wetlands currently and 762,000 km^2 historically (Table 13A-1). We assume that including these wet-but-not-wetland soils will decrease the estimated soil carbon density, but to what degree we do not know. However, just considering the differences in area will give large differences in the soil carbon pool. For example, Eswaran *et al.* (1995) estimated that wet mineral soils globally contain 108 Gt C to 1-m depth, whereas our estimate is 46 Gt C to 2-m depth (Table 13A-2).

For Alaska, many soil investigations have been conducted since the STATSGO soil data was coded. We updated STATSGO by calculating soil carbon densities from data obtained from the NRCS on 479 pedons collected in Alaska, and then we used this data for both FWMS wetlands and peatlands. For some of the Histosols, missing bulk densities were calculated using averages of measured bulk densities for the closest matching class in the USDA Soil Taxonomy (NRCS, 1999). A matching procedure was developed for relating sets of pedons to sets of STATSGO components. If there were multiple components for each map unit in STATSGO, the percentage of the component was used to scale area and carbon data. We compared matching sets of pedons to sets of components at the four top levels of the U.S. Soil Taxonomy: Orders, Suborders, Great Groups, and Subgroups. For example, the soil carbon for all pedons having the same soil order were averaged, and the carbon content was applied to all of the soil components of the same order (e.g., Histosol pedons are used to characterize Histosol components). At the Order level, all components were matched with pedon data. At the suborder level, pedon data were not available to match approximately 20,000 km^2 (compared to the nearly 1,500,000-km^2 area of soil in the state), but the soil characteristics were more closely associated with the appropriate land areas than at the Order level. At the Great Group and Subgroup levels, pedon data were unavailable for much larger areas, even though the quality of the data when available became better. For this study, we used the Suborder-level matching. The resulting soil carbon density for Alaskan FWMS wetlands was 469 Mg C ha^{-1}, reflecting large areas of wetlands with a histic epipedon as noted above.

Peatland Soil Carbon Pools

The carbon pool of permafrost and non-permafrost peatlands in Canada had been previously estimated by Tarnocai *et al.* (2005) based upon an extensive database. Good soil-carbon density data are unavailable for peatlands in the United States, as the NRCS soil pedon information typically only goes to a maximum depth of between 1.5 to 2 m, and many peatlands are deeper than this. Therefore, we used the carbon density estimates of Tarnocai *et al.* (2005) of 1,441 Mg C ha^{-1} for Histosols and 1,048 Mg C ha^{-1} for Histels to estimate the soil carbon pool in Alaskan peatlands.

The importance of our using a smaller area of Alaskan peatlands becomes obvious here. Using the larger area from Kivinen and Pakarinen (1981), Halsey *et al.* (2000) estimated that Alaskan peatlands have a soil carbon pool of 71.5 Gt, almost 5-fold higher than our estimate. However, some of the difference in soil carbon between the two estimates can be accounted for by the 26 Gt C that we calculated resides in Alaskan FWMS wetlands (Table 13A-2).

Table 13A-2. Soil carbon pools (Gt) and fluxes (Mt yr^{-1}) of wetlands in North America and the world.

The peatlands of the conterminous United States are different in texture, and probably depth, from those in Canada and Alaska, so it is probably inappropriate to use the soil carbon densities for Canadian peatlands for those in the conterminous United States. For example, we compared the relative percentage of the Histosol suborders (excluding the small area of Folists, as they are predominantly upland soils) for Canada (Tarnocai, 1998), Alaska (updated STATSGO data, J. Moore, personal comm.), and the conterminous U.S. (NRCS, 1999). The relative percentage of Fibrists, Hemists, and Saprists, respectively, in Canada are 37%, 62%, and 1%, in Alaska are 53%, 27%, and 20%, and in the conterminous United States are 1%, 19%, and 80%. Using the STATSGO database (N. Bliss, query of NRCS STATSGO database, December 2005), the average soil carbon density for Histosols in the conterminous United States is 1,089 Mg C ha^{-1}, but this is an underestimate as many peatlands were not sampled to their maximum depth. Armentano and Menges (1986) reported average carbon density of conterminous U.S. peatlands to 1-m depth of 1,147 to 1,125 Mg C ha^{-1}. Malterer (1996) gave soil carbon densities of conterminous U.S. peatlands of 2,902 Mg C ha^{-1} for Fibrist, 1,874 Mg C ha^{-1} for Hemists, and 2,740 Mg C ha^{-1} for Saprists, but it is unclear how he derived these estimates. Batjes (1996) and Eswaran *et al.* (1995) gave average soil carbon densities to 1-m depth for global peatlands of 776 and 2,235 Mg C ha^{-1}, respectively. We chose to use an average carbon density of 1,500 Mg C ha^{-1}, which is in the middle of the reported range.

Estuarine Soil Carbon Pools

Tidal wetland soil carbon density was based on a country-specific analysis of data reported in an extensive compilation by Chmura *et al.* (2003). There were more observations for the United States (n = 75) than Canada (n = 34) or Mexico (n = 4), and consequently there were more observations of marshes than mangroves. The Canadian salt marsh estimate was used for Alaskan salt marshes and mud flats. In the conterminous United States and Mexico, country-specific marsh or mangrove estimates were used for mudflats. Although Chmura *et al.* (2003) reported some significant correlations between soil carbon density and mean annual temperature, scatter plots suggest the relationships are weak or driven by a few sites. Thus, we did not separate the data by region or latitude and used mean values for scaling. Chmura *et al.* (2003) assumed a 50-cm-deep profile for the soil carbon pool, which may be an underestimate.

Plant Carbon Pools

While extensive data on plant biomass in individual wetlands have been published, no systematic inventory of wetland plant biomass has been undertaken in North America. Nationally, the forest carbon biomass pool (including aboveground and belowground biomass) has been estimated to be 5.49 kg C m^{-2} (Birdsey, 1992), which we used for forested wetlands in the United States and Canada. This approach assumes that wetland forests do not have substantially different biomass carbon densities from upland forests. There is one regional assessment of forested wetlands in the southeastern United States, which comprise approximately 35% of the total forested wetland area in the conterminous United States. We utilized the southeastern U.S. regional inventory to evaluate this assumption; aboveground tree biomass averaged 125.2 m^3 ha^{-1} for softwood stands and 116.1 m^3 ha^{-1} for hardwood stands. Using an average wood density and carbon content, the carbon density for these forests would be 3.3 kg C m^{-2} for softwood stands and 4.2 kg C m^{-2} for hardwood stands. However, these estimates do not include understory vegetation, belowground biomass, or dead trees, which account for 49% of the total forest biomass (Birdsey, 1992). Using that factor to make an adjustment for total forest biomass, the range would be 4.9 to 6.6 kg C m^{-2} for the softwood and hardwood stands, respectively. Accordingly, the assumption of using 5.49 kg C m^{-2} seems reasonable for a national-level estimate.

The area of forested wetlands in Canada came from Tarnocai *et al.* (2005), for Alaska from Hall *et al.* (1994), and for the conterminous United States from Dahl (2000).

Since Tarnocai *et al.* (2005) divided Canadian peatland area into bog and fen, we used aboveground biomass for each community type from Vitt *et al.* (2000), and assumed that 50% of biomass is belowground. We used the average bog and fen plant biomass from Vitt *et al.* (2000) for Alaskan peatlands. For other wetland areas, we used an average value of 2,000 g C m^{-2} for non-forested wetland biomass carbon density (Gorham, 1991).

Tidal marsh root and shoot biomass data were estimated from a compilation in Table 8-7 in Mitsch and Gosselink (1993). There was no clear latitudinal or regional pattern in biomass, so we used mean values for each. Mangrove biomass has been shown to vary with latitude, so we used the empirical relationship from Twilley *et al*. (1992), for this relationship. We made a simple estimate using a single latitude that visually bisected the distribution of mangroves either in the United States (26.9^{o}) or Mexico

(23.5°). Total biomass was estimated using a root-to-shoot ratio of 0.82 and a carbon-mass-to-biomass ratio of 0.45, both from Twilley *et al.* (1992).

Plant biomass carbon data are presented in Table 13A-3.

Table 13A-3. Plant carbon pools (Gt) and fluxes (Mt yr^{-1}) of wetlands in North America and the world.

CARBON FLUXES

Peatland Soil Carbon Accumulation Rates

Most studies report the long-term apparent rate of carbon accumulation (LORCA) in peatlands based upon basal peat dates, but this assumes a linear accumulation rate through time. However, due to the slow decay of the accumulated peat, the true rate of carbon accumulation will always be less than the LORCA (Clymo *et al*., 1998), so most reported rates are inherently biased upwards. Tolonen and Turunen (1996) found that the true rate of peat accumulation was about 67% of the LORCA.

For estimates of soil carbon sequestration in conterminous U.S. peatlands, we used the data from 82 sites and 215 cores throughout eastern North America (Webb and Webb III, 1988). They reported a median accumulation rate of 0.066 cm yr^{-1} (mean = 0.092, sd = 0.085). We converted this value into a carbon accumulation rate of -1.2 Mg C ha^{-1} yr^{-1} by assuming 58% C (see NRCS Soil Survey Laboratory Information Manual, available on-line at http://soils.usda.gov/survey/nscd/lim/), a bulk density of 0.59 g cm^{-3}, and an organic matter content of 55%. **(Positive carbon fluxes indicate net fluxes to the atmosphere, whereas negative carbon fluxes indicate net fluxes into an ecosystem**.) The bulk density and organic matter content were the average from all Histosol soil map units greater than 202.5 ha (n = 5,483) in the conterminous United States from the National Soil Information System (NASIS) data base provided by S. Campbell (USDA NRCS, Portland, OR). For comparison, Armentano and Menges (1986) used soil carbon accumulation rates that ranged from -0.48 Mg C ha^{-1} yr^{-1} in northern conterminous U.S. peatlands to -2.25 Mg C ha^{-1} yr^{-1} in Florida peatlands.

Peatlands accumulate lesser amounts of soil carbon at higher latitudes, with especially low accumulation rates in permafrost peatlands (Ovenden, 1990; Robinson and Moore, 1999). The rates used in this report reflect this gradient, going from -0.13 to -0.19 to -1.2 Mg C ha^{-1} yr^{-1} in permafrost peatlands, non-permafrost Canadian and Alaskan peatlands, and peatlands in the conterminous United States and Mexico, respectively (Table 13A-2).

Freshwater Mineral-Soil Wetland Carbon Accumulation Rates

Many studies have estimated sediment deposition rates in FWMS wetlands, with an average rate of 1,680 g m^{-2} yr^{-1} (range 0 to 7,840) in a review by Johnston (1991). Assuming 7.7% carbon for FWMS wetlands (Batjes, 1996), this gives a substantial accumulation rate of -129 g C m^{-2} yr^{-1}. Johnston (1991) found many more studies that just reported vertical sediment accumulation rates, with an average of 0.69 cm yr^{-1} (range -0.6 to 2.6). If we assume a bulk density of 1.38 g cm^{-3} for FWMS wetlands (Batjes, 1996), this converts into an impressive accumulation rate of -733 g C m^{-2} yr^{-1}. For reasons discussed in the main chapter, we assumed a lower carbon sequestration rate in FWMS wetlands of -34 g C m^{-2} yr^{-1}.

