*DRAFT*

[*For Internal Expert/Peer Review*]

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

## PART III: LAND AND WATER SYSTEMS



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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 and/or advancing the frontier of knowledge.

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*

**Lead Authors: SOCCR Coordinating Team**

**Coordinating Team Members: Lisa Dilling[1], Anthony King[2], David Fairman[3], Richard Houghton[4], Gregg Marland[2], Adam Rose[5], Thomas Wilbanks[2], and Gregory Zimmerman[2]**

[1]University of Colorado, [2]Oak Ridge National Laboratory, [3]Consensus Building Institute, Inc., [4]Woods Hole Research Center, [5]The Pennsylvania State University

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 transferred large amounts of carbon from geological reservoirs to the atmosphere, primarily as the combustion product carbon dioxide ($CO_2$). Clearing of forests and plowing of grasslands for agriculture has also transferred carbon from plants and soils to the atmosphere as $CO_2$. The combined rate of transfer is far larger than can be balanced by the biological and geological processes which naturally remove $CO_2$ from the atmosphere and store the carbon in various terrestrial and marine reservoirs as part of the earth's carbon cycle. The result is a "piling up" of $CO_2$ in the atmosphere, and a dramatic increase in atmospheric $CO_2$ concentration. The atmospheric concentration of carbon dioxide 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 almost a third (32%) of the world's entire fossil fuel emissions in 2002 and more than quarter (27%) in 2003. North America is incontrovertibly a major source of atmospheric $CO_2$.

North America may also be an important sink. Many lines of scientific evidence point to the vegetation and soils of the Northern Hemisphere as a net sink for atmospheric carbon, removing $CO_2$ from the atmosphere and to some degree mitigating fossil-fuel sources. The contribution of North America to that sink is, however, highly uncertain. The mechanisms that might be responsible for a North American sink, including forest regrowth and sequestration in agricultural soils, are reasonably well known. However, their relative contributions, their magnitudes, and their future fates 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 about the North American carbon budget relevant to carbon management policy options and a broad range 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. The questions were identified through early and continuing dialogue with these stakeholder groups, including scientists, decision makers 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.

The questions and the answers provided by this Report are summarized below.

## What is the carbon cycle and why should we care?

The carbon cycle is the combination of 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. We are familiar with the cycling of water in precipitation, runoff, stream flow, and evaporation. Water delivered from the atmosphere in rain and snow evaporates from land, freshwater rivers and lakes, and the ocean, and condenses in the atmosphere to form clouds. These clouds generate rain or snow, and the cycle begins anew. Similarly, carbon cycles through the atmosphere, land and water, and over long periods of time, through the earth's rocky crust itself.

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 source of energy for our modern, post-industrial societies. Today, the cycling of carbon among atmosphere, land, freshwater and marine reservoirs over periods of years and decades determines the balance of the carbon budget observed at any particular time: how much carbon is stored in a reservoir, how much is coming in, how much is going out, and how fast the carbon pool is changing. Currently the global carbon budget is in imbalance, with carbon building up in the atmosphere as carbon dioxide, and human use of coal, petroleum and natural gas to fuel economies is responsible.

If vast quantities of water had been trapped underground for millions of years and then, in recent decades, 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. This has not happened for water, but it has happened for carbon. The 19th and especially 20th centuries saw a dramatic rise in the combustion of "fossil fuels," releasing into the atmosphere over *decades* quantities of carbon that had been stored in the earth system over thousands of *millennia*. During this same time, forests that had once absorbed very large quantities of carbon dioxide were being converted to agricultural cropland with carbon released to the atmosphere during clearing.

It is not surprising, then, that concentrations of carbon dioxide and other carbon compounds in the earth's atmosphere, such as methane, are increasing. This facts, together with patterns of human activity that are likely to continue trends in fossil fuel use and deforestation, raise concerns about imbalances in the carbon cycle and their implications.

Climate change is an obvious concern. Atmospheric carbon dioxide is the largest single forcing agent of climate change. However, the consequences of increasing atmospheric carbon dioxide extend beyond climate change alone. It is increasingly evident that elevated atmospheric carbon dioxide concentrations are responsible for increased acidification of the surface ocean, 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 that of climate change to care about the carbon cycle and the accumulation of carbon dioxide in the atmosphere.

Invariably, 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 and concentrations of carbon dioxide in the atmosphere. That management involves both reducing sources of atmospheric carbon dioxide like the combustion of fossil fuels, or enhancing sinks such as uptake and storage or sequestration in vegetation and soils. In either case, formulation of options by decision makers and successful management of the earth's carbon budget requires solid scientific understanding of the carbon cycle.

## How do North American carbon sources and sinks relate to the global carbon cycle?

North America is responsible for approximately 27% of the carbon dioxide emissions produced globally by fossil fuel combustion. The United States accounts for 86% of the North American total and approximately one quarter of the global total. In recent years, extraction of fossil fuels and their conversion into energy delivery forms (solid, liquid, gas, and electric) in North America released on the order of 2800 million metric tons (Mt) of $CO_2$ per year to the atmosphere, approximately 10% of total global emissions in 2003. Electricity generation is responsible for most (90-95%) of North America's energy extraction and conversion emissions. The transportation sector of North America released 2151 Mt $CO_2$ into the atmosphere in 2003, 40% of the total carbon emissions from worldwide transportation activity and about 9% of total global $CO_2$ emissions. The buildings sector in North America is responsible for the annual emission of 2712 Mt $CO_2$ or 9% of global fossil fuel emissions. U.S. buildings alone are responsible for more $CO_2$ emissions than total $CO_2$ emissions of any country in the world, except China. Most—approximately 64%—of the emissions from the building sector of North America are associated with the production of electricity used in buildings. Emissions from the North American building sector, excluding electricity, were about 4% of global total $CO_2$ emissions in 2003. In 2002, North American industry (excluding fossil fuel mining and processing) was responsible for the release of 826 Mt $CO_2$ into the atmosphere, or 16% of the 5200 Mt $CO_2$ emissions from global industry.

The carbon budget of North America is dominated by the fossil fuel emissions source; however, the vegetation and soils of North America and the surrounding coastal oceans are also a substantial net sink. Approximately 30% of North American fossil fuel emissions are offset by a smaller sink of 2170 Mt $CO_2$ per year. Most (60%) of that sink is caused by relatively young, growing forests in the United States and Canada which have re-colonized land formerly cleared of forests for agricultural use in past centuries. The *global* terrestrial sink is quite uncertain, estimated as somewhere in the range of 2200 to 8433 Mt $CO_2$ per year during the 1990s, with the actual sink likely near 4000 Mt $CO_2$ per year. Thus, North America is probably responsible for at least half of the global terrestrial sink, but could account for as little as a quarter to nearly all of it.

Both as a source and a sink, North America is a major, even dominant component of the global carbon cycle. And it is clear that the North American carbon budget of the next few decades will continue to be dominated by the large sources from fossil fuel emissions as the trends responsible for current emissions continue into at least the near future. Consequently, the global carbon cycle will continue to be dominated by a large fossil fuel source 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 important contribution 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.

Because North America's carbon budget is such a substantial part of the global carbon budget, options and measures taken to manage the North American carbon budget will have important global consequences. North America has many opportunities for decreasing emissions, including changes to the energy system, increasing energy efficiency, 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 with international cooperation. This provides maximum coverage of $CO_2$ emissions and carbon sinks. It also allows 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?

### *The Sources*

The primary source of carbon in North America is the release of $CO_2$ during the combustion of fossil fuels. The North American fossil fuel source is three times larger than the net sink of land and water systems and dominates the net carbon balance of the continent. Fossil fuel carbon emissions in the United States, Canada and Mexico totaled 1856 Mt C (6805 Mt $CO_2$) in 2003 and have increased at an average rate of approximately 1% per year for the last 30 years. The United States was responsible for 85% of North America's fossil fuel emissions in 2003, Canada for 9% and Mexico 6%.

U.S. emissions dominate North American emissions and continue to grow at close to the North American average rate of ~1.0% per year, but U.S. per capita emissions have been roughly constant for the past 30 years, while the carbon intensity of the U.S. economy has decreased at a rate of ~2% per year. U.S. 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). Also, a large part of the decline in the carbon intensity of the U.S. economy was 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 carbon emissions per dollar of economic activity only 15% that of manufacturing. This implies that emissions growth is essentially decoupled from economic growth. Also, 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.

Electricity generation is the single largest contributor to the North American fossil-fuel source, accounting for approximately 40% of the total North American fossil fuel source. Again, U.S. emissions dominate. In 2003, electricity generation in the United States alone released 2409 Mt $CO_2$ to the atmosphere, 35% of total North American fossil fuel emissions for that year.

The transportation sector of North America released 2120 Mt $CO_2$ into the atmosphere in 2003, 31% of total North American emissions. Most (87%) of that source is from the United States. Transportation energy use in North America and the associated $CO_2$ emissions have grown substantially and relatively steadily over the past forty years. Growth has been most rapid in Mexico, the country most dependent upon road transport. Carbon emissions from the transportation sector 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 are 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.

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). The trend in the buildings sector over the last decade has been towards growth, with emissions from energy use in buildings in the United States and Canada (including the use of natural gas, wood, and other fuels as well as electricity) increasing 30% since 1990, corresponding to an annual growth rate of 2.1%. In the United States, the major drivers of energy consumption growth in the buildings sector are growth in commercial floor space and increase in the size of the average home. Carbon emissions from buildings will grow with energy consumption, which in turn will increase with population and income. Furthermore, the shift from large extended- to nuclear-family and single-occupant households means an increase in the number of households per unit population—each with its own heating and cooling systems and electrical appliances. Certain electrical appliances (such as space cooling/conditioning equipment) once considered a luxury are now becoming commonplace. 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 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 fossil fuel source. 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 toward less energy-intensive industries and dissemination of more energy efficient equipment kept the rate of energy demand growth lower than industrial GDP growth.

### *The Sinks*

Approximately 30% of North American fossil fuel emissions are offset by a natural sink of 592 Mt C per year caused by a variety of factors, including forest regrowth, fire suppression, and agricultural soil conservation. The sink currently absorbs 506 Mt C per year in the United States and 134 Mt C per year in Canada. Mexican ecosystems create a net source of 48 Mt C per year, mostly as a consequence of ongoing deforestation. The coastal ocean surrounding North America is also a small net source of carbon to the atmosphere (19 Mt C per year)

The primary carbon sink in North America is that of growing forests in the United States and Canada that have re-colonized land formerly cleared of forests for agricultural use in past centuries. Forest regrowth transfers carbon from the atmosphere, and it accumulates primarily in aboveground vegetation, with about a third accumulating as dead organic carbon in the soil. The suppression of forest fires also increases net carbon storage in forest biomass. The forest sink is by far the largest single component of the net North American sink, currently responsible for approximately 358 Mt C per year, or 60% of the total. As the recovering forests mature, however, net carbon uptake and the size of the sink decline; the estimated forest sink in Canada declined by nearly a third between 1990 and 2003.

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 level of approximately 80 million hectares burned per year, and this favors shrubs and trees in competition with grasses. The resulting sink has been estimated at 120 Mt C per year (20% of the North American sink), but the uncertainty around this estimate is greater than 100%. Woody encroachment might actually represent a small source of atmospheric carbon, or the sink might be twice the current estimate.

Wood products and wetlands are sinks of comparable size, 67 and 70 Mt C per year, respectively, or about 12% each of the total North American sink. Wood products create a carbon sink because they accumulate both in use (e.g., furniture, house frames, etc.) and in landfills. The wetland sink is primarily a consequence of peat accumulation in Canada’s extensive frozen and unfrozen wetlands and of sedimentation and the accompanying carbon sequestration in mineral soils of Canadian and U.S. wetlands. Drainage of peatlands in the United States has created a net source of 5 Mt C per year, and the very large reservoir of carbon in North American wetlands (the single largest carbon pool of any North American ecosystem) is vulnerable to release to the atmosphere in response to climate change and drainage for development, shifting this moderate sink to a potentially large source.

Agricultural lands in North America are currently nearly neutral with respect to carbon. Although mineral soils are estimated to be sequestering currently 6–15 Mt C per year, cultivation of organic soils releases 5–10 Mt C per year. The net is an approximate carbon balance for agricultural soils in Canada and a small sink 6 Mt C year or even source (1.5 Mt C per year) in the United States. 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. The effects of climate are uncertain.

Conversion of agricultural and wildlands to cities and other human settlements affect carbon sinks mainly by replacing biological ecosystems with built land cover. Growth of urban and suburban trees in North America are a part of the forest sink discussed above, but the rates of carbon sequestration in the vegetation and soils of settlements are uncertain and probably relatively small, certainly in comparison to fossil fuel emissions these areas. Thus, settlements in North America are almost certainly a source of atmospheric carbon, and 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 CO2 or other changes in the carbon cycle on the land and oceans of North America?

The consequences of a carbon cycle imbalance and the buildup of $CO_2$ in the atmosphere $CO_2$ extend beyond climate change alone. Ocean acidification and "$CO_2$ fertilization" of land plants are foremost among these direct, non-climatic effects.

The increasing concentration of $CO_2$ in the atmosphere has already made the world's oceans more acid. 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.

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, considerable uncertainty 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 or net uptake and storage by terrestrial ecosystem. 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. It is thus far from clear that elevated concentrations of atmospheric $CO_2$ have led to terrestrial carbon sequestration or will do so at the continental scale in the future.

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 carbon cycle of the land and ocean of North America as part of purposeful carbon management will consequently affect these other processes and cycles. For example, an increase in organic carbon in soils is likely to increase both the availability of nitrogen for plant growth and enhance the water holding capacity of the soil. 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 carbon cycle change or management on the interwoven systems of North America 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)?

Addressing imbalances in the North American and global carbon cycles requires options and measures focused on reducing carbon emissions. 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.

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 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, 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 and environmental management. 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 $CH_4$ or $N_2O$.

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.

## How can we improve the application of scientific information to decision support for carbon management and climate decision making?

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. Public concern, voluntary action and governmental efforts to regulate carbon emissions have heightened demand for basic data on the carbon cycle, models that link natural and social systems, and physical, economic and political analysis of specific carbon management options. There appears to 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 both incremental and major departures from existing practice with the goal 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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# Chapter 1. What is the Carbon Cycle and Why Do We Care?

## *An Introduction to the Purpose, Scope, and Structure of the State of the Carbon Cycle Report (SOCCR)*

**Lead Authors: SOCCR Coordinating Team**

**Coordinating Team Members: Lisa Dilling[1], Anthony King[2], David Fairman[3], Richard Houghton[4], Gregg Marland[2], Adam Rose[5], Thomas Wilbanks[2], and Gregory Zimmerman[2]**

[1]University of Colorado, [2]Oak Ridge National Laboratory, [3]Consensus Building Institute, Inc., [4]Woods Hole Research Center, [5]The Pennsylvania State University

## 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 (a) carbon-based life on and near the earth's surface, (b) carbon in the earth's atmosphere, and (c) water in the ocean. Stated in oversimplified terms, plants consume carbon dioxide from the atmosphere through photosynthesis and create sugars and other carbohydrates, which animals and humans use for food and shelter 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**. Reservoirs (in black) are gigatons [1 Gt = one billion ($1 \times 10^9$) metric tons] of carbon, and exchanges between reservoirs (in purple) are Gt carbon per year. *Illustration courtesy NASA Earth Science Enterprise*.

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. 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 that transfer, 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 or imbalance, but also provides insight into causes of any imbalance and steps that might be taken to manage that imbalance. Currently, the global carbon budget is in imbalance; and human use of coal, petroleum, and natural gas to fuel economies is responsible.

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 19$^{th}$ and 20$^{th}$ centuries, human societies found that these fossils had great value as energy sources for economic growth; and the 20$^{th}$ 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 *millenia*. During this same time, forests that had once absorbed very large quantities of carbon dioxide each year shrank in their extent.

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 ($CO_2$) 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 research is in its early stages, and 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).

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 and the atmospheric imbalance in the carbon budget.

## Other Implications of an Imbalance in the Carbon Budget

We do not yet have a full understanding of the consequences of this imbalance, 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 acidification of the surface ocean, 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.

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 vs 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). The North American sink is thus a substantial fraction, perhaps on the order of 30–60%, of the global terrestrial sink estimated to be in the range of 600 to 2300 Mt C $yr^{-1}$ and primarily in the extra-tropical Northern Hemisphere (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 transferred to oceans and land by carbon cycle processes and thus 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, but significant uncertainties remain (Caspersen *et al.*, 2000; Schimel *et al*., 2000; Houghton 2002). The future of the current North American terrestrial sink is highly uncertain, and it depends on which mechanisms are the dominant drivers.

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.

## 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. 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.

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, although such support 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.

Perhaps the most widely studied examples of scientist–stakeholder communication and dialogue have occurred through various types of scientific assessments. For example, Cash and Clark (2001) and Cash *et al.* (2003) found that the most effective[1] scientific assessments generally shared three interdependent characteristics, which they termed credibility, saliency, and legitimacy. Credibility is obviously essential if a scientific assessment is to be viewed as technically authoritative. The credibility of an assessment depends on the scientific scope and rigor of the process and on the scientific stature of its participants (Parson, 2003).

[1] The effectiveness of scientific syntheses and assessments is evaluated using a variety of criteria, including effects on policies, management options, research agendas, and attitudes of key constituencies (Cash and Clark, 2001; Parson 2003). These are not the only possible effectiveness criteria, but they provide an appropriate emphasis on the effectiveness of scientifically credible information that can be easily communicated to stakeholders and that they find useful for policy and management.

Saliency, according to Cash and Clark, is the extent to which an assessment is perceived as relevant and useful to stakeholders. Ensuring saliency requires early and ongoing dialogue with stakeholders to make sure that the questions posed within the scientific community are also important to the stakeholder community, and to educate the stakeholder community about the importance of scientific issues that they might otherwise overlook.

Cash and Clark (2001) defined legitimacy as the "perceived fairness of the assessment process." The legitimacy of a scientific assessment requires not only the contributions of scientific experts who represent a range of technical viewpoints, but also the substantive involvement of stakeholder representatives to ensure that the assessment is perceived as fair by their constituencies.

A common conclusion in analyses of scientific assessments is that the initial design and context are critically important (Cash and Clark, 2001; Farrell *et al.*, 2001; Parson 2003). The community and institutional mandate for an assessment have a strong influence on the eventual success of the process. The initial "framing" of the issues and questions to be addressed affects many decisions about the organization of the assessment, communication among participants, prioritization of goals, and ultimate effectiveness (Farrell *et al.*, 2001). The framing process requires great care because it may predetermine not only *who* gets to pose the questions, but also *how* the questions are posed.

How the assessment is delivered is as important as how it is defined. A potential pitfall in scientific assessment is to focus solely on producing a written report of findings, without understanding the importance of ongoing communication and social interaction that are critical for effective outcomes (Cash and Clark, 2001). Our proposed approach pays considerable attention to the ongoing process required to produce the SAP 2.2, with the explicit goal of ensuring that the SAP 2.2 is not only scientifically credible but also easily accessible, credible, and relevant to decision makers and other stakeholders. Transparency of the process will be a high priority through all stages.

Analysis of previous scientific assessments has emphasized that credibility, saliency, and legitimacy are inter-connected. As Parson (2003) put it, "Assessments that command little attention or respect by virtue of the collective stature of their participants; that draw no clear scientific judgments or conclusions about present knowledge except that more research is needed; that present no cogent new ways to understand the issue; and whose reports are both useless to scientists and inaccessible to lay persons, can expect to have no influence on policy, however high the quality of their work on other dimensions."

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.

## REFERENCES

Cash D., and W. Clark, 2001: From science to policy: Assessing the assessment process. Faculty Research Working Paper 01-045, Cambridge, MA, Kennedy School of Government, Harvard University. http://ksgnotes1.harvard.edu/Research/wpaper.nsf/RWP/RWP01-045.

Cash D., W. Clark, F. Alcock., N. Dickson., N. Eckley., D. Guston, J. Jäger, and R. Mitchell, 2003: Knowledge systems for sustainable development, In: *Proceedings of the National Academy of Sciences* 100, 8086–8091.

Casperson, J. P., S.W. Pacala, J. C. Jenkins, G. C. Hurtt, P. R. Moorcraft, and R. A. Birdsey, 2000: Contributions of land-use history to carbon accumulation in U.S. Forests, *Science* 290, 1148–1151.

CCSP, 2003: *Strategic Plan for the U.S. Climate Change Science Program: A report by the Climate Change Science Program and the Subcommittee on Global Change Research*, Climate Change Science Program Office, Washington, D.C.

DOE. 1997. Technology opportunities to reduce greenhouse gas emissions. U.S. Department of Energy, Washington, D.C.

Dilling, L., S. C. Doney, J. Edmonds, K. R. Gurney, R. C. Harriss, D. Schimel, B. Stephens, and G. Stokes, 2003: The role of carbon cycle observations and knowledge in carbon management, *Annual Review of Environment and Resources*, 28, 521–58.

Fan, S., M. Gloor, J. Mahlman, S. Pacala, J. Sarmiento, T. Takahashi, and P. Tans, 1998: A large terrestrial carbon sink in North America implied by atmospheric and oceanic carbon dioxide data and models, *Science*, 282, 442–446.

Farrell A., S. D. VanDeveer, and J. Jager, 2001: Environmental assessments: four under-appreciated elements of design, *Global Environmental Change*, 11, 311–333.

Friedlingstein, P., P. Cox, R. Betts, L. Bopp, W. von Bloh, V. Brovkin, S. Doney, M. Eby, I. Fung, B. Govindasamy, J. John, C. Jones, F. Joos, T. Kato, M. Kawamiya, W. Knorr, K. Lindsay, H. D. Matthews, T. Raddatz, P. Rayner, C. Reick, E. Roeckner, K.-G. Schnitzler, R. Schnur, K. Strassmann, S. Thompson, A.J. Weaver, C. Yoshikawa, and N. Zeng, 2006. Climate-carbon cycle feedback analysis, results from the $C^4$MIP model intercomparison, *Journal of Climate*, in press.

Goodale, C. L., *et al.*, 2002: Forest carbon sinks in the Northern Hemisphere, *Ecological Applications*, 12, 891–899.

Gurney, K. R., *et al.*, 2002: Towards robust regional estimates of $CO_2$ sources and sinks using atmospheric transport models, *Nature*, 415, 626–630.

Hoffert, M. I., K. C. Caldeira, G. Benford, *et al.*, 2002 : Advanced technology paths to global climate stability: energy for a greenhouse planet, *Science*, 298, 981–87.

Houghton, R. A., J. L. Hackler, and K. T. Lawrence, 1999: The U.S. carbon budget: contributions from land-use change, *Science*, 285, 574–578.

Houghton, R. A., 2002: Magnitude, distribution and causes of terrestrial carbon sinks and some implications for policy, *Climate Policy*, 2, 71–88.

IPCC, 2001: *Climate Change 2001: The Scientific Basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change* [J. T. Houghton, Y. Ding, D. J. Griggs, M. Noguer, P. J. van der Linden, *et al.* (eds.)]. Cambridge, United Kingdom, and New York, Cambridge University Press, 881 pp.

JGOFS Continental Margins Working Group, 2001: *Continental Margins Working Group Report, PI Meeting*, July 16–20. http://www1.whoi.edu/mzweb/cm_rpt_98.html

Marland, G., T., A. Boden, and R. J. Andres, 2003: Global, regional, and national $CO_2$ emissions. In: *Trends: A Compendium of Data on Global Change*, Carbon Dioxide Information Analysis Center, Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Parson, E.A., 2003: *Protecting the Ozone Layer*. Oxford University Press, Oxford, UK.

Schimel, D. S., J. Melillo, H. Tian, A. D. McGuire, D. Kicklighter, *et al.*, 2000: Contribution of increasing $CO_2$ and climate to carbon storage by ecosystems in the United States, *Science*, 287, 2004–6.

Turner, D. P., G. J. Koerper, M. E. Harmon, and J. J. Lee, 1995: A carbon budget for forests of the conterminous United States, *Ecological Applications*, 5, 421–436.

Wofsy, S. C., and R. C. Harriss, 2002: *The North American Carbon Program. A Report of the NACP Committee of the U.S. Carbon Cycle Science Steering Group*, U.S. Global Change Research Program, Washington, D.C.

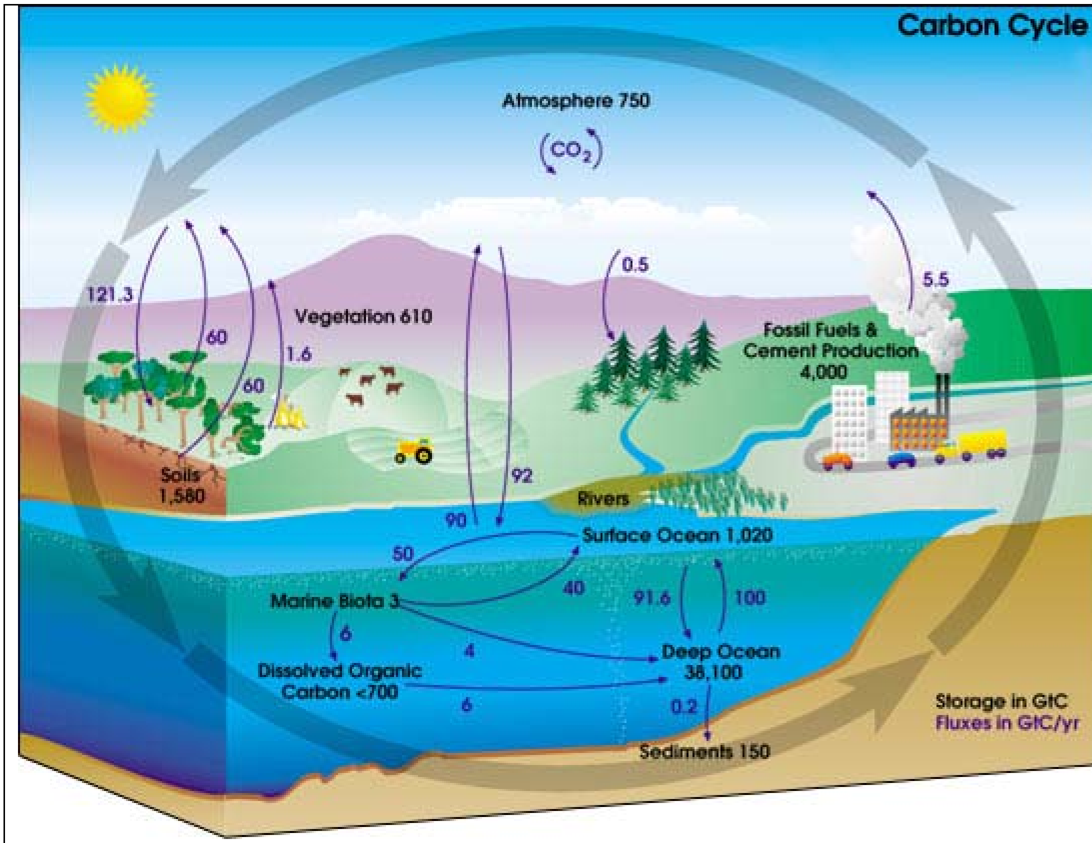


**Figure 1-1. The global carbon cycle**. Reservoirs (in black) are gigatons [1 Gt = one billion ($1 \times 10^9$) metric tons] of carbon, and exchanges between reservoirs (in purple) are Gt carbon per year. *Illustration courtesy NASA Earth Science Enterprise*.

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

**Coordinating Lead Author: Christopher B. Field[1]**

**Lead Authors: Jorge Sarmiento[2] and Burke Hales[3]**

[1]Carnegie Institution, [2]Princeton University, [3]Oregon State University

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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.
- While emissions (a carbon source) dominate the carbon budget of North America, these emissions are partially offset by 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. This sink is most likely caused by relatively young, growing forests which have re-colonized land formerly cleared of forests for agricultural use in past centuries.
- Global carbon dioxide emissions have increased for the last 30 years. In comparison, North American carbon dioxide emissions have increased at an average rate of approximately 1% per year for the last 30 years.
- 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. North America has many opportunities for decreasing emissions, including changes to the energy system, increasing energy efficiency, investments in forest planting and agricultural soil management, biomass energy, and geological sequestration.

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# 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 48% ± 5% of the carbon dioxide emitted to the atmosphere from fossil-fuel combustion and forest clearing has remained there (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 240 ± 40 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 $1 \times 10^{15}$ 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 carbon cycle.**

**Figure 2-2. Atmospheric $CO_2$ concentration from 1850 to 2005.** The data prior to 1957 are from the Siple ice core (Friedli *et al*., 1986). The data since 1957 are from continuous atmospheric sampling at the Mauna Loa Observatory (Hawaii) (Keeling *et al*., 1976; Thoning *et al*., 1989).

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 (Orr *et al*., 2005).

## The Background or 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 or unmanaged 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 100–200 Gt of atmospheric carbon, approximately 20 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, with the difference burned in wildfires or stored as plant biomass or soil organic carbon. The second comprises the ocean carbon cycle: about 92 ± 5 Gt of atmospheric carbon dissolves annually in the oceans, and about 90 Gt moves from the oceans to the atmosphere. The rest remains in the ocean as a mix of dissolved $CO_2$, bicarbonate ($HCO_3^{-}$), carbonate ($CO_3^{=}$), and organic matter.

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 0.45 ± 0.1 Gt C $yr^{-1}$ transferred to the oceans 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 (Keeling and Stephens, 2001; Sigman and Boyle, 2000). 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 plant growth, especially in the period 354 to 290 million years ago, the Carboniferous. During this period, 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. It also led to a near doubling of atmospheric oxygen (Falkowski *et al*., 2005).

## Anthropogenic Perturbations

Since the beginning of the industrial revolution, or about 1850, 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 1850 through 2004 are just over 300 ± 30 Gt (Marland and Rotty, 1984; Andres *et al*., 1999). Land use change during this period, mostly from the clearing of forests, added another 160 ± 160 Gt (DeFries *et al*., 1999; Houghton, 1999). 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 1850, with atmospheric $CO_2$ rising by 31% (i.e., from 287 ppm to 377 ppm).

Together, the three major countries of North America (Canada, Mexico, and the United States) accounted, in 2003, for carbon emissions from fossil-fuel combustion of approximately 1.83 ± 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. The carbon intensity of the United States' economy, at 0.15 metric ton of emitted carbon per $1000 (in 1995 dollars) of GDP (measured as PPP or Purchasing Power Parity), in 2003 was close to the world's average of 0.14 tC/$1000 [DOE EIA (U.S. Department of Energy, 2005)]. Canada's carbon intensity is somewhat higher at 0.19 tC/$1000, and Mexico's is somewhat lower at 0.12 tC/$1000. Rich countries with substantially lower carbon intensity include Japan, France, the United Kingdom, and Germany. Rich countries with higher carbon intensity include Australia and New Zealand [DOE EIA (U.S. Department of Energy, 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. Carbon intensity in both China and India is high. In 2003, carbon intensity in China was 4.6 times greater than that in the United States. The carbon intensity in India was 3.4 times that in the United States [DOE EIA (U.S. Department of Energy, 2005)].

Carbon emissions from North America have grown by about 1.0% per year for the last 30 years, substantially slower than the growth in GDP (Fig. 2-3). Slower growth in emissions than GDP characterizes many of the world's richest countries, including Canada and the United States. Since 1980, emissions growth has been only slightly slower than GDP growth in Mexico, a pattern typical of rapidly industrializing countries (Fig. 2-3). More rapid growth in GDP than in emissions can result from decreasing both the energy intensity of the economy (through, for example, more efficient manufacturing and increasing the role of the service sector) and the carbon intensity of the energy system (through, for example, replacing coal with natural gas in power plants or replacing fossil power plants with wind power plants) (Sathaye, 2004). It is not clear whether, in the absence of policy, historical trends in the energy intensity of GDP and the carbon intensity of the energy system will continue.

**Figure 2-3. GDP in 2000 U.S. dollars vs fossil-fuel carbon emissions (Mt C $yr^{-1}$).** Data from EIA (2005). Each arrow shows the sequence from 1980 to 2003 for a country. Note that carbon emissions per unit GDP decelerate as a country gains wealth. The lines in the figure show the slopes associated with the different ratios of GDP and emissions growth (the y-intercept of the dotted and dashed lines are not informative and were chosen only to keep from obscuring the arrows).

## 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 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. To date, estimates of continental-scale fluxes based on eddy flux must be regarded as preliminary.

Inventories, based on measuring trees on land (Birdsey and Heath, 1995) or carbon in 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 midlatitudes 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 passive tracers (Matsumoto *et al*., 2004) ($^{14}C$ 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). 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, 1999). 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 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 (Masarie and Tans, 1995). The $^{13}C$ in $CO_2$ and $O_2$ 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 (Siegenthaler and Oeschger, 1987). 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 460 ± 100 Gt carbon added to the atmosphere by human actions since 1850, only about 180 ± 5 Gt remain. The “missing carbon” was 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 is now in the oceans (Sabine *et al*., 2004a). This leaves about 100 Gt that must be stored on land. 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 (Gloor *et al*., 2003; Matear and McNeil, 2003; Matsumoto *et al*., 2004; Takahashi *et al*., 2002; Gurney *et al*., 2003). 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 (Fig. 2-4). 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).

**Figure 2-4. The spatial distribution of ocean $CO_2$ exchange from 1992–1996 for several regions and measurement approaches.** Tak99 and Tak02 are from (Takahashi *et al*., 2002) $\Delta pCO_2$ estimates, T3L1 and T3L2 are from (Gurney *et al*., 2003; Gurney *et al*., 2004), Fwd is from predictive ocean models, JI is from the ocean atmosphere ocean inversions of (Jacobson *et al*., 2006). The far right column is the sum of the individual ocean basins toward the left [from (Jacobson *et al*., 2006)].

On land 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. 2-5). Fire likely plays a large role in this variability (van der Werf *et al*., 2004).

**Figure 2-5. 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 approximately 80% of the carbon released to the atmosphere by human activities (Joos and Prentice, 2004). 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 (Goodale *et al*., 2002; Chen *et al*., 2003; Kurz and Apps, 1999)]. 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 $yr^{-1}$. 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).

Recent carbon storage in North America probably results from a number of different processes. Chen *et al*. (Chen *et al*., 2003) argue that Canadian forests are a small sink because processes tending to increase tree growth, including elevated atmospheric $CO_2$ and deposition of biologically available nitrogen, are more than compensating effects of recent disturbances. Kurz and Apps (Kurz and Apps, 1999) reach the opposite conclusion, that recent disturbances make Canadian forests a net carbon source. In the United States, forest regrowth is outpacing recent harvesting and disturbance (Birdsey and Heath, 1995). Some of this is a consequence of a profound historical shift in the location of United States agriculture.

Much of the Eastern United States was cleared for agriculture in the 18th century, only to be abandoned as agriculture moved to the Great Plains, the Southwest, and the West in the 19th and 20th centuries (Ramankutty and Foley, 1999). As a consequence, large areas once cleared for agriculture are currently regrowing forests (Caspersen *et al*., 2000). Increasing carbon in previously harvested forests has several drivers beyond the shift in agriculture, including changes in harvesting and management practices (Harmon *et al*., 1996; Goodale *et al*., 2002) and fire suppression (Calkin *et al*., 2005; Mouillot and Field, 2005). The processes sequestering carbon have been partially offset by processes that release stored carbon, including unusually high wildfire years [United States—(Mouillot and Field, 2005)], insect outbreaks [Canada—(Kurz and Apps, 1999)] , and storm damage [Europe—(Janssens *et al*., 2003)]. The heat wave and drought in Europe in the summer of 2003 led to a large loss of carbon, driven largely by decreased plant growth (Ciais *et al*., 2005).

Several other processes probably contribute to recent carbon sinks in the United States (Table 2-1), though they are difficult to quantify with confidence. These include the thickening of vegetation in woodland and shrubland areas, the burial of organic matter in lakes and reservoirs (Stallard, 1998), increases in the soil carbon in managed grassland and agricultural soils (Asner *et al*., 2003), and storage of carbon in durable products (e.g. houses and furniture) and waste in landfills (Pacala *et al*., 2001).

**Table 2-1. Sinks of carbon for 1980--90 in the coterminous United States (Gt C $yr^{-1}$).**

Some of the recent carbon storage in North America may be a consequence of increased atmospheric $CO_2$ (Schimel *et al*., 2000; Melillo *et al*., 2003), nitrogen deposition (Holland *et al*., 1997), or climate changes that have increased the length of the frost-free season in many locations (Myneni *et al*., 1997; Hicke *et al*., 2002). The evidence in support of the first two mechanisms comes from empirical and modeling studies. It is clear that plant growth in many terrestrial ecosystems is limited by either atmospheric $CO_2$ or biologically available nitrogen (Melillo *et al*., 2003). It is much less clear, however, that increased availability of either resource has led to carbon sequestration. Recent studies include many examples in which experimental treatment with elevated $CO_2$ leads to consistent increases in plant growth (e.g., Norby *et al*., 2005), but others in which elevated $CO_2$ has little effect on plant growth (Shaw *et al*., 2002), leads to an initial stimulation but limited long-term effects (Oren *et al*., 2001), or increases carbon losses as well as gains (Hungate, 1997; Schlesinger and Lichter, 2001). Evidence on the role of changes in the length of the growing season comes from field-based, satellite, and modeling studies (Myneni *et al*., 1997; Nemani *et al*., 2003). Recent evidence indicates that negative effects of dry summers can offset much or all of the effects of earlier springs (Angert *et al*., 2005).

To the extent that current carbon sink in North America reflects the regrowth of previously harvested forest, it is a one-time phenomenon and not a permanent feature of the carbon cycle. Similarly, a sink from effective fire suppression in the second half of the 20th century may have already saturated or even reversed, as large accumulations of highly flammable fuels amplify the challenge of current and future fire management. Sinks from $CO_2$ fertilization (Hungate *et al*., 2003), increased nitrogen deposition, and altered management of agricultural lands (Smith, 2004) could continue for some time, but they too will eventually saturate (Gruber *et al*., 2004).

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. Currently, as much as 50% of this may be offset by carbon uptake in plants and soils (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 (Hoffert *et al*., 2002; Caldeira *et al*., 2004; Pacala and Socolow, 2004). Many of these are in the area of increasing the efficiency of energy generation, the transportation system, building stocks, and manufacturing technologies. Others are in the area of replacing carbon-emitting energy technologies with nonemitting technologies, including solar, wind, biomass, and nuclear. Still others are in the area of sequestration, including both geological and biological components. Finally, there are many opportunities in conservation, in directing the economy and personal preferences away from carbon-intensive activities. Capitalizing on the opportunities in all four of these areas will require dedicated research, financial support, creativity, and an interested public (Raupach *et al*., 2004). Nothing about the status of the unmanaged carbon cycle provides a justification for decreasing the commitment to progress in all of these areas.

## REFERENCES

Andres, R. J., D. J. Fielding, G. Marland, T. A. Boden, N. Kumar, and A. T. Kearney. 1999. Carbon dioxide emissions from fossil-fuel use, 1751–1950. Tellus Series B Chemical and Physical Meteorology **51**:759–765.

Angert, A., S. Biraud, C. Bonfils, C. C. Henning, W. Buermann, J. Pinzon, C. J. Tucker, and I. Fung. 2005. Drier summers cancel out the $CO_2$ uptake enhancement induced by warmer springs. Proceedings of the National Academy of Sciences of the United States of America **102**:10823–10827.

Asner, G. P., S. Archer, R. F. Hughes, R. J. Ansley, and C. A. Wessman. 2003. Net changes in regional woody vegetation cover and carbon storage in Texas Drylands, 1937-1999. Global Change Biology **9**:316–335.

Bacastow, R., and C. D. Keeling. 1973. Atmospheric carbon dioxide and radiocarbon in the natural carbon cycle. II, Changes from A.D. 1700 to 2070 as deduced from a geochemical reservoir. Pages 86–135 *in* G. M. Woodwell and E. V. Pecan, editors. Carbon and the Biosphere. U.S. Dept. of Commerce, Springfield, Virginia.

Bachelet, D., R. P. Neilson, T. Hickler, R. J. Drapek, J. M. Lenihan, M. T. Sykes, B. Smith, S. Sitch, and K. Thonicke. 2003. Simulating past and future dynamics of natural ecosystems in the United States. Global Biogeochemical Cycles **17**:1045.

Baker, D. F., R. M. Law, K. R. Gurney, P. Rayner, P. Peylin, A. S. Denning, P. Bousquet, L. Bruhwiler, Y. H. Chen, P. Ciais, I. Y. Fung, M. Heimann, J. John, T. Maki, S. Maksyutov, K. Masarie, M. Prather, B. Pak, S. Taguchi, and Z. Zhu. 2006. TransCom 3 inversion intercomparison: Impact of transport model errors on the interannual variability of regional $CO_2$ fluxes, 1988–2003. Global Biogeochemical Cycles **20**:GB1002.

Baldocchi, D., and R. Valentini. 2004. Geographic and temporal variation of carbon exchange by ecosystems and their sensitivity to environmental perturbations. Pages 295-316 *in* C. B. Field and M. R. Raupach, editors. The Global Carbon Cycle: Integrating Humans, Climate, and the Natural World. Island Press, Washington.

Birdsey, R. A., and L. S. Heath. 1995. Carbon changes in U.S. forests. Pages 56-70 *in* L. A. Joyce, editor. Productivity of America's Forests and Climate Change, General Technical Report RM-GTR-271. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado.

Bousquet, P., P. Peylin, P. Ciais, C. L. Quéré, P. Friedlingstein, and P. P. Tans. 2000. Regional Changes in Carbon Dioxide Fluxes of Land and Oceans Since 1980. Science **290**:1342–1346.

Caldeira, K., G. Morgan, D. Baldocchi, P. Brewer, C.-T. A. Chen, G.-J. Nabuurs, N. Nakicenovic, and G. P. Robertson. 2004. A portfolio of carbon management options. Pages 103–130 *in* C. B. Field and M. R. Raupach, editors. The Global Carbon Cycle: Integrating Humans, Climate, and the Natural World. Island Press, Washington.

Caldeira, K., and M. E. Wickett. 2003. Anthropogenic carbon and ocean pH. Nature **425**:365–365.

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

Caspersen, J. P., S. W. Pacala, J. C. Jenkins, G. C. Hurtt, P. R. Moorcroft, and R. A. Birdsey. 2000. Contributions of land-use history to carbon accumulation in U.S. forests. Science **290**:1148–1151.

Chen, J. M., W. Ju, J. Cihlar, D. Price, J. Liu, W. Chen, J. Pan, A. Black, and A. Barr. 2003. Spatial distribution of carbon sources and sinks in Canada's forests. Tellus Series B Chemical and Physical Meteorology **55B**:622–641.

Ciais, P., M. Reichstein, N. Viovy, A. Granier, J. Ogee, V. Allard, M. Aubinet, N. Buchmann, C. Bernhofer, A. Carrara, F. Chevallier, N. De Noblet, A. D. Friend, P. Friedlingstein, T. Grunwald, B. Heinesch, P. Keronen, A. Knohl, G. Krinner, D. Loustau, G. Manca, G. Matteucci, F. Miglietta, J. M. Ourcival, D. Papale, K. Pilegaard, S. Rambal, G. Seufert, J. F. Soussana, M. J. Sanz, E. D. Schulze, T. Vesala, and R. Valentini. 2005. Europe-wide reduction in primary productivity caused by the heat and drought in 2003. Nature **437**:529–533.

Cramer, W., A. Bondeau, F. I. Woodward, I. C. Prentice, R. A. Betts, V. Brovkin, P. M. Cox, V. A. Fisher, J. A. Foley, A. D. Friend, and C. Kucharik. 2001. Global response of terrestrial ecosystem structure and function to

$CO_2$ and climate change: results from six dynamic global vegetation models. Global Change Biology **7**:357–373.

Cramer, W., D. W. Kicklighter, A. Bondeau, B. M. III, G. Churkina, B. Nemry, A. Ruimy, A. L. Schloss, J. Kaduk, and participants of the Potsdam NPP Model Intercomparison. 1999. Comparing global models of terrestrial net primary productivity (NPP): overview and key results. Global Change Biology **5 supplement**:1–15.

DeFries, R. S., C. B. Field, I. Fung, J. Collatz, and L. Bounoua. 1999. Combining satellite data and biogeochemical models to estimate global effects of human-induced land cover change on carbon emissions and primary productivity. Global Biogeochemical Cycles **13**:803–815.

DOE EIA (U.S. Department of Energy, Energy Information Administration). 2005.

Dukes, J. 2003. Burning buried sunshine: Human consumption of ancient solar energy. Climatic Change **61**:31–44.

Enting, I. G. 2002. Inverse Problems in Atmospheric Constituent Transport. Cambridge University Press, London.

Falkowski, P. G., M. E. Katz, A. J. Milligan, K. Fennel, B. S. Cramer, M. P. Aubry, R. A. Berner, M. J. Novacek, and W. M. Zapol. 2005. The Rise of Oxygen over the Past 205 Million Years and the Evolution of Large Placental Mammals. Science **309**:2202–2204.

Friedli, H., H. Lötscher, H. Oeschger, U. Siegenthaler, and B. Stauffer. 1986. Ice core record of $^{13}C/^{12}C$ ratio of atmospheric $CO_2$ in the past two centuries. Nature **324**:237–238.

Giampietro, M., S. Ulgiati, and D. Pimentel. 1997. Feasibility of large-scale biofuel production: Does an enlargement of scale change the picture? Bioscience **47**:587–600.

Gloor, M., N. Gruber, J. Sarmiento, C. L. Sabine, R. A. Feely, and C. Rodenbeck. 2003. A first estimate of present and preindustrial air-sea $CO_2$ flux patterns based on ocean interior carbon measurements and models - art. no. 1010. Geophysical Research Letters **30**:1010–1010.

Goodale, C. L., M. J. Apps, R. A. Birdsey, C. B. Field, L. S. Heath, R. A. Houghton, J. C. Jenkins, G. H. Kohlmaier, W. Kurz, S. R. Liu, G. J. Nabuurs, S. Nilsson, and A. Z. Shvidenko. 2002. Forest carbon sinks in the Northern Hemisphere. Ecological Applications **12**:891–899.

Graham, P. J. 2003. Potential for climate change mitigation through afforestation: An economic analysis of fossil fuel substitution and carbon sequestration benefits. Agroforestry Systems **59**:85–95.

Greenblatt, J. B., and J. L. Sarmiento. 2004. Variability and climate feedback mechanisms in ocean uptake of $CO_2$. Pages 257-275 *in* C. B. Field and M. R. Raupach, editors. The Global Carbon Cycle: Integrating Humans, Climate, and the Natural World. Island Press, Washington, D.C.

Gruber, N., P. Friedlingstein, C. B. Field, R. Valentini, M. Heimann, J. E. Richey, P. Romero-Lankao, E.-D. Schulze, and C.-T. A. Chen. 2004. The vulnerability of the carbon cycle in the 21st century: An assessment of carbon-climate-human interactions. Pages 45-76 *in* C. B. Field and M. R. Raupach, editors. The Global Carbon Cycle: Integrating Humans, Climate, and the Natural World. Island Press, Washington.

Gurney, K. R., R. M. Law, A. S. Denning, P. J. Rayner, D. Baker, P. Bousquet, L. Bruhwiler, Y. H. Chen, P. Ciais, S. M. Fan, I. Y. Fung, M. Gloor, M. Heimann, K. Higuchi, J. John, E. Kowalczyk, T. Maki, S. Maksyutov, P. Peylin, M. Prather, B. C. Pak, J. Sarmiento, S. Taguchi, T. Takahashi, and C. W. Yuen. 2003. TransCom 3

$CO_2$ inversion intercomparison: 1. Annual mean control results and sensitivity to transport and prior flux information. Tellus Series B-Chemical and Physical Meteorology **55**:555–579.

Gurney, K. R., R. M. Law, A. S. Denning, P. J. Rayner, B. C. Pak, D. Baker, P. Bousquet, L. Bruhwiler, Y. H. Chen, P. Ciais, I. Y. Fung, M. Heimann, J. John, T. Maki, S. Maksyutov, P. Peylin, M. Prather, and S. Taguchi. 2004. Transcom 3 inversion intercomparison: Model mean results for the estimation of seasonal carbon sources and sinks. Global Biogeochemical Cycles **18**:GB1010.

Hansen, J., L. Nazarenko, R. Ruedy, M. Sato, J. Willis, A. Del Genio, D. Koch, A. Lacis, K. Lo, S. Menon, T. Novakov, J. Perlwitz, G. Russell, G. A. Schmidt, and N. Tausnev. 2005. Earth's energy imbalance: confirmation and implications. Science **308**:1431–1435.

Harmon, M. E., S. L. Garman, and W. K. Ferrell. 1996. Modeling historical patterns of tree utilization in the Pacific Northwest: carbon sequestration implications. Ecological Applications **6**:641–653.

Heath, J., E. Ayres, M. Possell, R. D. Bardgett, H. I. J. Black, H. Grant, P. Ineson, and G. Kerstiens. 2005. Rising atmospheric $CO_2$ reduces sequestration of root-derived soil carbon. Science (Washington D C) **309**:1711–1713.

Hicke, J. A., G. P. Asner, J. T. Randerson, C. J. Tucker, S. O. Los, R. A. Birdsey, J. C. Jenkins, C. Field, and E. Holland. 2002. Satellite-derived increases in net primary productivity across North America 1982–1998. Geophysical Research Letters **29**.

Hoffert, M. I., K. Caldeira, G. Benford, D. R. Criswell, C. Green, H. Herzog, A. K. Jain, H. S. Kheshgi, K. S. Lackner, J. S. Lewis, H. D. Lightfoot, W. Manheimer, J. C. Mankins, M. E. Mauel, L. J. Perkins, M. E. Schlesinger, T. Volk, and T. M. L. Wigley. 2002. Advanced Technology Paths to Global Climate Stability: Energy for a Greenhouse Planet. Science **298**:981–987.

Holland, E. A., B. H. Braswell, J.-F. Lamarque, A. Townsend, J. Sulzman, J.-F. Müller, F. Dentener, G. Brasseur, H. L. II, J. E. Penner, and G.-J. Roelofs. 1997. Examination of spatial variation in atmospheric nitrogen deposition and its impact on the terrestrial ecosystems. Journal of Geophysical Research **106**:15,849–815,866.

Houghton, R. A. 1999. The annual net flux of carbon to the atmosphere from changes in land use 1850–1990. Tellus **51B**:298–313.

Houghton, R.A., J.L. Hackler, and K.T. Lawrence. 1999. The U.S. carbon budget: contributions from land-use change. Science **285**:574–578.

Hungate, B., J. Dukes, M. Shaw, Y. Luo, and C. Field. 2003. Nitrogen and climate change. Science **302**:1512–1513.

Hungate, B. A., Holland, E.A., Jackson, R.B., Chapin III, F.S., Mooney, H.A., and Field, C.B. 1997. The fate of carbon in grasslands under carbon dioxide enrichment. Nature **388**:576–579.

Jacobson, A. R., S. E. Mikaloff-Fletcher, N. Gruber, J. L. Sarmiento, M. Gloor, and TransCom Modelers. 2006. A joint atmosphere-ocean inversion for surface fluxes of carbon dioxide. Global Biogeochem Cycles **(submitted)**.

Janssens, I., A. Freibauer, P. Ciais, P. Smith, G. Nabuurs, G. Folberth, B. Schlamadinger, R. Hutjes, R. Ceulemans, E. Schulze, R. Valentini, and A. Dolman. 2003. Europe's terrestrial biosphere absorbs 7 to 12% of European anthropogenic $CO_2$ emissions. Science **300**:1538–1542.

Joos, F., and I. C. Prentice. 2004. A paleo perspective on the future of atmospheric $CO_2$ and climate. Pages 165–186 *in* C. B. Field and M. R. Raupach, editors. The Global Carbon Cycle: Integrating Humans, Climate, and the Natural World. Island Press, Washington.

Keeling, C. D., R. B. Bacastow, A. E. Bainbridge, C. A. Ekdahl, P. R. Guenther, and L. S. Waterman. 1976. Atmospheric carbon dioxide variations at Mauna Loa Observatory, Hawaii. Tellus **28**:538–551.

Keeling, R. F., S. C. Piper, and M. Heimann. 1996. Global and hemispheric $CO_2$ sinks deduced from changes in atmospheric $O_2$ concentration. Nature **381**:218–221.

Keeling, R. F., and B. B. Stephens. 2001. Antarctic sea ice and the control of Pleistocene climate instability. Paleoceanography **16**:112–131.

Kirschbaum, M. U. F. 2003. To sink or burn? A discussion of the potential contributions of forests to greenhouse gas balances through storing carbon or providing biofuels. Biomass and Bioenergy **24**:297–310.

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

Law, R. M., Y.-H. Chen, K. R. Gurney, and M. Transcom. 2003. TransCom 3 $CO_2$ inversion intercomparison: 2. Sensitivity of annual mean results to data choices. Tellus Series B Chemical and Physical Meteorology **55B**:580–595.

Le Quéré, C., and N. Metzl. 2004. Natural processes regulating the ocean uptake of $CO_2$. Pages 243–256 *in* C. B. Field and M. R. Raupach, editors. The Global Carbon Cycle: Integrating Humans, Climate, and the Natural World. Island Press, Washington.

Marland, G., and R. M. Rotty. 1984. Carbon dioxide emissions from fossil fuels: a procedure for estimation and results for 1950–1982. Tellus **36B**:232–261.

Martin, J. H. 1990. Glacial-interglacial $CO_2$ change: the iron hypothesis. Paleoceanography **5**:1–13.

Masarie, K. A., and P. P. Tans. 1995. Extension and integration of atmospheric carbon dioxide data into a globally consistent measurement record. Journal of Geophysical Research (Atmospheres) **100**:11593–11610.

Matear, R. J., and B. I. McNeil. 2003. Decadal accumulation of anthropogenic $CO_2$ in the Southern Ocean: A comparison of CFC-age derived estimates to multiple-linear regression estimates. Global Biogeochemical Cycles **17**:1113.

Matsumoto, K., J. L. Sarmiento, R. M. Key, O. Aumont, J. L. Bullister, K. Caldeira, J. M. Campin, S. C. Doney, H. Drange, J. C. Dutay, M. Follows, Y. Gao, A. Gnanadesikan, N. Gruber, A. Ishida, F. Joos, K. Lindsay, E. Maier-Reimer, J. C. Marshall, R. J. Matear, P. Monfray, A. Mouchet, R. Najjar, G. K. Plattner, R. Schlitzer, R. Slater, P. S. Swathi, I. J. Totterdell, M. F. Weirig, Y. Yamanaka, A. Yool, and J. C. Orr. 2004. Evaluation of ocean carbon cycle models with data-based metrics. Geophysical Research Letters **31**:L07303–07304.

Melillo, J. M., C. B. Field, and B. Moldan, editors. 2003. Interactions of the Major Biogeochemical Cycles: Global Change and Human Impacts. Island Press, Washington, D.C.

Mitchell, J. F. B., D. J. Karoly, G. C. Hegerl, F. W. Zwiers, M. R. Allen, and J. Marengo. 2001. Detection of climate change and attribution of causes. Pages 695–738 *in* J. T. Houghton, Y. Ding, D. J. Griggs, M. Noguer,

P. J. v. d. Linden, X. Dai, K. Maskell, and C. A. Johnson, editors. Climate Change 2001: The Scientific Basis. Cambridge University Press, Cambridge.

Mouillot, F., and C. B. Field. 2005. Fire history and the global carbon budget: a 1 degree × 1 degree fire history reconstruction for the 20th century. Global Change Biology **11**:1–23.

Myneni, R. B., C. D. Keeling, C. J. Tucker, G. Asrar, and R. R. Nemani. 1997. Increased plant growth in the northern high latitudes from 1981–1991. Nature **386**:698–702.

Nemani, R. R., C. D. Keeling, H. Hashimoto, W. M. Jolly, S. C. Piper, C. J. Tucker, R. B. Myneni, and S. W. Running. 2003. Climate-driven increases in global terrestrial net primary production from 1982 to 1999. Science **300**:1560–1563.

Newsam, G. N., and I. G. Enting. 1988. Inverse problems in atmospheric constituent studies: I. Determination of surface sources under a diffusive transport approximation. Inverse Problems **4**:1037–1054.

Norby, R. J., E. H. DeLucia, B. Gielen, C. Calfapietra, C. P. Giardina, J. S. King, J. Ledford, H. R. McCarthy, D. J. P. Moore, R. Ceulemans, P. De Angelis, A. C. Finzi, D. F. Karnosky, M. E. Kubiske, M. Lukac, K. S. Pregitzer, G. E. Scarascia-Mugnozza, W. H. Schlesinger, and R. Oren. 2005. Forest response to elevated $CO_2$ is conserved across a broad range of productivity. PROCEEDINGS OF THE NATIONAL ACADEMY OF SCIENCES OF THE UNITED STATES OF AMERICA **102**:18052–18056.

Oren, R., D. S. Ellsworth, K. H. Johnsen, N. Phillips, B. E. Ewers, C. Maier, K. V. R. Schafer, and e. al. 2001. Soil fertility limits carbon sequestration by forest ecosystems in a $CO_2$-enriched atmosphere. Nature **411**:469–472.

Orr, J. C., V. J. Fabry, O. Aumont, L. Bopp, S. C. Doney, R. A. Feely, A. Gnanadesikan, N. Gruber, A. Ishida, F. Joos, R. M. Key, K. Lindsay, E. Maier-Reimer, R. Matear, P. Monfray, A. Mouchet, R. G. Najjar, G. K. Plattner, K. B. Rodgers, C. L. Sabine, J. L. Sarmiento, R. Schlitzer, R. D. Slater, I. J. Totterdell, M. F. Weirig, Y. Yamanaka, and A. Yool. 2005. Anthropogenic ocean acidification over the twenty-first century and its impact on calcifying organisms. Nature **437**:681–686.

Pacala, S., and R. Socolow. 2004. Stabilization Wedges: Solving the Climate Problem for the Next 50 Years with Current Technologies. Science **305**:968–972.

Pacala, S. W., G. C. Hurtt, D. Baker, P. Peylin, R. A. Houghton, R. A. Birdsey, L. Heath, E. T. Sundquist, R. F. Stallard, P. Ciais, P. Moorcroft, J. P. Caspersen, E. Shevliakova, B. Moore, G. Kohlmaier, E. Holland, M. Gloor, M. E. Harmon, S. M. Fan, J. L. Sarmiento, C. L. Goodale, D. Schimel, and C. B. Field. 2001. Consistent land- and atmosphere-based U.S. carbon sink estimates. Science **292**:2316–2319.

Petit, J. R., J. Jouzel, D. Raynaud, N. I. Barkov, J.-M. Barnola, I. Basile, M. Bender, J. Chappellaz, M. Davis, G. Delaygue, M. Delmotte, V. M. Kotlyakov, M. Legrand, V. Y. Lipenkov, C. Lorius, L. Pépin, C. Ritz, E. Saltzman, and M. Stievenard. 1999. Climate and atmospheric history of the past 420,000 years from the Vostok ice core, Antarctica. Nature **399**:429–436.

Peylin, P., P. Bousquet, C. Le Quere, S. Sitch, P. Friedlingstein, G. McKinley, N. Gruber, P. Rayner, and P. Ciais. 2005. Multiple constraints on regional $CO_2$ flux variations over land and oceans. Global Biogeochemical Cycles **19**:GB1011.

Prentice, I. C. 2001. The carbon cycle and atmospheric carbon dioxide. *in* Climate Change 2001: The Scientific Basis (Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change).

Ramankutty, N., and J. A. Foley. 1999. Estimating historical changes in global land cover: Croplands from 1700 to 1992. Global Biogeochem. Cycles **13**:997–1028.

Raupach, M., J. Canadell, D. Bakker, P. Ciais, M. J. Sanz, J. Fang, J. Melillo, P. Romero-Lankao, J. Sathaye, D. Schulze, P. Smith, and J. Tschirley. 2004. Interactions between $CO_2$ stabilization pathways and requirements for a sustainable earth system. Pages 131–162 *in* C. B. Field and M. R. Raupach, editors. The Global Carbon Cycle: Integrating Humans, Climate, and the Natural World. Island Press, Washington.

Rodenbeck, C., S. Houweling, M. Gloor, and M. Heimann. 2003. $CO_2$ flux history 1982–2001 inferred from atmospheric data using a global inversion of atmospheric transport. Atmospheric Chemistry and Physics **3**:1919–1964.

Rousteenoja, K., T. R. Carter, K. Jylha, and H. Tuomenvirta. 2003. Future Climate in World Regions: An Intercomparison of Model-Based Projections for the New IPCC Emissions Scenarios. Finnish Environment Institute, Helsinki.

Running, S. W., R. R. Nemani, F. A. Heinsch, M. S. Zhao, M. Reeves, and H. Hashimoto. 2004. A continuous satellite-derived measure of global terrestrial primary production. Bioscience **54**:547–560.

Sabine, C. L., R. A. Feely, N. Gruber, R. M. Key, K. Lee, J. L. Bullister, R. Wanninkhof, C. S. Wong, D. W. R. Wallace, B. Tilbrook, F. J. Millero, T. H. Peng, A. Kozyr, T. Ono, and A. F. Rios. 2004a. The oceanic sink for anthropogenic $CO_2$. Science **305**:367–371.

Sabine, C. L., M. Heiman, P. Artaxo, D. C. E. Bakker, C.-T. A. Chen, C. B. Field, N. Gruber, C. LeQuéré, R. G. Prinn, J. E.Richey, P. Romero-Lankao, J. A. Sathaye, and R. Valentini. 2004b. Current status and past trends of the carbon cycle. Pages 17–44 *in* C. B. Field and M. R. Raupach, editors. The Global Carbon Cycle: Integrating Humans, Climate, and the Natural World. Island Press, Washington.

Sathaye, J. A. 2004. Near- and long-term climate change mitigation potential. Pages 405–418 *in* C. B. Field and M. R. Raupach, editors. The Global Carbon Cycle: Integrating Humans, Climate, and the Natural World. Island Press, Washington.

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

Schimel, D. S., J. I. House, K. A. Hibbard, P. Bousquet, P. Ciais, P. Peylin, B. H. Braswell, M. J. Apps, D. Baker, A. Bondeau, J. Canadell, G. Churkina, W. Cramer, A. S. Denning, C. B. Field, P. Friedlingstein, C. Goodale, M. Heimann, R. A. Houghton, J. M. Melillo, B. Moore, D. Murdiyarso, I. Noble, S. W. Pacala, I. C. Prentice, M. R. Raupach, P. J. Rayner, R. J. Scholes, W. L. Steffen, and C. Wirth. 2001. Recent patterns and mechanisms of carbon exchange by terrestrial ecosystems. Nature **414**:169–172.

Schlesinger, W. H., and J. Lichter. 2001. Limited carbon storage in soil and litter of experimental forest plots under increased atmospheric $CO_2$. Nature **411**:466–469.

Shaw, M. R., E. S. Zavaleta, N. R. Chiariello, E. E. Cleland, H. A. Mooney, and C. B. Field. 2002. Grassland Responses to Global Environmental Changes Suppressed by Elevated $CO_2$. Science **298**:1987–1990.

Siegenthaler, U., and H. Oeschger. 1987. Biospheric $CO_2$ emissions during the past 200 years reconstructed by deconvolution of ice core data. Tellus **39B**:140–154.

Sigman, D. M., and E. A. Boyle. 2000. glacial/interglacial variations in atmospheric carbon dioxide. Nature **407**:859–869.

Smith, P. 2004. Engineered biological sinks on land. Pages 479-492 *in* C. B. Field and M. R. Raupach, editors. The Global Carbon Cycle: Integrating Humans, Climate, and the Natural World. Island Press, Washington.

Stallard, R. F. 1998. Terrestrial sedimentation and the carbon cycle: Coupling weathering and erosion to carbon burial. Global Biogeochemical Cycles **12**:231–258.

Takahashi, T., S. C. Sutherland, C. Sweeney, A. Poisson, N. Metzl, B. Tilbrook, N. Bates, R. Wanninkhof, R. A. Feely, C. Sabine, J. Olafsson, and Y. Nojiri. 2002. Global sea-air $CO_2$ flux based on climatological surface ocean pCO(2), and seasonal biological and temperature effects. Deep-Sea Research Part Ii-Topical Studies in Oceanography **49**:1601–1622.

Tarantola, A. 1987. Inverse Problem Theory: Methods for Data Fitting and Model Parameter Estimation. Elsevier, New York.

Thoning, K. W., P. P. Tans, and W. D. Komhyr. 1989. Atmospheric carbon dioxide at Mauna Loa Observatory 2. Analysis of the NOAA GMCC data, 1974–1985. J. Geophys. Research **94**:8549–8565.

van der Werf, G. R., J. T. Randerson, G. J. Collatz, L. Giglio, P. S. Kasibhatla, A. F. Arellano, S. C. Olsen, and E. S. Kasischke. 2004. Continental-scale partitioning of fire emissions during the 1997 to 2001 El Nino/La Nina period. Science **303**:73–74.

Wofsy, S. C., M. L. Goulden, J. W. Munger, S.-M. Fan, P. S. Bakwin, B. C. Daube, S. L. Bassow, and F. A. Bazzaz. 1993. Net exchange of $CO_2$ in a mid-latitude temperate forest. Science **260**:1314–1317.

**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 carbon cycle.**

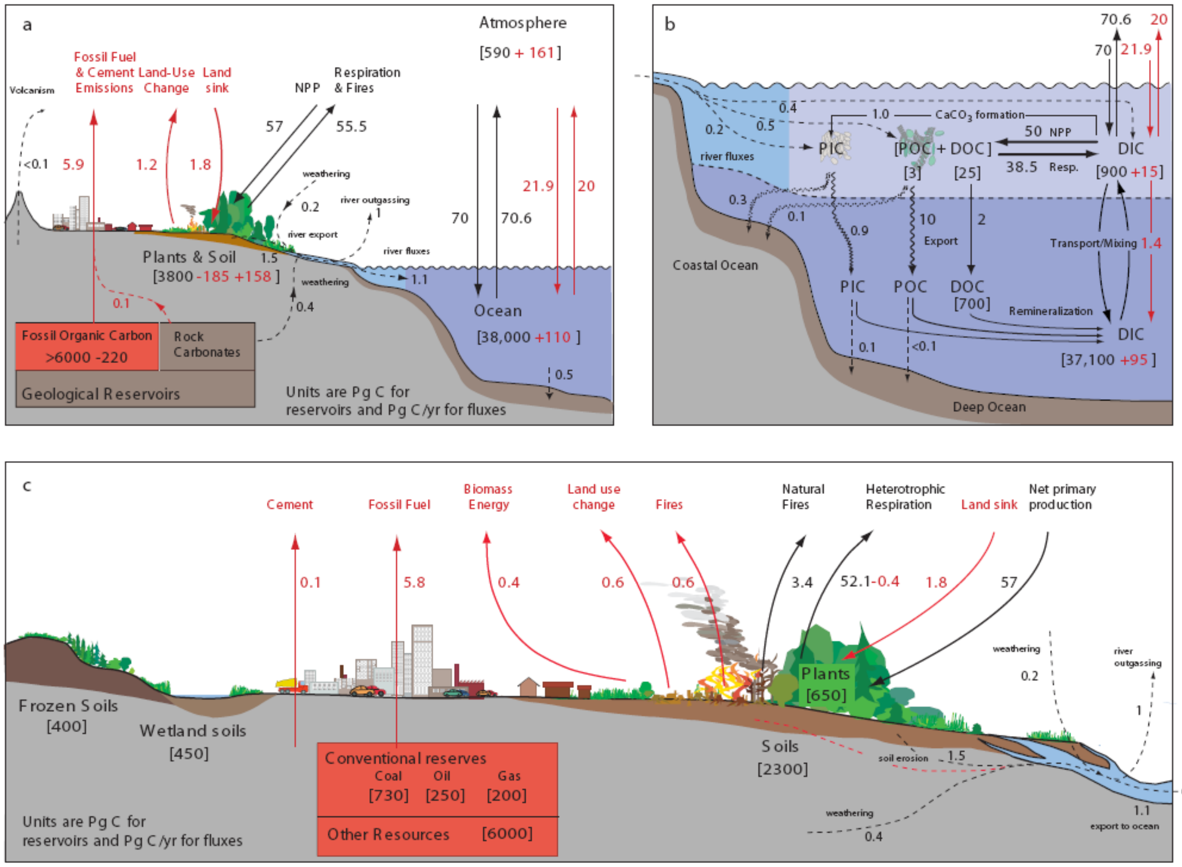
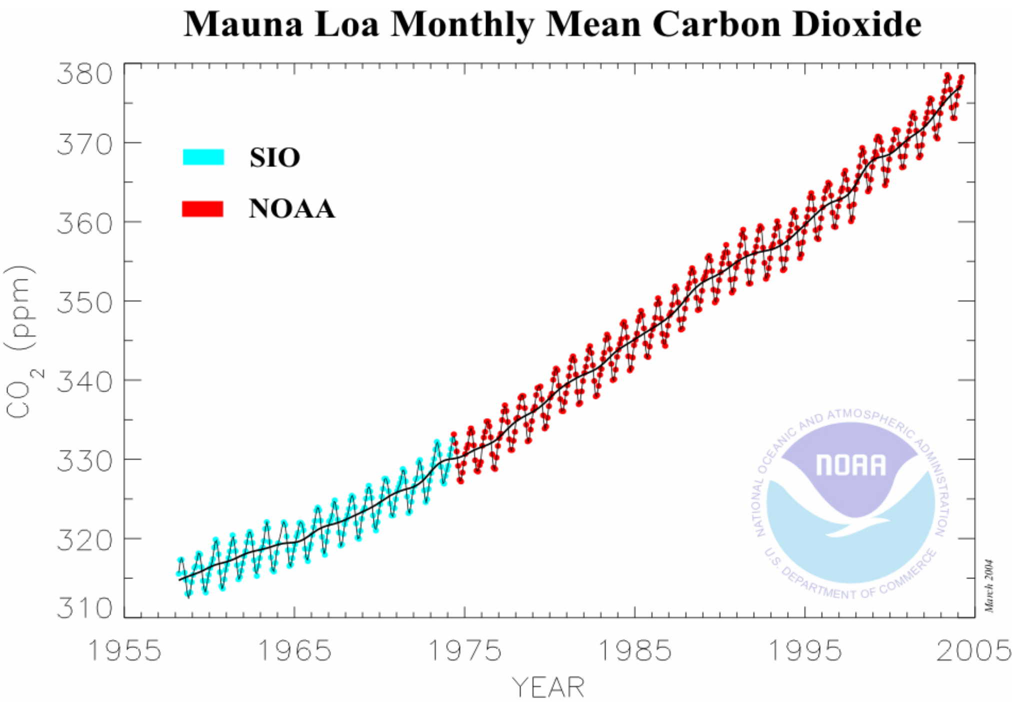


**Fig. 2-2. Atmospheric $CO_2$ concentration from 1850 to 2005.** The data prior to 1957 are from the Siple ice core (Friedli *et al*., 1986). The data since 1957 are from continuous atmospheric sampling at the Mauna Loa Observatory (Hawaii) (Keeling *et al*., 1976; Thoning *et al*., 1989).



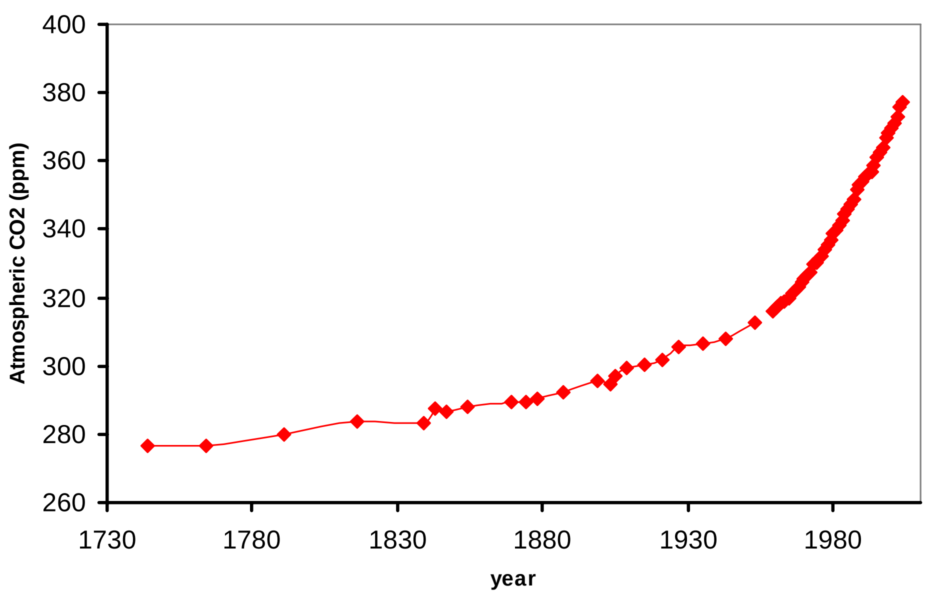



**Figure 2-3. GDP in 2000 U.S. dollars vs fossil-fuel carbon emissions (Mt C yr$^{-1}$).** Data from EIA (2005). Each arrow shows the sequence from 1980 to 2003 for a country. Note that carbon emissions per unit GDP decelerate as a country gains wealth. The lines in the figure show the slopes associated with the different ratios of GDP and emissions growth (the y-intercepts of the dotted and dashed lines are not informative and were chosen only to keep from obscuring the arrows).