Agriculture typically increases sedimentation rates by 10- to 100-fold, and 90% of sediments are stored within the watershed, or about 3 Gt yr^{-1} in the United States (Meade *et al*., 1990, as cited in Stallard, 1998), as cited in Stallard, 1998). Converting this to 1.5% C equates to -45 Mt C yr^{-1}, part of which will be stored in wetlands and is well within our estimated storage rate in FWMS wetlands (Table 13A-2).

Estuarine Carbon Accumulation Rates

Carbon accumulation in tidal wetlands was assumed to be entirely in the soil pool. This should provide a reasonable estimate because marshes are primarily herbaceous, and mangrove biomass should be in steady state unless the site was converted to another use. An important difference between soil carbon sequestration in tidal and non-tidal systems is that tidal sequestration occurs primarily through burial driven by sea level rise. For this reason, carbon accumulation rates can be estimated well with data on changes in soil surface elevation and carbon density. Rates of soil carbon accumulation were calculated from Chmura *et al.* (2003) as described for the soil carbon pool (above). These estimates are based on a variety of methods, such as ^{210}Pb dating and soil elevation tables, which integrate vertical soil accumulation rates over periods of time ranging from 1–100 yr. The soil carbon sequestered in estuarine wetland sediments is likely to be a mixture of both allochthonous and autochthonous sources. However, without better information, we assumed that in situ rates of soil carbon sequestration in estuarine wetlands is representative of the true landscape-level rate.

Extractive Uses of Peat

Use of peat for energy production is, and always has been, negligible in North America, as opposed to other parts of the world (WEC, 2001). However, Canada produces a greater volume of horticultural and agricultural peat than any other country in the world (WEC, 2001). Currently, 124 km^2 of Canadian peatlands have been under extraction now or in the past (Cleary *et al*., 2005). A life-cycle analysis by these authors estimated that as of 1990 Canada emitted 0.9 Mt yr^{-1} of CO_2-C equivalents through peat extraction. The U.S. production of horticultural peat is about 19% of Canada's (Joosten and Clarke, 2002), which assuming a similar life-cycle as for Canada, suggests that the United States produces 0.2 Mt of CO_2-C equivalents through peat extraction.

Methane Fluxes

Moore *et al.* (1995) reported a range of methane fluxes from 0 to 130 g CH_4 m^{-2} yr^{-1} from 120 peatland sites in Canada, with the majority <10 g CH_4 m^{-2} yr^{-1}. They estimated a low average flux rate of 2 to 3 g CH_4 m^{-2} yr^{-1}, which equaled an emission of 2–3 Mt CH_4 yr^{-1} from Canadian peatlands. We used an estimate of 2.5 g CH_4 m^{-2} yr^{-1} for Canadian peatlands and Alaskan freshwater wetlands (Table 13A-4).

Table 13A-4. Methane fluxes (Mt yr^{-1}) from wetlands in North America and the world.

To our knowledge, the last synthesis of field measurements of methane emissions from wetlands was done by Bartlett and Harriss (1993). We supplemented their analysis with all other published field studies (using chamber or eddy covariance techniques) we could find that reported annual or average daily methane fluxes in the conterminous United States (Table 13A-5). We excluded a few studies that used cores or estimated diffusive fluxes.

Table 13A-5. Methane fluxes measured in the conterminous United States.

In cases where multiple years from the same site were presented, we took the average of those years. Similarly, when multiple sites of the same type were presented in the same paper, we took the average. Studies were separated into freshwater and estuarine systems.

In cases where papers presented both an annual flux and a mean daily flux, we calculated a conversion factor [annual flux/(average daily flux × 10^3)] to quantify the relationship between those two numbers (Table 13A-5). When we looked at all studies (n = 30), this conversion factor was 0.36, suggesting that there is a 360-day emission season. There was surprisingly little variation in this ratio, and it was similar in freshwater (0.36) and estuarine (0.34) wetlands. In contrast, previous syntheses used a 150-day emission season for temperate wetlands (Matthews and Fung, 1987; Bartlett and Harriss, 1993). While substantial winter methane emissions have been found in some studies, it is likely that flux data from most studies have a non-normal distribution with occasional periods of high flux rates that are better captured with annual measurements.

Using the conversion factors for freshwater and estuarine wetlands, we estimated average annual fluxes from the average daily fluxes. For freshwater wetlands, the calculated average annual flux rate was

38.6 g CH_4 m^{-2} yr^{-1} (n = 74), which is slightly larger than the average actual measured flux rate of 32.1 g CH_4 m^{-2} yr^{-1} (n = 32). For estuarine wetlands, the average calculated annual flux rate was 9.8 g CH_4 m^{-2} yr^{-1} (n = 25), which is smaller than the average measured flux rate of 16.9 g CH_4 m^{-2} yr^{-1} (n = 13). However, if we remove one outlier, the average measured flux rate is 10.2 g CH_4 m^{-2} yr^{-1}.

Finally, we combined both approaches. In cases where a paper presented an annual value, we used that number. In cases where only an average daily number was presented, we used that value corrected with the appropriate conversion factor. For conterminous U.S. wetlands, FWMS Canadian wetlands, and Mexican wetlands, we used an average flux of 36 g CH_4 m^{-2} yr^{-1}, and for estuarine wetlands, we used an average flux of 10.3 g CH_4 m^{-2} yr^{-1}.

Plant Carbon Fluxes

For ecosystems at approximately steady state, plant biomass should be reasonably constant on average because plant production is roughly balanced by mortality and subsequent decomposition. We assumed insignificant plant biomass accumulation in freshwater and estuarine marshes because they are dominated by herbaceous plants that do not accumulate carbon in wood. Sequestration in plants in relatively undisturbed forested wetlands in Alaska and many parts of Canada is probably small, although there may be substantial logging of Canadian forested wetlands for which we do not have data. Similarly, no data was available to evaluate the effect of harvesting of woody biomass in Mexican mangroves on carbon fluxes.

Tree biomass carbon sequestration averages -140 g C m^2 yr^{-1} in U.S. forests across all forest types (Birdsey, 1992). Using the tree growth estimates from the southeastern U.S. regional assessment of wetland forests (Brown *et al*., 2001) yields an even lower estimate of sequestration in aboveground tree biomass (approx. -50.2 g C m^2 yr^{-1}). We used this lower value and area estimates from Dahl (2000) to estimate that forested wetlands in the conterminous U.S. currently sequester -10.3 Mt C yr^{-1}.

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Table 13A-1. Current and historical area of wetlands in North America and the world (×10^3 km^2). Historical refers to approximately 1800, unless otherwise specified.

	Permafrost peatlands	Non-permafrost peatlands	Mineral-soil freshwater	Salt marsh	Mangrove	Mudflat	**Total**
Canada							
Current	422[a]	714[a]	159[b]	0.4[c]	0	6[d]	1301
Historical	424[e]	726[f]	359[g]	1.3[b]	0	7[h]	1517
Alaska							
Current	89[i]	43[i]	556[j]	1.4[c]	0	7[k]	696
Historical	89	43	556	1.4	0	7	696
Conterminous United States							
Current	0	93[l]	312[m]	18[c]	3[c]	2[n]	428
Historical	0	111[i]	762[o]	20[p]	4[n]	3[n]	899
Mexico							
Current	0	10[p]	21[p]	0	5[c]	ND[q]	36
Historical	0	45[p]		0	7[h]	ND	52
North America							
Current	511	861	1,047	20	8	15	2,461
Historical	513	894[r]	1,706[r]	23	11	17	3,164
Global							
Current	3,443[s]		2,315[t]	22[u]	181[v]	ND	~6,000
Historical	4,000[w]		5,000[x]	26[y]	ND	ND	~9,000[x]

[a]Tarnocai *et al.* (2005).

[b]National Wetlands Working Group (National Wetlands Working Group, 1988).

[c]Mendelssohn and McKee (2000).

[d]Estimated from the area of Canadian salt marshes and the ratio of mudflat to salt marsh area reported by Hanson and Calkins (1996).

[e]Accounting for losses due to permafrost melting in western Canada (Vitt *et al*., 1994). This is an underestimate, as similar, but undocumented, losses have probably also occurred in eastern Canada and Alaska.

[f]9000 km^2 lost to reservoir flooding (Rubec, 1996), 250 km^2 to forestry drainage (Rubec, 1996), 124 km^2 to peat harvesting for horticulture (Cleary *et al*., 2005), and 16 km^2 to oil sands mining (Turetsky *et al*., 2002). See note e for permafrost melting estimate.

[g]Rubec (1996).

[h]Assumed same loss rate as the conterminous United States since 1954 (Dahl, 2000).

[i]Historical area from NRCS soil inventory (Bridgham *et al*., 2000), except Alaska inventory updated by N. Bliss from a February 2006 query of the STATSGO database. Less than 1% wetland losses have occurred in Alaska (Dahl, 1990).

[j]Total freshwater wetland area from (Hall *et al*., 1994) minus peatland area.

[k]Hall (1994).

[l]Historical area from Bridgham *et al.* (2000) minus losses in Armentano and Menges (1986).

[m]Overall freshwater wetland area from Dahl (2000) minus peatland area.

[n]Dahl (2000). Historical area estimates are only from the 1950s.

[o]Total historical wetland area from Dahl (1990) minus historical peatland area minus historical estuarine area.

[p]Spiers (1999).

[q]ND indicates that no data are available.

[r]Assuming that historical proportion of peatlands to total wetlands in Mexico was the same as today.

[s]Bridgham *et al.* (2000) for the United States, Tarnocai *et al.* (2005) for Canada, Joosten and Clarke (2002) for the rest of world. Recent range in literature 2,974,000–3,985,000 km^2 (Matthews and Fung, 1987; Aselmann and Crutzen, 1989; Maltby and Immirzi, 1993; Bridgham *et al*., 2000; Joosten and Clarke, 2002).

[t]Average of 2,289,000 km^2 from Matthews and Fung (1987) and 2,341,000 km^2 Aselmann and Crutzen (1989).

[u]Chmura *et al.* (2003). Underestimated because no inventories were available for the continents Asia, South America and Australia which are mangrove-dominated but also support salt marsh.

[v]Spalding (1997).

[w]Range from 3,880 to 4,086 in Maltby and Immirzi (1993).

[x]Approximately 50% loss from Moser *et al.* (1996).

[y]Assumed.

Table 13A-2. Soil carbon pools (Gt) and fluxes (Mt yr^{-1}) of wetlands in North America and the world. "Sequestration in current wetlands" refers to carbon sequestration in extant wetlands; "oxidation in former wetlands" refers to emissions from wetlands that have been converted to non-wetland uses or conversion among wetland types due to human influence; "historical loss in sequestration capacity" refers to the loss in the carbon sequestration function of wetlands that have been converted to non-wetland uses; "change in flux from wetland conversions" is the sum of the two previous fluxes. Positive flux numbers indicate a net flux into the atmosphere, whereas negative numbers indicate a net flux into the ecosystem**.**

	Permafrost peatlands	Non-perma-frost peatlands	Mineral-soil freshwater	Salt marsh	Mangrove	Mudflat	**Total**
Canada							
Pool Size in Current Wetlands	44.2[a]	102.9[a]	4.6[b]	0.0[c]	0.0	0.1[d]	151.8
Sequestration in Current Wetlands	-5.5[e]	-13.6[e]	-5.1[f]	-0.1	0.0	-1.2[d]	-25.5
Oxidation in Former Wetlands	0.2[g]		0.0[h]	0.0[i]	0.0	0.0	0.2
Historical Loss in Sequestration Capacity	0.0[e]	0.2[e]	6.5[f]	0.2	0.0	0.3	7.2
Change in Flux From Wetland Conversions	0.4		6.5	0.2	0.0	0.3	7.4
Alaska							
Pool Size in Current Wetlands	9.3[j]	6.2[j]	26.0[k]	0.0	0.0	0.1	41.7
Sequestration in Current Wetlands	-1.1[e]	-0.8[e]	-18.0[f]	-0.3	0.0	-1.6	-21.9
Oxidation in Former Wetlands	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Historical Loss in Sequestration Capacity	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Change in Flux From Wetland Conversions	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Conterminous United States							
Pool Size in Current Wetlands	0	14.0[l]	5.1[k]	0.4	0.1	0.1	19.7
Sequestration in Current Wetlands	0	-11.6[m]	-10.1[f]	-3.9	-0.5	-0.5	-26.6
Oxidation in Former Wetlands	0	18.0[n]	0.0[h]	0.0	0.0	0.0	18.0
Historical Loss in Sequestration Capacity	0	2.1[m]	14.5[f]	0.3	0.0	0.1	17.1
Change in Flux from Wetland Conversions	0	20.1	14.6	0.3	0.0	0.1	35.2
Mexico							
Pool Size in Current Wetlands	0.0	1.5[l]	0.3[k]	0.0	0.1	ND*	1.9
Sequestration in Current Wetlands	0	-1.6[o]	-0.7[f]	0.0	-1.6	ND	-3.9

Oxidation in Former Wetlands	0	ND	ND	0.0	0.0	0.0	ND
Historical Loss in Sequestration Capacity	0	ND	ND	0.0	0.5	ND	0.5
Change in Flux from Wetland Conversions	0	ND	ND	0.0	0.5	ND	0.5
North America							
Pool Size in Current Wetlands	53.5	124.6	36.0	0.4	0.2	0.3	215.1
Sequestration in Current Wetlands	-6.6	-27.6	-33.9	-4.3	-2.1	-3.3	-77.8
Oxidation in Former Wetlands	18.2		0.0	0.0	0.0	0.0	18.2
Historical Loss in Sequestration Capacity	0	2.3	21.0	0.5	0.5	0.5	24.8
Change in Flux from Wetland Conversions	20.5		21.1	0.5	0.5	0.5	43.1
Global							
Pool Size in Current Wetlands	462[p]		46[q]	0.4[r]	5.0[r]	ND	513
Sequestration in Current Wetlands	-55[s]		-75[f]	-4.6[r]	-38.0[r]	ND	-173
Oxidation in Former Wetlands	205[t]		ND	0	0	0	205
Historical Loss in Sequestration Capacity	16[t]		87[f]	0.8[u]	12.7[v]	ND	116
Change in Flux From Wetland Conversions	221[t]		> 87[w]	0.8	12.7	ND	321

*ND indicates that no data are available.

[a]Tarnocai *et al.* (2005).

[b]Tarnocai (1998).

[c]Rates calculated from Chimura *et al.* (2003); areas from Mendelssohn and McKee (2000).

[d]Assumed the same carbon density and accumulation rates as the adjacent vegetated wetland ecosystem (mangrove data for Mexico and salt marsh data elsewhere).

[e]Assumed carbon accumulation rate of 0.13 Mg C ha^{-1} yr^{-1} for permafrost peatlands and 0.19 Mg C ha^{-1} yr^{-1} non-permafrost peatlands. Reported range of long-term apparent accumulation rates from 0.05-0.35 (Ovenden, 1990; Maltby and Immirzi, 1993; Trumbore and Harden, 1997; Vitt *et al*., 2000; Turunen *et al*., 2004).

[f]Potential rate calculated as the average sediment accumulation rate of 1680 g m^{-2} yr^{-1} (range 0–7840) from Johnston (1991) times 7.7% C (CV = 109) (Batjes, 1996). We assumed that all sequestered soil C was of allochthonous origin and decomposition was 25% slower in wetlands than in the uplands from which the sediment was eroded (see text).

[g]Sum of -0.24 Mt C yr^{-1} from horticulture removal of peat (Cleary *et al*., 2005) and 0.10 Mt C yr^{-1} from increased peat sequestration due to permafrost melting (Turetsky *et al*., 2002).

[h]Assumed that the net oxidation of 8.6% of the soil carbon pool (Euliss *et al*., 2006) over 50 yr after conversion to non-wetland use.

[i]Assumed that conversion of tidal systems is caused by fill and results in burial and preservation of SOM define SOM rather than oxidation.

[j]Soil carbon densities of 1,441 Mg C ha^{-1} for Histosols and 1,048 Mg C ha^{-1} for Histels (Tarnocai *et al*., 2005).

[k]Soil carbon density of 162 Mg C ha^{-1} for the conterminous United States and Mexico and 468 Mg C ha^{-1} for Alaska based upon NRCS STATSGO database and soil pedon information.

[l]Assumed soil carbon density of 1,500 Mg C ha^{-1}.

[m]Webb and Webb (1988).

[n]Estimated loss rate as of early 1980s (Armentano and Menges,1986). Overall wetlands losses in the United States have declined dramatically since then (Dahl, 2000) and probably even more so for Histosols, so this number may still be representative.

[o]Using peat accumulation rate of 1.6 Mg C ha^{-1} (range 1.0–2.25) (Maltby and Immirzi, 1993).

[p]From Maltby and Immirzi (1993). Range of 234 to 679 Gt C (Gorham, 1991; Maltby and Immirzi, 1993; Eswaran *et al*., 1995; Batjes, 1996; Lappalainen, 1996; Joosten and Clarke, 2002).

[q]Soil carbon density of 199 Mg C ha^{-1} (Batjes, 1996).

[r]Chmura *et al.* (2003).

[s]Joosten and Clarke (2002) reported range of -40 to -70 Mt C yr^{-1} . Using the peatland estimate in Table 13A-1 and a C accumulation rate of 0.19 Mg C ha^{-1} yr^{-1}, we calculate a global flux of -65 Mt C yr^{-1} in peatlands.

[t]Current oxidative flux is the difference between the change in flux and the historical loss in sequestration capacity from this table. The change in flux is from Maltby and Immirzi (1993) (reported range 176 to 266 Mt C yr^{-1}) and the historical loss in sequestration capacity is from this table for North America, from Armentano and Menges (1986) for other northern peatlands, and from Maltby and Immirzi (1993) for tropical peatlands.

[u]Assumed that global rates approximate the North America rate because most salt marshes inventoried are in North America.

[v]Assumed 25% loss globally since the late 1800s.

[w]> sign indicates that this a minimal loss estimate.

Table 13A-3. Plant carbon pools (Gt) and fluxes (Mt yr^{-1}) of wetlands in North America and the world. Positive flux numbers indicate a net flux into the atmosphere, whereas negative numbers indicate a net flux into the ecosystem.

<table>
<tr><th></th><th>Permafrost peatlands</th><th>Non-perma-frost peatlands</th><th>Mineral-soil freshwater</th><th>Salt marsh</th><th>Mangrove</th><th>Total</th></tr>
<tr><td><u>Canada</u></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Pool Size in Current Wetlands</td><td colspan="2">1.4[a]</td><td>0.3[b]</td><td>0.0[c]</td><td>0.0</td><td>1.7</td></tr>
<tr><td>Sequestration in Current Wetlands</td><td>0.0</td><td colspan="2">ND*</td><td>0.0</td><td>0.0</td><td>0.0</td></tr>
<tr><td><u>Alaska</u></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Pool Size in Current Wetlands</td><td colspan="2">0.4[a]</td><td>1.1[d]</td><td>0.0</td><td>0.0</td><td>1.5</td></tr>
<tr><td>Sequestration in Current Wetlands</td><td>0.0</td><td>0.0</td><td>0.0</td><td>0.0</td><td>0.0</td><td>0.0</td></tr>
<tr><td><u>Conterminous United States</u></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Pool Size in Current Wetlands</td><td>0.0</td><td colspan="2">1.5[d]</td><td>0.0</td><td>0.0</td><td>1.5</td></tr>
<tr><td>Sequestration in Current Wetlands</td><td>0.0</td><td colspan="2">-10.3[e]</td><td>0.0</td><td>0.0</td><td>-10.3</td></tr>
<tr><td><u>Mexico</u></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Pool Size in Current Wetlands</td><td>0.0</td><td>0.0[b]</td><td>0.0[b]</td><td>0.0</td><td>0.1</td><td>0.1</td></tr>
<tr><td>Sequestration in Current Wetlands</td><td>0.0</td><td>ND</td><td>ND</td><td>0.0</td><td>ND</td><td>0.0</td></tr>
<tr><td><u>North America</u></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Pool Size in Current Wetlands</td><td colspan="3">4.8</td><td>0.0</td><td>0.1</td><td>4.9</td></tr>
<tr><td>Sequestration in Current Wetlands</td><td>0.0</td><td colspan="2">-10.3</td><td>0.0</td><td>ND</td><td>-10.3</td></tr>
<tr><td><u>Global</u></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Pool Size in Current Wetlands</td><td colspan="2">6.9[b]</td><td>4.6[b]</td><td>0.0[f]</td><td>4.0[g]</td><td>15.5</td></tr>
<tr><td>Sequestration in Current Wetlands</td><td>0.0</td><td>ND</td><td>ND</td><td>0.0</td><td>ND</td><td>ND</td></tr>
</table>

*ND indicates that no data are available.

[a]Biomass for non-forested peatlands from Vitt *et al.* (2000), assuming 50% of biomass is belowground. Forest biomass density from Birdsey (1992) and forested area from Tarnocai *et al.* (2005) for Canada and from Hall *et al.* (1994) for Alaska.

[b]Assumed 2000 g C m^{-2} in aboveground and belowground plant biomass (Gorham, 1991).

[c]Biomass data from Mitsch and Gosselink (1993).

[d]Biomass for non-forested wetlands from Gorham (1991). Forest biomass density from Birdsey (1992), and forested area from Hall *et al*. (1994) for Alaska and Dahl (2000) for the conterminous U.S..

[e]50 g C m^{-2} yr^{-1} sequestration from forest growth from a southeastern U.S. regional assessment of wetland forest growth (Brown *et al*., 2001).
[f]Assumed that global pools approximate those from North America because most salt marshes inventoried are in North America.
[g]Twilley *et al*. (1992).