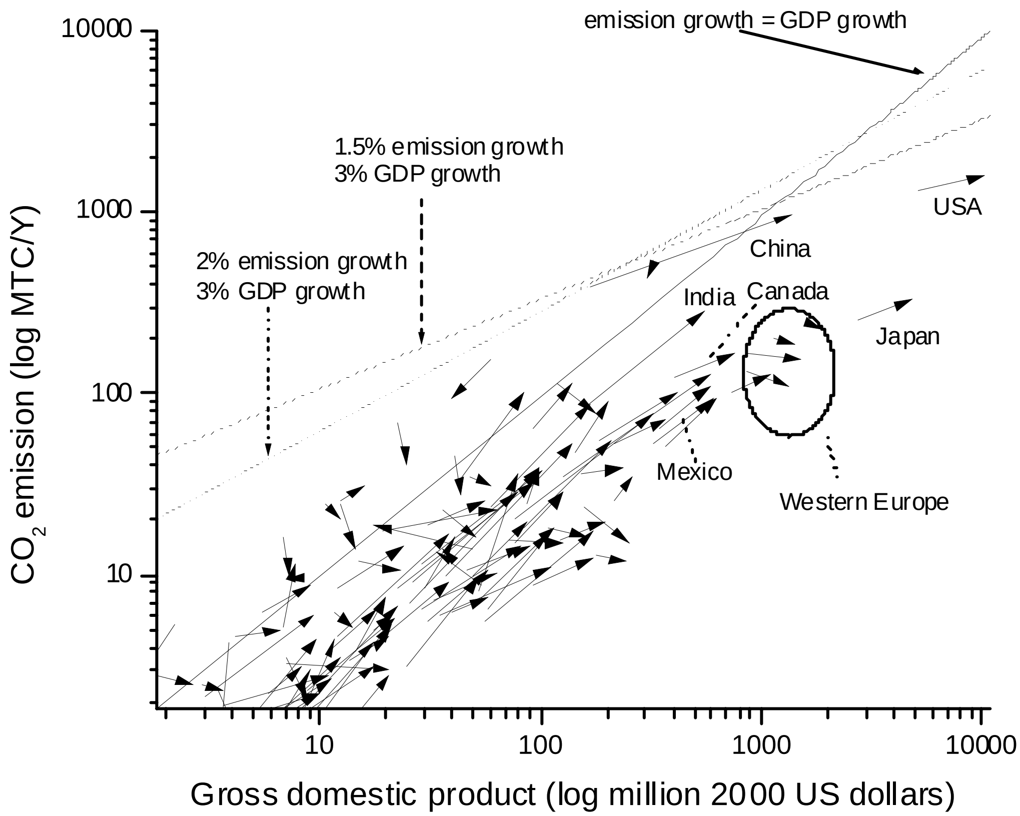


**Figure 2-4. The spatial distribution of ocean $CO_2$ exchange from 1992–1996, for several regions and measurement approaches.** Tak99 and Tak02 are from (Takahashi *et al.***,** 2002) $\Delta pCO_2$ estimates, T3L1 and T3L2 are from (Gurney *et al.***,** 2003**;** Gurney *et al.***,** 2004), Fwd is from predictive ocean models, JI is from the ocean atmosphere ocean inversions of (Jacobson *et al.***,** 2006). The far right column is the sum of the individual ocean basins toward the left [from (Jacobson *et al.***,** 2006)].

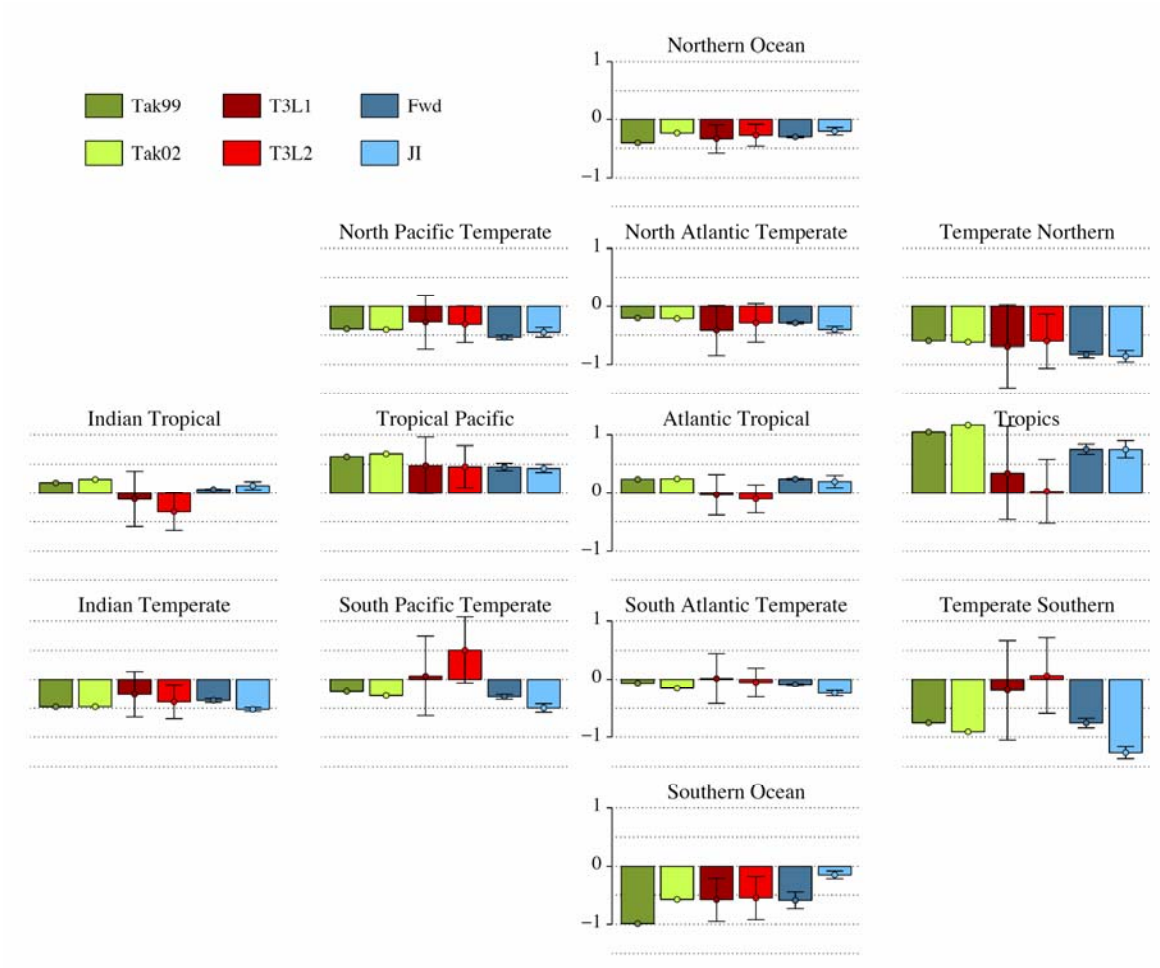
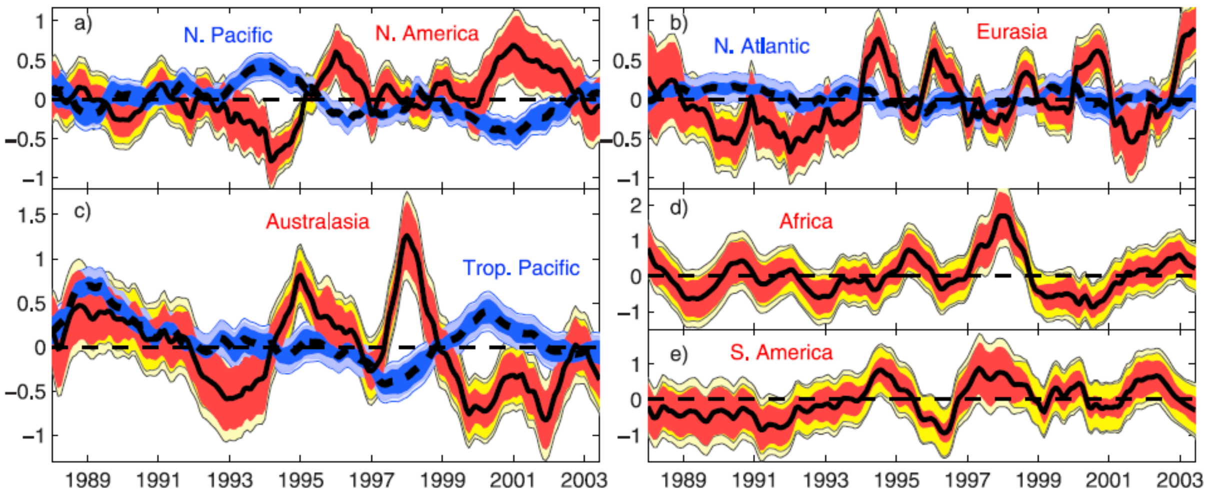


**Figure 2-5. The 13-model mean $CO_2$ flux interannual variability (Gt C $yr^{-1}$) for several continents (solid lines) and ocean basins (dashed lines)** (a) North Pacific and North America, (b) Atlantic north of 15ºN and Eurasia, (c) Australasia and Tropical Pacific, (d) Africa, and (e) South America (note the different scales for Africa and South America) [from (Baker *et al*., 2006)].



# Chapter 3. The North American Carbon Budget Past and Present

**Coordinating Lead Author: Stephen Pacala[1]**

**Lead Authors: Richard Birdsey,[2] Scott Bridgham,[3] Richard T. Conant,[4] Kenneth Davis,[5] Burke Hales,[6] Richard Houghton,[7] J. C. Jenkins,[8] Mark Johnston,[9] Gregg Marland,[10] Keith Paustian,[4] and Steven C. Wofsy[11]**

**Contributing Authors: John Caspersen,[12] Robert Socolow,[13] and Richard S. J. Tol[14]**

[1]Department of Ecology and Evolutionary Biology, Princeton University, [2]USDA Forest Service, [3]Center for Ecology and Evolutionary Biology, University of Oregon, [4]Natural Resource Ecology Laboratory, Colorado State University, [5]Department of Meteorology, The Pennsylvania State University, [6]College of Oceanic and Atmospheric Sciences, Oregon State University, [7]Woods Hole Research Center, [8]The Rubenstein School of Environment and Natural Resources, Gund Institute for Ecological Economics, University of Vermont, [9]Saskatchewan Research Council, [10]Department of Engineering, Physics and Mathematics, Mid Sweden University, [11]Atmospheric and Environmental Science (FAS), Harvard University, [12]Faculty of Forestry, University of Toronto, [13]Department of Mechanical and Aerospace Engineering and Princeton Environmental Institute, Princeton University, [14]Research Unit Sustainability and Global Change, Hamburg University

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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 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 carbon. 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.

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## INTRODUCTORY SUMMARY

### 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 16.5% of the global land area, 7.4% of the global population, and 25.0% of global GDP (EIA, 2005).

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

The United States is the world's largest emitter in absolute terms, with approximately one-quarter of the global total. 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 ton 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 continue to grow at close to the North American average rate of ~1.0% per year, but U.S. per capita emissions have been roughly constant for the past 30 years, while the carbon intensity of the U.S. economy has decreased at a rate of ~2% per year (see Figs. 3-1 to 3-3).

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). Also, a large part of the decline in the carbon intensity of the U.S. economy was 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 carbon emissions per dollar of economic activity only 15% that of manufacturing (Figs. 3-3b to 3-3c). This implies that emissions growth is essentially decoupled from economic growth. Also, 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 speaks to the issue of our technological readiness to achieve an emissions target. For example, a portfolio of technologies able to reduce the 1% annual growth in emissions to 0%, might be insufficient if carbon emissions were to begin rising at the ~3% growth rate of GDP (Pacala and Socolow, 2004).

## Carbon Sinks (see Table 3-1 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 (i.e., 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 are small net emitters to the atmosphere at the continental scale (19 Mt C $yr^{-1}$) (see Chapter 15). However, much of the $CO_2$ absorbed from or emitted to the air by coastal waters is part of the natural carbon cycle of the oceans, and so coastal sea-air exchanges should also be excluded from the continental carbon sink.

As reported in Chapter 2, all of the world's continents collectively absorbed a net of approximately 1500 Mt C $yr^{-1}$ of atmospheric $CO_2$ during the 1990s. However, because this value includes the losses of 1000–2000 Mt C $yr^{-1}$ caused by tropical deforestation (Archard *et al*., 2002; DeFries *et al*., 2002; Houghton, 2003b), carbon sinks during the 1990s actually totaled 2500–3500 Mt C $yr^{-1}$. North America's net absorption of more than 700 Mt C $yr^{-1}$ thus represents 20–30% of the global total on 16.5% of the global land area. Similarly, the United States was responsible for 17–24% of the global total despite having only 6.5% of the land area (Table 3-1). 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.**

The magnitude of the North American sink documented in Table 3-1 offers the possibility that significant carbon mitigation 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 value reported in Table 3-1. 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 four significant components of the carbon budget in Canada and six in Mexico (see Table 3-1).

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 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.

## NORTH AMERICAN FOSSIL FUEL EMISSIONS

Fossil fuel emissions currently dominate the net carbon balance in the United States, Canada, and Mexico (Fig. 3-1, Table 3-1). Fossil emissions are more than three times larger than the net carbon sink in the United States, marginally larger than the net sink in Canada, and twice as large as the net deforestation source in Mexico. 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).

Carbon dioxide emissions continue to grow in North America at close to their 30-year average of 1.0% per year. Figure 3-2 shows the growth of GDP and $CO_2$ emissions in more than 100 countries from 1980 (tail of each arrow) until 2003 (arrow head). The vertical distance between the solid diagonal line and the average position of an arrow is inversely related to the country's relative carbon intensity. Note that the United States is no outlier in this respect. Also, the slope of an arrow shows the rate of emissions growth relative to the rate of economic growth—the flatter the slope, the faster the country's carbon intensity is decreasing. Thus, countries vertically close to the line have higher carbon intensities than countries far from the line. Note that the United States has a flatter slope than many countries including Japan, but that several other industrialized counties actually have growing GDP and declining emissions (the circled arrows).

**Figure 3-2. GDP in 2000 U.S. dollars vs fossil fuel carbon emissions (Mt C $yr^{-1}$).** Data from EIA (2005). Each arrow shows the sequence from 1980 to 2003 for a country. Note that carbon emissions per unit GDP decelerate as a country gains wealth. The lines in the figure show the slopes associated with the different ratios of GDP and emissions growth (the y-intercepts of the dotted and dashed lines are not important; we moved the lines representing different ratios of GDP and emissions growth to higher y-intercepts so as not to obscure the data summarized by the arrows).

Historical decreases in U.S. carbon intensity began early in the 20th century and continue despite the approximate stabilization of per capita emissions (Fig. 3-3a). 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. 3-3b). Because the service sector has a much lower carbon intensity than manufacturing (a factor of 6.5 in 2002; compare Figs. 3-3b and 3-3c), 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. 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. 3-3b and 3-3c). 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. 3-3c; 30-year average of 1.0% per year), while emissions from transportation are growing faster than the population but slower than GDP (Fig. 3-3c; 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 3-3. (a) 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 (red 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). (b) Historical U.S. GDP divided among the manufacturing, services and agricultural sectors. *Source*: Mitchell (1998) and WRI (2005). (c) Historical U.S. carbon emissions divided among the residential, services, manufacturing, and transportation sectors. *Source*: EIA (2005).

## NORTH AMERICAN CARBON SINK

Appendix 3A contains an overview of the historical development of the sinks in U.S. and Canadian ecosystems and the source from ongoing deforestation in Mexico. The remainder of this chapter focuses on current values. To estimate non-fossil sources and sinks, we rely exclusively on 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. 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).

The current emissions of carbon by the United States, Canada, Mexico, and North America are listed in Table 3-1, and the much larger current stocks of ecosystem carbon are listed in Table 3-2 (note the change of units from millions of tons of carbon per year in Table 3-1 to billions of tons of carbon in Table 3-2). **In Table 3-1, a negative number indicates a carbon sink, and a positive number indicates a carbon source.**

**Table 3-2. Carbon stocks in North America in billions of tons.**

## 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) plus 23 Mt C $yr^{-1}$ from urban and suburban trees (Chapter 14). The total estimate of the carbon sink in forested ecosystems is –259 Mt C $yr^{-1}$ and includes a sink of 90 Mt C $yr^{-1}$ 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 now 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. 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). 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.

The two studies of 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.

## 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 1.5 to –6 Mt C $yr^{-1}$ in the United States.

## Wetlands

Peatlands are wetlands that have accumulated deep soil carbon deposits over thousands of years because decomposition in them is less than plant productivity. Thus, wetlands form the largest carbon pool of any North American ecosystem (Table 3-2). 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 report is on carbon fluxes without a consideration of the radiative forcing of different greenhouse gases [i.e., global warming potential (GWP)]. However, wetlands are naturally an important source of methane ($CH_4$). The GWP of a gas depends on its instantaneous radiative forcing and its lifetime in the atmosphere, with methane having GWPs of 1.9 and 16.9 $CO_2$-C equivalents on 500-year and 20-year time frames, respectively (Ramaswamy et al., 2001). 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 9). 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 reservoirs, alluvium, and colluvium 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 preindustrial 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

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.

## REFERENCES


Aldy, J. E. (2005). "An Environmental Kuznets Curve Analysis of US State Level Carbon Dioxide Emissions," Journal of Environment and Development, 14(1), 58–72.

Ang, B. W., and F. Q. Zhang (2000). "A Survey of Index Decomposition Analysis in Energy and Environmental Studies," Energy, 25, 1149_1176.

Archard, F., H. D. Eva, H-J. Stibig, P. Mayaux, J. Gallego, T. Richards, and J-P. Malingreau. 2002. Determination of Deforestation Rates of the World's Humid Tropical Forests. Science, 297, 999–1002.

Baldocchi, D., E. Falge, L. H. Gu, R. Olson, D. Hollinger, S. Running, P. Anthoni, C. Bernhofer, K. Davis, R. Evans, J. Fuentes, A. Goldstein, G. Katul, B. Law, X. H. Lee, Y. Malhi, T. Meyers, W. Munger, W. Oechel, K.T. Paw U, K. Pilegaard, H. P. Schmid, R. Valentini, S. Verma, T. Vesala, K. Wilson, and S. Wofsy. 2001. FLUXNET: A new tool to study the temporal and spatial variability of ecosystem-scale carbon dioxide, water vapor, and energy flux densities, Bull. Am. Meterol. Soc., 82, 2415–2434.

Barford, C.C., S. C. Wofsy, M. L. Goulden, J. W. Munger, E. H. Pyle, S. P. Urbanski, L. Hutyra, S. R. Saleska, D. Fitzjarrald, and K. Moore (2001). Factors controlling long- and short-term sequestration of atmospheric CO2 in a mid-latitude forest, Science, 294, 1688–1691.

Birdsey, R. A., and L. S. Heath. _____. "Carbon Changes in U.S. Forests," in L. A. Joyce (ed.), Productivity of America's Forests and Climate Change, General Technical Report RM-GTR-271 (Fort Collins, CO: USDA Forest Service, 1995).

Birdsey, Richard A., and George M. Lewis. 2003. Current and historical trends in use, management, and disturbance of U.S. forestlands, in J. M. Kimble, L. S. Heath, and R. A. Birdsey (eds.), The potential of U.S. forest soils to sequester carbon and mitigate the greenhouse effect, CRC Press LLC, New York, 15–33.

Bradley, B. A., R. A. Houghton, J. F. Mustard, and S. P. Hamburg. ____. Invasive grass reduces carbon stocks in shrublands of the Western U.S. In prep. [In press in time for this report.]

Cairns, M. A., P. K. Haggerty, R. Alvarez, B. H. J. De Jong, and I. Olmsted. 2000. Tropical Mexico's Recent Land-Use Change: A Region's Contribution to the Global Carbon Cycle, Ecological Applications, 10(5), 1426–1441.

Canadell, J. G., H. A. Mooney, D. D. Baldocchi, J. A. Berry, J. R. Ehleringer, C. B. Field, S. T. Gower, D. Y. Hollinger, J. E. Hunt, R. B. Jackson, S. W. Running, G. R. Shaver, W. Steffen, S. E. Trumbore, R. Valentini, B. Y. Bond. 2000. Carbon metabolism of the terrestrial biosphere: A multitechnique approach for improved understanding, Ecosystems, 3, 115–130.

Casler, S. D., and A. Z. Rose (1998). "Carbon Dioxide Emissions in the US Economy," Environmental and Resource Economics, 11(3–4), 349–363.

Caspersen, J. P., S. W. Pacala, J. C. Jenkins, G. C. Hurtt, P. R. Moorcroft, and R. A. Birdsey. 2000. Contributions of land-use history to carbon accumulation in U.S. forests, Science, 290, 1148–1151.

Cook, B. D., K. J. Davis, W. Wang, A. Desai, B. W. Berger, R. M. Teclaw, J. G. Martin, P. V. Bolstad, P. S. Bakwin, C. Yi, and W. Heilman. 2004. Carbon exchange and venting anomalies in an upland deciduous forest in northern Wisconsin, USA, Agricultural and Forest Meteorology, 126, 271–295.

Cox, P. M., R. A. Betts, C. D. Jones, S. A. Spall, and I. J. Totterdell. 2000. Acceleration of global warming due to carbon-cycle feedbacks in a coupled climate model, Nature, 408, 184–187.

Curtis, P. S., P. J. Hanson, P. Bolstad, C. Barford, J. C. Randolph, H. P. Schmid, K. B. Wilson (2002). Biometric and eddy-covariance based estimates of annual carbon storage in five eastern North American deciduous forests, Agricultural and Forest Meteorology, 113, 3–19.

Davis, W. B., A. H. Sanstad, and J. G. Koomey (2002), "Contributions of Weather and Fuel Mix to Recent Declines in US Energy and Carbon Intensity," Energy Economics, 25, 375–396.

Defries, R. S., R. A. Houghton, M. C. Hansen, C. B. Field, D. Skole, and J. Townshend. 2002. Carbon emissions from tropical deforestation and regrowth based on satellite observations for the 1980s and 1990s, PNAS, 99(22), 14256–14261.

Ehman, J. L., H. P. Schmid, C. S. B. Grimmond, J. C. Randolph, P. J. Hanson, C. A. Wayson, F. D. Cropley (2002). An initial intercomparison of micrometerological and ecological inventory estimates of carbon exchange in a mid-latitude deciduous forest, Global Change Biol., 8, 575–589.

EIA (2005). Historical Data Overview, Energy Information Agency, U.S. Department of Energy, http://www.eia.doe.gov/overview_hd.html; ftp://ftp.eia.doe.gov/pub/oiaf/1605/cdrom/pdf/ggrpt/057304.pdf

Environment Canada. 2005. Canada's Greenhouse Gas Inventory 1990–2003: Initial Submission. Greenhouse Gas Division, Environment Canada, Ottawa, ON. Available at: http://unfccc.int/national_reports/annex_i_ghg_inventories/national_inventories_submissions/items/2761.php.

Fan, S.-M., M. Gloor, J. Mahlman, S. Pacala, J. Sarmiento, T. Takahashi, and P. Tans. 1998. Atmospheric and Oceanic CO2 Data and Models Imply a Large Terrestrial Carbon Sink in North America, Science, 282, 442–446.

Golove, W. H. and L. J. Schipper (1998). "Long-Term Trends in US Manufacturing Energy Consumption and Carbon Dioxide Emissions," Energy, 21(7/8), 683–692.

Goodale, C. L., M. J. Apps, R. A. Birdsey, C. B. Field, L. S. Heath, R. A. Houghton, J. C. Jenkins, G. H. Kohlmaier, W. Kurz, S. Liu, G. J. Nabuurs, S. Nilsson, and A. Z. Shvidenko. 2002. Forest Carbon Sinks in the Northern Hemisphere, Ecological Applications, 12(3), 891–899.

Gough, C. M., P. S. Curtis, J. G. Vogel, H. P. Schmid, and H. B. Su (in review). Annual carbon storage from 1999 to 2003 in a Northern hardwood forest assessed using eddy-covariance and biometric methods, Agricultural and Forest Meteorology.

Greening, L. A., W. B. Davis, L. Schipper, and M. Khrushch (1997). "Comparison of Six Decomposition Methods: Application to Aggregate Energy Intensity for Manufacturing in 10 OECD Countries," Energy Economics, 19(3), 375–390.

Greening, L. A., W. B. Davis, and L. Schipper (1998). "Decomposition of Aggregate Carbon Intensity for the Manufacturing Sector: Comparison of Declining Trends from 10 OECD Countries for the Period 1971–1993," Energy Economics, 20(1), 43–65.

Greening, L. A., M. Ting, and W. B. Davis (1999). "Decomposition of Aggregate Carbon Intensity for Freight: Trends from 10 OECD Countries for the Period 1971–1993," Energy Economics, 21(4), 331–361.

Greening, L. A., M. Ting, and T. J. Krackler (2001). "Effects of Changes in Residential End-Uses on Aggregate Carbon Intensity: Comparison of 10 OECD Countries for the Period 1970 through 1993," Energy Economics, 23(2), 153–178.

Greening, L. A. (2004). "Effects of Human Behaviour on Aggregate Carbon Intensity of Personal Transportation: Comparison of 10 OECD Countries for the Period 1970–1993," Energy Economics, 26(1), 1–30.

Grossman, G. M., and A. B. Krueger (1995). "Economic Growth and the Environment," Quarterly Journal of Economics, 60(2), 353–375.

Guo, L. B., and R. M. Gifford. 2002. Soil carbon stocks and land use change: a meta analysis, Global Change Biology, 8(4), 345–360.

Gurney, K. R., R. M. Law, A. S. Denning, P. J. Rayner, B. C. Pak, D. Baker, P. Bousquet, L. Bruhwiler, Y. H. Chen, P. Ciais, I. Y. Fung, M. Heimann, J. John, T. Maki, S. Maksyutov, P. Peylin, M. Prather, and S. Taguchi. 2004. Transcom 3 inversion intercomparison: Model mean results for the estimation of seasonal carbon sources and sinks, Global Biogeochemical Cycles, 18, GB1010.

Horst, T. W., and J. C. Weil. 1994. How far is far enough? The fetch requirements for micrometeorological measurement of surface fluxes, Journal of Atmospheric & Oceanic Technology, 11, 1018–1025.

Houghton, R. A., J. L. Hackler, and K. T. Lawrence. 1999. The U.S. carbon budget: contributions from land-use change, Science, 285, 574–578.

Houghton, R. A., and J. L. Hackler. 2000. Changes in terrestrial carbon storage in the United States. 1. The roles of agriculture and forestry, Global Ecology and Biogeography, 9, 125–144.

Houghton, R. A., J. L. Hackler, and K. T. Lawrence. 2000. Changes in terrestrial carbon storage in the United States. 2. The role of fire and fire management, Global Ecology and Biogeography, 9, 145–170.

Houghton, R. A. 2003a. Revised estimates of the annual net flux of carbon to the atmosphere from changes in land use and land management 1850–2000, Tellus B, 55(2), 378–390.

Houghton, R. A. 2003b. Why are estimates of the terrestrial carbon balance so different? Global Change Biology, 9(4), 500–509.

Hungate, B. A., J. S. Dukes, M. R. Shaw, Y. Luo, and C. B. Field. 2003. Nitrogen and Climate Change. Science, 302, 1512–1513.

Hurtt, G. C., S. W. Pacala, P. R. Moorcroft, J. Caspersen, E. Shevliakova, R. A. Houghton, and B. Moore III. 2002. Projecting the future of the U.S. carbon sink, Proceedings of the National Academy of Sciences, 99, 1389–1394.

Jackson, R. B., J. L. Banner, E. G. Jobbagy, W. T. Pockman, and D. H. Wall. 2002. Ecosystem carbon loss with woody plant invasion of grasslands, Nature, 418(6898), 623–626.

Kahn, M. E. (2003). "The Geography of US Pollution Intensive Trade: Evidence from 1958 to 1994," Regional Science and Urban Economics, 33, 383–400.

Körner, C., R. Asshoff, O. Bignucolo, S. Hättenschwiler, S. G. Keel, S. Peláez-Riedl, S. Pepin, R. T. W. Siegwolf, and G. Zotz. 2005. Carbon Flux and Growth in Mature Deciduous Forest Trees Exposed to Elevated CO2. Science, 309, 1360–1362.

Kulshreshtha, S. N., B. Junkins, and R. Desjardins. 2000. Prioritizing greenhouse gas emission mitigation measures for agriculture, Agricultural Systems, 66(3), 145–166.

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

Law, B. E., P. E. Thornton, J. Irvine, P. M. Anthoni, and S. Van Tuyl (2001). Carbon storage and fluxes in ponderosa pine forests at different developmental stages, Global Change Biol., 7, 755–777.

Lenzen, M., M. Wier, C. Cohen, H. Hayami, S. Pachauri, and R. Schaeffer (2006). "A Comparative Multivariate Analysis of Household Energy Requirements in Australia, Brazil, Denmark, India and Japan," Energy, 31, 181–207.

Lindmark, M. (2004). "Patterns of Historical CO2 Intensity Transitions among High and Low Income Countries," Explorations in Economic History, 41, 426–447.

Luo, Y., D. Hui, and D. Zhang. 2006. Elevated Carbon Dioxide Stimulates Net Accumulations of Carbon and Nitrogen in Terrestrial Ecosystems: A Meta-Analysis, Ecology (forthcoming in the 1st issue).

Masera, O. R., M. J. Ordonez, and R. Dirzo. 1997. Carbon Emissions from Mexican Forests: Current Situation and Long-Term Scenarios, Climate Chante, 35, 265–295.

Maddison, A. (2003). The World Economy: Historical Statistics, OECD, Paris.

Marland, G., T. A. Boden, and R. J. Andres (2005). "Global, Regional and National CO2 Emissions" in Trends: A Compendium of Data on Global Change, Oak Ridge National Laboratory, Oak Ridge, http://cdiac.esd.ornl.gov/trends/emis/em_cont.htm

Mitchell, B. R. (1998). International Historical Statistics: The Americas 1750–1993, 4th Edition, Stockton Press, New York.

Oren, R., D. S. Ellsworth, K. H. Johnsen, N. Phillips, B. E. Ewers, C. Maier, K. V. R. Schäfer, H. McCarthy, G. Hendrey, S. G. McNulty, and G. G. Katul. 2001. Soil fertility limits carbon sequestration by forest ecosystems in a CO2-enriched atmosphere, Nature, 411, 469–478.

Pacala, S. W., G. C. Hurtt, R. A. Houghton, R. A. Birdsey, L. Heath, E. T. Sundquist, R. F. Stallard, D. Baker, P. Peylin, P. Moorcroft, J. Caspersen, E. Shevliakova, M. E. Harmon, S.-M. Fan, J. L. Sarmiento, C. Goodale, C. B. Field, M. Gloor, and D. Schimel. 2001. Consistent Land- and Atmosphere-Based U.S. Carbon Sink Estimates. Science, 292(5525), 2316–2320.

Pacala, S. W., and R. H. Socolow. 2004. Stabilization Wedges: Solving the Climate Problem for the next 50 Years with Current Technologies, Science, 305(5686), 968–972.

Ramaswamy, V., O. Boucher, J. Haigh, D. Hauglustaine, J. Haywood, G. Myhre, T. Nakajima, G. Y. Shi, and S. Solomon. 2001. Radiative forcing of climate change, in Climate Change 2001: The Scientific Basis, contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change (J. T. Houghton, J. T. Y. Ding, D. J. Griggs, M. Noguer, P. J. van der Linden, X. Dai, K. Maskell, and C. A. Johnson, eds.), 349–416, Cambridge: Cambridge University Press.

Rothman, D. S. (1998). "Environmental Kuznets Curves—Real Progress or Passing the Buck? A Case for Consumption-Based Approaches," Ecological Economics, 25, 177–194.

Selden, T. M., and D. Song (1994). "Environmental Quality and Development—Is There a Kuznets Curve for Air Pollution Emissions?," Journal of Environmental Economics and Management, 27, 147–162.

Skog, K. E., and G. A. Nicholson. 1998. Carbon cycling through wood products: the role of wood and paper products in carbon sequestration, Forest Products Journal 48, 75–83 (http://www.fpl.fs.fed.us/documnts/pdf1998/skog98a.pdf).

Skog, Kenneth E., Kim Pingoud, and James E. Smith. 2004. A method countries can use to estimate changes in carbon stored in harvested wood products and the uncertainty of such estimates, Environmental Management, 33, Supplement 1: S65–S73.

Smith, W. Brad, Patrick D. Miles, John S. Vissage, and Scott A. Pugh. 2004. Forest Resources of the United States, 2002, Gen. Tech. Rep. NC-241, St. Paul, MN, U.S. Department of Agriculture, Forest Service, p. 137.

Smith, James E., and Linda S. Heath. 2005. Land use change and forestry and related sections (Excerpted), in: U.S. Environmental Protection Agency, Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990–2003, EPA 430-R-05-003. Available at http://yosemite.epa.gov/oar/globalwarming.nsf/content/ResourceCenterPublicationsGHGEmissionsUSEmissionsInventory2005.html

Stallard, R. F. 1998. Terrestrial sedimentation and the carbon cycle: Coupling weathering and erosion to carbon burial, Global Biogeochemical Cycles, 12(2), 231.

Suri, V., and D. Chapman (1998). "Economic Growth, Trade and Energy: Implications for the Environmental Kuznets Curve," Ecological Economics, 25(2), 195–208.

Verma, S. B., A. Dobermann, K. G. Cassman, D. T. Walters, J. M. Knops, T. J. Arkebauer, A. E. Suyker, G. G. Burba, B. Amos, H. S. Yang, D. Ginting, K. G. Hubbard, A. A. Gitelson, and E. A. Walter-Shea (2005). Annual carbon dioxide exchange in irrigated and rainfed maize-based agroecosystems, Agricultural and Forest Meteorology, 131, 77–96.

Wofsy, S. C., M. L. Goulden, J. W. Munger, S.-M. Fan, P. S. Bakwin, B. C. Daube, S. L. Bassow, and F. A. Bazzaz. 1993. Net exchange of CO2 in a mid-latitude forest, Science, 260, 1314–1317.

WRI (2005). EarthTrends—The Environmental Information Portal, World Resource Institute http://earthtrends.wri.org/

**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] | –283 |
| 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 sink *** | –506 | –134 | 48 | –592 |
| Net horizontal exports (negative) or imports (positive) | | | | |
| Wood products**** | 14[e] | –74[c] | –1[d] | –61 |
| Agriculture products*** | –65[k] | ND | ND | –65 |
| Rivers to ocean** | –35[k] | ND | ND | –35 |
| Total net absorption** (Sink plus exports) | –592 | –208 | 47 | –753 |
| Net absorption (negative) or emission (positive) by coastal waters **** | ND | ND | ND | 19[l] |

Uncertainty:

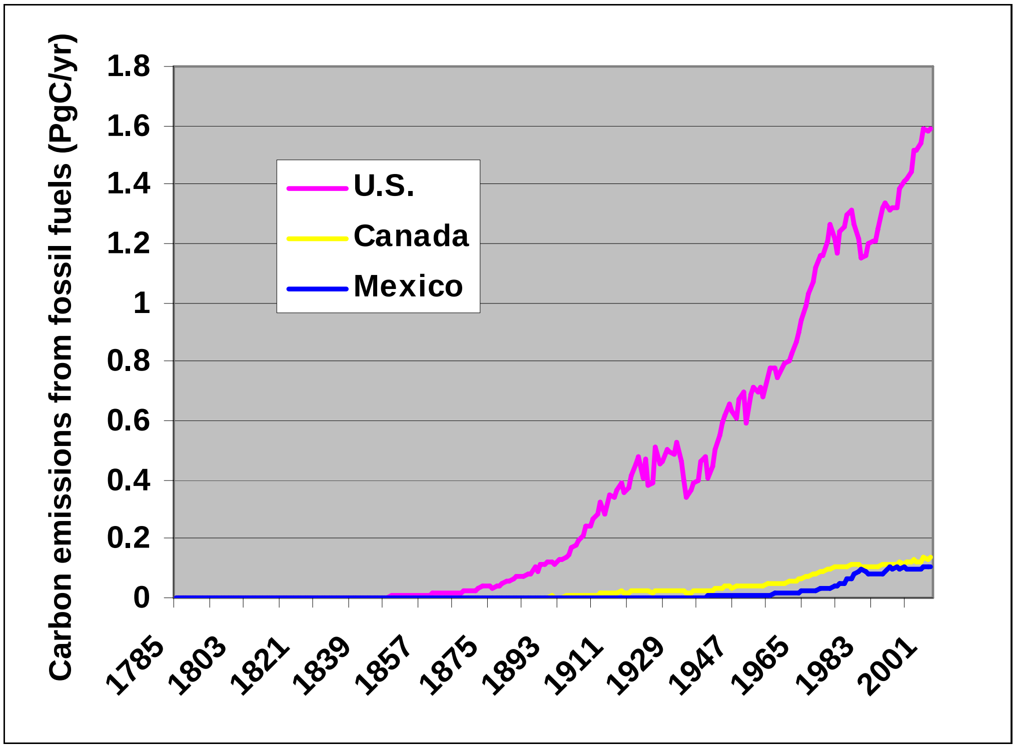
*****(95% confidence within 10%)
****(95% confidence within 25%)
***(95% confidence within 50%)
**(95% confidence within 100%)
*(95% confidence bounds >100%)
ND = No data available
[a]http://www.eia.doe.gov/env/inlenv.htm
[b]Smith and Heath (2005) for above ground carbon, but including 23 Mt C/yr$^{-1}$ 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
[i]Chapter 13
[j]Stallard, 1998; Pacala *et al*. (2001)
[k]Pacala *et al*. (2001)
[l]Chapter 15

**Table 3-2. 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 |

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

1




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

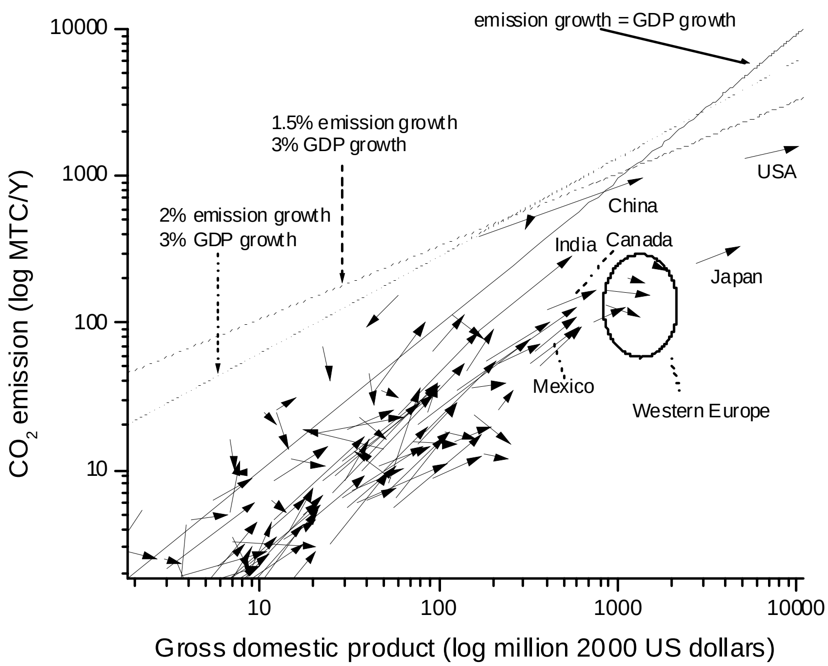
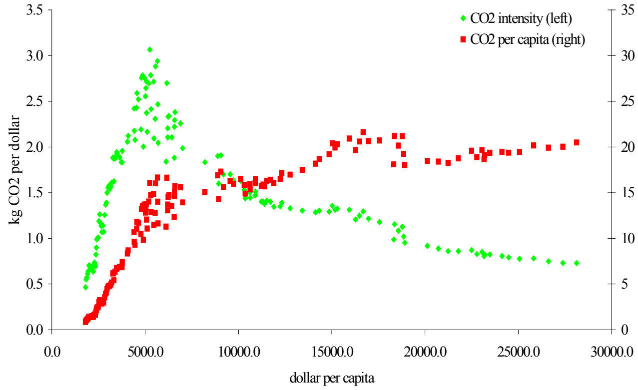
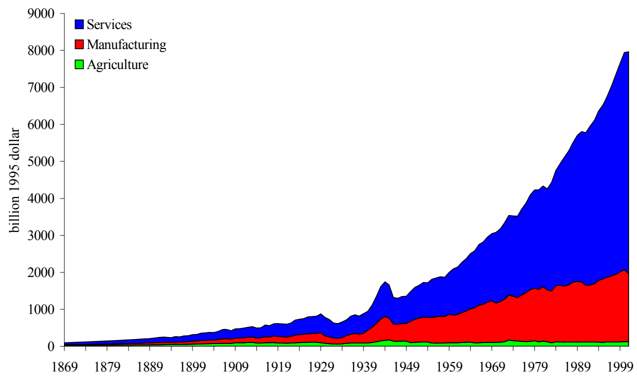


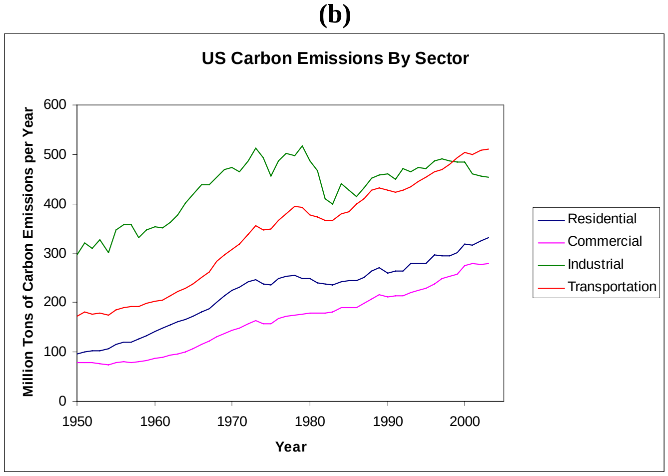

**Fig. 3-2. GDP in 2000 U.S. dollars vs fossil fuel carbon emissions (Mt C/yr$^{-1}$).** Data from EIA (2005). Each arrow shows the sequence from 1980 to 2003 for a country. Note that carbon emissions per unit GDP decelerate as a country gains wealth. The lines in the figure show the slopes associated with the different ratios of GDP and emissions growth (the y-intercepts of the dotted and dashed lines are not important; we moved the lines representing different ratios of GDP and emissions growth to higher y-intercepts so as not to obscure the data summarized by the arrows).