Table 13A-4. Methane fluxes (Mt yr^{-1}) from wetlands in North America and the world

	Permafrost peatlands	Non-perma-frost peatlands	Mineral-soil freshwater	Salt marsh	Mangrove	Mudflat	**Total**
Canada							
CH_4 Flux in Current Wetlands	1.1[a]	2.1[b]	5.7	0.0	0.0	0.0[c]	8.9
Historical change in CH_4 Flux	0.0	0.3	-7.2	0.0	0.0	0.0	-6.9
Alaska							
CH_4 Flux in Current Wetlands	0.2	0.1	1.4	0.0	0.0	0.1	1.8
Historical change in CH_4 Flux	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Conterminous United States							
CH_4 Flux in Current Wetlands	0.0	3.4	11.2	0.1	0.0	0.0	14.7
Historical change in CH_4 Flux	0.0	-0.6	-16.2	0.0	0.0	0.0	-16.8
Mexico							
CH_4 Flux in Current Wetlands	0.0	0.4	0.7	0.0	0.0	ND*	1.1
Historical change in CH_4 Flux	0.0	-0.5		0.0	0.0	ND	-0.5
North America							
CH_4 Flux in Current Wetlands	1.3	5.9	19.1	0.1	0.1	0.1	26.5
Historical change in CH_4 Flux	0.0	-24.2		0.0	0.0	0.0	-24.2
Global							
CH_4 Flux in Current Wetlands	14.1[d]	22.5[d]	68.0[d]	0.1[e]	1.4	ND	164[f]
Historical change in CH_4 Flux	-3.6		-79	0.0[g]	-0.5	ND	-83

*ND indicates that no data are available.

[a]Used CH_4 flux of 2.5 g m^{-2} yr^{-1} (range 0 to 130, likely mean 2–3) (Moore and Roulet, 1995) for Canadian peatlands and all Alaskan freshwater wetlands. Used CH_4 flux of 36.0 g m^{-2} yr^{-1} for Canadian freshwater mineral-soil wetlands and all U.S. and Mexican freshwater wetlands and 10.3 g m^{-2} yr^{-1} for estuarine wetlands—from synthesis of published CH_4 fluxes for the United States (see Table 13A-5).

[b]Includes a 17-fold increase in CH_4 flux (Kelly *et al*., 1997) in the 9000 km^2 of reservoirs that have been formed on peatlands (Rubec, 1996) and an estimated CH_4 flux of 15 g m^{-2} yr^{-1} (Moore *et al*., 1998) from 2,630 km^2 of melted permafrost peatlands (Vitt *et al*., 1994).

[c]Assumed trace gas fluxes from unvegetated estuarine wetlands (i.e., mudflats) was the same as adjacent wetlands.

[d]Bartlett and Harriss (1993).

[e]Assumed that global rates approximate the North America rate because most salt marshes area is in North America.

[f]Ehhalt *et al.* (2001), range of 92 to 237 Mt yr^{-1}.

[g]Assumed a conservative 25% loss since the late 1800s.

Table 13A-5. Methane fluxes measured in the conterminous United States. The conversion factor is the ratio of the daily average flux to the measured annual flux × 10^3. The calculated annual flux was determined based upon the average conversion factor for freshwater (FW) and saltwater wetlands (SW). The measured annual flux was used if that was available; otherwise, the calculated annual flux was used.

Habitat	State	Method[a]	Salt/ Fresh	Daily Average Flux (mg CH_4 m^{-2} d^{-1})	Measured Annual Flux (g CH_4 m^{-2} yr^{-1})	Conversion Factor	Calculated Annual Flux (g CH_4 m^{-2} yr^{-1})	Used Annual Flux (g CH_4 m^{-2} yr^{-1})	Reference
Fens	CO	C	FW		40.7			40.7	Chimner and Cooper (2003)
Wet Alpine Meadow	CO	C	FW	0.1			0.0	0.0	Neff *et al.* (1994)
Lake - Average	CO	C	FW	25.4			9.2	9.2	Smith and Lewis (1992)
Wetland - Average	CO	C	FW	28.3			10.3	10.3	Smith and Lewis (1992)
Nuphar Bed	CO	C	FW	202.1			73.6	73.6	Smith and Lewis (1992)
Tundra - Carex Meadow	CO	C	FW	2.8			1.0	1.0	West *et al.* (1999)
Tundra - Acomastylis Meadow	CO	C	FW	-0.5			-0.2	-0.2	West *et al.* (1999)
Tundra - Kobresia Meadow	CO	C	FW	-0.8			-0.3	-0.3	West *et al.* (1999)
Moist Grassy	CO	C	FW	6.1	1.9	0.32	2.2	1.9	Wickland *et al.* (1999)
Moist Mossy	CO	C	FW	1.5	0.5	0.33	0.5	0.5	Wickland *et al.* (1999)
Wetland	CO	C	FW		41.7			41.7	Wickland *et al.* (1999)
Hardwood Hammock	FL	C	FW	0.0			0.0	0.0	Bartlett *et al.* (1989)
Dwarf Cypress / Sawgrass	FL	C	FW	7.5			2.7	2.7	Bartlett *et al.* (1989)
Spikerush	FL	C	FW	29.4			10.7	10.7	Bartlett *et al.* (1989)
Sawgrass < 1m	FL	C	FW	38.8			14.1	14.1	Bartlett *et al.* (1989)
Sawgrass/Spkerush/Periphyton	FL	C	FW	45.1			16.4	16.4	Bartlett *et al.* (1989)
Swamp Forest	FL	C	FW	68.9			25.1	25.1	Bartlett *et al.* (1989)
Sawgrass > 1m	FL	C	FW	71.9			26.2	26.2	Bartlett *et al.* (1989)
Sawgrass	FL	C	FW	107.0			38.9	38.9	Burke *et al.* (1988)
Pond Open Water	FL	C	FW	624.0			227.1	227.1	Burke *et al.* (1988)
Everglades - Cladium	FL	C	FW	45.4			16.5	16.5	Chanton *et al.* (1993)
Everglades - Typha	FL	C	FW	142.9			52.0	52.0	Chanton *et al.* (1993)
Wet Prairie (Marl)	FL	C	FW	87.0			31.6	31.6	Happell *et al.* (1993)
Wet Prairie (Marl)	FL	C	FW	27.4			10.0	10.0	Happell *et al.* (1993)
Marsh (Marl)	FL	C	FW	30.0			10.9	10.9	Happell *et al.* (1993)
Marsh (Marl)	FL	C	FW	49.6			18.0	18.0	Happell *et al.* (1993)
Marsh (Peat)	FL	C	FW	45.4			16.5	16.5	Happell *et al.* (1993)

Marsh (Peat)	FL	C	FW	13.0			4.7	4.7	Happell *et al.* (1993)
Marsh (Peat)	FL	C	FW	163.6			59.6	59.6	Happell *et al.* (1993)
Marsh (Peat)	FL	C	FW	20.4			7.4	7.4	Happell *et al.* (1993)
Wet Prairie / Sawgrass	FL	C	FW	61.0			22.2	22.2	Harriss *et al.* (1988)
Wetland Forest	FL	C	FW	59.0			21.5	21.5	Harriss *et al.* (1988)
Cypress Swamp - Flowing Water	FL	C	FW	67.0			24.4	24.4	Harriss and Sebacher (1981)
Open Water Swamp	FL	C	FW	480.0			174.7	174.7	Schipper and Reddy (1994)
Waterlily Slough	FL	C	FW	91.0			33.1	33.1	Schipper and Reddy (1994)
Cypress Swamp - Deep Water	GA	C	FW	92.3			33.6	33.6	Harriss and Sebacher (1981)
Bottotmand Hardwoods/ Swamps	GA	C	FW		23.0			23.0	Pulliam (1993)
Swamp Forest	LA	C	FW	146.0			53.1	53.1	Alford *et al.* (1997)
Freshwater Marsh	LA	C	FW	251.0			91.4	91.4	Alford *et al.* (1997)
Fresh	LA	C	FW	587.0	213.0	0.36	213.6	213.0	DeLaune *et al.* (1983)
Fresh	LA	C	FW	49.0	18.7	0.38	17.8	18.7	DeLaune *et al.* (1983)
Sphagnum Bog	MD	C	FW	-1.1			-0.4	-0.4	Yavitt et al. (1990)
Bog	MI	C	FW	193.0			70.2	70.2	Shannon and White (1994)
Bog	MI	C	FW	28.0			10.2	10.2	Shannon and White (1994)
Beaver Meadow	MN	C	FW		2.3			2.3	Bridgham *et al.* (1995)
Open Bogs	MN	C	FW		0.0			0.0	Bridgham *et al.* (1995)
Bog (Forested Hummock)	MN	C	FW	10.0	3.5	0.35	3.6	3.5	Dise (1993)
Bog (Forested Hollow)	MN	C	FW	38.0	13.8	0.36	13.8	13.8	Dise (1993)
Fen Lagg	MN	C	FW	35.0	12.6	0.36	12.7	12.6	Dise (1993)
Bog (Open Bog)	MN	C	FW	118.0	43.1	0.37	42.9	43.1	Dise (1993)
Fen (Open Poor Fen)	MN	C	FW	180.0	65.7	0.37	65.5	65.7	Dise (1993)
Poor Fen	MN	C	FW	242.0			88.1	88.1	Dise and Verry (2001)
Sedge Meadow	MN	C	FW		11.7			11.7	Naiman *et al.* ((1991)
Submergent	MN	C	FW		14.4			14.4	Naiman *et al.* (1991)
Deep Water	MN	C	FW		0.5			0.5	Naiman *et al.* (1991)
Poor Fen	MN	T	FW		14.6			14.6	Shurpali and Verma (1998)
Submerged Tidal	NC	C, E	FW	144.8			52.7	52.7	Kelly *et al.* (1995)
Banks Tidal	NC	C, E	FW	20.1			7.3	7.3	Kelly *et al.* (1995)
Tidal Marsh	NC	C	FW	3.0	1.0	0.34	1.1	1.0	Megonigal and Schlesinger (2002)
Tidal Marsh	NC	C	FW	3.5	2.3	0.65	1.3	2.3	Megonigal and Schlesinger (2002)
Prairie Marsh	NE	T	FW		64.0			64.0	Kim *et al.* (1998)
Poor Fen	NH	C	FW	503.3	110.6	0.22	183.2	110.6	Carroll and Crill (1997)
Poor Fen	NH	C	FW		69.3			69.3	Frolking and Crill (1994)