**(a)**




**(b)**




**(c)**

**Fig. 3-3. (a) 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 (red 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). (b) Historical U.S. GDP divided among the manufacturing, services, and agricultural sectors. *Source*: Mitchell (1998), WRI (2005). (c) Historical U.S. carbon emissions divided among the residential, services, manufacturing, and transportation sectors. *Source*: EIA (2005).

# 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 (Fig. 3-2). 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).

# 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. 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 better agreement. 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).

**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) |

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

**Coordinating Lead Author: Erik Haites[1]**

**Lead Authors: Ken Calderia,[2] Patricia Romero Lankao,[3] Adam Rose,[4] and Tom Wilbanks[5]**

**Contributing Authors: Skip Laitner,[6] Richard Ready,[4] and Roger Sedjo[7]**

[1]Margaree Consultants, Inc., [2]Carnegie Institution, [3]Metropolitan Autonomous University—Xochimilco,
[4]The Pennsylvania State University, [5]Oak Ridge National Laboratory,
[6]U.S. Environmental Protection Agency, [7]Resources for the Future

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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 fee; (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 fee 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.

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## INTRODUCTION

This chapter provides an overview of measures that can reduce $CO_2$ and $CH_4$ 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

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.

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 electricity and hydrogen become the major energy carriers. They are 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.

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

- efficiency improvement,

[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.

- 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 directly or indirectly.[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.

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[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 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, 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, use wood waste and sawdust as fuel. Ethanol, currently produced mainly from corn, is blended with gasoline. The $CO_2$ emission reduction achieved depends on whether the biomass used is replaced, on the fossil-fuel energy used to produce the fuel, and the carbon content of the fuel displaced.

Carbon-free energy sources include hydro,[7] wind, solar, biomass, geothermal, and nuclear fission. 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.

## 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.

[7] Reservoirs for hydroelectric generation produce $CO_2$ and methane emissions, so such sources are not totally carbon free.

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 molecules 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.[8] 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 to safely and cost effectively integrate $CO_2$ 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 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 and used as an energy source. Methane emissions also occur during production of coal, oil, and natural gas. Such emissions usually can be flared (though this generates $CO_2$) or collected for use as an energy source. 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.

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

## 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, 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 the soil type. 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.[9] 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.

## INTEGRATED COMPARISON OF OPTIONS

As is clear from the previous sections, there are thousands of options to reduce emissions of or to sequester $CO_2$. To help them decide which options to implement, policy makers need to know which 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 on other measures as well, 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.

[9]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%.

Changes to the age structure of the population, increases in per capita incomes, and other factors can affect the potential for some options as well. 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 near 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 standardized to a common unit of measurement—2004 U.S. dollars per metric ton of carbon.[10]

**Table 4-1. Standardized cost estimates [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. So the estimated scale of

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

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 mitigation 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 individual chapters, along with some estimates of their effects 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 a permit 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 Annex I (industrialized countries only) case for the U.S. economy.

The task of choosing the "best" combination of options may seem daunting given the numerous options and the associated cost ranges. This combination will depend on several factors including the emission target, the emitters covered, the compliance period, and the ancillary benefits 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 permit trading, that cover a large number of emitters and allow them to choose the lowest cost reduction options.

## POLICY OPTIONS

### Overview

Stabilizing the carbon cycle will require very substantial reductions and increased sequestration of $CO_2$ emissions. 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. Policies will also need to encourage research and development of technologies that can reduce emissions even further in the long term.

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). A portfolio of options will need to be implemented, including greater efficiency in the production and use of energy; expanded use of renewable energy technologies; technologies for removing carbon from fossil fuels and sequestering it in geological formations; various changes in forestry, agricultural, and land use practices; and possibly other approaches, some of which are currently very controversial, such as nuclear power and certain types of "geoengineering."

Because $CO_2$ has a long atmospheric residence time,[11] immediate action to reduce emissions and increase sequestration allows its atmospheric concentration to be stabilized at a lower level.[12] 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.

## 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). Information and voluntary programs are generally not environmentally effective[13] (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 emissions trading system 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.

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[11] $CO_2$ 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)

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

[13] 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).

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

The choice of policy instrument needs to consider 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, and employment, or in more technical measures of economic well-being (e.g., "welfare measures" such as "compensating variation"). 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.[14] The cost can have competitiveness impacts if some emitters in other jurisdictions are not subject to similar policies. But the societal benefits, such as improved public health and reduced environmental damage, may exceed 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).[15] 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.[16] A policy that raises revenue

[14]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.

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

[16]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.

(an emissions tax or auctioned allowances) has a lower macroeconomic cost than a policy that does not, if the revenue is used to reduce existing distortionary taxes 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.

$CO_2$ 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.

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).[17] In some cases, such as refrigerators, standards have been used successfully to drive technology development.

## 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 carbon uptake. Policies that affect crop choice (support payments, crop insurance, disaster relief) and

[17]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.

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. The tradable credits can be sold to sources subject to a $CO_2$ emissions trading program or offset requirement.[18] 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);
- 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). 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 help are needed to create a market for less 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

[18] 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). Also, forests could create carbon offset credits that could be exchanged in tradable carbon systems. Some offset credits might be viewed as temporary. However, there are many circumstances where temporary credits would be valuable additions to a carbon reduction portfolio.

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.[19]

## 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;
- 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.

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 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 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.

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

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[19] 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.

- Encourage adoption of cost-effective emission reduction and sink enhancement measures. An emissions trading program or emission fee 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.
- Adopt appropriate regulations to complement the emissions trading program or emission fee 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.

## REFERENCES

Bohm, P. and C. Russell. 1986. "Comparative Analysis of Alternative Policy Instruments," in A. Kneese and J. Sweeney (eds.), *Handbook of Natural Resource and Energy Economics*, Vol. 2. New York: Elsevier.

Congressional Budget Office (CBO). 2003: "The Economic Costs of Fuel Economy Standards Versus a Gasoline Tax," Congress of the United States, Washington, DC.

DOE. 2006. www.fossil.energy.gov/programs/sequestration/overview.html Accessed on March 27, 2006. US Department of Energy.

Energy Information Administration (EIA), 2003. *International Energy Outlook: 2003*, DOE/EIA-0484(2003), U.S. Department of Energy, Washington, D.C., May.

Energy Modeling Forum. 2002. Stanford University, Palo Alto, CA.

Environmental Protection Agency (EPA). 2005. *Greenhouse Gas Mitigation Potential in U.S. Forestry and Agriculture*. Washington, DC: U.S. Environmental Protection Agency. 154 p.

Feng, Hongli, C.L. Kling, L.A. Kurkalova and S. Secchi 2003. "Subsidies! The Other Incentive-Based Instrument: The Case of the Conservation Reserve Program," Working Paper 03-WP 345, Center for Agricultural and Rural Development, Iowa State University, Ames, Iowa, October.

Greene, D.L., P.D. Patterson, M. Singh and J. Li. 2005. "Feebates, Rebates and Gas Guzzler Taxes: A Study of Incentives for Increased Fuel Economy." Energy Policy 33(6), 757-776.

Goulder, L. 2004, *Induced Technological Change and Climate Policy*, Pew Center on Global Climate Change, Washington, DC.

Herzog, H. 1999. "The Economics of CO2 Capture," P Reimer, B Eliasson, A Wokaum, eds., *Greenhouse Gas Control Technologies*, Oxford: Elsevier Science Ltd., pp. 101-106.

Hoffert, M.I., K. Calderia, A. K. Jain, E. F. Haites, L. D. D. Harvey, S. D. Potter, M. E. Schlesinger, S. H. Schneider, R. G. Watson, T. M. L. Wigley and D. J. Wuebbles, 1998. "Energy Implications of Future Stabilization of Atmospheric $CO_2$ Content," *Nature*, 395, 881-884.

Hoffert, M.I., K. Caldeira, G. Benford, D.R. Criswell, C. Green, H. Herzog, A.K. Jain, H.S. Kheshgi, K.S. Lackner, J.S. Lewis, H.D. Lightfoot, W. Manheimer, J.C. Mankins, M.E. Mauel, L.J. Perkins, M.E. Schlesinger, T. Volk, and T.M.L. Wigley, 2002. "Advanced Technology Paths to Global Climate Stability: Energy for a Greenhouse Planet," *Science*, 298, pp. 981-987.

Intergovernmental Panel on Climate Change (IPCC) 2000a. *Emissions Scenarios*, Special Report of Working Group III of the IPCC, Cambridge University Press, Cambridge, U.K.

Intergovernmental Panel on Climate Change (IPCC) 2000b. *Land Use, Land-Use Change, and Forestry*, Special Report of the IPCC, Cambridge University Press, Cambridge, U.K.

Intergovernmental Panel on Climate Change (IPCC) 2001a. *Climate Change 2001: The Scientific Basis*, Contribution of Working Group I to the Third Assessment report of the IPCC, Cambridge University Press, Cambridge, U.K.

Intergovernmental Panel on Climate Change (IPCC) 2001b. *Climate Change 2001: Mitigation*, Contribution of Working Group III to the Third Assessment report of the IPCC, Cambridge University Press, Cambridge, U.K.

Intergovernmental Panel on Climate Change (IPCC) 2005. *IPCC Special Report on Carbon Dioxide Capture and Storage, Summary for Policymakers*, Approved by the 8th Session of IPCC Working Group III, Montreal, Canada.

Jaccard, M., J. Nyboer, and B. Sadownik. 2002. *The Cost of Climate Policy*. Vancouver, UBC Press.

Jaccard, M., J. Nyboer, C. Bataille, and B. Sadownik. 2003. "Modeling the Cost of Climate Policy: Distinguishing Between Alternative Cost Definitions and Long-Run Cost Dynamics," *The Energy Journal* 24(1): 49-73.

Jaccard, M., R. Loulou, A. Kanudia, J. Nyboer, A. Bailie, and M. Labriet. 2003. "Methodological Contrasts in Costing GHG Abatement Policies: Optimization and Simulation Modeling of Micro-economic Effects in Canada," *European Journal of Operations Research* 145 (1): 148-164.

Lewandrowski, J., M. Sperow, M. Peters, M. Eve, C. Jones, K. Paustian, and R. House. 2004. Economics of Sequestering Carbon in the U.S. Agricultural Sector. Tech. Bull. 1909. Washington, DC: U.S. Department of Agriculture, Economic Research Service. 61 p.

Marland, G., B.A. McCarl, and U.A. Schneider, 2001. "Soil Carbon: Policy and Economics," *Climatic Change* 51(1), 101-117.

Marland, G., B.A. McCarl, and U.A. Schneider, 2001. "Soil Carbon: Policy and Economics," *Climatic Change,* 51(1), 101-117.

Martin, N., E. Worrell, M. Ruth, L. Price, R. N. Elliott, A. M. Shipley, J. Thorne. 2001. *Emerging Energy-Efficient Industrial Technologies*: New York State Edition. LBNL Report Number 46990, published by American Council for an Energy-Efficient Economy (ACEEE).

Nabuurs, G.J., A.V. Dolman, E. Verkaik, P.J. Kuikman, C.A. van Diepen, A. Whitmore, W. Daamen, O. Oenema, P. Kabat and G.M.J. Mohren, 2000. "Article 3.3 and 3.4 of the Kyoto Protocol – consequences for industrialized countries' commitment, the monitoring needs and possible side effect," *Environmental Science and Policy*, v. 3, n. 2/3, pp. 123-134.

National Academies, 2004. The Hydrogen Economy: Opportunities, Costs, Barriers, and R&D Needs, The National Academies Press, Washington, D.C.

Committee on Alternatives and Strategies for Future Hydrogen Production and Use, National Research Council, National Academy of Engineering

National Commission on Energy Policy (NCEP), 2005. Ending the Energy Stalemate: A Bipartisan Strategy to Meet America's Energy Challenge, Washington, DC.

OECD, 2003a. *Technology Innovation, Development and Diffusion*, OECD and IEA Information Paper, COM/ENV/EPOC/IEA/SLT(2003)4, Paris.

OECD, 2003b. Voluntary Approaches for Environmental Policy: Effectiveness, Efficiency and Usage in Policy Mixes, Paris.

Parry, I.W.H., R. Willians and L.H. Goulder, 1999. "When Can Carbon Abatement Policies Increase Welfare? The Fundamental Role of Distorted Factor Markets," *Journal of Environmental Economics and Management*, 37(1), 52-84.

Raupach, M. Canadell J.G, Bakker D.C., Ciais P, Sans M.J. Fank J.Y. Melillo, J.M., Romero Lankao P., Sathaye, J.A., Schulze E.D., Smith P., and Tschirley J. 2004. "Atmospheric Stabilization in the Context of Carbon-Climate-Human Interactions" in *Toward $CO_2$ Stabilization: Issues, Strategies, and Consequences*, C. Field and M. Raupach (eds.) Washington, D.C.: Island Press.

Sedjo, R.A., 2001. "Forest 'Sinks' as a Tool for Climate-Change Policymaking: A Look at the Advantages and Challenges." *Resources*, Spring 2001, Issue 143: 21-23.

Sedjo, R.A. and S.K. Swallow, 2002. "Voluntary Eco-Labeling and the Price Premium," Land Economics, 87(2), 272-284, May.

Stavins, Robert N. and Kenneth R. Richards. 2005. *The cost of U.S. forest-based carbon sequestration*. Arlington, VA: The Pew Center on Global Climate Change. 40 p. [available at www.pewclimate.org]

Tietenberg, T. 2000. *Environmental and Natural Resource Economics*, 5th Ed. New York: Addison-Wesley.

Tietenberg, T. and D. Wheeler. 2001. "Empower the Community: Information Strategies for Pollution Control," in H. Folmer, H. L. Gabel, S. Gerking, and A. Rose (eds.), *Frontiers of Environmental Economics*, Edward Elgar, Cheltenham, UK.

(US DOE/EIA) U.S. Department of Energy, Energy Information Administration. 2003: Analysis of S.139, the Climate Stewardship Act of 2003. SR/OIAF/2003-02, Washington, DC.

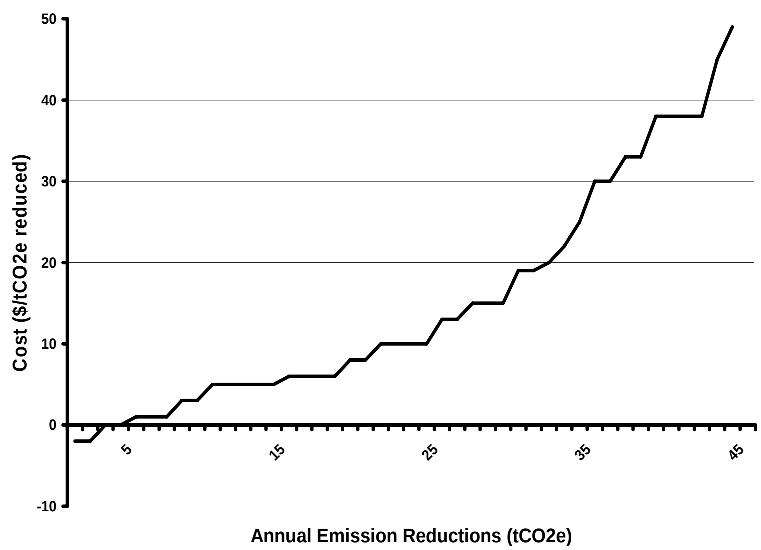
Worrell, E., L.K. Price, C. Galitsky. 2004. *Emerging Energy-efficient Technologies in Industry: Case Studies of Selected Technologies*. Environmental Technologies Division, Lawrence Berkeley Laboratory, University of California at Berkeley.

***[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.




**Hypothetical emission reduction supply curve.**

The curve shows the estimated unit cost ($/t $CO_2$e) and annual emission reduction (t $CO_2$e) 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 [annualized cost in 2004 constant U.S. dollars per metric ton of carbon (t C)]**

| Option/applicable date(s) | Annualized 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), Stavins and Richards (2005), EPA (2005) |
| Forest management/2010–2110 | $4 to 109/t C | 8 to 94 | |
| Biofuels/2010–2110 | $109 to181/t C | 123 to 169 | |
| 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); Martin *et al*. (2001); Jaccard *et al*. (2002, 2003a, 2003b); Worrel *et al*. (2004); DOE (2006) |
| Energy efficiency | $0 to 180/t C | 12% to 20% | |
| Process change | $92 to 180/t C | 20% | |
| Fuel substitution | $0 to 92/t C | 10% | |
| $CO_2$ capture and storage | $180 to 367/t C | 30% | |
| Waste management | | | |
| Reduction of fugitives | $0 to 180/t C | 90% | Hertzog (1999), Jaccard *et al*. (2002) |
| $CO_2$ capture and storage | >$367/t C | 30% | |
| Entire U.S. economy | | | |
| No trading | $102 to 548/t C | Marginal cost | EMF (2000) |
| Annex I trading | $19 to 299/t C | Marginal cost | EMF (2000) |
| Global trading | $7 to 164/t C | Marginal cost | EMF (2000) |

# Chapter 5. How Can We Improve the Application of Scientific Information to Decision Support Related to Carbon Management and Climate Decision-Making?

**Coordinating Lead Authors: Lisa Dilling[1] and Ronald Mitchell[2]**

**Lead Author: David Fairman[3]**

**Contributing Authors: Myanna Lahsen,[4] Susanne Moser,[5] Anthony Patt,[6] Chris Potter,[7] Charles Rice,[8] and Stacy VanDeveer[9]**

[1]University of Colorado/National Center for Atmospheric Research (NCAR); [2]University of Oregon; [3]Consensus Building Institute, Inc.; [4]Affiliated with University of Colorado, on location in Brazil; [5]Institute for the Study of Science and the Environment, NCAR; [6]Boston University; [7]National Aeronautics and Space Administration, Ames; [8]Kansas State University; [9]University of New Hampshire

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## KEY FINDINGS

- Information is lacking on emerging needs and demands for carbon cycle related data and analyses across scales and sectors. In fact, carbon management is a relatively new concept for most decision-makers and members of the public.
- Improving the usefulness of carbon science in North America will require more explicit commitments to generate decision-relevant information.
- Research on the production of policy-relevant scientific information suggests a number of options, from co-production of knowledge to uses of modeling tools in decision support structures and certain uses of “boundary organizations.”
- A number of initiatives to improve understandings 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).
- Further participatory pilot experiments should be considered to enhance interactions between climate change scientists and parties involved in carbon management activities and decisions.

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## INTRODUCTION: THE CHALLENGE OF "USABLE" CARBON SCIENCE

Humans have been inadvertently altering the Earth's carbon cycle since at least the dawn of agriculture, and more rapidly since the industrial revolution. Recent climate science has shown that these influences are large enough to cause significant climate change (IPCC, 2001). In response, environmental advocates, business executives, and policy-makers have increasingly recognized the need for deliberate management of the carbon cycle. Effective carbon management would seem to require that relevant, appropriate science be communicated to the wide variety of people whose decisions affect carbon cycling. Yet, thus far, carbon cycle science has rarely been organized or conducted in ways that directly support decision making on managing carbon emissions, sequestration, and impacts. There are two main reasons: (1) carbon cycle science has been conducted primarily as basic science[1] and (2) non-scientists have only recently begun to demand carbon cycle information for decision making. As a result, the emerging efforts to consciously manage carbon occur in the virtual absence of information and insights on whether these efforts are appropriate, sufficient, or implemented effectively relative to the needs to reduce carbon emissions and atmospheric concentrations (Dilling *et al*., 2003). To make carbon cycle science more relevant to public and private decision makers, scientists and decision makers will need to clarify what information is most needed 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 short, they will need to collaborate to make carbon cycle science and analysis more "usable" (Mitchell *et al*., forthcoming; Cash *et al*., 2003). Such a component of more "applied" or "solutions-oriented" research could be combined with a basic science portfolio to 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?

The first question to address then is what might we consider "decision support?" There are many different uses of the term. We adopt the definition of decision support included in the U.S. Climate Change Science Program (CCSP) Strategic Plan: "Decision support resources refers to 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? Most people constantly if 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 in these sectors to begin

[1] Carbon cycle research has been applied to agricultural soil management for a number of years; however, the focus has been on improving agricultural productivity, not limiting carbon concentrations in the atmosphere.

making more conscious choices to limit carbon emissions.[2] Whether driven by normative commitments to averting climate change, by political pressures or requirements to reduce carbon emissions, or by economic opportunities and consumer pressures, actors in these sectors are beginning to seek out information that can help them achieve their specific carbon-related goals, including those that relate to or affect the carbon cycle and the climate.[3] Even in countries and economic sectors where no consensus exists on the need to manage carbon, some entities have begun to experiment with carbon-limiting practices and investments in anticipation of a carbon-constrained future.

As part of the process of 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 needs of these stakeholders for information and decision support capabilities, a preliminary review of their interests and activities suggests that there are many stakeholders potentially 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 delineated needs for applied research (Government of Canada, 2005). A few prominent stakeholders in the U.S. are actively using carbon science to move forward with voluntary emissions offset programs such as the Chicago Climate Exchange, which brokers, among other mechanisms, agricultural carbon credits in partnership with the Iowa Farm Bureau.[4] Cities and states, including large regional partnerships on the east and west coasts, are beginning to show interest in managing emissions and carbon-related science (Text Box 1). In addition to these select visible, active stakeholders for carbon-related information, there may be many other potential stakeholders in the U.S. across sectors and scales (Text Box 1). Whether or not interest in carbon information emerges broadly in these constituencies may well depend on whether and how mandatory policies involving carbon management evolve, and what incentives might be put in place. In Europe, for example, mandatory carbon emissions policies have resulted in intense interest in carbon science from interested stakeholders who are directly affected by such policies (Schröter *et al*., 2005).

[2] For examples, see Text Box 1
[3] 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/]
[4] www.iowafarmbureau.com/special/carbon/default.aspx

In the U.S., the federal carbon science enterprise does not have many mechanisms to assess emerging demands for carbon information across scales and sectors. Thus far, federally-funded carbon science has focused predominantly on basic research in order to elucidate some of the fundamental uncertainties in the global carbon cycle and local and regional processes affecting the exchange of carbon (Dilling, in review). Most of the effort at the U.S. federal level is organized under the Climate Change Science Program (CCSP). Almost two-thirds of this effort is managed by the National Aeronautics and Space Administration and the National Science Foundation, whose missions are explicitly focused on basic research, not decision support per se (U.S. Climate Change Science Program, 2006; Dilling, in review). There are research efforts at a relatively lower level of investment at the Department of Energy and the U.S. Department of Agriculture under the CCSP[5] 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 between these programs may enhance the ability of CCSP carbon-related research to serve decision support needs.

Until perhaps the past decade, carbon management as a concept was not widely recognized—even now, most members of the public do not know the term "carbon sequestration" or understand its potential implications (Shackley *et al*., 2005; Curry *et al*., 2004). In more recent years, however, the carbon cycle science community has increasingly recognized that it may have more direct relevance to issues of policy and decision making, calling for "coordinated rigorous, interdisciplinary research that is strategically prioritized to address societal needs" (Sarmiento and Wofsy, 1999). The North American Carbon Program's (NACP) "Implementation Plan" lists decision support as one of four organizing questions (Denning *et al*., 2005).

As stated in that same plan, however, little is known in the scientific community about the likely users of decision support information that might emerge from a program such as the NACP. 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).

There are two programs within the CCSP framework that may inform this question of how to link carbon science to user needs more explicitly in the coming years. NASA has an Applications program

[5] 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.

that seeks to find uses for its data and modeling products using a "benchmarking systems" approach, and USDA and DOE have invested significant resources in science that might inform future carbon sequestration efforts and carbon accounting in agriculture and forests. Conducted as separate efforts, the programs have not yet been integrated into a broader framework aimed at making carbon cycle science more useful to decision makers within the CCSP carbon research agenda, but certainly may contribute to such a strategy if developed.

Improving the usefulness of carbon science in North America will require more explicit commitments by scientific research 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 may be helpful both in refining research agendas and accelerating 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 that have examined the creation and use of knowledge for decision making have found that information 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 decision makers believe is fair, unbiased and respectful of divergent views and interests) (Mitchell *et al*., forthcoming; Cash *et al*., 2003). Even the most technically and intellectually rigorous science may fail to influence decision makers if it does not address the decisions they face, or if it is conducted in a way that they perceive as biased or unresponsive to their concerns.

Research on the production of policy-relevant scientific information suggests strategies to maintain the integrity of the research endeavor while increasing its policy relevance. Although communicating results more effectively is important, generating science that is more applicable to decision making may require modifying the way scientific information is produced. Carbon cycle scientists and carbon decision makers will need to develop methods for interaction that work best in their specific application. At their core, all of these strategies promote scientist-stakeholder interaction 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*., forthcoming). Such processes work best when they enhance the research and its utility while preserving the credibility of both scientists and stakeholders. Transparency and 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 nine regional partnerships across the U.S., university researchers partner with local operational agencies and others that might incorporate climate information in decision making. New research is developed in consultation with all partners in an ongoing, iterative process (Lemos and Morehouse, 2005).
- *Institutional experimentation and adaptive behavior* (*e.g., adaptive management)*: Adaptive management is a powerful concept that acknowledges the inherent uncertainty of responses of natural systems to human management, and seeks to periodically assess the outcomes of management decisions and adjust policy decisions and new actions accordingly, a form of deliberate "learning by doing" (c.f. Holling 1978). Adaptive management principles have been applied for resources with multiple interests at stake, such as management of large river systems as well as forests in the Pacific Northwest (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, salient and legitimate played a significant role in the successful implementation of the Montreal Protocol which phased out the use of ozone-depleting substances. The presence of a highly credible scientific and technical assessment process with diverse participation from academics and industry scientists is credited as a key factor in the Protocol's success (Parson, 2003).
- *Mediated modeling*: Shared tools can facilitate scientist-user interactions, help diverse groups orient around a problem and illuminate common assumptions as well as differences. Mediated modeling involves a guided process in which participants from a wide variety of perspectives jointly construct a computer model that can be used in solving complex environmental problems, or envisioning a shared future. The process has been successfully used for watershed management, endangered species management and a host of other difficult environmental issues (Van den Belt, 2004).
- *Carbon modeling tools as decision support*: As carbon management within the United States is increasingly considered at the national level, some federal agencies have begun to develop decision support tools to help estimate carbon sequestration in various ecosystems and under various land use scenarios. These pilot-phase tools are available online and feature a customizable user interface (see examples such as the NASA Ames CQUEST, Carbon Query and Evaluation Support Tools, http://geo.arc.nasa.gov/website/cquestwebsite/; the U.S. Forest Service COLE, Carbon Online Estimator, http://ncasi.uml.edu/COLE/; and Colorado State COMET-VR, CarbOn Management Evaluation Tool, http://www.cometvr.colostate.edu/).

Over time, well-structured scientist-stakeholder interaction can bring substantial benefits to both scientists and decision makers (Moser, 2005). Scientists learn to identify research questions that are both scientifically interesting and relevant to decisions, and to frame their answers in ways that audiences are more likely to find compelling. Non-scientists learn more about what questions science can and cannot answer. They also 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).

Such interactive approaches to research also 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 consumes considerable time and effort. Involving private sector firms may require scientists accustomed to working in an open informational environment to navigate in a world in which much information is proprietary. Scientists may also choose not to pursue 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).

On the stakeholder side, some may lack the financial resources, expertise, time, and other capacities needed for meaningful participation. Some will distrust scientists in general and government-sponsored science in particular due to cultural, institutional, historical, or other factors. Some may reject participation in open and public processes in which they must interact with those with whom they disagree politically or compete economically. In some cases, stakeholders will try to manipulate research questions and findings to serve their political or economic interests. Perhaps most importantly, stakeholders often show little interest in diverting their time (or that of their employees) 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 exist to a large extent as quite separate social and professional realms, with quite different traditions, norms, codes of behavior, and reward systems. The boundaries that exist between them serve many useful functions but may also inhibit the transfer of useful knowledge across those boundaries. According to Guston (2001), a boundary organization is an institution that "straddles the shifting divide" between politics and science. Boundary organizations are accountable to both sides of the boundary and involve professionals from each, as well as those serving in a mediating role. Such "boundary spanning" individuals and organizations can often facilitate the uptake of science by translating scientific findings so that stakeholders find them more user-friendly and by stimulating adjustments in research agendas and approach. Boundary organizations can exist at a variety of scales and for a wide variety of purposes. Cooperative agricultural extension services and NGOs that successfully convert large-scale scientific understandings of weather, aquifers, or pesticides into locally-tuned guidance to farmers are classic examples of boundary organizations (Cash, 2001). The International Research Institute for Climate Prediction focuses on seasonal-to-interannual scale climate research and modeling so that their research results are useful to farmers, fishermen, and public health officials (e.g., Agrawala *et al*., 2001). The Subsidiary Body for Scientific and Technological Advice (SBSTA) of the United Nations Framework Convention on Climate Change serves also as a boundary organization at an international level. The SBSTA serves as a link between information and assessments provided by expert sources (such as the IPCC) and the Conference of the Parties (COP), which focuses on setting policy.[6] The University of California Berkeley Digital Library Project Calflora project has sought to ensure that an extensive database on plants is designed and implemented in ways that support environmental planning (Van House *et al*., 2003).

And of course, there are other significant challenges to the use of knowledge, even when created through self-conscious efforts like those just delineated. People fail to integrate new research and information in their decisions for many reasons. Besides obstacles already mentioned, people often are not motivated to use information that implies or 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 that it gets used but it will make it possible and more likely that it will.

## RESEARCH NEEDS TO ENHANCE DECISION SUPPORT FOR CARBON MANAGEMENT

There is likely to be substantial and growing demand for detailed analysis of carbon management issues and options across major economic sectors, nations and levels of government in North America. This is especially likely in jurisdictions that place policy constraints on carbon budgets, such as within the states comprising the Regional Greenhouse Gas Initiative, or the State of California. Although some new

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

efforts are underway in parts of agencies, carbon cycle science, at least in the U.S., could be organized and carried out in ways that better meet this potential demand in a more systematic fashion. As noted by the National Research Council (2004), effective implementation of the goals of the program, as a part of the Climate Change Science Program, "requires focused research to develop decision support resources and methods." While such recommendations were stated for the whole of the program, they are pertinent to carbon-related science as one of the major components.

The process of creating information to support decision making should be significantly different from the process of creating "basic" or "fundamental" scientific knowledge. The primary driver for such "use-inspired" research is societal need, not scientific curiosity alone (Stokes, 1997). To improve the application of scientific information to support carbon and climate-related decisions, scientists and non-scientist carbon managers need to improve their joint understanding of the top priority questions facing carbon-related decision making. They also need to collaborate more effectively in undertaking research and interpreting results in order to answer those questions. The scale of information provided and its specificity to regional or local concerns are often important considerations for the salience of information (Cash and Moser, 2000).

As a first step, a formal process could be developed "for gathering requirements and understanding the problems for which research can inform decision makers outside the scientific community," including the formation of a decision support working group (Denning *et al*., 2005). To move forward on creating an effective decision component of the CCSP program, the NRC recommends organizing a variety of deliberative activities, such as workshops, focus groups, working panels, and citizen advisory groups, with the goals being 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). The current status of decision support activities across the CCSP will be assessed by several other SAP processes, complementary to this one, specifically SAP 5.1, 5.2, and 5.3 (organized under the heading of "Explore the uses and identify the limits of evolving knowledge to manage risks and opportunities related to climate variability and change").

## SUMMARY AND CONCLUSIONS

The carbon cycle is influenced through deliberate and inadvertent decisions on the part of diverse and spatially dispersed actors, located in many different sectors and at different scales. Scientific information and analysis can lead to better-informed decision making across many sectors and levels of action, if decision makers recognize that information and analysis as relevant and legitimate. To make carbon cycle science more useful to decision makers, we suggest the following steps, to be initiated by leaders in the scientific and program level carbon science community:

- 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 both incremental and major departures from existing practice with the goal 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 and communication experts.
- Consider initiating participatory pilot research projects and identifying existing boundary organizations (or establishing new ones) to bridge carbon management and carbon science.

## REFERENCES

Adler, P., R. Barrett, M. Bean, J. Birkoff, C. Ozawa, and E. Rudin. 1999. Managing Scientific and Technical Information in Environmental Cases: Principles and Practices for Mediators. Tucson, AZ: U.S. Institute for Environmental Conflict Resolution.

Agrawala S., K. Broad, and D. H. Guston. 2001. Integrating climate forecasts and societal decision making: Challenges to an emergent boundary organization. Science, Technology and Human Values: 26(4):454–477.

Apps, M., J. Canadell, M. Heimann, V. Jaramillo, D. Murdiyarso, D. Schimel, and M. Manning. 2003. Expert meeting report: IPCC meeting on current understanding of the processes affecting terrestrial carbon stocks and human influences upon them. Geneva Switzerland, July 21–23, 2003. Available at: http://www.ipcc.ch/pub/carbon.pdf.

Cash, D.W. 2001. "In order to aid in diffusing useful and practical information:" Agricultural Extension and boundary organizations. Science, Technology and Human Values 26:431–453.

Cash, D., S. C. Moser. 2000. "Linking global and local scales: designing dynamic assessment and management processes." *Global Environmental Change* 10:109–120.

Cash, D., W. Clark, F. Alcock, N. Dickson, N. Eckley, D. Guston, J. Jaeger, and R. Mitchell. 2003. Knowledge systems for development. Proceedings of the National Academy of Science, 100 (14):8086–8091.

Curry, T., D. Reiner, S. Ansolabehere, and H. Herzog. *How Aware is the Public of Carbon Capture and Storage?* Presented at the Seventh International Conference on Greenhouse Gas Control Technologies, Vancouver, Canada, September 2004. Available at http://sequestration.mit.edu/ bibliography/policy.html.

Denning, A.S. *et al*. (2005). Science Implementation Strategy for the North American Carbon Program. Report of the NACP Implementation Strategy Group of the U.S. Carbon Cycle Interagency Working Group. Washington, DC: U.S. Carbon Cycle Science Program, 68 pp. Available online: http://www.nacarbon.org/nacp/documents.html.

Dilling, L. Towards science in support of decision making: Characterizing the supply of carbon cycle science. Environmental Science and Policy, in review 2005.

Dilling, L., S. C. Doney, J. Edmonds, K. R. Gurney, R. C. Harriss, D. Schimel, B. Stephens, G. Stokes. 2003. The role of carbon cycle observations and knowledge in carbon management. Ann. Rev. Environ. Resour. 28:521–58.

Douglas, M., and A. Wildavsky. 1984. Risk and Culture. Berkeley: University of California Press.

Ehrmann, J., and B. Stinson. 1999. Joint fact-finding and the use of technical experts, in L. Susskind, J. T. Larmer, and S. McKearnan, eds. The Consensus Building Handbook. Thousand Oaks, CA: Sage Publications.

Farrell, A., and J. Jaeger, eds. 2005. Assessments of regional and global environmental risks: designing processes for the effective use of science in decision-making. Washington, DC: Resources for the Future.

Gibbons, M., C. Limoges, and H. Nowotny. 1994. The new production of knowledge: the dynamics of science and research in contemporary societies. London: Sage.

Government of Canada, 2005. Project Green: Moving Forward on Climate Change: A Plan for Honoring our Kyoto Commitment. http://www.climatechange.gc.ca/english/ newsroom/2005/plan05.asp.

Guston, D. H. 2001**.** Boundary Organizations in Environmental Policy and Science: An Introduction**.** *Science, Technology, & Human Values*, Vol. 26, No. 4, Special Issue: Boundary Organizations in Environmental Policy and Science (Autumn 2001), pp. 399–408.

Holling, C. S., editor. 1978. Adaptive environmental assessment and management. John Wiley, New York, New York, USA.

Holling, C. (1995). What barriers? What Bridges? Barriers and Bridges to the renewal of ecosystems and institutions. L. Gunderson, C. Holling and S. Light. New York, Columbia University Press: 593 pp.

Intergovernmental Panel on Climate Change (IPCC). 2000. *Land Use, Land-Use Change, and Forestry. Special Rep. IPCC*. ed. R. T. Watson, I. R. Noble, B. Bolin, N. H. Ravindranath, D. J. Verardo, D. J. Dokken. Cambridge, UK/New York: Cambridge Univ. Press. 377 pp.

Intergovernmental Panel on Climate Change (IPCC). 2001. Climate Change 2001: The scientific basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change. ed. J. T. Houghton, Y. Ding, D. J. Griggs, M. Noguer, P. J. van der Linden, *et al*. Cambridge UK/New York. Cambridge University Press. 881 pp. Available from http://www.ipcc.ch/.

Lahsen, M.. 1998. “The Detection and Attribution of Conspiracies: The Controversy Over Chapter 8.” In *Paranoia Within Reason: A Casebook on Conspiracy as Explanation. Late Editions 6, Cultural Studies for the End of the Century*, edited by George E. Marcus. Chicago: University of Chicago Press.

Lahsen, M.. Forthcoming. “International Science, National Policy: The Politics of Carbon Cycle Science in Brazil.” *Climatic Change*.

Lemos, M. C., and B. J. Morehouse. 2005. The co-production of science and policy in integrated climate assessments. Global Environmental Change. 15:57–68

Martinez, J., and A. Fernandez-Bremauntz, eds. 2004 Cambio climatico: Una vision desde Mexico. Secretaria de Medio Ambiente y Recursos Naturales, Instituto Nacional de Ecologia. Mexico City, Mexico.

Mitchell, R. B., W. C. Clark, D. W. Cash, and F. Alcock. 2004. "Science, Scientists, and the Policy Process: Lessons from Global Environmental Assessments for the Northwest Forest." In Forest Futures: Science, Politics and Policy for the Next Century. Ed. K. Arabas and J. Bowersox. Rowman and Littlefield, 95–111.

Mitchell, R. B., W. C. Clark, D. W. Cash, and N. M. Dickson, eds. forthcoming. Global environmental assessments: information and influence. Cambridge, MA: The MIT Press.

Moser, S. 2005. *Stakeholder Involvement in the First U.S. National Assessment of the Potential Consequences of Climate Variability and Change: An Evaluation, Finally.* In: Public Participation in Environmental Assessment and Decision Making, National Research Council, Committee on Human Dimensions of Global Change, NAS/NRC, Washington, DC, refereed, forthcoming.

National Research Council. 1999. Making Climate Forecasts Matter. Washington, DC. National Academy Press.

National Research Council, Committee to Review the U.S. Climate Change Science Program Strategic Plan. 2004. Implementing climate and global change research: A review of the final U.S. Climate Change Science Program Strategic Plan. Washington, DC. National Academy Press.

National Research Council, Roundtable on Science and Technology for Sustainability. 2005. Knowledge-action systems for seasonal to interannual climate forecasting: Summary of a workshop. Washington, DC. National Academy Press.

Parson, E. A. 2003. Protecting the Ozone Layer. Oxford University Press, Oxford, UK.

Patt, A., P. Suarez, and C. Gwata. 2005a. Effects of seasonal climate forecasts and participatory workshops among subsistence farmers in Zimbabwe. Proceedings of the National Academy of Sciences of the United States of America 102:12673–12678.

Patt, A. G., R. Klein, and A. de la Vega-Leinert. 2005b. Taking the uncertainties in climate change vulnerability assessment seriously. Comptes Rendus Geosciences, 337:411–424.

Pulwarty, R. S., and K. T. Redmond. 1997. Climate and salmon restoration in the Columbia River Basin: The role and usability of seasonal forecasts. Bulletin of the American Meteorological Society 78 (3):381–396

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

Sarmiento, J. L., and S. C. Wofsy. 1999. A U.S. Carbon Cycle Science Plan: A report of the Carbon and Climate Working Group. U.S.GCRP, Washington, DC. Available online at: http://www.nacarbon.org/nacp/documents.html.

Schröter, D. *et al*. 2005. Ecosystem Service Supply and Vulnerability to Global Change in Europe. Science, 310(5752):1333–1337.

Shackley, S., C. McLachlan, and C. Gough. (2005) The public perception of carbon dioxide capture and storage in the UK: results from focus groups and a survey Climate Policy 4 377–398.

Stokes, D. E. 1997. Pasteur's Quadrant: Basic science and technological innovation. Brookings Institution Press. Washington, DC.

U.S. Climate Change Science Program. 2003. Strategic Plan for the U.S. Climate Change Science Program. Available online at: www.climatescience.gov, last Accessed February 20, 2006.

U.S. Climate Change Science Program. 2006. Our Changing Planet: The US Climate Change Science Program for Fiscal Year 2006. A Report by the Climate Change Science Program and the Subcommittee on Global Change Research. Washington, DC. Available online at: www.climatescience.gov, last accessed February 23, 2006.

U.S. Department of State, 2004. U.S. Climate Change Policy: The Bush Administration's Actions on Global Climate Change. Fact Sheet Released by the White House, Office of the Press Secretary Washington, DC November 19, 2004, posted at http://www.state.gov/g/oes/rls/fs/2004/38641.htm

Van den Belt, M. 2004. Mediated modeling: A systems dynamic approach to environmental consensus building. Island Press, Wash. DC, 296 pp.

Van House, N. A. 2003. Digital Libraries and Collaborative Knowledge Construction. In Digital Library Use: Social Practice in Design and Evaluation. Editors: Ann P. Bishop, Barbara P. Buttenfield, and Nancy A. Van House. MIT Press, 271–295.

Yaniv, I. 2004. Receiving other people's advice: influence and benefit. Organizational Behavior and Human Decision Processes, 93:1–13.

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**Sectors Expressing Interest and/or Participating in the SAP 2.2 Process.** This list of sectors is not an exhaustive list nor is it based on a rigorous assessment, it is meant to demonstrate the wide variety of potential stakeholders with an 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 demonstrated significant interest in carbon management as ways 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 non-profits might 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 on 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:*** 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

**Coordinating Lead Author: G. Marland[1,2]**

**Contributing Authors: R. J. Andres,[3] T. J.Blasing,[1] T. A. Boden,[1] C. T. Broniak,[4] J. S. Gregg,[5] L. M. Losey,[6] and K. Treanton[7]**

[1]Environmental Sciences Division, Oak Ridge National Laboratory, [2]Ecotechnology Program, Mid Sweden University, [3]Department of Space Studies, University of North Dakota, [4]Oregon State University, [5]Department of Geography, University of Maryland, [6]Department of Space Studies, University of North Dakota, [7]International Energy Agency

## 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 fuel is burned. Fossil fuels are composed primarily of compounds of hydrogen and carbon. 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 continuing to grow as population grows and demand for energy grows. Looking back from the end of 2005, fully half of the $CO_2$ released from fossil-fuel burning globally has occurred since 1980 (Figure 1).

**Figure 1. Cumulative global emissions of $CO_2$ from fossil-fuel combustion and cement manufacture from 1751 to 2002.**

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 manufacturing now releases less than 1% of the $CO_2$ released by fossil-fuel combustion. However, cement manufacturing is the largest anthropogenic (of human origin) 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 in 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 manufacturing in North America, and discusses some scenarios for emissions in the future. 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 that 38.4% of $CO_2$ emissions from North America come from enterprises whose primary business is to provide electricity and heat and another 30.8% 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 processes motivating their use.

## Estimating $CO_2$ Emissions

If we have estimates of the consumption of fossil fuels, it is relatively straightforward to estimate the amount of $CO_2$ released to the atmosphere when they 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 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 products such as highway asphalt, lubricants, waxes, solvents, and plastics and thus may not soon be converted to $CO_2$. Some of these long-lived carbon-containing products will release the carbon they contain to the atmosphere as $CO_2$ during use or during processing of the materials as 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 ways they deal with the carbon in these carbon-containing 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, petroleum, 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 billion J for coal, 20 kg C per billion J for petroleum, and 15 kg C per billion J 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 kg C per billion J of coal (measured on a net heating value basis[1]), with a standard error of the mean at 2.09%. The value is slightly higher for lignite and brown coals (26.23 kg C per billion J ±2.33%, also shown in Figure 1). 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 from measures of fossil-fuel use. The differences between the values in Table 1 and those in Figure 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.** To make them easier to distinguish, data for lignites and brown coals are shown on the left axis, and 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).

**Table 1. A sample of the coefficients used for estimating $CO_2$ emissions from the amount of fuel burned** (from IPCC, 1996).

Data on fossil-fuel production, trade, consumption, and so on 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

[1]"Net heating value" is the heat release measured when fuel is burned at constant pressure so that the water is released as water vapor. This is distinguished from the "gross heating value," which is the heat release measured when the fuel is burned at constant volume so that the water is released as liquid water. The difference is essentially the heat of vaporization of the water and is related to the hydrogen content of the fuel.

estimates of fuel consumption. All of the emissions estimates in this chapter are in terms of 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 countries like the United States—with good data on fuel production, trade, and consumption—the uncertainty in national emissions of $CO_2$ is probably on the order of ±5% or less. In fact, the U.S. Environmental Protection Agency (EPA) (2005) suggests its 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 its estimates for Canada are within –4 to 0 %. The Mexican National Report (Mexico, 2001) does not provide estimates of uncertainty, but our analyses using 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 are inclined to 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.

## The Magnitude of National and Regional $CO_2$ Emissions

Figure 3 shows that from the beginning of the fossil-fuel era (1750 in these graphs) to the end of 2002, there were 93.5 Gt C released as $CO_2$ from fossil-fuel consumption (and cement manufacturing) 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 is 31.5 % of the global total. Among all countries, the United States, Canada, and Mexico rank as the first, eighth, and eleventh largest emitters of $CO_2$ from fossil-fuel consumption, respectively (for 2002) (Marland *et al.*, 2005). 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 manufacturing as a function of time, for the three countries of North America and for the sum of the**

[2]The carbon 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 carbon in the $CO_2$. The numbers can be easily converted back and forth using the ratio of the molecular masses, i.e. (mass of carbon) × (44/12) = (mass of $CO_2$).

**three.** Figure 3a is for the United States, Figure 3b is for Canada, Figure 3c is for Mexico, and Figure 3d 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.*, 2005).

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

The long time series of emissions estimates illustrated in Figures 2 and 3 are from CDIAC (Marland *et al.*, 2005). 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, 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, http://unfccc.int). The latter two sets of estimates are based on data on actual fuel combustion 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. However, they do represent different manipulations of these 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. 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 earlier.

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 manufacturing 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 a correction is made for non-fuel usage of fossil-fuel resources, the conversion from mass or volume to energy units, and/or the way estimates of carbon content are derived. Because the national estimates from CDIAC do not include emissions from the non-fuel uses of

[3]IEA 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.

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 difference among the multiple sources: the emissions from Mexico in 1990. Losey (2004) has suggested, based on other criteria, that there is an inaccuracy in the UN energy data set for Mexican natural gas for the three-year period 1990–1992; these kinds of analyses result in reexamination of some of the fundamental data.

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

IEA (2005, p. 1.4) has systematically compared its estimates with those reported to the UNFCCC by the different countries, and it finds that the differences for most developed countries are within 5%. 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 the 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
- a different split between what is identified as emissions from energy and emissions from industrial processes.

## Emissions by Month and/or State

With interest increasing in the details and processes of the global carbon cycle, there is also 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 there is considerable variation in United States 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.

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

**Figure 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).

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 United States 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. It 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.** Monthly values are shown where estimates are justified by the availability of monthly data on fuel consumption or sales (from Andres *et al.*, 2005).

## Emissions by Economic Sector

To understand how $CO_2$ emissions from fossil-fuel use enter and 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 in this brief discussion. 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 more detail at how the fuels are used and at the economic sectors in which the fuels are used and from which the $CO_2$ is emitted. This is the goal of the next four chapters of this volume.

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 additional 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 (Figure 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 such as climate, population density, and access to resources (Blasing *et al.*, 2005b; Neumayer, 2004).

Chapters 6 through 9 of this volume 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 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 numeric differences, however, the 4 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 50% of its electricity from coal, Mexico gets 73% from petroleum and natural gas, and Canada gets 60% from hydroelectric stations. Partially as a reflection of this difference, 40% of United States $CO_2$ 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 2002, from IEA, 2004). 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 International Energy Agency allows us to summarize $CO_2$ emissions for the 3 countries according to sectors that closely correspond to the sectoral division of chapters 6 through 9 (Table 4).

**Figure 8. Per capita emissions of $CO_2$ from fossil-fuel consumption (and cement manufacturing) 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 manufacturing and gas flaring are not included) per unit of GDP for the United States, Canada, and Mexico and worldwide.**

**Figure 9. Per capita emissions of $CO_2$ from fossil-fuel consumption for the 50 U.S. states in 2000.** To demonstrate the range of values, values have been rounded to whole numbers of metric tons 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).

## CONCLUSION

There are a variety of reasons 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.

## REFERENCES

Andres, R.J., D.J.Fielding, G. Marland, T.A.Boden, N. Kumar, and A.T. Kearney, 1999. Carbon dioxide emissions from fossil-fuel use, 1751-1950. Tellus 51:759-765.

Andres, R.J., J.S. Gregg, L.M. Losey, and G. Marland, 2005. Monthly resolution fossil-fuel-derived carbon dioxide emissions for the countries of the North American Carbon Program, Proceedings of the Seventh International Carbon Dioxide Conference, Bloomfield, Colorado, September, 2005, pp. 157-158.

Blasing, T. J., C.T. Broniak, and G. Marland, 2005a. The annual cycle of fossil-fuel carbon dioxide emissions in the United States. Tellus 57B: 107-115. (data available at http://cdiac.esd.ornl.gov)

Blasing, T.J., C. Broniak, and G. Marland, 2005b. State-by-state carbon dioxide emissions from fossil-fuel use in the United States 1960-2000. Mitigation and Adaptation Strategies for Global Change 10: 659-674.

Environment Canada, 2005. Canada's Greenhouse Gas Inventory: 1990-2003, National Inventory Report – April 15, 2005, Greenhouse Gas Division, Environment Canada.

EPA, 2005. Inventory of U.S. greenhouse gas emissions and sinks: 1990-2003, EPA 430-R-05-003, United States Environmental Protection Agency, Washington D.C.

Gregg, J.S., 2005. Improving the temporal and spatial resolution of carbon dioxide emissions estimates from fossil-fuel consumption, A thesis submitted to the graduate faculty of the University of North Dakota, August, 2005, 404 pp. (data available at http://cdiac.esd.ornl.gov)

Gurney, K.R., Y.H Chen, T. Maki, S.R. Kawa, A. Andrews, and Z. Zhu, 2005. Sensitivity of atmospheric $CO_2$ inversion to seasonal and interannual variations in fossil-fuel emissions," J. Geophys. Res. 110 (D10), 10308.

IEA, 2005. $CO_2$ emissions from fuel combustion: 1971-2003. International Energy Agency, OECD/IEA, Paris.

IPCC (Intergovernmental Panel on Climate Change), 1997. revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories (3 volumes), IPCC Technical Support Unit, Bracknell,UK.

Losey, L.M., 2004. Monthly and seasonal estimates of carbon dioxide emissions from fossil fuel consumption in Canada, Mexico, Brazil, The United Kingdom, France, Spain, Italy, and Poland. A thesis submitted to the graduate faculty of the University of North Dakota, May, 2004, 328 pp. (data available at http://cdiac.essd.ornl.gov)

Marland, G., T. Boden, and R.J. Andres, 1995. Carbon dioxide emissions from fossil fuel burning: emissions coefficients and the global contribution of eastern European countries. Időjárás 99:157-170.

Marland, G., T.A. Boden, and R.J. Andres. 2005. Global, Regional, and National $CO_2$ Emissions. In Trends: A Compendium of Data on Global Change. Carbon Dioxide Information Analysis Center, Oak Ridge National Laboratory, U.S. Department of Energy, Oak Ridge, Tenn., U.S.A. (data available at http://cdiac.esd.ornl.gov)

Mexico, 2001. México: Segunda comunicación nacional ante la Convención Marco de las Naciones Unidas sobre el cambio climático. Comité intersecretarial sobre cambio climático, Secretaria de Medio Ambiente y Recursos Naturales (Semarnat), 374 pp., Mexico City.

Neumayer, E., 2004. National carbon dioxide emissions: geography matters. Area 36.1: 33-40.

**Table 1. A sample of the coefficients used for estimating $CO_2$ emissions from the amount of fuel burned**
(from IPCC, 1996)

| Fuel | Emissions coefficient (kg carbon/$10^9$ 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. 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 |
| | U.S. EPA | 1316 | U.S. EPA | 1478 | U.S. EPA | 1534 |
| Canada | CDIAC | 112 | CDIAC | 119 | CDIAC | 139 |
| | IEA | 117 | IEA | 136 | IEA | 145 |
| | U.S. EPA | 117 | U.S. EPA | 133 | U.S. EPA | 144 |
| Mexico | CDIAC | 99 | CDIAC | 96 | CDIAC | 100 |
| | IEA | 80 | IEA | 96 | IEA | 100 |
| | U.S. EPA | 81 | U.S. EPA | 96 | U.S. EPA | NA |

Notes:

These data have been multiplied by 12/44 to get the mass of carbon for the comparison here.

Many of these data were published in terms of the mass of $CO_2$ .

Values for the United States, Canada and Mexico represent consumption data as reported by CDIAC (Marland *et al.*, 2005), IEA (2005), and by the National Reports to the United Nations Framework Convention on Climate Change [United States (EPA, 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 manufacturing and gas flaring are not included) per unit of GDP for the United States, Canada, and Mexico and worldwide**

| 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 2002**

| Sector | United States | Canada | Mexico | North America |
|---|---|---|---|---|
| Energy extraction and conversion[a] | 46.8 | 36.0 | 49.4 | 46.1 |
| Transportation[b] | 31.2 | 28.3 | 28.7 | 30.8 |
| Industry[c] | 11.0 | 16.8 | 13.2 | 11.6 |
| Buildings[d] | 11.0 | 18.9 | 8.8 | 11.6 |

[a]The sum of three IEA categories, "public electricity and heat production," "unallocated autoproducers," and "other energy industries." (IEA, 2004)

[b]IEA category "transport." (IEA, 2004)

[c]IEA category "manufacturing industries and construction." (IEA, 2004)

[d]IEA category "other sectors." (IEA, 2004)

1

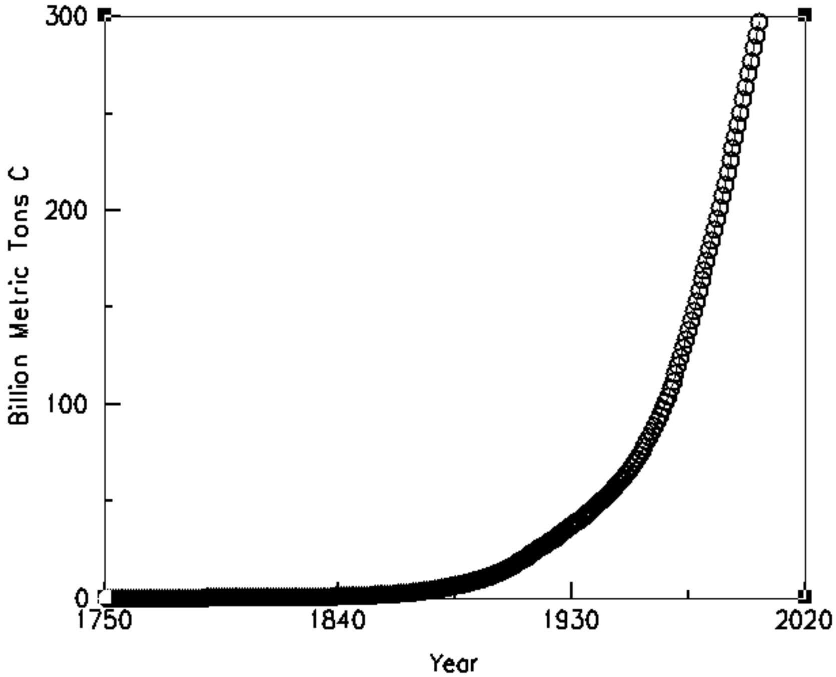


**Figure 1. Cumulative global emissions of $CO_2$ from fossil-fuel combustion and cement manufacture from 1751 to 2002.**

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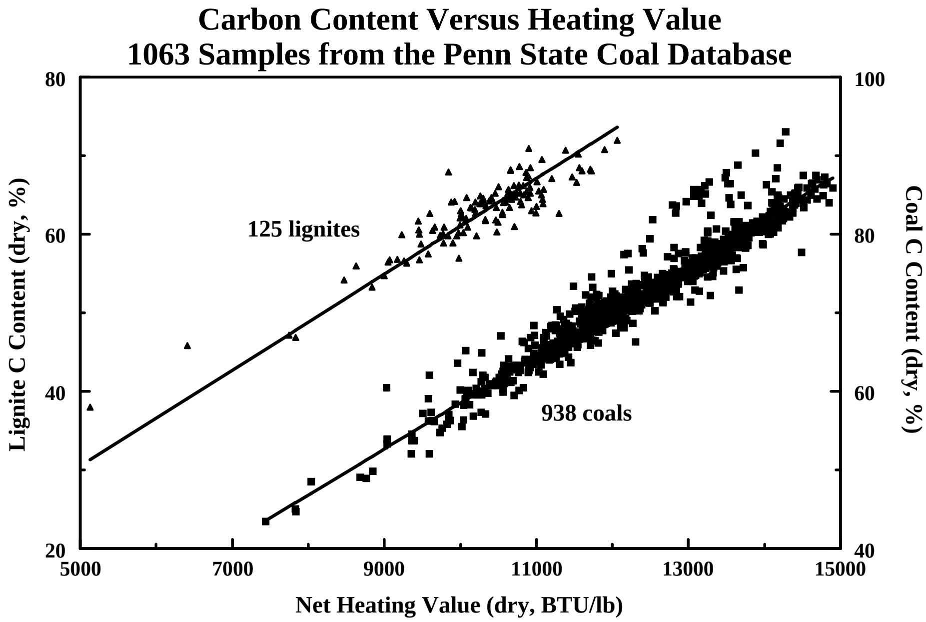



**Figure 2. The carbon content of coal varies with the heat content, shown here as the net heating value.** To make them easier to distinguish, data for lignites and brown coals are shown on the left axis, and data for hard coals are offset by 20% and shown on the right axis. Heating value is plotted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

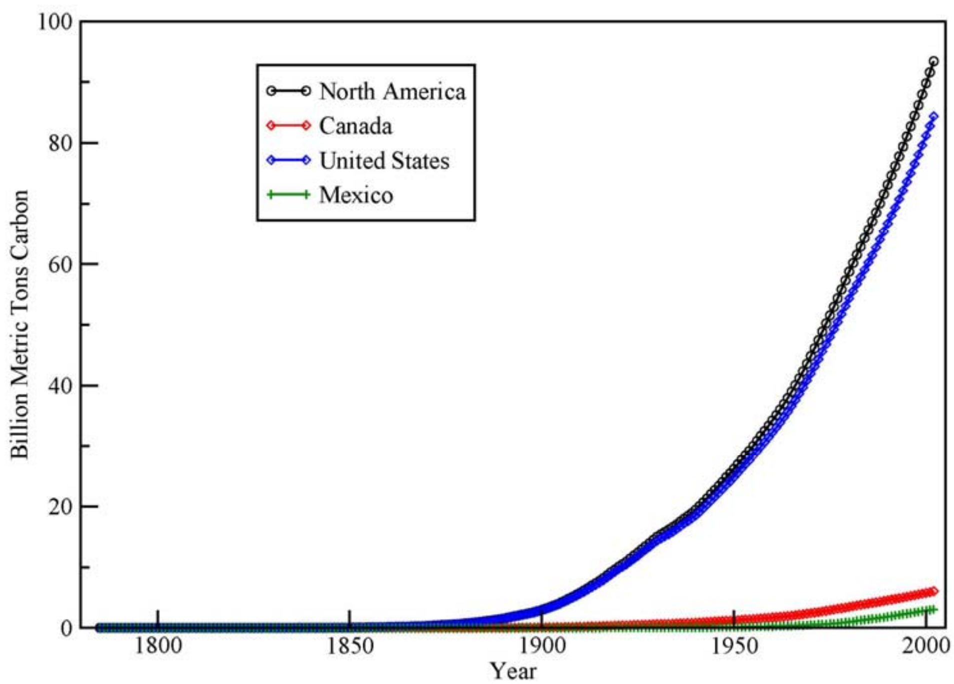
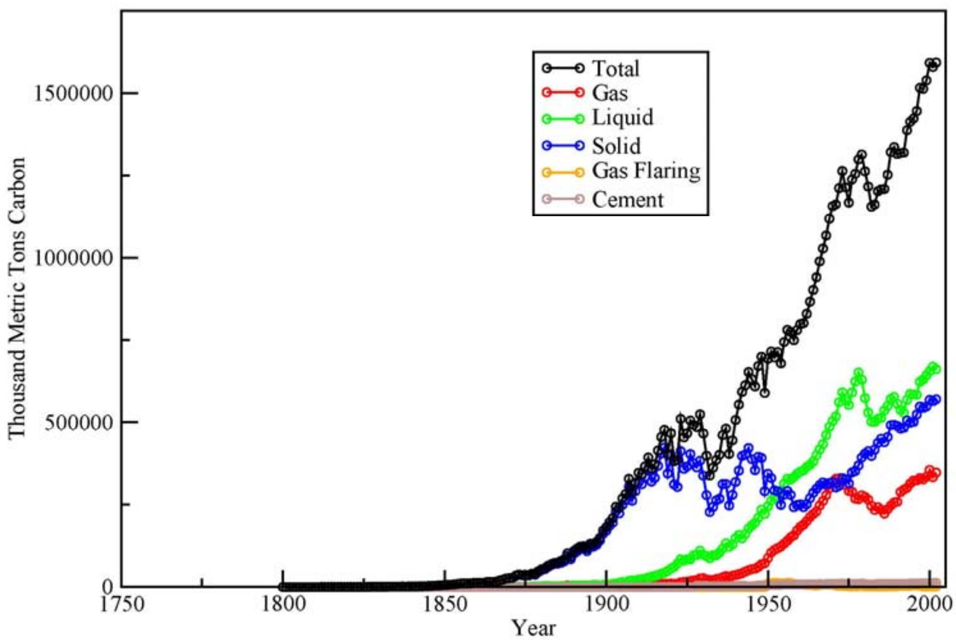



**Figure 3. The cumulative total of $CO_2$ emissions from fossil-fuel consumption and cement manufacturing as a function of time, for the three countries of North America and for the sum of the three.**

**(a)**




**(b)**

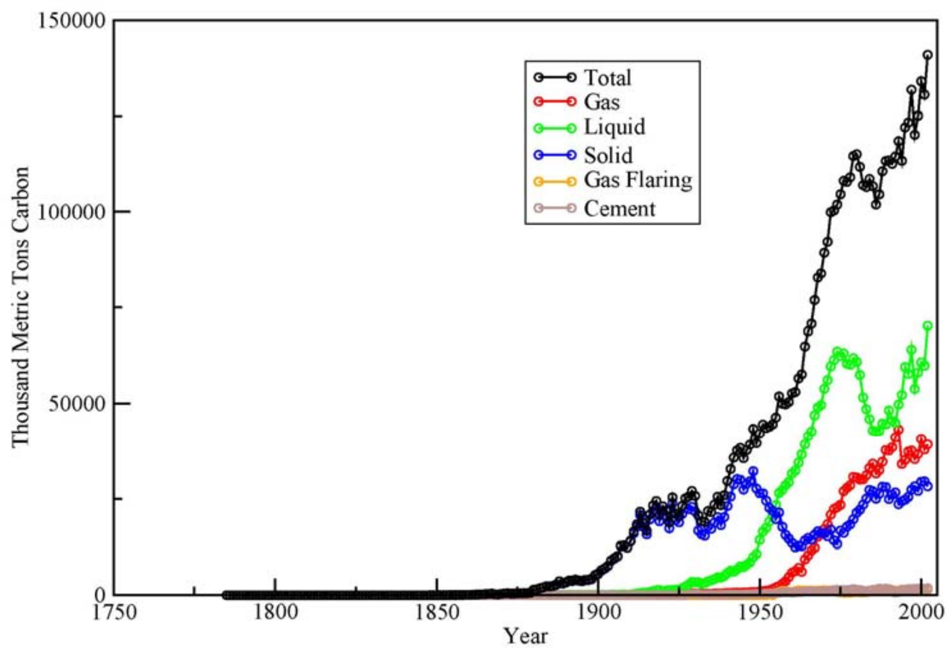
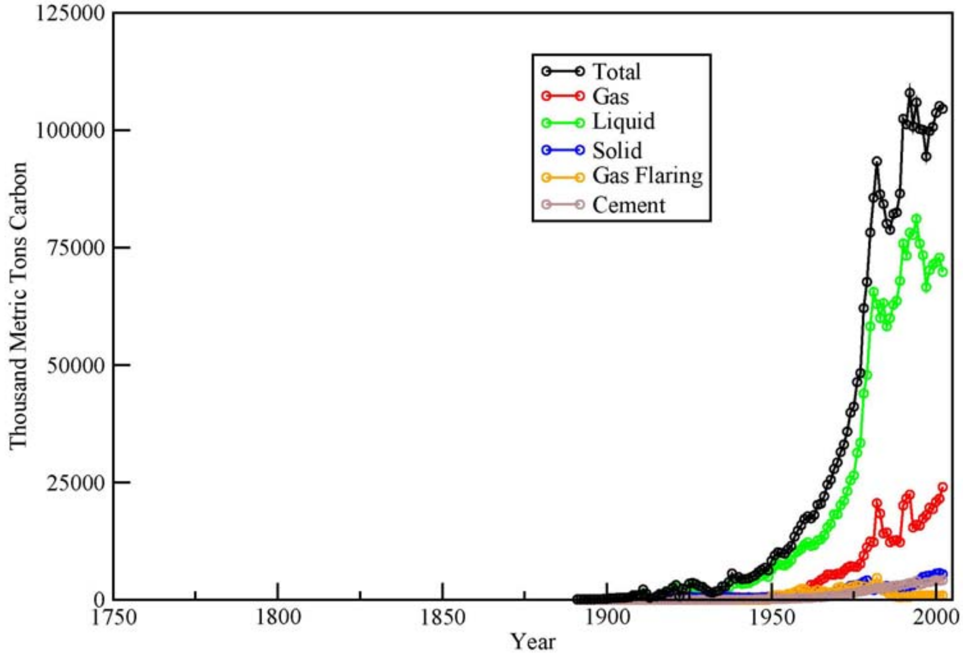



**Figure 4a and 4b. Annual emissions of $CO_2$ from fossil-fuel use by fuel type.** Figure 4a is for the United States, Figure 4b is for Canada, Figure 4c is for Mexico, and Figure 4d is for the sum of the three. Note that in order to illustrate the contributions of the different fuels, the four plots are not to the same vertical scale (from Marland et al. 2005).

**(c)**




**(d)**

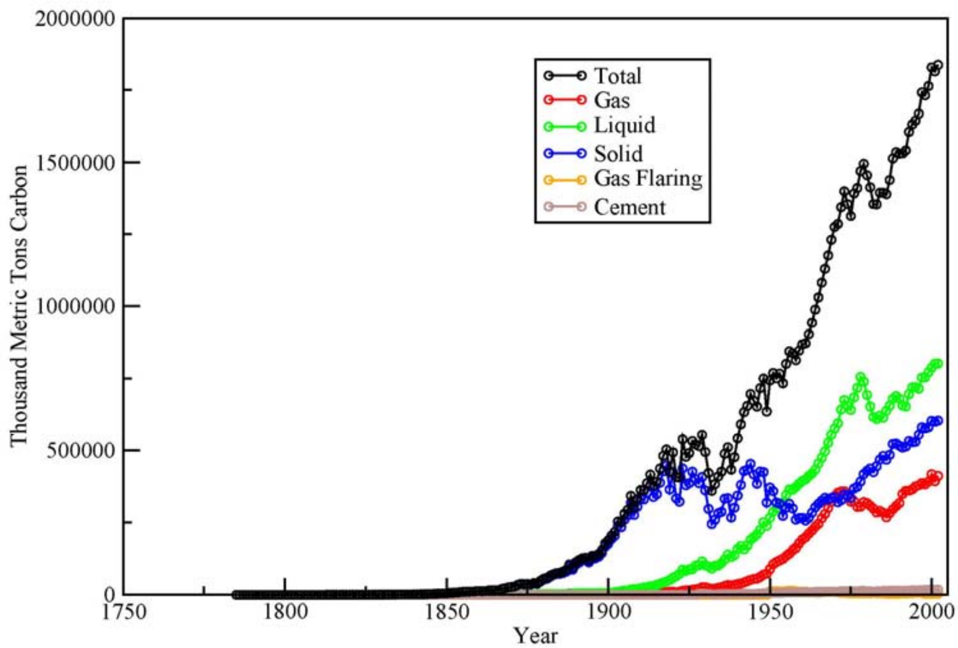



**Figure 4c and 4d. Annual emissions of $CO_2$ from fossil-fuel use by fuel type.** Figure 4a is for the United States, Figure 4b is for Canada, Figure 4c is for Mexico, and Figure 4d is for the sum of the three. Note that in order to illustrate the contributions of the different fuels, the four plots are not to the same vertical scale (from Marland et al. 2005).

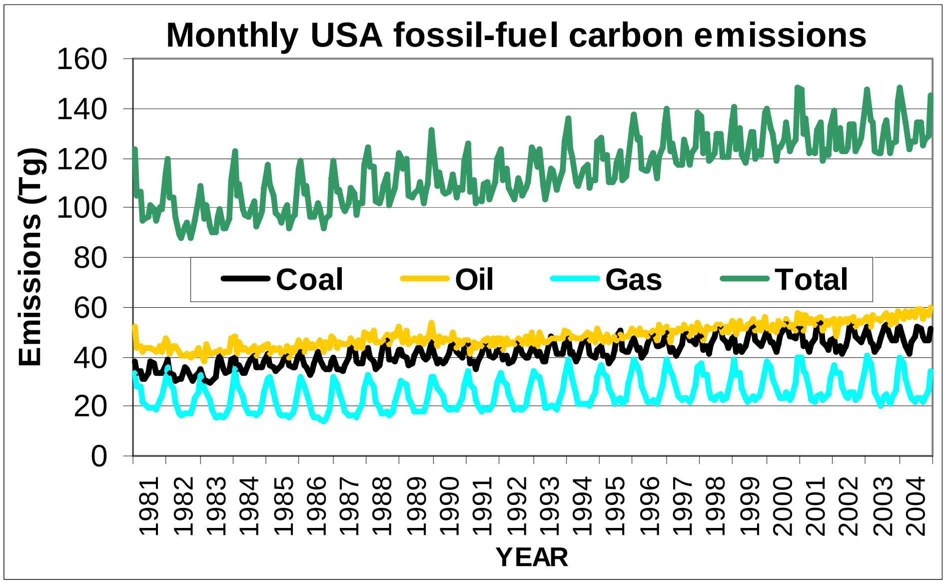



**Figure 5. Emissions of $CO_2$ from fossil-fuel consumption in the United States, by month.** Emissions from cement manufacturing are not included (from Blasing et al. 2005a).