Forested Peatland	NY	C	FW	0.6	0.2	0.37	0.2	0.2	Coles and Yavitt (2004)
Pools Forested Swamp	NY	C	FW	224.6	69.0	0.31	81.7	69.0	Miller *et al.* (1999)
Typha Marsh - Mineral Soils	NY	C	FW	344.4			125.3	125.3	Yavitt (1997)
Typha Marsh - Peat Soils	NY	C	FW	65.1			23.7	23.7	Yavitt (1997)
Typha Marsh - All soils	NY	C	FW	204.8			74.5	74.5	Yavitt (1997)
Cypress Swamp - Floodplain	SC	C	FW	9.9			3.6	3.6	Harriss and Sebacher (1981)
Swamp	VA	C	FW	470.3			171.2	171.2	Chanton *et al.* (1992)
Maple/gum Forested Swamp	VA	C	FW		0.5			0.5	Harriss *et al.* (1982)
Emergent Tidal Freshwater Marsh	VA	C	FW		96.2			96.2	Neubauer *et al.* (2000)
Oak Swamp (Bank Site)	VA	C	FW	117.0	43.7	0.37	42.6	43.7	Wilson *et al.* (1989)
Emergent Macrophytes (Peltandra)	VA	C	FW	155.0			56.4	56.4	Wilson *et al.* (1989)
Emergent Macrophytes (Smartweed)	VA	C	FW	83.0			30.2	30.2	Wilson *et al.* (1989)
Ash Tree Swamp	VA	C	FW	152.0			55.3	55.3	Wilson *et al.* (1989)
Bog	WA	C	FW	73.0			26.6	26.6	Lansdown *et al.* (1992)
Lowland Shrub and Forested Wetland	WI	T	FW		12.4			12.4	Werner *et al.* (2003)
Sphagnum Eriophorum (Poor Fen)	WV	C	FW	6.6			2.4	2.4	Yavitt *et al.* (1990)
Sphagnum Shrub (Fen)	WV	C	FW	0.1			0.0	0.0	Yavitt *et al.* (1990)
Polytrichum Shrub (Fen)	WV	C	FW	-0.1			0.0	0.0	Yavitt *et al.* (1990)
Sphagnum Forest	WV	C	FW	9.6			3.5	3.5	Yavitt *et al.* (1990)
Sedge Meadow	WV	C	FW	1.5			0.5	0.5	Yavitt *et al.* (1990)
Beaver Pond	WV	C	FW	250.0			91.0	91.0	Yavitt *et al.* (1990)
Low Gradient Headwater Stream	WV	C	FW	300.0			109.2	109.2	Yavitt *et al.* (1990)
Sphagnum-Eriophorum	WV	C	FW	52.1	19.0	0.37	18.9	19.0	Yavitt *et al.* (1993)
Polytrichum	WV	C	FW	41.1	15.0	0.37	15.0	15.0	Yavitt *et al.* (1993)
Sphagnum-Shurub	WV	C	FW	4.4	1.6	0.37	1.6	1.6	Yavitt *et al.* (1993)
Salt Marsh	DE	C	SW	0.5			0.2	0.2	Bartlett *et al.* (1985)
Red Mangroves	FL	C	SW	4.2			1.4	1.4	Bartlett *et al.* (1989)
Dwarf Red Mangrove	FL	C	SW	81.9			27.9	27.9	Bartlett *et al.* (1989)
High Marsh	FL	C	SW	3.9			1.3	1.3	Bartlett *et al.* (1985)
Salt Marsh	FL	C	SW	0.6			0.2	0.2	Bartlett *et al.* (1985)
Salt Water Mangroves	FL	C	SW	4.0			1.4	1.4	Harriss *et al.*(1988)
Salt Marsh	GA	C	SW	13.4			4.6	4.6	Bartlett *et al.* (1985)

Short Spartina Marsh - High Marsh	GA	C	SW	145.2	53.1	0.37	49.5	53.1	King and Wiebe (1978)
Mid Marsh	GA	C	SW	15.8	5.8	0.37	5.4	5.8	King and Wiebe (1978)
Tall Spartina Marsh - Low Marsh	GA	C	SW	1.2	0.4	0.34	0.4	0.4	King and Wiebe (1978)
Intermediate Marsh	LA	C	SW	**912**[b]					Alford *et al.* (1997)
Salt Marsh	LA	C	SW	15.7	5.7	0.36	5.4	5.7	DeLaune *et al.* (1983)
Brackish	LA	C	SW	267.0	97.0		91.1	97.0	DeLaune *et al.* (1983)
Salt Marsh	LA	C	SW	4.8	1.7	0.35	1.6	1.7	DeLaune *et al.* (1983)
Brackish	LA	C	SW	17.0	6.4	0.38	5.8	6.4	DeLaune *et al.* (1983)
Cypress Swamp - Floodplain	SC	C	SW	1.5			0.5	0.5	Bartlett *et al.* (1985)
Salt Marsh	SC	C	SW	0.4			0.1	0.1	Bartlett *et al.* (1985)
Salt Marsh	VA	C	SW	3.0	1.3	0.43	1.0	1.3	Bartlett *et al.* (1985)
Salt Marsh	VA	C	SW	5.0	1.2	0.24	1.7	1.2	Bartlett *et al.* (1985)
Salt Meadow	VA	C	SW	2.0	0.4	0.22	0.7	0.4	Bartlett *et al.* (1985)
Salt Marsh	VA	C	SW	-0.8			-0.3	-0.3	Bartlett *et al.* (1985)
Salt Marsh	VA	C	SW	1.5			0.5	0.5	Bartlett *et al.* (1985)
Salt Meadow	VA	C	SW	-1.9			-0.6	-0.6	Bartlett *et al.* (1985)
Tidal Salt Marsh	VA	C	SW	16.0	5.6	0.35	5.5	5.6	Bartlett *et al.* (1987)
Tidal Brackish Marsh	VA	C	SW	64.6	22.4	0.35	22.0	22.4	Bartlett *et al.* (1987)
Tidal Brackish/Fresh Marsh	VA	C	SW	53.5	18.2	0.34	18.2	18.2	Bartlett *et al.* (1987)
				FW Average =	32.1	0.36	38.6	36.0	
				FW n =	32	18	74	88	
				FW StError=	7.9	0.02	6.0	5.0	
				SW Average =	16.9	0.34	9.8	10.3	
				SW n =	13	12	25	25	

SW StError=	7.8	0.02		4.1	4.4

[a]C = chamber, T = tower, eddy covariance, E = ebulition measured separately.

[b]Outlier that was removed from further analysis.

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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 Mt C 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 Mt C yr^{-1}, 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 sequestration in 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 CO_2 from fossil fuel combustion, (2) alter plant and soil carbon cycling in converting wild lands to residential and urban land cover.
- A number of municipalities in Canada, Mexico, and the U.S. have made commitments to voluntary GHG 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 understandings of human and environmental systems in settled areas.

Activities in human settlements form the basis for much of North America's contribution to global 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.

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%. 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.

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 (Fig. 14-1).

Fig. 14-1. North America urban extents.

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 Mt (335–980 Mt C) with sequestration rates of 22.8 Mt C yr^{-1} (13.7–25.9 Mt C yr^{-1}) (Nowak and Crane, 2002). These estimates 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 t C ha^{-1}, and carbon sequestration ranged from 0.8 to 1.5 t C ha^{-1} yr^{-1}. 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 yr^{-1}. 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 ha^{-1} and 0.1 to 0.3 t $ha^{-1}yr^{-1}$, respectively, leading to estimates of 10 to 107 Mt C stored in urban trees in Mexico and 0.2 to 3.1 Mt C yr^{-1} sequestered. Estimates of historical trends are not available.

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 yr^{-1} (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 yr^{-1} (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).

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 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.

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.

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.

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).

Table 14-1. Increases in number of households and the total population of the United States, Canada, and Mexico between 1985 and 2000. (United Nations, 2002; United Nations Habitat, 2003).

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.

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 yr^{-1} (Imhoff *et al.*, 2004).

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 the components of urban energy balance 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 generally 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 semiarid 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 aboveground 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 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.

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 co-generation 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 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.

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.

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Table 14-1. Increases in number of households and the total population of the United States, Canada, and Mexico between 1985 and 2000. (United Nations, 2002; United Nations Habitat, 2003).

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

Figure 14-1. North America urban extents.

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Chapter 15. Coastal Oceans

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KEY FINDINGS

- The combustion of fossil fuels has increased CO_2 in the atmosphere, and by diffusion the oceans have absorbed an equivalent of 20-30% of the released CO_2 on an annual basis. The present annual uptake by the oceans of 1.3-2.3 Gt C 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 spatial variability. Existing data are insufficient, for example, to estimate the amount of anthropogenic carbon stored in the coastal regions of North America or to predict future scenarios.
- New air-sea flux observations reveal that on average, nearshore waters surrounding North America are neither a source nor a sink of CO_2 to the atmosphere. A small net source of CO_2 to the atmosphere of 19 Mt C yr^{-1} is estimated mostly from waters around the Gulf of Mexico and the Caribbean Sea, with a variation (standard deviation) around that number of $\pm$ 22 Mt C yr^{-1}. This equates to 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 CO_2 where terrestrially-derived organic material is decomposing, while reservoir systems may be storing carbon through sediment transport and burial.
- Options and measures for sequestration of carbon in the ocean include deep-sea injection of CO_2 and iron fertilization, although it is unresolved how important, feasible or acceptable any of these options might be for the North American region. Ocean carbon sequestration studies should be continued.

- Highly variable air-sea CO_2 fluxes in coastal areas may introduce errors in North American CO_2 fluxes calculated by atmospheric inversion methods. Reducing these errors will require ocean observatories utilizing fixed and mobile platforms with instrumentation to measure critical stocks and fluxes as part of coordinated national and international research programs. Ocean carbon sequestration studies should also be continued.

INVENTORIES (STOCKS AND FLUXES, QUANTIFICATION)

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 surrounding North America and considers how the underlying processes affect the air-sea fluxes. Aquatic stocks of living carbon are small relative to stocks in the terrestrial environments, 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 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 380 ppm. This increase in atmospheric concentrations is driving more CO_2 into the ocean with the present net air-sea CO_2 flux well constrained to about 1,800 ± 500 Mt C [1 Mt = one million (10^6) metric tons] or 1.8 ± 0.5 Gt C yr^{-1} [1 Gt = one billion (10^9) metric tons] from the atmosphere into the ocean (Figure 15-1 and Table 15-1) (See Chapter 2 for a description of how ocean carbon fluxes relate to the global carbon cycle). The uptake of this anthropogenically-driven 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 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.

Table 15-1. Climatological mean distribution of the net air-sea CO_2 flux (in Gt C yr^{-1}) over the global ocean (excluding coastal areas) in reference year 1995. Positive values indicate a source for atmospheric CO_2, and negative values indicate a sink. 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) for calculating the air-sea flux. The flux, however, depends on the wind speed and air-sea gas transfer rate parameterizations used, and varies by about ± 30% (Takahashi *et al*., 2002). 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 yr^{-1}

Figure 15-1. Global distribution of air-sea CO_2 flux. The map yields a total annual air-to-sea flux of 1.5 Gt C yr^{-1}. The white line represents zero flux and separates sources (yellow and red) and sinks (blue and purple). Negative values indicate that the ocean is a CO_2 sink for the atmosphere. The sources are primarily in the tropics (yellow and red) with a few areas of deep mixing at high latitudes. Updated from Takahashi *et al.* (2002).

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, 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).

Figure 15-2. In the top panel, mean air/sea CO_2 flux is calculated from shipboard measurements on a line perpendicular to the central California coast. 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 towards the California Current and into Monterey Bay. Phytoplankton growth and photosynthesis draw down CO_2 in seawater to low levels in the upwelled water. 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 seawater CO_2 is higher than the atmosphere, CO_2 is driven into the atmosphere. Conversely, where the level of surface CO_2 is lower than that of atmospheric CO_2, CO_2 is driven from the atmosphere into the ocean. The net sea/air flux on this spatial scale is near zero. DIC = dissolved inorganic carbon; POC = particulate organic carbon. Updated from Pennington *et al.* (in press).

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 1,000 Mt C yr^{-1}, about 38% of it as dissolved CO_2 (or 384 Mt C yr^{-1}), 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 yr^{-1} (Sarmiento and Sundquist, 1992). The Mississippi River, the seventh-largest in freshwater discharge in the world, delivers about 13 Mt C yr^{-1} as dissolved CO_2 (Cai, 2003). Organic matter in continental 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, 1997). 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 1,200 Mt C yr^{-1} and a large CO_2 sink of 900 Mt C yr^{-1} for the atmosphere. In contrast, Smith and Hollibaugh (1993) estimated a biological pump of about 200 Mt C yr^{-1} and concluded that the coastal oceans are a weak CO_2 sink of 100 Mt C yr^{-1}, 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).

North American Coastal Carbon

Two important types of North American coastal ocean environments can be identified: (1) river-dominated coastal 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 1,000 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.

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). Negative values indicate that the ocean is a CO_2 sink for the atmosphere.

Synthesis of Available North American Air-Sea Coastal CO_2 Fluxes

A large data set consisting of 550,000 measurements of the partial pressure of CO_2 (pCO_2) in surface waters has been assembled and analyzed (Figure 15-3; see Appendix 15A for details). pCO_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. CO_2 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 increasing with temperature and decreasing with photosynthesis. 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 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.

Figure 15-3. (A). Distribution of coastal CO_2 partial pressure measurements made between 1979 and 2004. (B). The distribution of the net air-sea CO_2 flux over 1° × 1° pixel areas (N-S 100 km, E-W 80 km) around North America. The flux (grams of carbon per square meter 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 sink. 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/CO2).

Figure 15-4. Estimated air-sea CO_2 fluxes (grams of carbon per square meter 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 (ΔpCO_2) values were calculated from the weekly mean atmospheric CO_2

concentrations in the GLOBALVIEW-CO2 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). Negative values indicate that the ocean is a CO_2 sink for the atmosphere. The ± uncertainties represent one standard deviation.

Table 15-3. Climatological mean annual air-sea CO_2 flux (grams of carbon per square meter 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.

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.

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 distributions of temperature, salinity, and pCO_2 in surface waters based on measurements made in for 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).

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 pCO_2 in surface waters during July–September 2005. The Columbia River plume (~46°N) and the upwelling of deep waters off the 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 (the fluxes are reported in grams of carbon per square meter per year so they can be compared to values throughout the chapter). Seawater is a CO_2 source for the atmosphere during the summer upwelling events, but biological uptake reduces levels very rapidly. The rapid fluctuations seen in (B) 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 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).

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.

In the coastal zone very high pCO_2 values (up to 2,600 µatm) are observed occasionally in areas within 10 km offshore of the barrier islands (see small red dots off the coasts of Georgia and Carolinas in Figures 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.

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.

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.

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 yr^{-1} (Table 15-4). Most of the flux (14 ± 9 Mt C yr^{-1}) occurs in the Gulf of Mexico and Caribbean Sea. The open oceans are a net CO_2 sink on an annual average (Table 15-4; Takahashi *et al.*, 2004). 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. 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.

Table 15-4. Areas (km^2) and mean annual air-sea CO_2 flux ($Mt\ C\ yr^{-1}$) over four ocean regions surrounding North America. Negative values indicate that the ocean is a CO_2 sink for the atmosphere. 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.

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 2,000 Mt C y^{-1} (or 2 Gt C yr^{-1}), 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 m^{-2} yr^{-1} or 0.7 g C m^{-2} day^{-1} 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 U.S. 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-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.

OPTIONS AND MEASURES

Two options for ocean carbon sequestration have been considered: (1) deep-sea injection of CO_2 (Brewer, 2003) and (2) ocean iron fertilization (Martin, 1990). The first might be viable in North American coastal waters, although cost and potential biological side effects are unresolved issues. The largest potential for iron fertilization resides in the equatorial Pacific and the Southern Ocean, although it could be considered for the open ocean waters of the Gulf of Alaska and offshore waters of coastal upwelling systems. However, there is still disagreement over how much carbon would be sequestered (Bakker *et al.*, 2001; Boyd *et al.*, 2000; Coale *et al.*, 2004; Gervais *et al.*, 2002) and what the potential side effects would be (Chisholm *et al.*, 2001).

R&D NEEDS VIS A 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, 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). 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.

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Table 15-1. Climatological mean distribution of the net air-sea CO_2 flux (in Gt C yr^{-1}) 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 $(\text{wind speed})^2$ 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 yr^{-1}

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 (10^6 km^2)	152.0	74.6	53.0	41.1	320.7
% of area	47	23	17	13	100

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 ($mol/m^{-2}\ yr^{-1}$)	Annual average ($mol\ m^{-2}\ yr^{-1}$)	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 meas.	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 meas.	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 = no data available

Table 15-3. Climatological mean annual air-sea CO_2 flux (g C m^{-2} yr^{-1}) 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		First offshore		Second offshore		Third offshore		Open ocean	
	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

Table 15-4. Areas (km^2) and mean annual air-sea CO_2 flux (Mt C yr^{-1}) 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 (km^2)					Mean air-sea CO_2 flux (10^{12} grams or Mt C yr^{-1})				
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

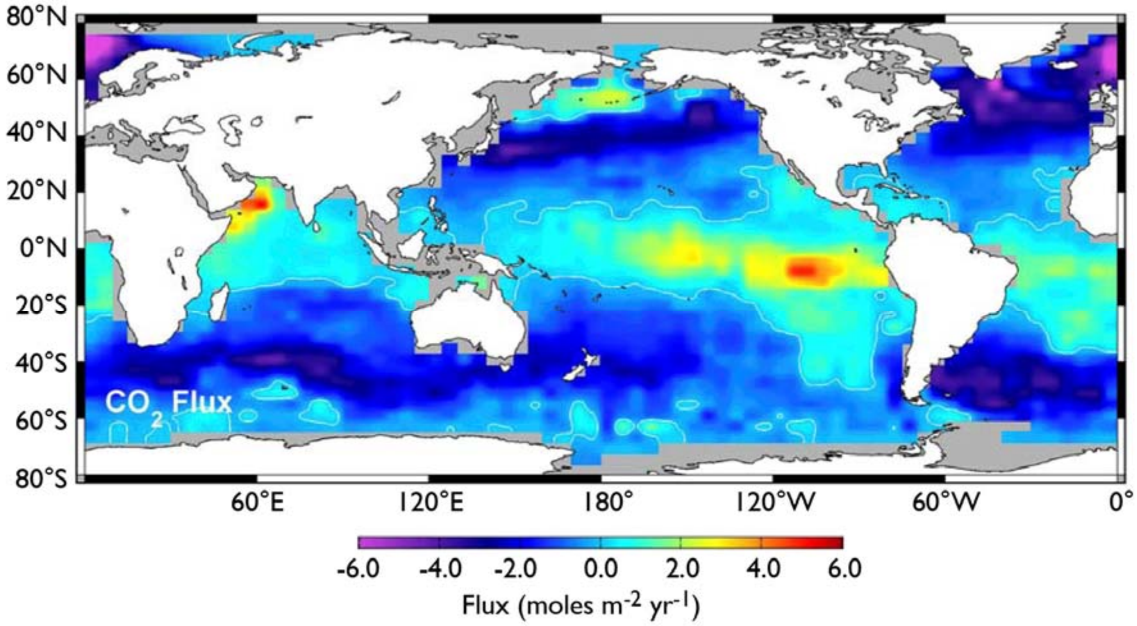


Figure 15-1. Global distribution of air-sea CO_2 flux. The white line represents zero flux and separates sources and sinks. The sources are primarily in the tropics (yellow and red) with a few areas of deep mixing at high latitudes. Updated from Takahashi *et al.* (2002).

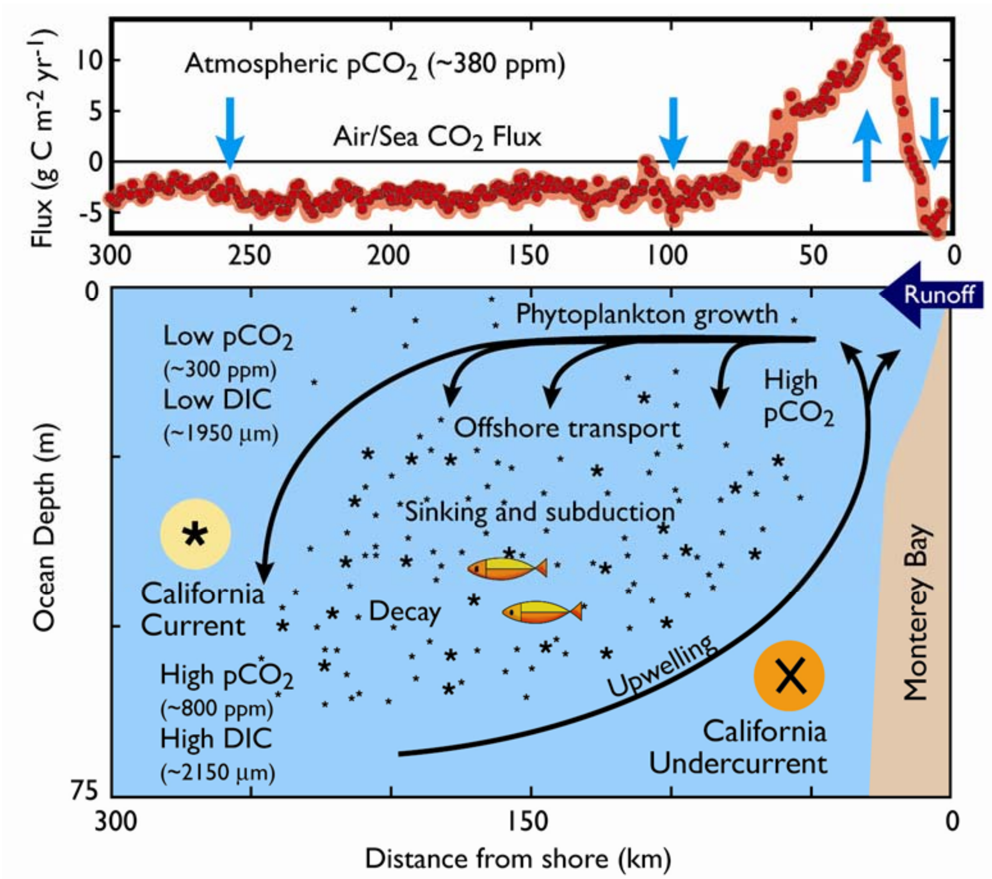
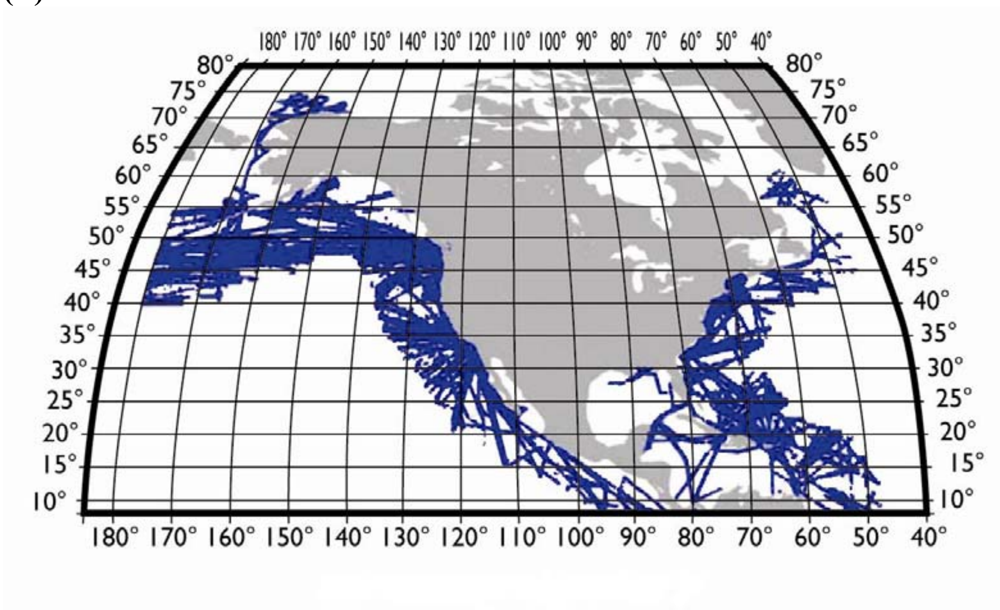
1
2


Figure 15-2. In the top panel, mean air-sea CO_2 flux is calculated from shipboard measurements on a line perpendicular to the central California coast. 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 atmosphericCO_2, diffusion drives CO_2 into the ocean. The net air-sea flux on this spatial scale is near zero. DIC = dissolved inorganic carbon; POC = particulate organic carbon. Updated from Pennington et al. (in press).