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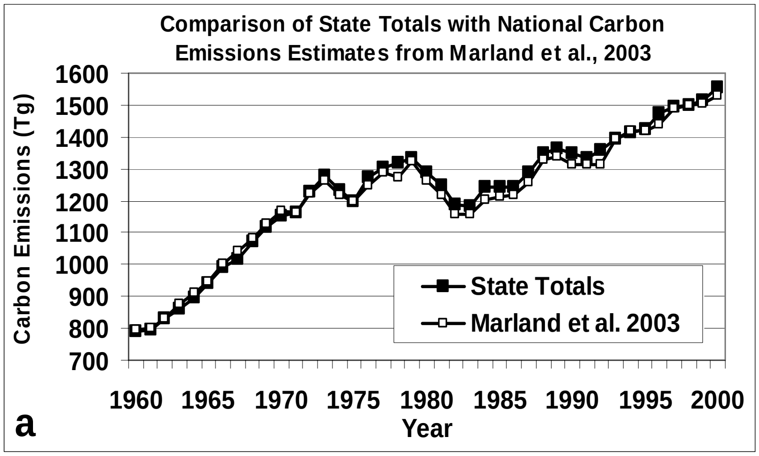
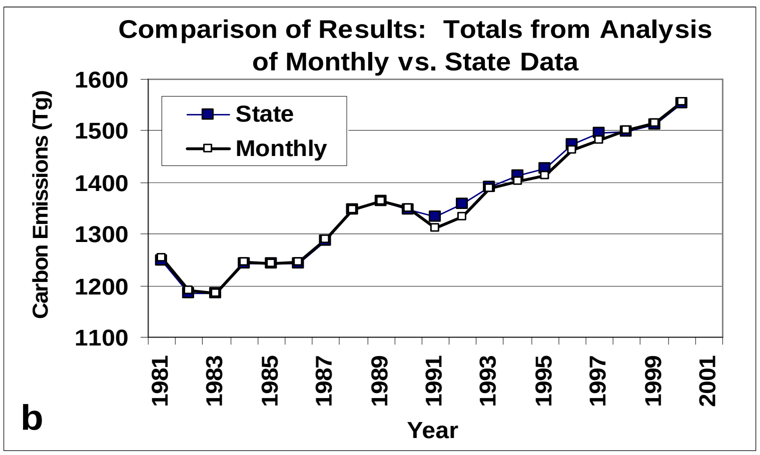



Estimates from DOE data on fuel consumption by state (black squares) vs estimates based on the UN Statistics Office data on apparent fuel consumption for the full United States (open squares).




Estimates based on DOE data on fuel consumption in the 50 U.S. states (black squares) vs estimates based on national fuel consumption for each of the 12 months (open 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).

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

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3

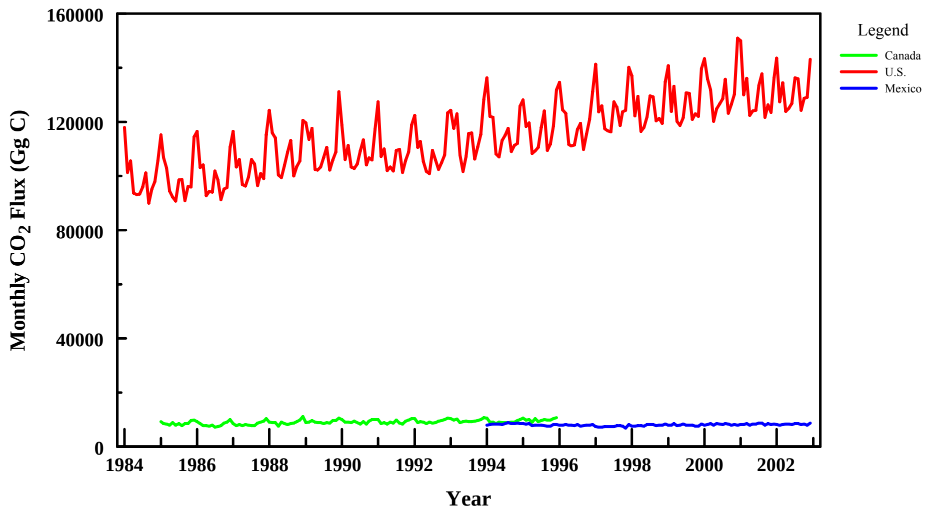


**Figure 7. $CO_2$ emissions from fossil-fuel consumption in North America, by month.** Monthly values are shown where estimates are justified by the availability of monthly data on fuel consumption or sales (from Andres et al. 2005).

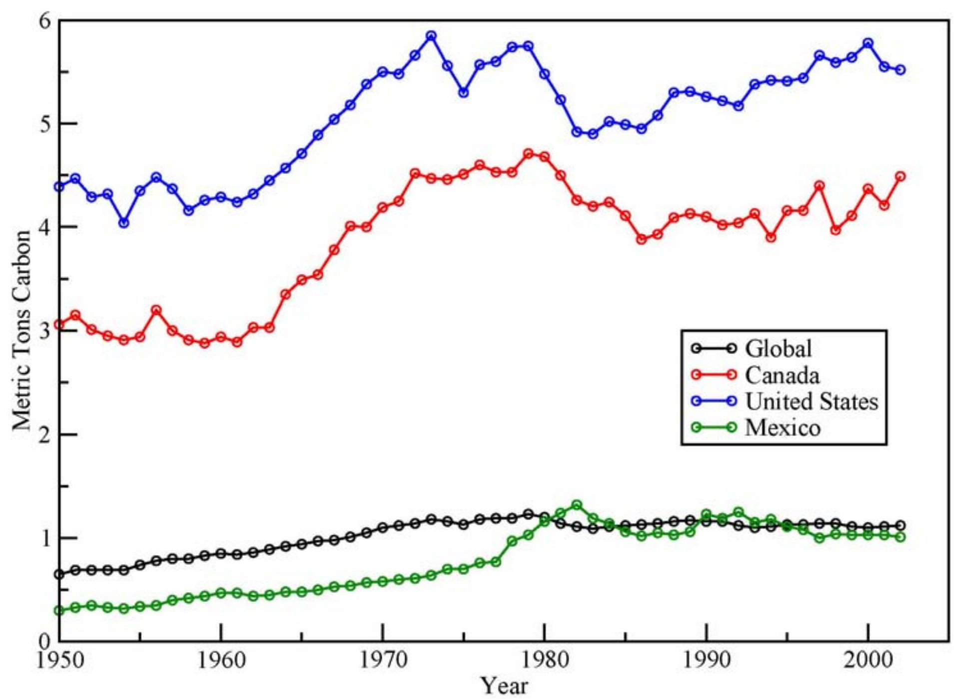


**Figure 8. Per capita emissions of $CO_2$ from fossil-fuel consumption (and cement manufacturing) in the United States, Canada, and Mexico and for the global total of emissions** (from Marland et al. 2005).

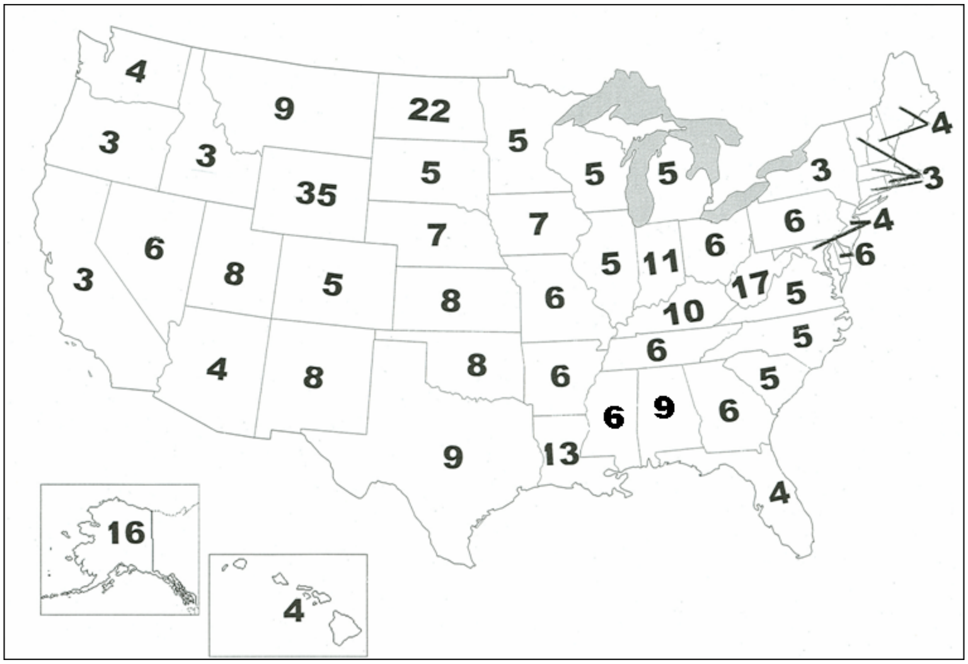


**Figure 9. Per capita emissions of $CO_2$ from fossil-fuel consumption for the 50 US states in 2000.** To demonstrate the range of values, values have been rounded to whole numbers of metric tons per capita. A large portion of the range for extreme values is related to the occurrence of coal resources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

# Chapter 6. Energy Extraction and Conversion

**Lead Author: Thomas J. Wilbanks[1]**

**Contributing Authors: Marilyn Brown,[1] Ken Caldeira,[2] Bill Fulkerson,[3] Eric Haites,[4] Steve Pacala,[5] and David Fairman[6]**

[1]Oak Ridge National Laboratory, [2]Carnegie Institution, [3]University of Tennessee, [4]Margaree Consultants, [5]Princeton University, and [6]Consensus Building Institute, Inc.

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**KEY FINDINGS**

- In recent years, extraction of primary energy sources and their conversion into energy delivery forms (solid, liquid, gas, and electric) in North America released on the order of 2800 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 most (90–95%) of North America's energy extraction and conversion emissions.
- Carbon dioxide emissions from energy extraction and conversion in North America are currently rising.
- The principal drivers behind carbon emissions from energy extraction and conversion are (1) the growing appetite for energy services such as comfort, convenience, mobility, and labor productivity, so closely related to economic and social progress, and (2) the market competitiveness of fossil energy sources compared with alternatives.
- Emissions from energy extraction and conversion in North America are projected to increase in the future. Projections vary among the countries, but increases approaching 50% or more appear likely. Projections for the United States., for example, indicate that $CO_2$ emissions from electricity generation alone will rise to about 3314 Mt $CO_2$ by 2025, a 45% increase over emissions in 2003, 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 extraction and conversion in North America appear dependent upon 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.
- Research and development priorities for managing carbon emissions from energy extraction and conversion 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 energy sources associated with carbon emissions.

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# INTRODUCTION

A significant component of the North American carbon cycle is the extraction of primary energy sources and their conversion into energy delivery forms (solid, liquid, gas, and electric) because both energy resource extraction and energy conversion activities in North America emphasize fossil fuels and their associated emissions of greenhouse gases. This chapter summarizes the knowledge bases related to 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, as when industry practices co-generation as an energy-efficiency strategy, and biomass energy extraction/conversion is directly related to agriculture and forestry. In addition, in addressing options and measures, policy alternatives are often directed at both supply and demand responses, i.e., involving not only emission reductions from supply systems 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 extraction (e.g., mining and oil/gas production) and conversion (e.g., electricity generation and refining) are one of the “big three” sectors accounting for most of total emissions from human systems, along with industry and transportation. The largest share of total emissions from energy supply (not including energy end use) are from (a) coal and other fossil fuel use in producing electricity and (b) fossil fuel conversion activities such as oil refining. Other emission sources are less well-defined but generally small, such as 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.

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, it seems more appropriate to report inventories by country in whatever forms are available than to try to construct a North American inventory that is consistent across all three major countries (which appears unattainable). 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 other fossil fuel production (Environment Canada, 2003).

### 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, $CO_2$ emissions from "energy industries" were 47.3 Mt $CO_2$; from electricity generation they totaled 101.3 Mt $CO_2$, and "fugitive" emissions from oil and gas production and distribution were between 1.9 and 2.6 Mt of $CH_4$, depending on the estimated "emission factor" (Government of Mexico, 2001). An estimate for, say, 2003 might be constructed by increasing these totals in proportion to 1998–2003 gross domestic product growth.

### 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 2003 account for 2409 Mt of $CO_2$, or 41% of a national total of 5870 (EIA, 2004). Emissions from oil refining, natural gas transmission, and other fossil energy supply activities are also substantial, though harder to document because they are grouped with aggregate industrial emissions. Oil refineries are known to be a major source of methane as well as $CO_2$ emissions; natural gas supply systems emit methane as well. For example, 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 scale (AAG, 2003).

### Carbon Sinks Associated with Energy Extraction and Conversion

Generally, energy extraction and conversion are based heavily on mining hydrocarbons from carbon sinks accumulated over millions of years, but carbon sequestration occurs in connection with energy production from biomass during plant growth. Limited strictly to energy sector applications, the total contribution of these sinks to the North American carbon cycle is potentially nontrivial but probably relatively small.

## 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.* In some cases reinforced by policy conditions, 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. 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 from 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., Figure 6.1). National trends and drivers are as follows.

**Figure 6.1. United States $CO_2$ emissions from electricity generation, 1990–2003, in million metric tons $CO_2$** (Source: EIA, 2004).

### Canada

Canada is the only Annex I country that 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 would increase 2000–2020 from about 90 Mt of $CO_2$ equivalent to about 150, while emissions from fossil fuel production would remain relatively stable at about 100 Mt.

### 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, 2005b).

### United States

The Energy Information Administration (EIA, 2005a) projects that $CO_2$ emissions from electricity generation in the United States will rise between 2003 and 2025 from about 2286 to about 3314 Mt, a 45% increase, with three-quarters of the increase associated with greater coal use in electric power plants. EIA projects that technology advances could reduce the increase by as much as 7%. Projections of other emissions from energy supply systems appear to be unavailable.

## 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 resource extraction, conversion, and energy production (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). Examples include very real 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 currently available or 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 wedges 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 wedge 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 some 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 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, although the prospects remain speculative at this time.

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 between about 100 and 325 Gt. 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 summary of policy options because 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).

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

**[TEXT BOX HERE]**

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. 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 global $CO_2$ 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. It estimated that the cost of emission reducing technologies for power generation, compared with coal-fired power, range from 3 to 8 cents/kWh, except for more expensive photovoltaic and solar thermal technologies. According to the IPCC report, based on DOE/EIA analyses in 2000, advanced coal generation technologies such as integrated gasification combined cycle technology would cost between 3 and 4 cents/kWh in the United States without $CO_2$ capture. $CO_2$ capture would raise costs to between 5 and 7 cents. Nuclear energy costs would rise 5 to 6 cents/kWh. Solar energy options would rise from 3 to 5 cents for wind power, 4 to 8 cents for biomass, and 9 to 25 cents for photovoltaics and thermal solar. 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 $CO_2$ 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 $CO_2$, while a $50 price would reduce emissions by 17 to 19%, or 306 to 332 Mt $CO_2$. 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; see page 3-19 in that report) 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,

- other 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 wedges of available technology to transform the capacity to manage carbon emissions from energy supply systems, if that is a high priority for North America.

## REFERENCES

AAG, 2003. Global Change and Local Places: Estimating, Understanding, and Reducing Greenhouse Gases, Association of American Geographers, Cambridge: Cambridge University Press.

Caldeira, K., *et al.*, 2005. Climate Change Technology Exploratory Research. Working paper. Washington, DC: Climate Policy Center.

EIA, 2004. Annual Energy Review, 2004. Washington: Energy Information Administration.

EIA, 2005a. Annual Energy Outlook, 2005. Washington: Energy Information Administration.

EIA, 2005b. International Energy Outlook, 2005. Washington: Energy Information Administration

Environment Canada, 2003. Canada’s Greenhouse Gas Inventory, 1990–2003 (see: http://www.ec.gc.ca/pdb/ghg/inventory_report/2003_report/ts_2_e.cfm).

Environment Canada, 2005. The Green Lane: Climate Change: The Greenhouse Gas Emissions Outlook to 2020 (see: http://www.ec.gc.ca/climate/overview_2020-e.html)

Government Accountability Office (GAO), 2004. Climate Change: Analysis of Two Studies of Estimated Costs of Implementing the Kyoto Protocol. Washington: January 2004.

Government of Canada, 2005. Project Green: Moving Forward on Climate Change, April 2005.

Government of Mexico, 2001. Second National Communication. Submitted to UNFCCC by the Secretaria de Medio Ambiente y Recursos, Naturales. Mexico City.

Hoffert, M. I. *et al*., 2002. Advanced Technology Paths to Global Climate Stability: Energy for a Greenhouse Planet, *Science*, 298 (1 November 2002): 981–87.

Interlaboratory Working Group, 1997. Scenarios of U.S. Carbon Reductions. Prepared by Lawrence Berkeley National Laboratory (LBNL-40533) and Oak Ridge National Laboratory (ORNL/CON-444) for the U.S. Department of Energy.

Interlaboratory Working Group, 2000. Scenarios for a Clean Energy Future. Prepared by Lawrence Berkeley National Laboratory (LBNL-44029) and Oak Ridge National Laboratory (ORNL/CON-476) for the U.S. Department of Energy.

IPCC, 2001. Climate Change, 2001: Mitigation. Contribution of Working Group III to the Third Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge: Cambridge University Press.

IPCC, 2006. Carbon Dioxide Capture and Storage. IPCC Special Report. Cambridge University Press.

Lewis, N., 2005. Global Energy Perspective, Paper presented to the U.S. DOE Laboratory Energy and Development Working Group (LERDWG), Washington, DC: September 2005.

National Academy of Sciences (NAS), 1992. Policy Implications of Greenhouse Warming: Mitigation, Adaptation, and the Science Base. Washington.

National Academy of Sciences (NAS), 1999. Our Common Journey: A Transition Toward Sustainability. Washington, DC: National Academy Press.

National Commission on Energy Policy, 2004. Ending the Energy Stalemate: A Bipartisan Strategy to Meet America's Energy Challenges, Washington, DC, NCEP, December 2004.

National Laboratory Directors, 1997. Technology Opportunities to Reduce U.S. Greenhouse Gas Emissions. Prepared for the U.S. Department of Energy.

Office of Technology Assessment (OTA), 1991. Changing By Degrees: Steps to Reduce Greenhouse Gases, OTA-0-482. Washington.

Pacala, S. and R. Socolow, 2004. Stabilization Wedges: Solving the Climate Problem for the Next 50 Years with Current Technologies, *Science*, 305 (13 August 2004): 968–72.

Pew Center on Global Climate Change, 2002. Climate Change Mitigation in Developing Countries. Report prepared by W. Chandler, *et al.* Washington.

Tremblay, A., 2004. Greenhouse Gas Emissions – Fluxes and Processes: Hydroelectric Reservoirs and Natural Environments, New York: Springer.

U.S. Climate Change Technology Program, (2005). *Strategic Plan: Draft for Public Comment*, September 2005. (see http://www.climatetechnology.gov/stratplan/draft/index.htm).

U.S. Department of Energy, 2004. National Energy Policy/Overview/Canada, in *Energy Trends*. (see http://energytrends.pnl.gov/Canada/ca004.htm).

Wilbanks, T., 1992. "Energy Policy Responses to Concerns about Global Climate Change," in S. Majumdar and others, eds., *Global Climate Change: Implications, Challenges and Mitigation Measures.* Easton, PA: Pennsylvania Academy of Sciences: 452–70.

***[BEGIN TEXTBOX]***

**THE CARBON MITIGATION INITIATIVE AT PRINCETON**

In September 2000, British Petroleum and Ford Motor Company established a partnership with the Princeton Environmental Institute to explore pathways for capturing and sequestering a large fraction of the carbon emissions from fossil fuels, with $20 million in industry funding over a ten-year period. This program assesses the potential of low-carbon energy technologies, studies the feasibility of long-term underground carbon storage, considers impacts of carbon dioxide on the carbon cycle, and analyzes possible pathways for carbon mitigation.

***[END TEXTBOX]***

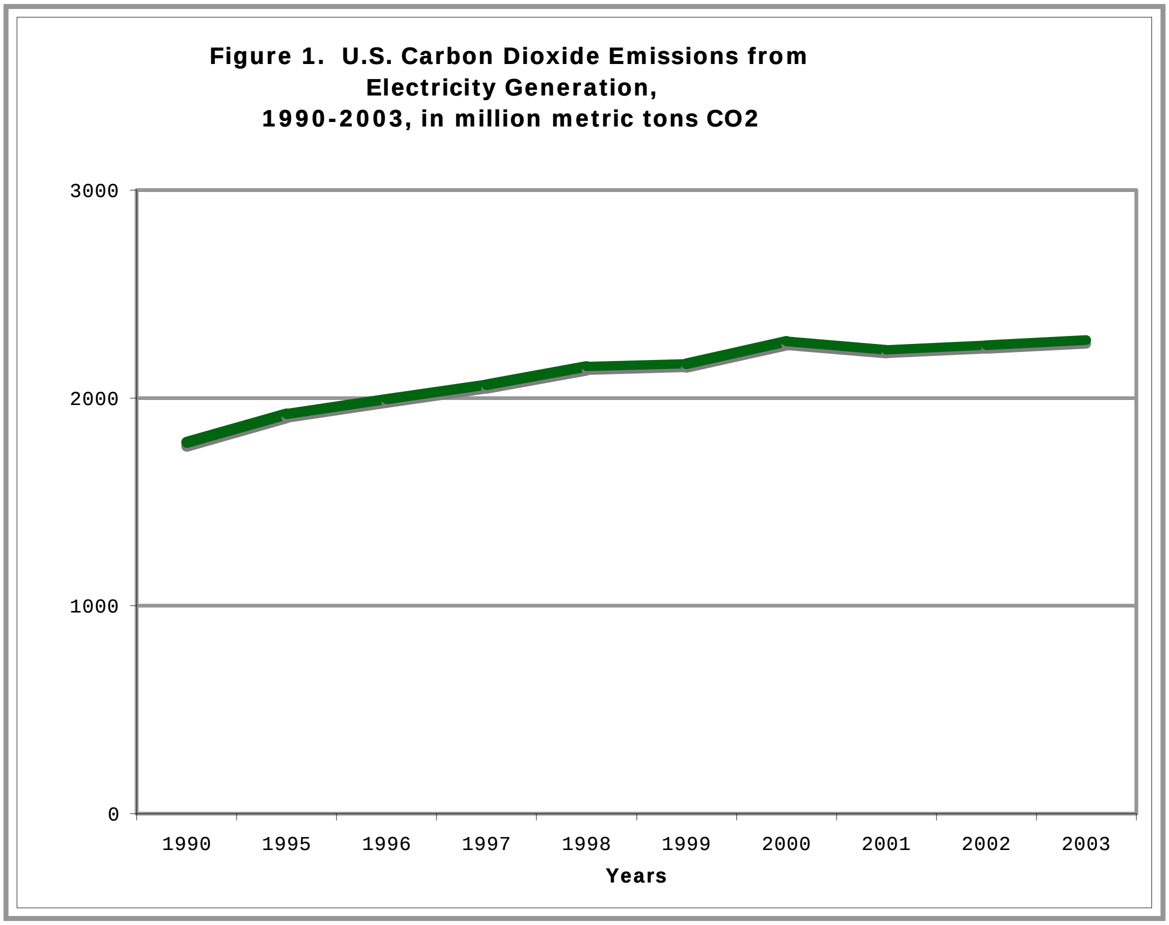


**Figure 6.1. United States $CO_2$ emissions from electricity generation, 1990–2003, in million metric tons $CO_2$** (Source: EIA, 2004).

# Chapter 7. Transportation

**Lead Author: David L. Greene[1]**

[1]Oak Ridge National Laboratory

**KEY FINDINGS**

- The transportation sector of North America released 2120 Mt of $CO_2$ into the atmosphere in 2003, 37% of the total $CO_2$ emissions from worldwide transportation activity and about 22% of total global $CO_2$ emissions.
- Transportation energy use in North America and the associated $CO_2$ 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 2151 Mt $CO_2$ in 2003 to 3149 Mt $CO_2$ 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 $CO_2$ 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 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. There is also a need for improved data, particularly the provision of data to complete the country-specific histories of emissions from transportation, and a consistent description of the accuracy of each country's data.

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% (5.36 Gt $yr^{-1}$) of total global carbon dioxide emissions from the combustion of fossil fuels (24.2 Gt $CO_2$) 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% (an estimated 2120 Mt $CO_2$ in 2005) of the total carbon emissions from worldwide transportation activity (an estimated 5846 Mt $CO_2$ in 2005) (Fulton and Eads, 2004).

Transportation activity is driven 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 reflected in carbon dioxide emissions from the North American transportation sector (Table 7-1).

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

**Table 7-1. Carbon dioxide 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 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, 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. 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 minor in comparison to the volume of $CO_2$ emissions. For example, in the United States, mobile sources including international bunker fuels produced only 132,000 Mt $CH_4$ (or 2.8 Mt $CO_2$ equivalents) in 2003. This is a tiny fraction of the 1770.4 Mt of $CO_2$ emitted by the transportation sector (see Tables 2-3, 2-4, and 2-7 in U.S. EPA, 2005). This chapter will therefore address only the carbon dioxide emissions from transportation activities.

## 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 (Fig. 7-1).

**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, 2003.**

## 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, 7-3b, and 7-3c. Freight activity by mode in Canada, Mexico, and 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. 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, 88% was by light-duty personal vehicles, roughly equally split between passenger cars and light trucks (Fig. 7-4a) (motorcycle pkt, about 0.2% of the total, is included with passenger car). Air travel claims almost 9%; 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 large 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

In all three countries, transportation energy use has grown substantially and relatively steadily. 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 is also visible, especially with respect to energy use for air travel.

**Figure 7-5a and 7-5b. Evolution of transport energy use in Mexico and 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 were effective in restraining 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 and 7-6b. Transport $CO_2$ emissions in Canada and the United States, 1990–2003.**

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). With rising per capita income comes increased vehicle ownership, 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).

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 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).

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 (2005). 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 does in the IEO 2005 Reference Case projection, carbon dioxide emissions would increase from 2151 Mt $CO_2$ in 2003 to 3149 Mt $CO_2$ 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.**

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 $CO_2$ emissions from transportation vehicles to 2050 by regions, WBCSD reference case projection.**

**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. 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 dioxide emissions by 10% from 826 Mt C in the reference case to 744 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 (CAFÉ) 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 1,419 Mt of $CO_2$ emissions, then a 10% reduction amounts to 142 Mt $CO_2$. 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 $21/t $CO_2$.

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 (510 Mt $CO_2$) 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 (246 Mt $CO_2$), or an implied cost of $12/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 sugar cane or cellulose or liquid hydrocarbon fuels produced via biomass gasification and synthesis, appear to be a promising near- and long-term option, while hydrogen could become an important energy carrier after 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 up 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). 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 7-4). The study's premises assumed no change in the year 2000 distribution and efficiency 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). 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 (5) 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.

**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 1752 Mt $CO_2$ in 1997 to 2567 Mt $CO_2$ 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 569 Mt $CO_2$ to 1998 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.

## 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. The North American Transportation Statistics project made a start at producing comprehensive and consistent estimates for all three countries. However, there are many items of missing data, and the country-specific time series are incomplete. 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. A consistent description of the accuracy of each country's data is also needed.

The most pressing research need is for comprehensive, consistent, and rigorous assessments of carbon emissions mitigation potential for North America. The lack of such studies for North America parallels a similar dearth of 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.

Highly promising technologies for reducing transportation GHG emissions include hybrid vehicles, plug-in hybrid vehicles capable of accepting electrical energy from the grid, and 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.

## REFERENCES

(CIA) Central Intelligence Agency, 2005: *The World Factbook*, http://www.cia.gov/cia/publications/factbook , Washington, DC, November 8.

(CBO) Congressional Budget Office, 2003: "The Economic Costs of Fuel Economy Standards Versus a Gasoline Tax," Congress of the United States, Washington, DC, December.

Davis, S.C. and S.W. Diegel, 2004: *Transportation Energy Data Book: Edition 24.* ORNL-6973, Oak Ridge National Laboratory, Oak Ridge, Tennessee, December.

Fulton, L. and G. Eads, 2004: "IEA/SMP Model Documentation and Reference Case Projection." World Business Council for Sustainable Development, http://www.wbcsd.ch/web/publications/mobility/smp-model-document.pdf , July.

General Motors Corporation, Argonne National Laboratory, ExxonMobil and Shell, 2001: *Well-to-Wheel Energy Use and Greenhouse Gas Emissions of Advanced Fuel/Vehicle Systems: North American Analysis, vol. 2.* Published by Argonne National Laboratory, Argonne, Illinois, June.

Government of Canada, 2005. *Moving Forward on Climate Change: A Plan for Honouring our Kyoto Commitment*, available at www.climatechange.gc.ca, Ottawa, Canada.

Greene, D.L. and A. Schafer, 2003: *Reducing Greenhouse Gas Emissions from U.S. Transportation*. Pew Center on Global Climate Change, Arlington, Virginia, May.

Greene, D.L., P.D. Patterson, M. Singh and J. Li, 2005: "Feebates, Rebates and Gas-guzzler Taxes: A Study of Incentives for Increased Fuel Economy." *Energy Policy*, **33**(6), 757-776.

Harrington, W. and V. McConnell, 2003: *Motor Vehicles and the Environment*. RFF Report, Resources for the Future, Washington, DC, April.

(IEA) International Energy Agency, 2005: *World Energy Outlook 2005*. OECD, Paris.

(IEA) International Energy Agency, 2004: *Biofuels for Transport*. OECD, Paris.

(IEA) International Energy Agency, 2001: *Saving Oil and Reducing $CO_2$ Emissions in Transport*, OECD, Paris.

(ILWG) Interlaboratory Working Group, 2000: *Scenarios for a Clean Energy Future*. ORNL/CON-476, LBNL-44029, Oak Ridge National Laboratory, Oak Ridge, TN, Lawrence Berkeley National Laboratory, Berkeley, CA, November.

Marland, G., T. Boden, and R.J. Andres, 2005: *Global $CO_2$ Emissions from Fossil Fuel Burning, Cement Manufacture and Gas Flaring, 1751-2002*, http://cdiac.esd.ornl.gov/ftp/ndp030/global.1751_2002.ems , November 8.

Moomaw, W.R. and J.R. Moreira, 2001: "Technological and Economic Potential of Greenhouse Gas Emissions Reduction." Chapter 3 in Metz, Davidson, Swart and Pan, eds., *Climate Change 2001: Mitigation*, Cambridge University Press, Cambridge, UK.

Nakićenović, N., A. Grűbler, and A. McDonald, 1998: *Global Energy Perspectives*, Cambridge University Press, Cambridge, UK.

(NAS) National Academy of Sciences, 2004: *The Hydrogen Economy*, National Academies Press, Washington, DC.

(NAS) National Academy of Sciences, 2002: *Effectiveness and Impact of Corporate Average Fuel Economy (CAFE) Standards*, National Academy Press, Washington, D.C.

(NATS) North American Transportation Statistics, 2005: Various tables, found at http://nats.sct.gob.mx/lib/series , a joint project of the U.S. Bureau of Transportation Statistics, Statistics Canada and Instituto Nacional de Estadistica Gegrafica Informatica (INEGI), Mexico.

(NRCan) Natural Resources Canada, 2006. "Comprehensive Energy Use Data Base Tables," Transportation Sector, Table 1 Secondary Energy Use by Source, Table 8, GHG Emissions by Transportation Mode, available on the Internet at http://oee.nrcan.gc.ca/corporate/statistics/neud/dpa/trends_tran_ca.cfm.

(NRTEE) National Round Table on the Environment and the Economy, 2005: *Economic Instruments for Long-term Reductions in Energy-based Carbon Emissions*. Renouf Publishing Co., Ltd., Ottawa, Ontario, Canada.

Patterson, P., D. Greene, E. Steiner, S. Plotkin, M. Singh, A. Vyas, M. Mintz, D. Santini, S. Folga, J. Moore, P. Reilly-Roe, K. Cliffe, R. Talbot, P. Khannna, V. Stanciulescu, 2003: *Joint DOE/NRCan Study of North American Transportation Energy Futures*. Energy Efficiency and Renewable Energy, U.S. Department of Energy, Washington, DC, May, available at www.eere.energy.gov/ba/pdfs/final_**2050**_pres.pdf .

Perlack, R.D., L.L. Wright, A.F. Turhollow, R.L. Graham, B.J. Stokes, D.C. Erbach, 2005. *Biomass as Feedstock for a Bioenergy and Bioproducts Industry: The Technical Feasibility of a Billion-Ton Annual Supply*, DOE/GO-102995-2135, U.S. Department of Energy, Washington, D.C., April.

Rodriíguez, H.M., 2005: "Perspectivas del Uso de los Hidrocarburos a Nevel México." Presentation, *Subsecretario de Hidrocarburos*, Mexico City, Mexico, April 14.

(U.S. DOE) U.S. Department of Energy, 2005: *Hydrogen, Fuel Cells and Infrastructure Technologies Program: Multi-Year Research, Development and Demonstration Plann*. DOE/GO-102003-1741, Energy Efficiency and Renewable Energy, available at www.eere.energy.gov , January.

(U.S. DOE/EIA) U.S. Department of Energy, Energy Information Administration, 2005a: *Annual Energy Review 2004*. DOE/EIA-0384(2004), Washington, DC, August. . Also available on line at www.eia.doe.gov .

(U.S. DOE/EIA)U.S. Department of Energy, Energy Information Administration, 2005b: *International Energy Outlook 2005*. DOE/EIA-0484(2005), Washington, DC.

(U.S. DOE/EIA) U.S. Department of Energy, Energy Information Administration, 2003: *Analysis of S.139, the Climate Stewardship Act of 2003*. SR/OIAF/2003-02, Washington, DC, June.

(U.S. DOT) U.S. Department of Transportation, 1998: *Transportation and Global Climate Change: A Review and Analysis of the Literature*. Federal Highway Administration, Washington, DC, June.

(U.S. EPA) U.S. Environmental Protection Agency, 2005: *Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2003*. EPA 430-R-05-003, Office of Atmospheric Programs, Washington, DC, April 15.

Wang, M.Q., 2005: Argonne Expert Addresses Energy and Environmental Impacts of Fuel Ethanol. *TransForum*, vol. 5, no 2, Transportation Technology R&D Center, Argonne National Laboratory, Argonne, Illiniois, November.

Weiss, M.A., J.B. Heywood, E.M. Drake, A. Schafer and F.F. AuYeung, 2000. "On the Road in 2020", Energy Laboratory Report #MIT EL 00-003, Energy Laboratory, Massachusetts Institute of Technology, Cambridge, Massachusetts, October.

Williams, R.H., 2005: *$CO_2$ Capture and Storage Strategies for Coal and Biomass to Reduce GHG Emissions for Synfuels*. Princeton Environmental Institute, Princeton University, Princeton, New Jersey, March.

(WBCSD) World Business Council for Sustainable Development, 2004: *Mobility 2030*. The Sustainable Mobility Project, Geneva Switzerland, available at www.wbcsd.org .

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

| | Carbon dioxide emissions ($Mt\ CO_2$) |
|---|---|
| North America | 2151 |
| Canada | 1865 |
| United States | 169 |
| Mexico | 117 |

Note: Summarized from Table 7-3 in this chapter.