(A)



(B)

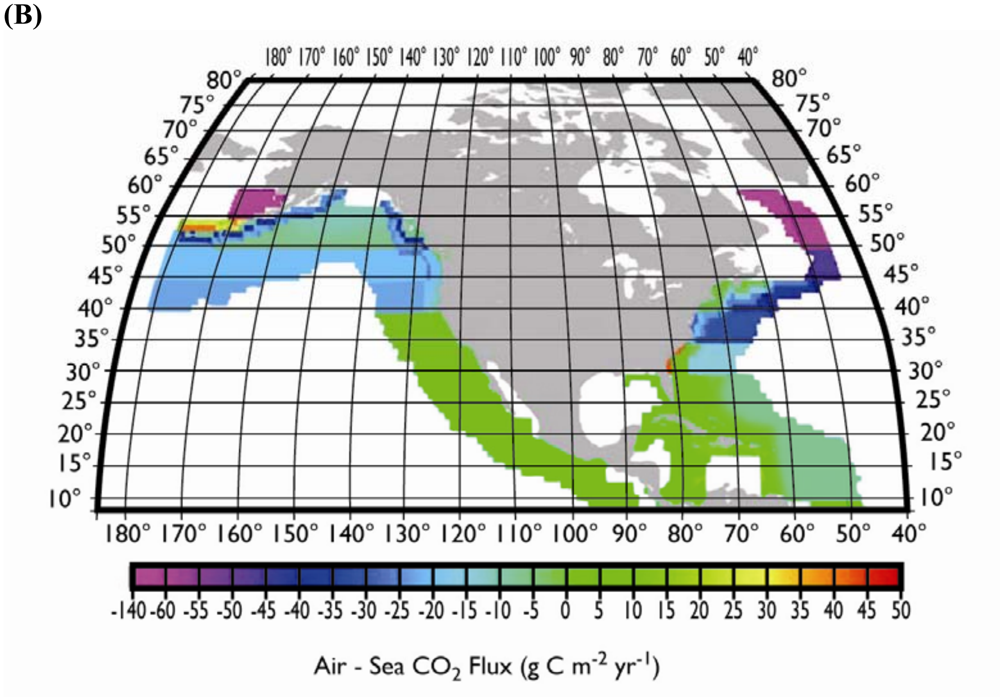
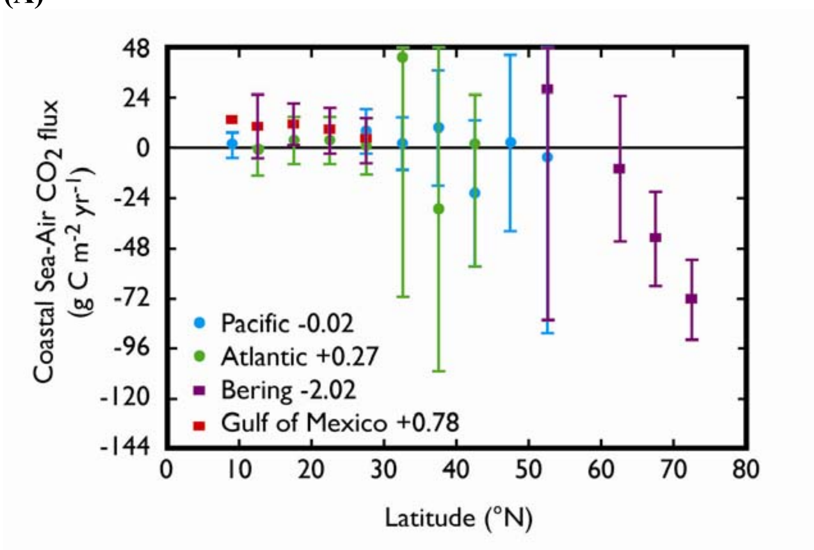


Figure 15-3. (A). Distribution of coastal CO_2 partial pressure measurements made between 1979 and 2004. (B). The distribution of the net air-sea CO_2 flux over 1° × 1° pixel areas (N-S 100 km, E-W 80 km) around North America. The flux (grams of carbon per square meter per year) represents the climatological 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 sink. 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/CO2).

(A)



(B)

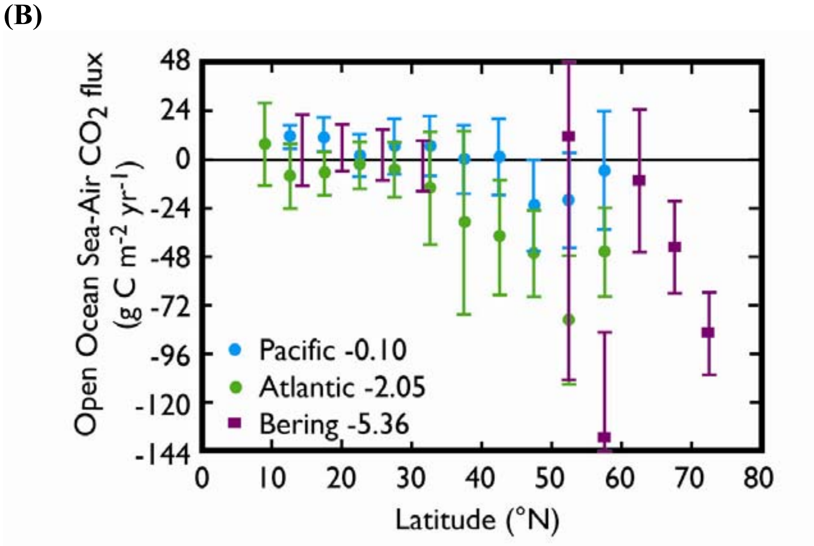
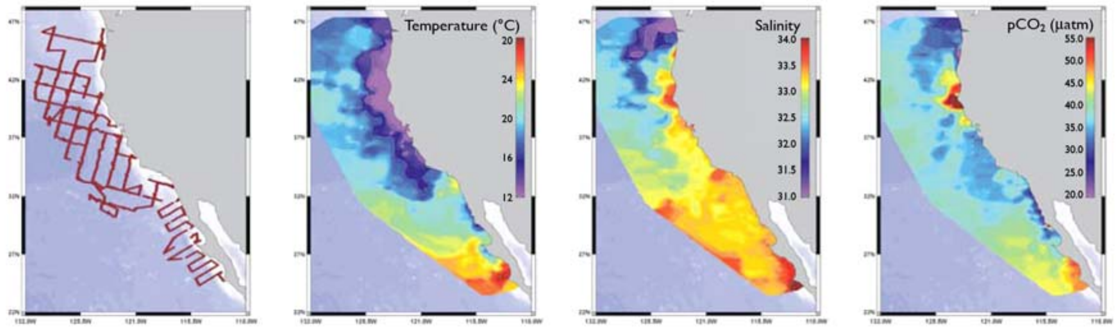


Figure 15-4. Estimated air-sea CO_2 fluxes (grams of carbon per square meter per year) from 550,000 seawater CO_2 partial pressure (pCO_2) observations made from 1979 to 2004 in ocean waters surrounding the North American continent. (A) Waters within one degree (about 80 km) of the coast and (B) open ocean waters between 300 and 900 km from the shore (see Figure 15-3B). The annual mean air-sea pCO_2 difference (delta pCO_2) values were calculated from the weekly mean atmospheric CO_2 concentrations in the GLOBALVIEW-CO2 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.

(A)



(B)

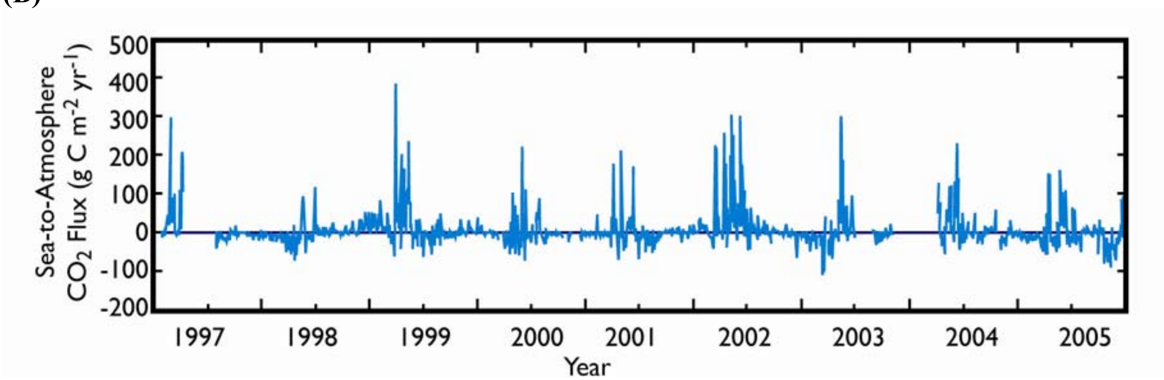


Figure 15-5. Time-space variability of coastal waters off the west coast of North America. (A) Quasi-synoptic distribution of the temperature, salinity, and pCO_2 in surface waters during July–September 2005. The Columbia River plume (~46°N) and the upwelling of deep waters off the Cape Mendocino (~40°N) 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.

Appendix 15A
Database and Methods

A database for pCO_2, temperature and salinity in surface waters within about 1,000 km from the shore of the North American continent has been assembled. About 550,000 seawater pCO_2 observations were made from 1979 to 2004 by the authors and collaborators of Chapter 15. The pCO_2 data have been obtained by a method using an infrared gas analyzer or gas-chromatograph for the determination of CO_2 concentrations in a carrier gas equilibrated with seawater at a known temperature and total pressure. The precision of pCO_2 measurements has been estimated to be about ± 0.7% on average. The quality-controlled data are archived at www.ldeo.columbia.edu/res/pi/CO2.

The zonal distribution of the surface water pCO_2, sea surface temperature (SST), and salinity data shows that the greatest variability is confined within 300 km from the shores of both the Atlantic and Pacific. Observations made in various years were combined into a single year and were averaged into 1° × 1° pixels (approximately N-S 100 km by E-W 80 km) for the analysis. Accordingly, the results represent a climatological mean condition over the past 25 years. Finer resolutions (10 × 10 km) may be desirable for some areas close to shore because of outflow of estuarine and river waters and upwelling. However, for this study, which is aimed at a broad picture of waters surrounding the continent, the fine scale measurements have been incorporated into the 1° × 1° pixels. In addition, data with salinities of less than 16.0 are considered to be inland waters and have been excluded from the analysis.

Climatological monthly and annual mean values for pCO_2 in each zone where computed first. Then the air-sea pCO_2 difference, which represents the thermodynamic driving potential for air-sea CO_2 gas transfer, was estimated using the atmospheric CO_2 concentration data. Finally, the net air-sea CO_2 flux was computed using transfer coefficients estimated on the basis of climatological mean monthly wind speeds using the (wind speed)^2 formulation of Wanninkhof (1992). The transfer coefficient depends on the state of turbulence above and below the air-sea interface and is commonly parameterized as a function of wind speeds (corrected to 10 m above the sea surface). However, selection of wind data is problematic because wind speeds vary with the time scale (hourly, diurnal, or seasonal). For example, fluxes calculated for the South Atlantic Bight from 6-h mean wind speeds in the NCEP/NCAR version 2 file (1° × 1° mean) were lower than those estimated using the monthly mean. This discrepancy suggests that ships used commonly for coastal carbon studies tend to be small and hence are rarely at sea under high wind conditions, so observations are biased toward lower winds. Taking into account that the observations have been made infrequently over multiple years, the gas transfer coefficients estimated from climatological mean monthly wind speeds may be more representative. The Schmidt number is computed using measured SST and climatological mean salinity (Da Silva *et al*. 1994). The flux values in a given month are then averaged to yield a climatological mean flux (and standard deviation) for each month. This procedure assumes implicitly that the seawater pCO_2 changes at much slower rates in space and time than the wind speed and that the seawater pCO_2 does not correlate with the wind speed.

REFERENCES

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Wanninkhof, R., 1992: Relationship between wind speed and gas exchange. *Journal of Geophysical Research*, **97**, 7373–7382.

GLOSSARY

anthropogenic	Human-induced; produced by or resulting from human activity
apparent consumption	The amount or quantity expressed by the following formula: production + imports – exports +/– changes in stocks
biomass	The mass of living organic matter (plant and animal) in an ecosystem. Biomass also refers to organic matter (living and dead) available on a renewable basis for use as a fuel. Biomass includes trees and plants (both terrestrial and aquatic), agricultural crops and wastes, wood and wood wastes, forest and mill residues, animal wastes, livestock operation residues, and some municipal and industrial wastes
carbon sequestration	The chemical conversion or physical restraining of carbon or carbon-bearing molecules so as to prevent their direct entry into the atmosphere or environment for some period of time. Often used narrowly to refer to increasing the carbon content of carbon pools in the biosphere and distinguished from physical or chemical collection of carbon followed by injection into geologic reservoirs, which is generally referred to as “carbon capture and storage.”
carbon cycle	The combination of the many different physical, chemical and biological processes that transfer carbon between storage pools or reservoirs in the atmosphere, plants, soils, freshwater systems, ocean and geological sediments
carbon equivalent	The amount of carbon in the form of carbon dioxide that would produce the same effect on the radiative balance of the Earth’s climate system. Applicable in this report to greenhouse gases such as methane (CH_4).
carbon intensity	The relative amount of carbon emitted per unit of energy or fuels consumed
CO_2 equivalent	The amount of carbon dioxide that would produce the same effect on the radiative balance of the Earth’s climate system as another greenhouse gas, such as methane (CH_4).
CO_2 fertilization	The phenomenon in which plant growth increases (and agricultural crop yields increase) due to the increased rates of photosynthesis of plant species in response to elevated concentrations of CO_2 in the atmosphere

decarbonization	Reduction in the use of carbon-based energy sources as a proportion of total energy supplies or increased use of carbon-based fuels with lower values of carbon content per unit of energy content.
dry climates	Climates where the ratio of mean annual precipitation to potential evapotranspiration is less than 1.0
ecosystem	A naturally occurring unit consisting of all biota (e.g., plants, animals, and microbes) in a given area, and the associated abiotic environments with which they interact through nutrient cycling and energy flows
energy intensity	The relative amount or ratio of the consumption of energy to the resulting amount of output, service or activity (i.e., expressed as energy per unit of output)
fossil fuels	Fuels such as coal, petroleum, and natural gas derived from the chemical and physical transformation (fossilization) of the remains of plants and animals that lived during the Carboniferous Period 360–286 million years ago
global warming potential (GWP)	A factor describing the radiative forcing impact (e.g., warming of the atmosphere) of one unit mass of a given greenhouse gas relative to the warming caused by a similar mass of carbon dioxide. Methane (CH_4), for example, has a GWP of 23.
greenhouse gases (GHGs)	Certain gases (including water vapor, carbon dioxide, methane, nitrous oxide, and halocarbons) which are "radiatively active" in the atmosphere in that they trap or absorb heat radiated from the earth's surface (i.e., heat that would otherwise be lost into space) thereby contributing to the potential warming of the air in the lower levels of the earth's atmosphere
measures	Actions or activities designed to reduce carbon emissions or otherwise manage the carbon budget
mitigation	A human intervention to reduce the sources of or to enhance the sinks of greenhouse gases
North America	The combined land area of Canada, the United States of America, and Mexico and their coastal waters
ocean acidification	The phenomenon in which the pH of the oceans becomes more acidic due to increased levels of CO_2 in the atmosphere which, in turn, increase the amount of dissolved CO_2 in sea water

option	A choice among a set of possible measures (*q.v.*) or alternatives
peatlands	Areas characterized as having an organic layer thickness of at least 30 cm (note, the current U.S. and Canadian soil taxonomies specify a minimum thickness of 40 cm)
permafrost	Soils or rocks that remain below 0° C for at least two consecutive years
pool/reservoir	Any natural region or zone, or any artificial holding area, containing an accumulation of carbon or carbon-bearing compounds or having the potential to accumulate such substances
private sector	Those entities, functions, and interest areas that are not directly associated with the "public sector" (*q.v.*); for example, carbon-related industry (including energy); transportation, agriculture, and forestry sectors; and climate policy and carbon management interest groups
public sector	The collective set of entities directly associated with the functions of federal, state, and/or local governments
sink	In general, any process, activity or mechanism which removes a greenhouse gas or a precursor of a greenhouse gas or aerosol from the atmosphere. In this report, a sink is any regime or pool in which the amount of carbon is increasing (i.e., is being accumulated or stored).
source	In general, any process, activity or mechanism which releases a greenhouse gas or a precursor of a greenhouse gas or aerosol into the atmosphere. In this report, a source is any regime or pool in which the amount of carbon is decreasing (i.e., is being released or emitted).
stocks	The amount or quantity contained in the inventory of a pool or reservoir
temperate zones	Regions of the earth's surface located above 30º latitude and below 66.5º latitude
trend	A systematic change over time
tropical zones	Regions located between the earth's equator and 30º latitude (this area includes subtropical regions)

uncertainty	A term used to describe the range of possible values around a best estimate, sometimes expressed in terms of probability or likelihood
wet climates	Climates where the ratio of mean annual precipitation to potential evapotranspiration is greater than 1.0
wetlands	Areas characterized by the presence of waterlogged conditions in the upper soil profile during at least part of the growing season and by plant species and soil conditions that reflect these hydrologic conditions

ACRONYMS AND ABBREVIATIONS

µatm	microatmosphere (a measure of pressure)
ACEEE	American Council for an Energy-Efficient Economy
CAFE	Corporate Average Fuel Economy
CAIT	Climate Analysis Indicators Tool
CAST	Council for Agricultural Science and Technology
CBO	U.S. Congressional Budget Office
CCSP	U.S. Climate Change Science Program
CCTP	Climate Change Technology Program
CDIAC	Carbon Dioxide Information Analysis Center
CEC	California Energy Commission
CH_4	methane
CIEEDAC	Canadian Industrial Energy End-Use Data and Analysis Centre
CO	carbon monoxide
CO_2	carbon dioxide
CO_2e	carbon dioxide equivalent
CO_3	carbonate
COP	Conference of Parties
DOC	dissolved organic carbon
DOE	U.S. Department of Energy
DOT	U.S. Department of Transportation
EIA	Energy Information Administration
EPA	U.S. Environmental Protection Agency
ESCOs	energy services companies
FAO	Food and Agriculture Organization
FWMS	freshwater mineral-soil
g	gram
GAO	U.S. Government Accountability Office
GDP	gross domestic product
GHG	greenhouse gas
Gt C	gigatons of carbon (billions of metric tons; i.e., petagrams)
GWP	global warming potential

ha	hectare
HCO_3	bicarbonate
ICLEI	International Council for Local Environmental Initiatives (now known as International Governments for Local Sustainability)
IOOS	Integrated Ocean Observing System
IPCC	Intergovernmental Panel on Climate Change
IWG	Interlaboratory Working Group
kg	kilogram
km	kilometer
L	liter
LEED	Leadership in Energy and Environment Design
m	meter
MAP	mean annual precipitation
mpg	miles per gallon
Mt C	megatons of carbon (millions of metric tons; i.e., teragrams)
N_2O	nitrous oxide (also, dinitrogen oxide)
NACP	North American Carbon Program
NAO	North Atlantic oscillation
NAS	U.S. National Academy of Sciences
NASA	National Aeronautics and Space Administration
NATS	North American Transportation Statistics
NCAR	National Center for Atmospheric Research
NCEP	National Centers for Environmental Prediction; National Commission on Energy Policy
NEE	net ecosystem exchange
NEP	net ecosystem productivity
NGO	non-governmental organization
NO_2	nitric oxide (also, nitrogen dioxide)
NOAA	National Oceanic and Atmospheric Administration
NO_x	oxides of nitrogen
NPP	net primary productivity
NRC	National Research Council
NRCS	National Resources Conservation Service

NSF	National Science Foundation
NWI	National Wetland Inventory
OCCC	Ocean Carbon and Climate Change
pCO_2	partial pressure of carbon dioxide
PDO	Pacific decadal oscillation
PET	potential evapotranspiration
PJ	petajoules
ppm	parts per million
PPP	purchasing power parity
RGGI	Regional Greenhouse Gas Initiative
SAP	Synthesis and Assessment Product
SBSTA	Subsidiary Body for Scientific and Technological Advice
SOCCR	State of the Carbon Cycle Report
UNFCCC	United Nations Framework Convention on Climate Change
USDA	U.S. Department of Agriculture
VOCs	volatile organic compounds
WBCSD	World Business Council for Sustainable Development
yr	year

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