**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 $CO_2$) |
|---|---|---|
| Gasoline | 20,923 | 1,314 |
| Diesel/distillate | 7,344 | 475 |
| Jet fuel/kerosene | 2,298 | 251 |
| Residual | 681 | 53 |
| Other fuels | 124 | 5 |
| Natural gas | 926 | 36 |
| Electricity | 36 | 3 |
| Unalloc./error | 466 | 0 |
| Total | 32,798 | 2,137 |
| | | |
| **United States** | | |
| Gasoline | 18,520 | 1,146 |
| Diesel/distillate | 6,193 | 393 |
| Jet fuel/kerosene | 1,986 | 229 |
| Residual | 612 | 48 |
| Other fuels | 50 | 1 |
| Natural gas | 748 | 35 |
| Electricity | 20 | 3 |
| Unalloc./error | 466.2 | |
| Total | 28,595.2 | 1,855 |

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

| Canada | | |
|---|---|---|
| Gasoline | 1,337 | 96 |
| Diesel/distillate | 704 | 51 |
| Jet fuel/kerosene | 206 | 16 |
| Residual | 66 | 5 |
| Other fuels | 17 | 1 |
| Natural gas | 178 | 0 |
| Electricity | 12 | 0 |
| Unalloc./error | 0 | |
| Total | 2,518 | 169 |

NRCan, 2005, Tables 1 and 8.

| Mexico | | |
|---|---|---|
| Gasoline | 1,066 | 72 |
| Diesel/distillate | 447 | 31 |
| Jet fuel/kerosene | 106 | 7 |
| Residual | 4 | 0 |
| Other fuels | 57 | 3 |
| Natural gas | 1 | 0 |
| Electricity | 4 | |
| Unalloc./error | | |
| Total | 1,685 | 114 |

*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. Electricity is assumed to produce no carbon emissions in end 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 $CO_2$) |
|---|---|---|
| Road | 25,830 | 1,698 |
| Air | 2,667 | 194 |
| Rail | 751 | 50 |
| Waterborne | 1,386 | 68 |
| Pipeline | 990 | 57 |
| | 0 | 84 |
| Total | 31,624 | 2,151 |
| | | |
| United States | | |
| Road | | |
| Light vehicles | 17,083 | 1,113 |
| Heavy vehicles | 5,505 | 350 |
| Air | 2,335 | 171 |
| Rail | 655 | 43 |
| Waterborne | 1,250 | 58 |
| Pipeline/other | 986 | 47 |
| Internatl./Bunker | | 84 |
| Total | 27,814 | 1,865 |
| *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 | 87 |
| Heavy vehicles | 491 | 46 |
| Air | 226 | 16 |
| Rail | 74 | 6 |
| Waterborne | 103 | 8 |
| Pipeline/other | | 7 |
| Total | 2,126 | 169 |
| *Source*: NRCan, 2005; Tables 1 and 8. | | |
| | | |
| Mexico | | |
| Road | 1,518 | 102 |
| Light vehicles | | |
| Heavy vehicles | | |
| Air | 107 | 7 |
| Rail | 22 | 2 |
| Waterborne | 33 | 2 |
| Electric | 4 | 4 |
| Total | 1,684 | 117 |
| *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 $CO_2$ emissions from transportation vehicles to 2050 by regions, WBCSD reference case projection**

| | 2000 | 2005 | 2010 | 2015 | 2020 | 2025 | 2030 | 2035 | 2040 | 2045 | 2050 |
|---|---|---|---|---|---|---|---|---|---|---|---|
| OECD North America | 1995.6 | 2119.7 | 2285.0 | 2447.4 | 2594.4 | 2706.1 | 2814.8 | 2917.9 | 3021.1 | 3125.6 | 3232.5 |
| OECD Europe | 1146.3 | 1224.1 | 1314.7 | 1395.7 | 1438.1 | 1474.6 | 1510.7 | 1525.5 | 1540.2 | 1554.9 | 1569.7 |
| OECD Pacific | 489.2 | 499.5 | 521.9 | 542.2 | 560.5 | 574.4 | 589.0 | 603.7 | 620.1 | 637.7 | 656.4 |
| FSU | 176.9 | 203.7 | 234.1 | 274.3 | 324.2 | 361.1 | 401.2 | 444.0 | 484.4 | 523.2 | 561.5 |
| Eastern Europe | 84.1 | 92.7 | 103.3 | 115.6 | 130.2 | 142.0 | 154.6 | 172.2 | 191.9 | 214.4 | 240.4 |
| China | 251.9 | 314.8 | 394.9 | 488.8 | 599.0 | 702.7 | 826.8 | 967.8 | 1130.0 | 1316.2 | 1530.0 |
| Other Asia | 360.6 | 412.5 | 480.0 | 554.6 | 639.4 | 715.8 | 806.1 | 913.1 | 1037.7 | 1182.5 | 1350.1 |
| India | 137.6 | 163.9 | 199.6 | 242.1 | 292.0 | 338.8 | 395.2 | 457.8 | 534.2 | 628.1 | 743.5 |
| Middle East | 215.3 | 236.7 | 261.5 | 288.6 | 323.5 | 355.2 | 387.0 | 417.3 | 447.1 | 476.5 | 505.6 |
| Latin America | 348.2 | 397.8 | 467.0 | 543.1 | 630.5 | 703.1 | 792.0 | 892.2 | 1008.6 | 1141.2 | 1290.2 |
| Africa | 159.4 | 181.0 | 211.7 | 248.8 | 293.7 | 337.2 | 378.1 | 419.0 | 464.3 | 516.8 | 579.5 |
| **Total—All regions** | **5364.9** | **5846.3** | **6473.6** | **7141.4** | **7825.4** | **8411.1** | **9055.5** | **9730.3** | **10479.7** | **11317.1** | **12259.4** |

*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)**

| Management option | Carbon emission ($Mt\ CO_2$) 2000 | Reduction potential per mode/fuel (%) | | Transportation sector reduction potential (%) | |
|---|---|---|---|---|---|
| | | 2015 | 2030 | 2015 | 2030 |
| **Research, development and demonstration** | | | | | |
| Light-duty vehicles (LDVs) | 1061 | 11[b] | 38[b] | 7[b] | 23[b] |
| Heavy trucks | 294 | 11[b] | 24[b] | 2[b] | 4[b] |
| Commercial aircraft | 196 | 11[b] | 27[b] | 1[b] | 3[b] |
| **Efficiency standards** | | | | | |
| Light-duty vehicles | 1061 | 9 | 31 | 6 | 18 |
| Heavy trucks | 294 | 9 | 20 | 2 | 3 |
| Commercial aircraft | 196 | 9 | 22 | 1 | 2 |
| **Replacement and alternative fuels** | | | | | |
| Low-carbon replacement fuels (~10% of LDV fuel) | 100 | 30 | 100 | 2 | 7 |
| Hydrogen fuel (All LDV fuel) | 1061 | 1 | 6 | 1 | 4 |
| **Pricing policies** | | | | | |
| Low-carbon replacement fuels (~10% of LDV fuel) | 100 | 30 | 100 | 2 | 6 |
| Carbon pricing (All transportation fuel) | 1792 | 3 | 6 | 3 | 6 |
| Variabilization (All highway vehicle fuel) | 1355 | 8 | 12 | 6 | 9 |
| **Behavioral** | | | | | |
| Land use and infrastructure (2/3 of highway fuel) | 903 | 5 | 10 | 3 | 5 |
| System efficiency (25% LDV fuel) | 265 | 2 | 5 | 0 | 1 |
| Climate change education (All transportation fuel) | 1792 | 1 | 2 | 1 | 2 |
| Fuel economy information (All LDV fuel) | 1061 | 1 | 2 | 1 | 1 |
| **Total** | **1792** | | | **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.]

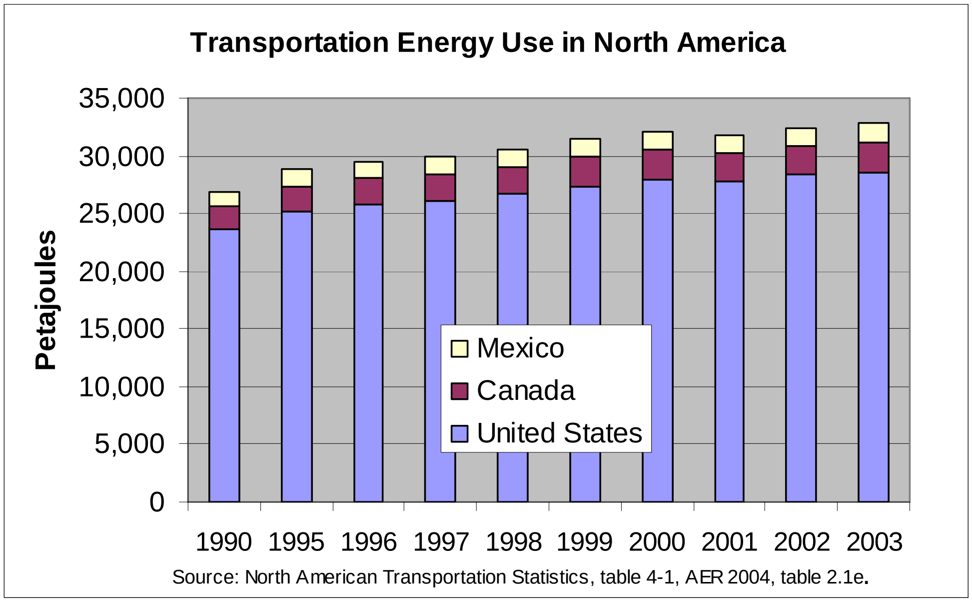


**Fig. 7-1. Transportation energy use in North America, 1990–2003.**

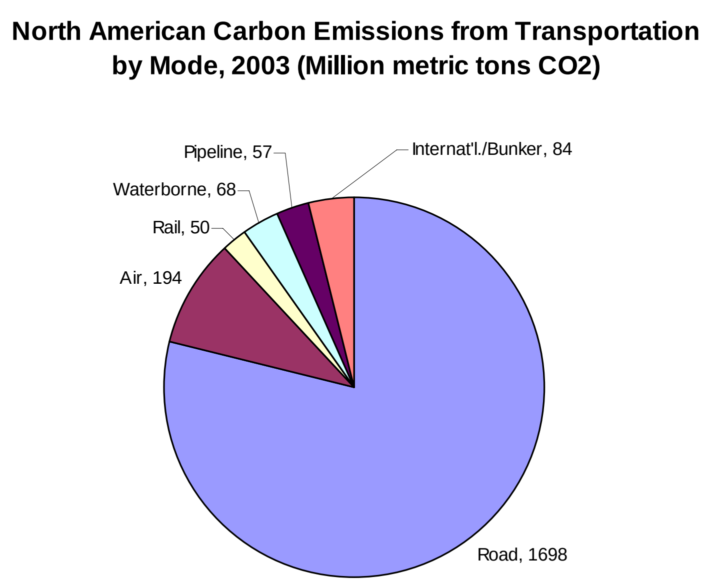


**Fig. 7-2. North American carbon emissions from transportation by mode (million metric tons $CO_2$) 2003.** *Source*: Table 7-3, this chapter.

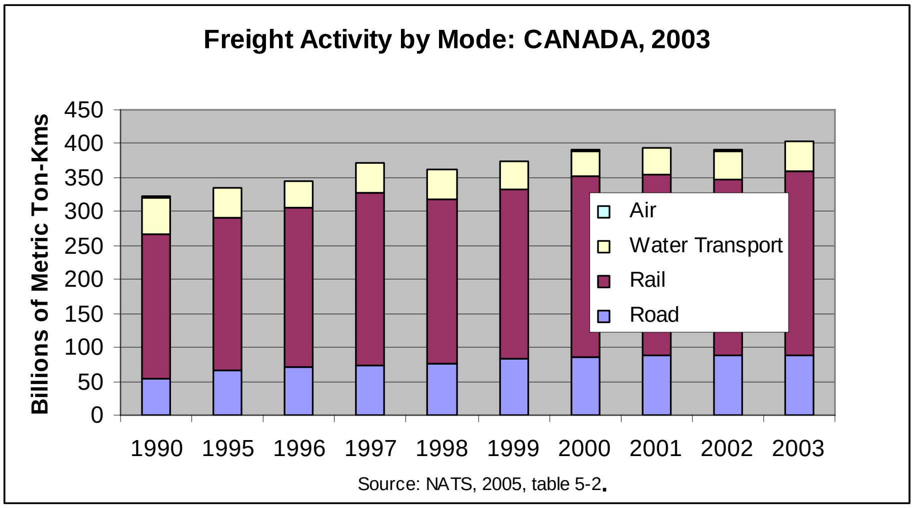


**Fig. 7-3a. Freight activity by mode in Canada, Mexico, and the United States.**

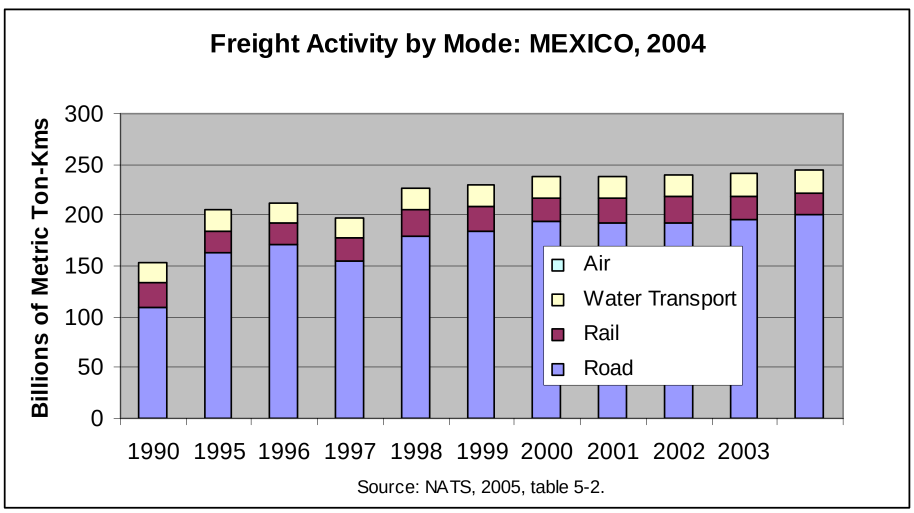


**Fig. 7-3b. Freight activity by mode in Canada, Mexico, and the United States.**

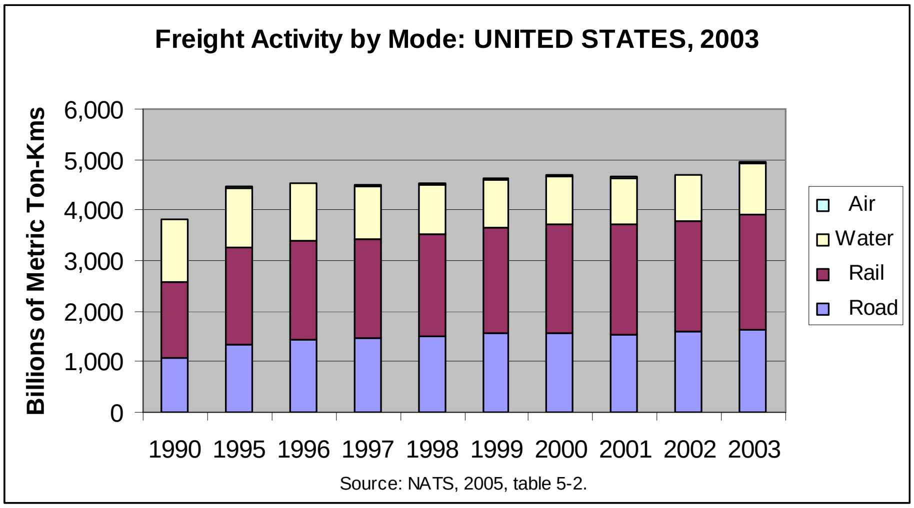


**Fig. 7-3c. Freight activity by mode in Canada, Mexico and the United States.**

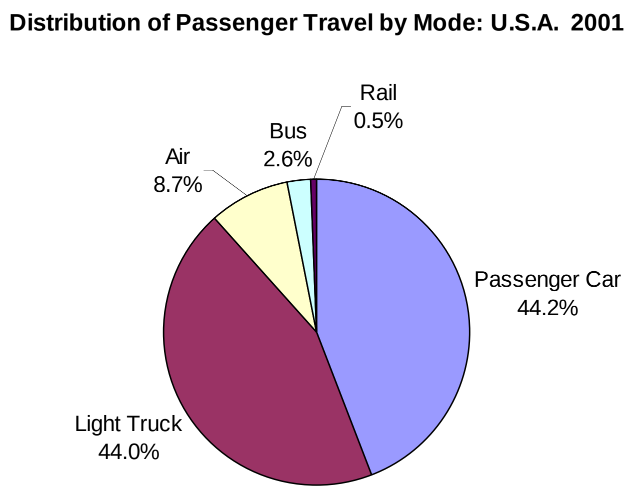


**Fig. 7-4a. Distribution of passenger travel in the United States by mode.** *Source*: Table 8-1 in NATS, 2005.

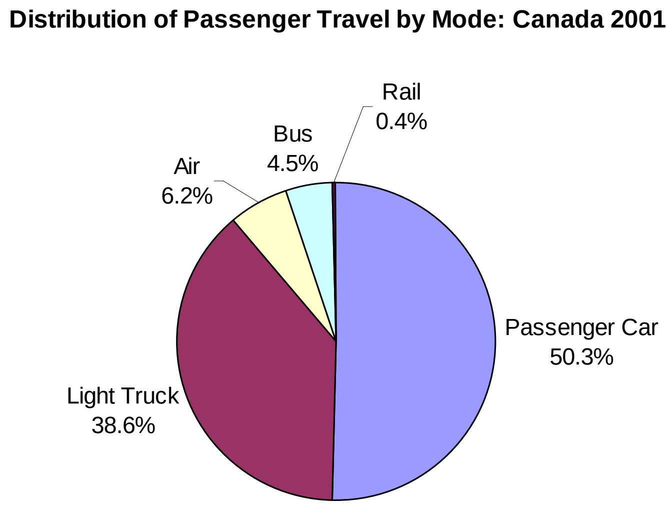


**Fig. 7-4b. Distribution of passenger travel by mode in Canada.** *Source*: Table 8-1 in NATS, 2005.

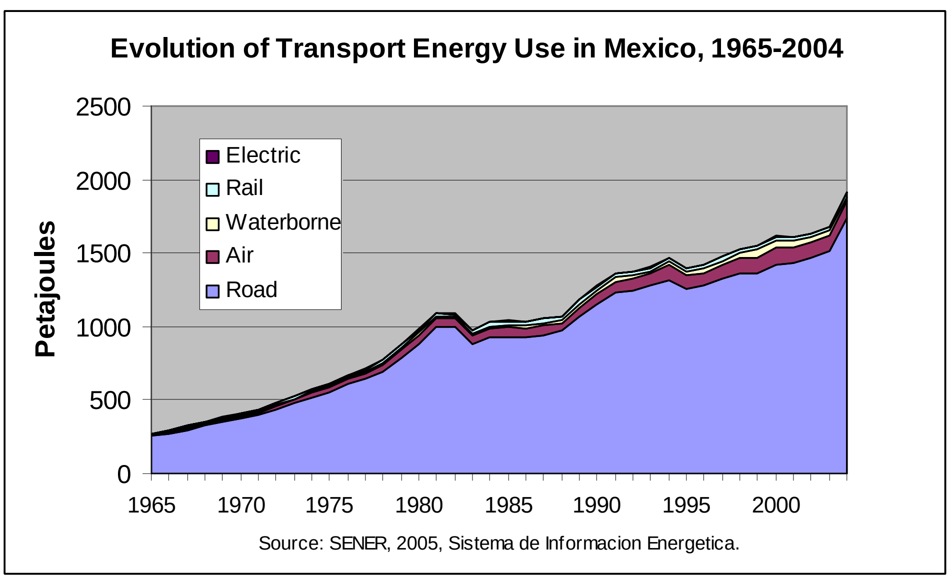


**Fig. 7-5a. Evolution of transport energy use in Mexico and the United States.**

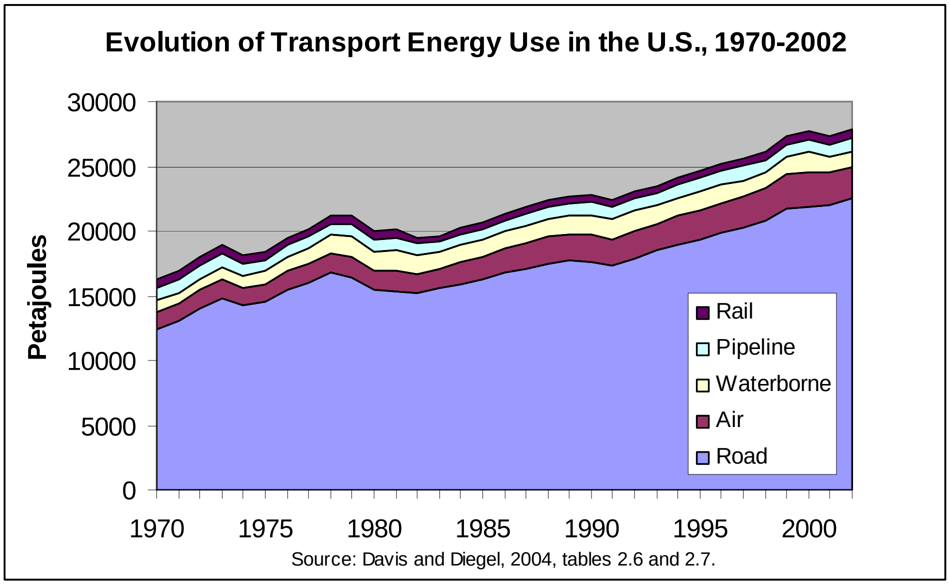


**Fig. 7-5b. Evolution of transport energy use in Mexico and the United States.**

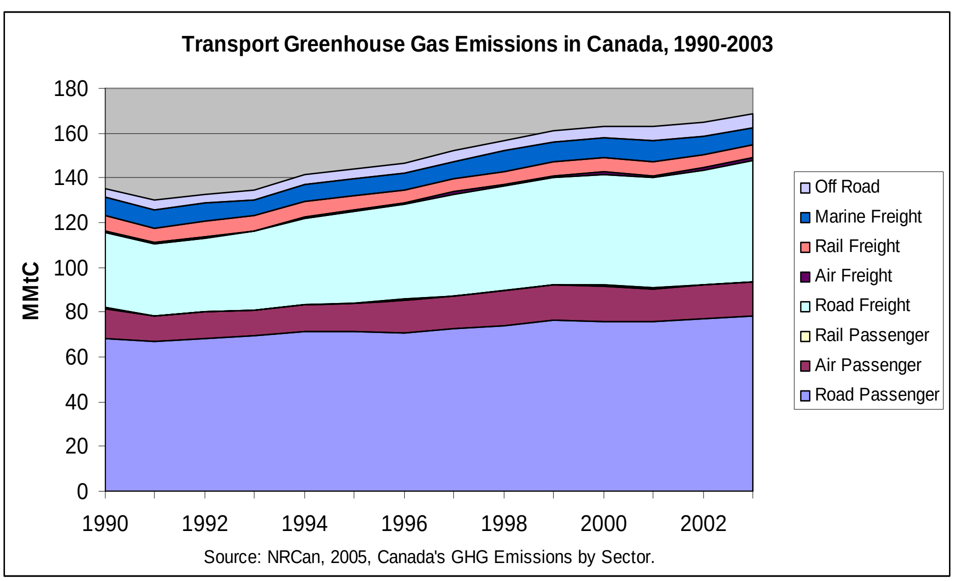


**Fig. 7-6a. Transport $CO_2$ emissions in Canada and the United States, 1990–2003.**

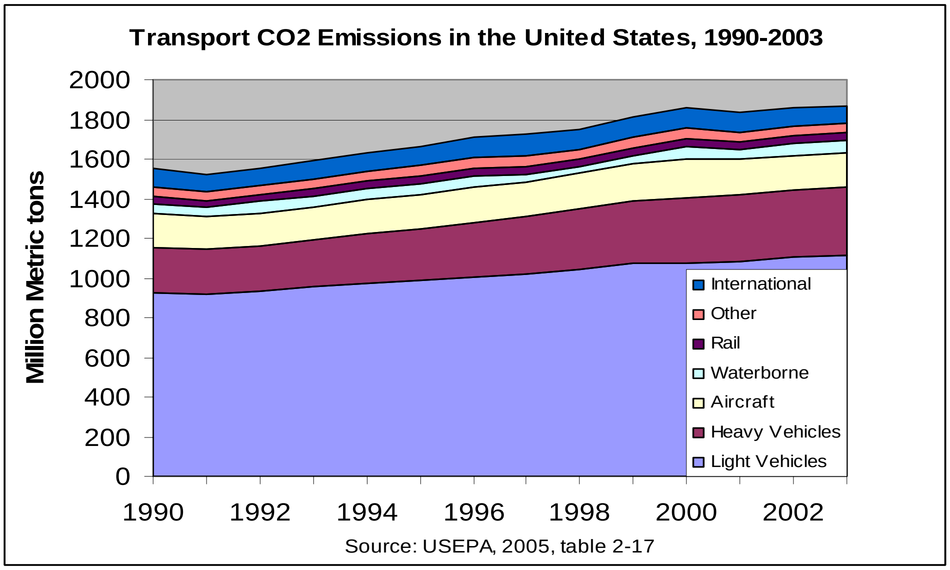


**Fig. 7-6b. Transport $CO_2$ emissions in Canada and the United States, 1990–2003.**

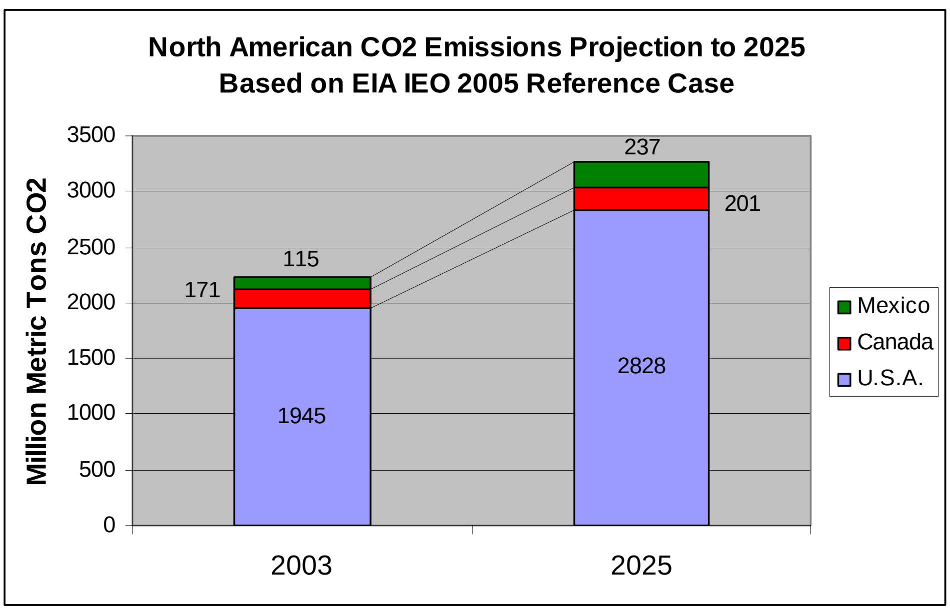


**Fig. 7-7. Projected carbon dioxide emissions from the North American transport sector in 2025.** *Source*: Fulton and Eads, 2004.

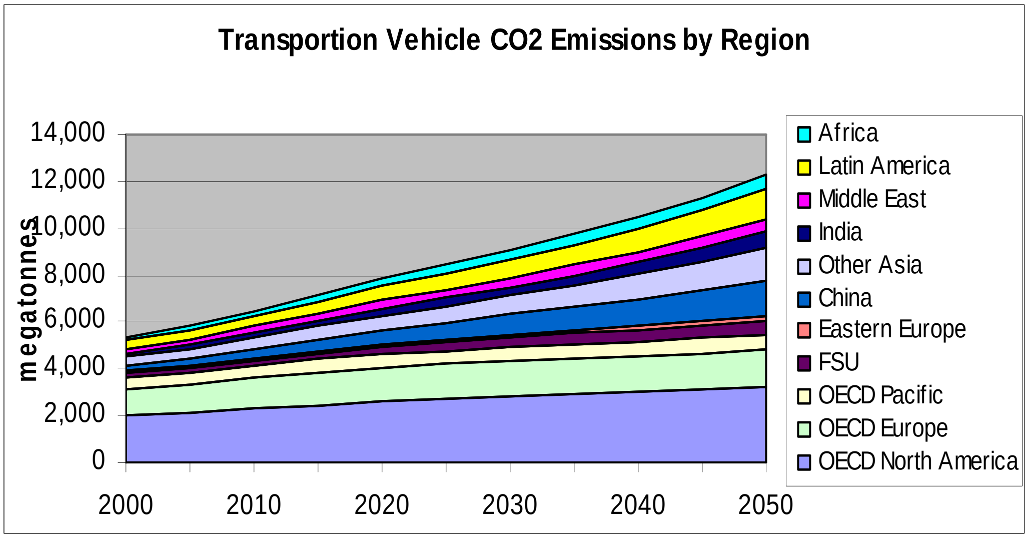


**Fig. 7-8. WBCSD projections of world transportation vehicle $CO_2$ emissions to 2050.**
*Source*: Fulton and Eads, 2004.

# Chapter 8. Industry and Waste Management

**Lead Author: John Nyboer[1]**

**Contributing Authors: Mark Jaccard[2] and Ernst Worrell[3]**

[1]Canadian Industrial Energy End-Use Data and Analysis Centre (CIEEDAC), Simon Fraser University, [2]Simon Fraser University, [3]Lawrence Berkeley National Laboratory

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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 accounts for 13.4 Mt of $CH_4$ (282 Mt $CO_2$e), 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.

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## INTRODUCTION

This chapter presents two components of the carbon cycle. The first section assesses carbon flows through industry (manufacturing, construction, and industry process emissions, but excluding fossil fuel mining, and processing).[1] The second section assesses municipal waste disposal (primarily landfills) for its impact on the fate of carbon and the release of methane and other carbon-based gases.

In 2002, industry was responsible for 5220.6 Mt of $CO_2$, 21% of anthropogenic $CO_2$ emissions to the atmosphere worldwide (this includes 4322.9 Mt from fuel combustion and 897.7 Mt from the industrial processes described later in this chapter). North America's industry contributed 758.7 Mt of combustion-sourced emissions and 66.8 Mt of process emission for a total of 826 Mt, 16% of global totals. The manufacturing industry and its process emissions contributed only 12% of total North American GHG emissions, lower than in many other parts of the world, because of the high $CO_2$ intensity of the continent's transportation sector and the significance of heating and cooling energy demands. But with North America's population at 6.8% of the world total, the continent's industry contributed a proportionally larger share of total industrial emissions than the rest of the world (see comparative tables and graphs, 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% during that period (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 energy demand growth lower than industrial GDP

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[1]This includes direct flows only. Indirect carbon flows, such as the carbon released due to electricity generation, are not associated with the industry that consumed the electricity but with power generation (see Chapter 6).

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

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. $CO_2$ is generated from aerobic metabolism in waste removal and storage processes. Because this $CO_2$ arises from biological material, it is considered neutral in terms of GHG emissions. Methane ($CH_4$), released from anaerobic activity at waste treatment plants and landfill sites, forms a substantial portion of carbon emissions to the atmosphere. Given its much higher rating as a GHG gas in terms of 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 $SF_6$.**

Landfills are not "efficient" aerobic or anaerobic digesters; substantial sequestration of carbon occurs.[5] Data on carbon buried in landfills are poor. The Environmental Protection Agency (EPA) used 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 and are assumed to remain in the landfill site as sequestered carbon. In all, more than 80% of the carbon entering a landfill site may be sequestered, depending on moisture, aeration, and other conditions of the site. In another paper, Bogner and Spokas (1993) estimate that "in general, more than 75% of the carbon deposited in landfills remains in sedimentary storage."

## INDUSTRY CARBON CYCLE

Carbon may enter industry as a fuel, providing energy for the completion of industrial processes, or as a feedstock where the carbon becomes entrained in the final product generated by that industry. Thus, carbon exits industrial sites either as a constituent of a product or as a waste. Carbon in the waste stream can be distinguished as atmospheric and non-atmospheric, the former being comprised of process and combustion-related emissions. Process emissions refer to $CO_2$ emitted as a result of the transformation of the material inputs to the production process (i.e., a non-combustive source). For example, cement

[3] Decomposition analyses assess the changes in the overall energy consumption or emissions release in terms of the major factors affecting such a change. Changes in energy consumption, for example, can result from increase industry activity, changes in relative productivity to or from more intense industry subsectors, or changes in material or energy efficiency in industrial processes.

[4] While not carbon-based, $N_2O$ from sewage treatment is shown in Fig. 2 to show its relative importance as a GHG.

[5] IPCC guidelines currently do not provide a methodology for addressing landfill sequestration. Such guidelines are to be included in the 2006 guidelines.

production involves the calcination of lime, a process that chemically alters limestone to form calcium oxide and releases $CO_2$. Combustion-related emissions such as $CO_2$ occur when carbon-based fuels provide thermal energy to industrial processes.

## Overview of Carbon Inputs and Outputs

Relatively speaking, industry as it is here defined 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 in the form of biomass (primarily wood), limestone, soda ash, oil products, coal/coke, natural gas, and natural gas liquids. These inputs are converted to dimension lumber and other wood products, various types of paper and paperboard, cement and lime, glass, and a host of chemical products, plastics, and fertilizers based on oil, coal, natural gas, and natural gas liquids.

While the bulk of the input raw material leaves the industrial site as a product, some of the carbon leaves the process as $CO_2$ (e.g., from limestone in cement production), and some is converted to fuel combusted in the plant. Waste wood (or hog fuel) and black liquor, a product generated in the production of chemical pulps, are burned to provide process heat or steam for digesting wood chips in the production of chemical pulp and for drying paper or wood products. In some cases, electricity is cogenerated from this biomass energy. Chemical processes utilizing natural gas or natural gas liquids often generate off-gases that can be mixed with conventional fuels to 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 in a process known as "reduction." In most of the literature, such carbon is considered an input to the process even though it acts as a fuel (i.e., it unites with oxygen to form $CO_2$ and releases heat, just as it would in combustion processes). For example, in metal smelting and refining processes, a carbon-based reductant is used to separate oxygen from the metal atoms. Coke, a product of the destructive distillation of high-quality coal, enters a blast furnace with iron ore to strip off the oxygen associated with the iron, leaving pig iron at the bottom of the blast furnace with $CO_2$ exiting at the top. Carbon anodes in electric arc furnaces in steel mills and in the 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 (wood, paper, chemical products), as a waste product (waste wood, pulp mill sludge), or as a gas, usually $CO_2$. The carbon in the commodities generated may, in turn, be utilized by other industries to be released as another commodity or as a waste product or emission.

Process emissions are $CO_2$ emissions that occur as a result of the process itself—the calcining of limestone releases about 0.5 Mt $CO_2$ per metric ton of clinker (unground cement) or about 0.8 Mt per metric ton of lime,[6] depending on the degree to which limestone or dolomite is used as a feedstock.[7] The oxidation of carbon anodes generates about 1.5 Mt $CO_2$ in the production of a metric ton of aluminium.[8] Striping hydrogen from methane to make ammonia releases about 1.6 Mt $CO_2$ per metric ton of ammonia.

Combustion of carbon-based fuels results in the release of $CO_2$ to the atmosphere. In many cases, the combustion process is not complete, and other carbon-based compounds may also be released, such as carbon monoxide, methane, or mixtures of more complex carbon products known as volatile organic compounds (VOCs). 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 industrial sites as per the industry categories in Sect. 2.2. Numbers for the full North American budget are defined in the figure. Comparable diagrams for the individual countries are presented in Appendix 8A. On the left side of Fig. 8-2, all carbon-based material is accounted for by industry sector, 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 input to industry, as well as some of the feedstock materials, leave in the waste stream as emissions from fuel combustion, process emissions, or as other products and waste. The potential for carbon capture and storage is 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

Energy-intensive industries differ significantly in their carbon cycle dynamics. Figure 8-2 shows the current contribution to the carbon cycle of different industries.

[6]In these industries, more $CO_2$ is generated from the process of limestone transformation than from the fossils fuels combusted to drive the transformation.

[7]The calcination of limestone also takes place in the iron and steel, pulp and paper, glass and sugar industries.

[8]Ceramic anodes may soon be available to aluminum producers and significantly reduce $CO_2$ release in the production of aluminum.

### Pulp and Paper

While pulp and paper products are quite energy-intensive, much of the energy used in their production is obtained from biomass. By using biomass waste, such as hog fuel and black liquor, some types of pulp mills (and associated paper plants) are energy self-sufficient. These plants could be considered carbon neutral because the capture of carbon as forests grow is assumed to offset the $CO_2$ released from such activities.[9] However, fuel handling difficulties and air quality concerns (especially from particulates) can arise from the use of biomass as a fuel depending on the location.

### Cement, Lime, and Other Nonmetallic Minerals

Cement and lime production requires the calcination of limestone, which releases $CO_2$. This process emission is releases about 0.78 Mt $CO_2$ per metric ton of lime calcined.

$$\underset{\text{calcium carbonate}}{CaCO_3} \rightarrow \underset{\text{calcium oxide}}{CaO} + \underset{\text{carbon dioxide}}{CO_2}$$

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 during the calcining process.

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

$$\underset{\text{sodium bicarbonate}}{2NaHCO_3} \rightarrow \underset{\text{sodium carbonate}}{Na_2CO_3} + \underset{\text{carbon dioxide}}{CO_2} + \underset{\text{water}}{H_2O}$$

### Nonferrous Metal Smelting and Iron and Steel Smelting

Often metal smelting requires the reduction of metal oxides to obtain the pure metal. In such operations, a reductant, a substance that can carry away the oxygen from the metal, is required. Typically, the reductant is carbon, usually in the form of coke. The coke is added to the hot metal oxide, as in the case of iron, zinc, or magnesium, to generate the reduced metal and $CO_2$. Such reduction processes

[9] Based on guidelines from the United Nations Framework Convention on Climate Change, biomass-based industries such as pulp and paper are deemed, in effect, to be carbon neutral in so far as biomass is concerned (UNFCCC, IPCC guidelines).

generate relatively pure streams of $CO_2$ (with some CO), which improves the potential for capture and storage.

In electric arc furnaces, carbon anodes decompose to $CO_2$ as they melt the scrap iron and steel feed in steel mini-mills. In a Hall-Heroult cell in the aluminium industry, a carbon anode oxidizes when an electric current forces oxygen from aluminium oxide (alumina) in the production of aluminium, with $CO_2$ as a by-product.

### Metal and Nonmetal Mining

Mining involves the extraction of ore and its transformation into a more concentrated form. This involves milling (grinding) the ore after it has been transported from the mine site and removing mineral-bearing material from the ground rock. Much of the process involves grinding and separating, most of which is done through the action of devices driven by electric motors. Thus, a large proportion of the activity in mining operations requires electricity rather than fossil fuels directly—although fossil fuels may have produced the electricity. Some processes, like the sintering or agglomeration of iron ore and the liquid extraction of potash, use a considerable amount of fossil fuels directly. And, of course, much of the movement of the ores from mine to mill is accomplished by diesel-driven motorized vehicles.

### Chemical Products

This diverse group of industries includes energy-intensive electrolytic processes as well as the consumption of large quantities of natural gas and natural gas liquids (hydrocarbon liquids found with natural gas) as a feedstock to produce commodities like ammonia, methanol, and hydrogen from natural gas and monomers such as ethylene and propylene from natural gas liquids. These products provide the feedstock for many synthetic resins and plastics. Some chemical processes, such as the production of ammonia, generate fairly pure streams of $CO_2$ suitable for capture and sequestration.

### Forest Products

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

### Other Manufacturing

Most of the remaining industries, while economically important, each play a relatively minor role in the carbon cycle because they are not energy intensive and use little biomass—the exceptions being the food industry, the beverage industry, and some textile industries. Industries in this group include the automotive industry, electronic products, leather and allied products, fabricated metals, furniture and related products, and plastics and rubber products. In aggregate, however, these various industries contribute significantly to total industrial $CO_2$ emissions.

## 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 industries are further disaggregated into energy-intensive industries, growing at 1.5% per year, and non-energy intensive industries, growing at 2.9% per year. The slower growth in the energy-intensive industries is reflected by an expected decline of 1.6% per year in the energy intensity of United States industrial output over the forecast (EIA forecast 2005).

The International Energy Agency reviewed energy consumption and emissions during the last 30 years to identify and project underlying trends in carbon intensity.[10] 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 fossil fuels. The shift from coal and refined petroleum products to natural gas and electricity[11] 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.

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

[11]Emissions associated with electricity are considered "indirect" to industry and are allocated to the electricity supply sector. Thus, a shift to electricity is like a shift from coal to natural gas, moving from a more $CO_2$ intense energy supply to a less intense one (in this case, to 0 $CO_2$/unit). However, one shifts the allocation of associated $CO_2$ to another sector as well; there is no net reduction in $CO_2$ emitted, unless the supply sector chooses to generate electricity from a less $CO_2$ intense source.

## 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 reduces energy consumption per unit of output by 26 to 95% (see Table 8-1). Further work on materials substitution also holds promise for reduced energy consumption and 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.

**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, both generic and industry-specific. These include efficient Hall-Heroult cell retrofits in aluminium production, black liquor gasification in kraft pulp production, and shape casting in iron and steel industries. Worrel *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 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), as shown in Table 8-2. Of the 25 Mt generated in Canada, the contribution from each of these sources is about the same (Statistics Canada, 2004). In Mexico, as much as 20% of wastes are not systematically collected, and no disaggregated data are available (EPA, 2005).

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

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, a slightly lower percentage in Canada. Incineration can reduce the waste stream in a given location by up to 80%, but this ensures that more of the carbon reaches the atmosphere as opposed to being buried in solid form (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 for electricity generation and heat production.

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[12] depending on the conditions of the landfill site, as well as a trace amount of carbon monoxide (which soon becomes $CO_2$ in the atmosphere) 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,[13] a relatively small amount compared to total other (sub)sectors in this chapter. One should consider this derived estimate highly uncertain and not of the same calibre as other emissions data provided here. Also recall that, in the context of IPCC assessment guidelines, $CO_2$ emissions from biological sources are considered GHG-neutral and that these emissions are from biomass.

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). However, the complexity of this assessment and the data required to support it from the multiple sources prevented any further assessment from taking place for this report.

Fugitive methane gases are the result of anaerobic digestion and can be captured and, like incineration, used to generate power and steam. 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 $CO_2$

[12]When based on gas volumes, this would mean that roughly equivalent amounts of carbon are released to the atmosphere in $CO_2$ as in $CH_4$.

[13]14 Mt of $CH_4$ (see Table 8-3) are equivalent, volume wise at standard temperature and pressure, to 38 Mt of $CO_2$.

released from such combustion is considered biological in origin (i.e., the methane used arose from biological material). Thus, only methane emissions, at 21 times the $CO_2$ global warming potential, are included as part of GHG inventories.[14] Their combustion greatly alleviates the net contribution to GHG emissions and provides power or steam that might prevent the combustion of fossil fuels elsewhere for these purposes.

## COSTS RELATED TO CONTROLLING ANTHROPOGENIC IMPACTS ON THE CARBON CYCLE

The subject of defining costs associated with reducing anthropogenic impacts on the carbon cycle is one of the most contentious of issues in any carbon-focussed analysis. Different modelling approaches to cost assessments (top-down, bottom-up, applicable discount rates, social costing, cost effectiveness, no regrets, etc.), different understandings of what costs actually include (risk, option values, welfare, intangibles, etc.), different values associated with energy demand in different countries (accessibility, availability, infrastructure, resource type and size), the number of possible actions and technologies included in the analysis, and the perspective on technology development all have an impact on how one evaluates costs. Should analysts consider only historical responses to energy prices, production and demand elasticities, income changes and the like? Does one consider only technology options and their strict financial costs? Should one review producers' or consumers' welfare issues associated with new technologies? Are there local, national, international issues to be broached?

How might one reduce emissions in industrial and waste sectors? Methods of reduction can be classified as:

- reducing process/fugitive emissions or converting currently released emissions (e.g., reduce process emissions from cement, lime production, capture or prevent fugitive emissions leaks from pipelines or combustion of emissions such as methane from landfills, cogeneration using landfill offgases);
- energy efficiency, including combined heat and power;
- process change (materials efficiency, recycling, substitution between materials or between materials and energy);
- fuel substitution; and
- carbon capture and storage.

Variation within industries is significant, but some simple allocation of a broad range of costs can be attributed to potential reductions over a set time period. We suggest the cost categories ("A" through "D")

[14]Theoretically, one should assume a factor of 20 because, were the methane released as $CO_2$, it would be considered to have no net GHG effect.

shown in Table 8-3. The table contains estimates of the percentage reduction at the grouped cost levels. The costs represented here are not drawn from a single source but, rather, are the authors' estimates based on a long history of interaction with cost reported in various documents.

**Table 8-3. Approximate costs and reductions potential**

When looking at cost numbers like this, one should remember that, for each $10 cost increment per Mt $CO_2$ (or $2.73 per Mt C), gasoline prices would increase about 2.4¢/L (9¢/U.S. gal). Diesel fuel cost would be slightly higher, at nearly 2.7¢/L (10¢/U.S. gal). At this rate, costs per GJ (slightly smaller than one million BTU) vary by fuel: coal would rise about 90¢/GJ, depending on type, HFO by 73¢, and natural gas by 50¢. Were one to use these fuels to generate electricity at a 35% efficiency rate, the cost increase in coal-fired electricity generation would be about 0.8¢/kWh, about 0.65¢/kWh for HFO fired electricity, and about 0.45¢/kWh if natural gas was used.

Of course, as the cost of carbon increases one can always obtain greater reductions, but the return on these expenditures becomes marginal or insignificant and so are not included in the cells of Table 8-3. If two cost categories are shown in a cell (e.g., A/B) and the quantity reduced ($\%Q_{red}$) as 15/20 in the neighbouring cell, the value associated with the second portrays the marginal addition that may be made at that increased expenditure level. In this example, spending up to $25/t $CO_2$ may reduce emissions by 15% and with a further expenditure of up to $50/t $CO_2$ would add a further 20% for a total of 35% were all expenditures to reduce emissions made (see "Metal Smelt" in Table 8-3).

Because not all actions are applicable to all industries, as one aggregates to an "all industry" level (top line in the table), the total overall emissions reduction level may be less than any of the individual industries sited. We provide an approximation of each industry, but if potentials for a certain option are not available in that industry (e.g., process change), this lowers the average for the aggregate category.

## Some Explanatory Notes

The five categories are not independent, and thus, reductions are not additive across categories. We have tried to isolate somewhat what reductions might be like were one to focus only on that particular category. Data come from a variety of sources and often focus on more than one of the following aspects (Hertzog, 1999; Martin *et al.*, 2001; Jaccard *et al.*, 2002; Jaccard *et al.*, 2003; Jaccard, Nyboer, *et al.*, 2003; Worrel *et al.*, 2004; DOE, 2006).

**Process and Fugitives:** Process and fugitive reductions are only available in certain industries. For example, cement and lime calcination, ammonia production, and others (see above) have process emissions that one may be able to control or manipulate. Because wood products industries burn a lot of wood waste, fugitives (methane and VOCs) are higher than in other industries and reduction potentials exist. In this particular example, fugitive emission and reduction potential are small compared to those possible in petroleum refining and upstream gas and oil.

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. In an EPA study (EPA, 2003a), estimates of between 40% and 60% of methane available for capture may generate net economic benefits.

**Energy Efficiency**: While efficiency is important for more than just $CO_2$ reduction, depending on how one views the advent and penetration of new technologies, potentials for reduction are limited if one does not consider changing processes or switching fuels; that is, the potential for emissions reductions from efficiency improvements is strongly linked with these other two avenues. For example, using DRI processes in iron smelting, moving to Cermet anodes in electric arc furnaces in iron and steel and aluminium smelting industries or shifting to hydrometallurgic processes from pyrometallurgic ones in nonferrous metal smelting can significantly improve efficiencies and lower both combustion and process GHG emissions; we include them here as an efficiency improvement even though they may be considered a process change and fall under the next column in Table 8-3.

Because so much emphasis is placed on this particular avenue to reductions, we define it here as a separate category even though it is difficult to disaggregate from fuel switching and process change. Modeling from a more technical, strict end-use approach using technology possibility curves or similar factors for efficiency improvement over time tends to show higher potentials than when one uses hybrid approaches that try to assess the impacts of costs on technology choice (and thus energy demand); we have portrayed the outcome of the latter and provide 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; Worrel *et al.*, 2004).

**Process Change**: Process change in its broader sense is difficult to estimate; it 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 be different than before the change was made. In pulp production, for example, one could move away from kraft pulp and increase production ratios (the kraft process only converts one-half the tree into pulp), but will market acceptability for the end product be unaffected? Reducing the actual clinker content of ground cement can radically alter emissions levels. Numerous substitution possibilities exist in the rather diverse Other Manufacturing section (carpet recycling, alternative uses for plastics, etc.).

**Fuel Substitution**: As mentioned, it is difficult to isolate fuel substitution and efficiency improvement because fuel types do contain inherent qualities that affect efficiency. While fuel substitution is an important method of reducing carbon, this is beneficial only if options to move to less carbon-intense fuels exist. In pulp and paper, one can move to biomass, but the industry already depends on at least one-half of its energy from biomass. Some operating pulp and paper plants are totally self-sufficient. Even so, further potential exists especially in combination with energy efficiency improvement such as cogeneration. Most of the cement and lime, in Canada at least, is produced using coal or coke, allowing for reductions were they to move to biomass, waste fuels, natural gas, or even oil. In some industries, like mining, the bulk of the energy used is electricity, and direct reduction opportunities are small.

**Carbon Capture and Storage (CC&S)**: In one sense, all industries and landfills could invoke CC&S but the methods to accomplish this are not well understood and/or the costs are very high. For example, one could introduce an oxygen stream into all combustion devices such that the exhaust steam is $CO_2$ rich, suitable for capture and storage. Some industries, like cement production (nonmetal minerals), are reasonable candidates for capture, but transport of the $CO_2$ for storage may prohibit implementation (see particularly Hertzog, 1999; DOE, 2006).

## REFERENCES

Barlaz, Morton A., and Robert K. Ham. 1990 “The Use of Mass Balances for Calculation of the Methane Potential of Fresh and Anaerobically Decomposed Refuse,” in *Proceedings from the GRCDA 13th Annual International Landfill Gas Symposium, March 27–29, 1990* (Silver Spring, MD: GRCDA—The Association of Solid Waste Management Professionals) 1990, p. 235.

Barlaz, M. 1994 *Measurement of the Methane Potential of the Paper, Yard Waste, and Food Waste Components of Municipal Solid Waste*. Unpublished paper, Department of Civil Engineering, North Carolina State University, 1994.

Bogner, J., and K. Spokas. 1993. *Landfill $CH_4$: Rates, Fates, and Role in the Global Carbon Cycle.* Chemosphere, Vol. 26, Nos. 1–4, pp. 369–386.

California Environmental Protection Agency. 2003. *Environmental Technologies and Service Opportunities in the Baja California Peninsula*. International Affairs Unit.

Canadian Industrial Energy End-Use Data and Analysis Centre. 2005. *Development of Energy Intensity Indicators for Canadian Industry: 1990–2004*, Simon Fraser University, Vancouver.

Energy Information Administration. 2005. International Energy Outlook, 2005.

EPIC (Environment and Plastics Industry Council). 2002. Opportunities for Reducing Greenhouse Gas Emissions through Residential Waste Management. Prepared by Environment and Plastics Industry Council.

Environmental Protection Agency. 2003a. *International Analysis of Methane and Nitrous Oxide Abatement Opportunities: Report to Energy Modeling Forum, Working Group 21*.

Environmental Protection Agency. 2003b. *Municipal Solid Waste in the United States: 2003 Facts and Figures*.

Environmental Protection Agency. 2005. Inventory of U.S. Greenhouse Gas Emissions and Sinks, 1990–2003.

Hershkowitz, Allen. 1997. *Too Good to Throw Away: Recycling's Proven Record.* National Resources Defense Council. February 1997.

Herzog, H. 1999. "The Economics of $CO_2$ Capture," P Reimer, B Eliasson, A Wokaum, eds., *Greenhouse Gas Control Technologies*, Oxford: Elsevier Science Ltd., pp. 101–106 (1999).

International Energy Agency. 2004. *30 Years of Energy Use in IEA Countries.*

IPCC (Intergovernmental Panel on Climate Change). 2001. *Third Assessment Report, Volume 3 Mitigation*, Chapter 3.

Jaccard, M., J. Nyboer, and B. Sadownik. 2002. *The Cost of Climate Policy*. Vancouver, UBC Press.

Jaccard, M., J. Nyboer, C. Bataille, and B. Sadownik. 2003. *Modeling the Cost of Climate Policy: Distinguishing Between Alternative Cost Definitions and Long-Run Cost Dynamics.* The Energy Journal 24(1): 49–73.

Jaccard, M., R. Loulou, A. Kanudia, J. Nyboer, A. Bailie, and M. Labriet. 2003. *Methodological contrasts in costing GHG abatement policies: Optimization and simulation modeling of micro-economic effects in Canada.* European Journal of Operations Research 145 (1): 148–164.

Martin, N., E. Worrell, M. Ruth, L. Price, R. N. Elliott, A. M. Shipley, and J. Thorne. 2001. *Emerging Energy-Efficient Industrial Technologies: New York State Edition.* LBNL Report Number 46990, published by American Council for an Energy-Efficient Economy (ACEEE).

Mohareb, A. K., M. Warith, and R. M. Narbaitz. 2003. Strategies for the municipal solid waste sector to assist Canada in meeting its Kyoto Protocol commitments. Environmental Review 12: 71–95.

Statistics Canada. 2004. *Human Activity and the Environment*. Statistics Canada, Cat no.16-201-XIE.

DOE. 2006. www.fossil.energy.gov/programs/sequestration/overview.html Accessed on March 27, 2006. U.S. Department of Energy.

Worrell, Ernst, Lynn K. Price, and Christina Galitsky. 2004. *Emerging energy-efficient technologies in industry: Case studies of selected technologies.* Environmental Technologies Division, Lawrence Berkeley Laboratory, University of California at Berkeley.

**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 recycled | 47.3 | 4.3 | – |
| 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 | – |

*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**

| Sector | Reduction of fugitives | | Energy efficiency | | Process change | | Fuel substitution | | Carbon Capture and Storage | |
|---|---|---|---|---|---|---|---|---|---|---|
| | 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.

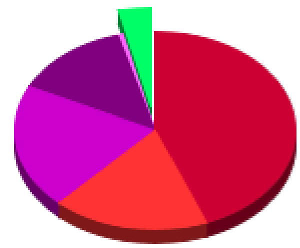
**The “Cost Categories” are as follows:**

**$CO_2$-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

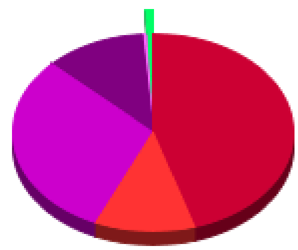
**World**

| Sector | | $MtCO_2$ | | % |
|---|---|---|---|---|
| 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** | | |



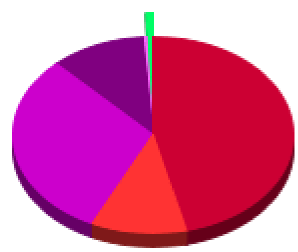
**North America (w/ Mexico)**

| Sector | | $MtCO_2$ | | % |
|---|---|---|---|---|
| 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** | | |



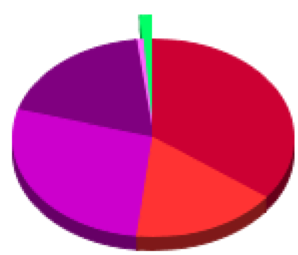
**United States of America**

| Sector | | $MtCO_2$ | | % |
|---|---|---|---|---|
| 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 | | $MtCO_2$ | | % |
|---|---|---|---|---|
| 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** | | |



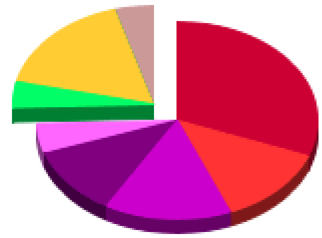
**Mexico**

| Sector | | $MtCO_2$ | | % |
|---|---|---|---|---|
| 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).

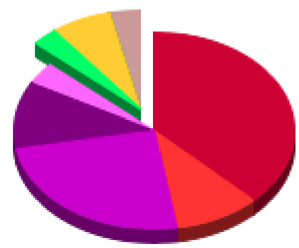
**World**

| Sector | | $MtCO_2$ | | % |
|---|---|---|---|---|
| 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** | | |



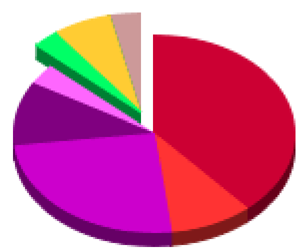
**North America (w/ Mexico)**

| Sector | | $MtCO_2$ | | % |
|---|---|---|---|---|
| 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** | | |



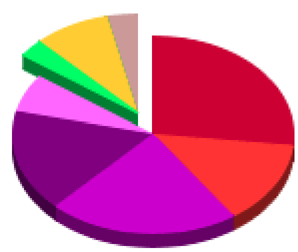
**United States of America**

| Sector | | $MtCO_2$ | | % |
|---|---|---|---|---|
| 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** | | |



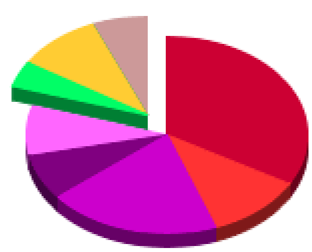
**Canada**

| Sector | | $MtCO_2$ | | % |
|---|---|---|---|---|
| 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 | | $MtCO_2$ | | % |
|---|---|---|---|---|
| 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).

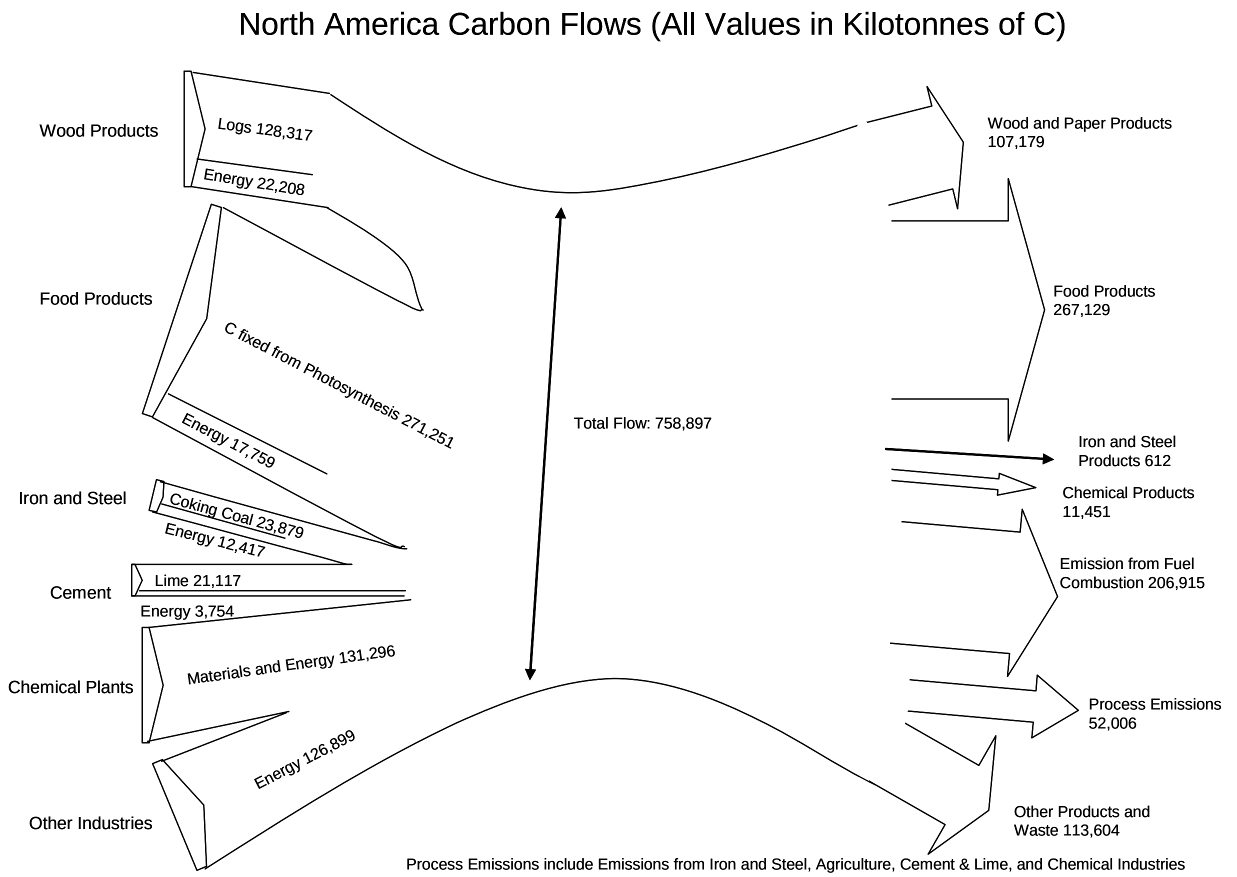



**Fig. 8-2. Carbon flows for Canada, the United States and Mexico combined.** Values in kilotons carbon can be converted to kilotons $CO_2$ equivalents by multiplying by 44/12, the ratio of carbon dioxide mass to carbon mass. Comparable diagrams for the individual countries are in Appendix 8A**.** *Source*: 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.

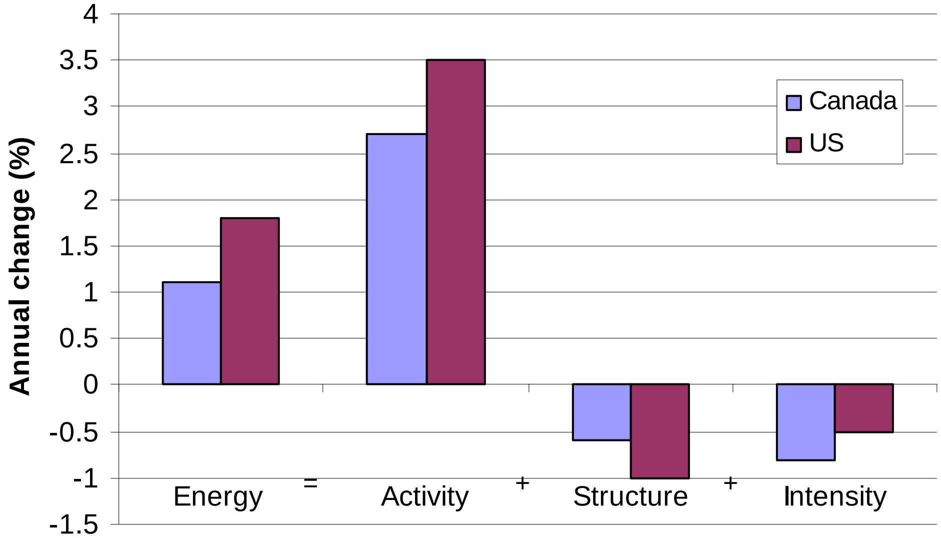
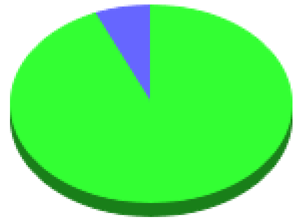



**Fig. 8-3. Decomposition of energy use, manufacturing sector, 1990–1998.** *Source*: IEA, 2004.

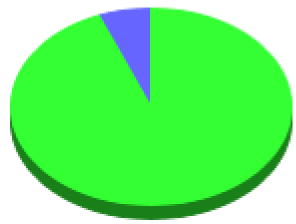
**World**

| Gas | $MtCO_2$ | % |
|---|---|---|
| $CH_4$ | 1,386.4 | 93.5 |
| $N_2O$ | 97.2 | 6.5 |
| **Total** | **1,483.6** | |



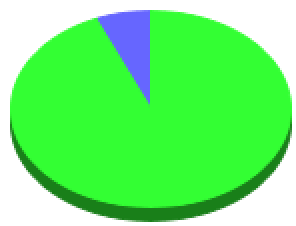
**North America (w/ Mexico)**

| Gas | $MtCO_2$ | % |
|---|---|---|
| $CH_4$ | 281.8 | 93.9 |
| $N_2O$ | 18.2 | 6.1 |
| **Total** | **300.0** | |



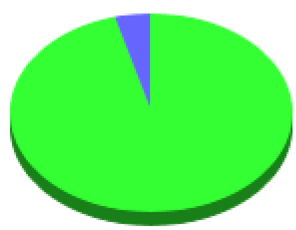
**United States of America**

| Gas | $MtCO_2$ | % |
|---|---|---|
| $CH_4$ | 227.7 | 93.6 |
| $N_2O$ | 15.6 | 6.4 |
| **Total** | **243.3** | |



**Canada**

| Gas | $MtCO_2$ | % |
|---|---|---|
| $CH_4$ | 23.2 | 95.8 |
| $N_2O$ | 1.0 | 4.2 |
| **Total** | **24.2** | |



**Mexico**

| Gas | $MtCO_2$ | % |
|---|---|---|
| $CH_4$ | 31.0 | 95.2 |
| $N_2O$ | 1.6 | 4.8 |
| **Total** | **32.5** | |

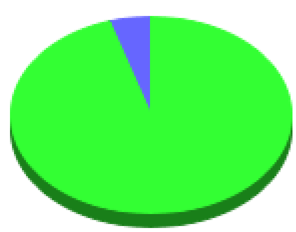


**Fig. 8-4. GHG emissions by gas from waste in 2000.** *Source*: Climate Analysis Indicators Tool (CAIT) Version 3.0 (Washington, D.C.: World Resources Institute, 2005).

# 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.**

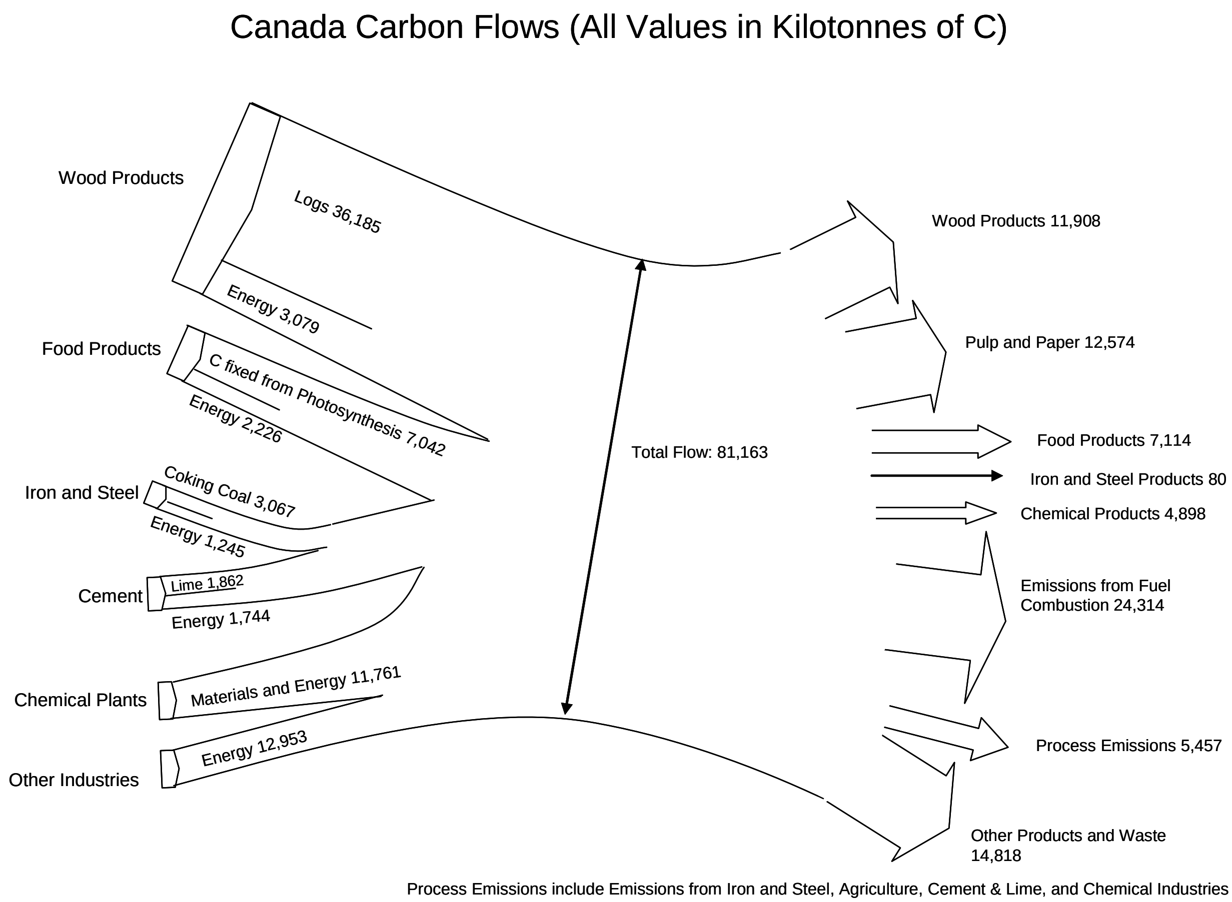



**Fig. 8A-1. Carbon flows, Canada.** *Source*: Energy data from Statistics Canada Industrial Consumption of Energy survey, conversion coefficients and process emissions from Environment Canada, *Canada GHG Inventory, 2002*. Production data from Statistics Canada, CANSIM Table 002-0010, Tables 303-0010,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

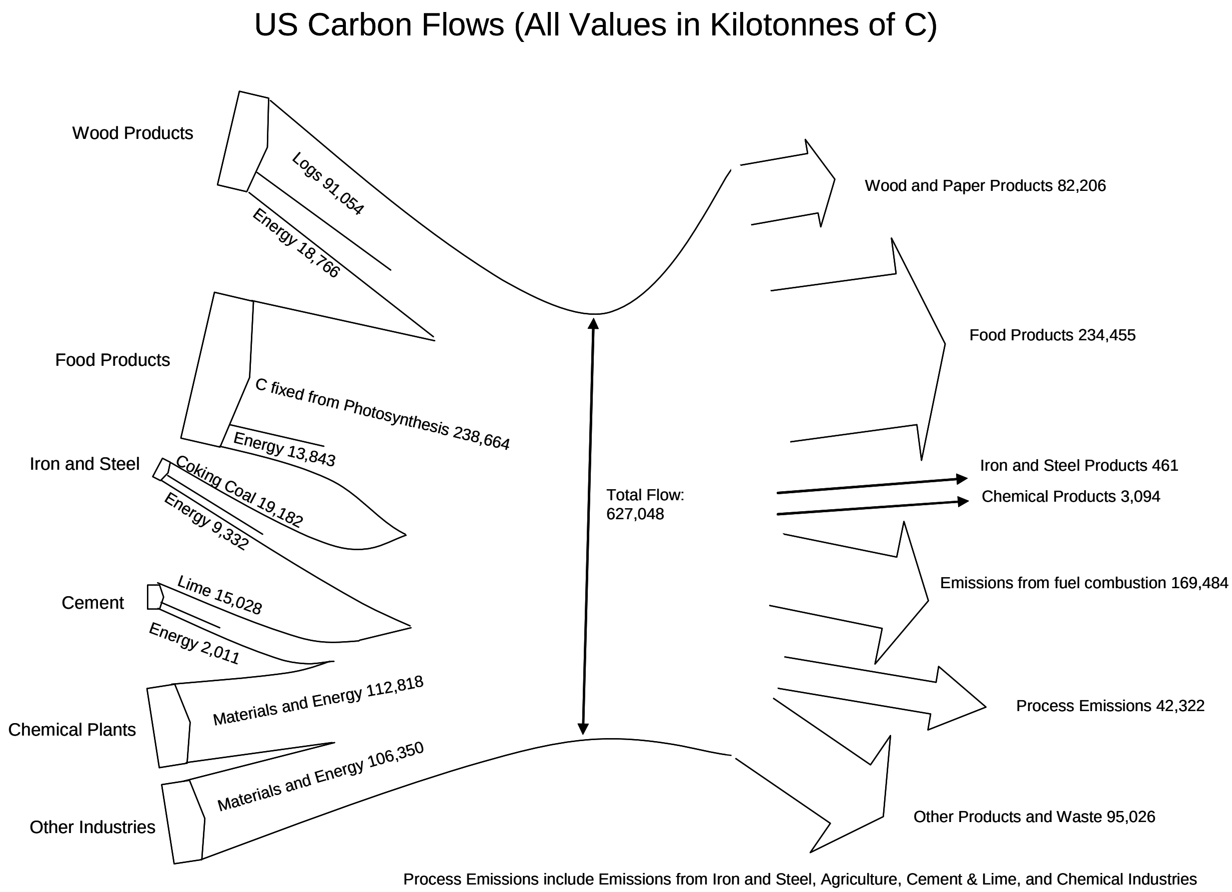



**Fig. 8A-2. Carbon flows, United States.** *Source*: Energy data from IEA Oil Information 2004, IEA Coal Information 2005, IEA Natural Gas Information 2004. Process emissions: EPA, U.S. Emissions Inventory. Production of forestry products: USDA Database; FO-2471000 and -2472010, U.S. Timber Production, 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.

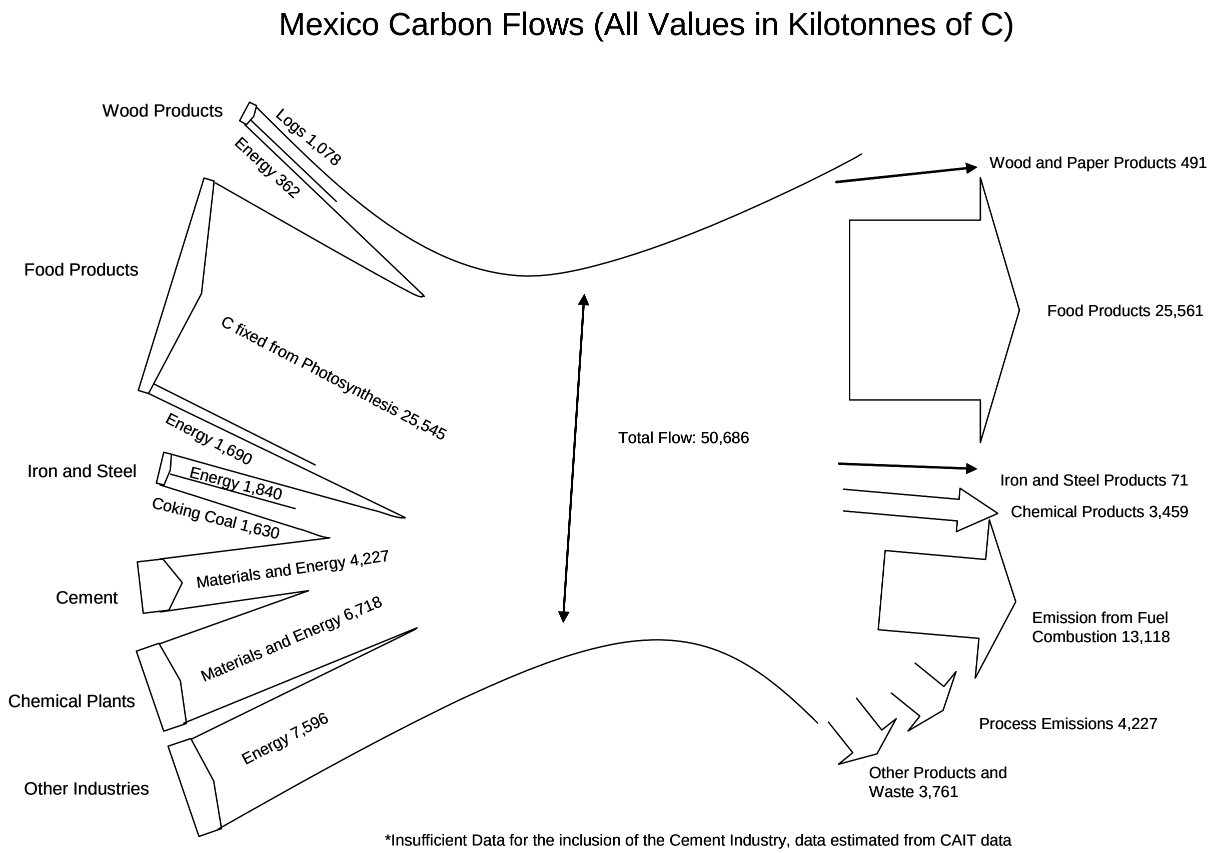



**Fig. 8A-3. Carbon flows, Mexico.** *Source*: Energy data from IEA Oil Information 2004, IEA Coal Information 2005, IEA Natural Gas Information 2004. Process emissions: EPA, U.S. Emissions Inventory. Production of forestry products: USDA Database; FO-2471000, -2472010, -2482000, -2483040, -6342000, -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

**Lead Author: James E. McMahon[1]**

[1]Lawrence Berkeley National Laboratory

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**KEY FINDINGS**

- The buildings sector of North America was responsible for the annual emission of 2,712 Mt $CO_2$ in 2003, which is 36% of total North American $CO_2$ emissions and 9% of global emissions. U.S. buildings alone are responsible for more $CO_2$ emissions than total $CO_2$ emissions of any country in the world, except China.
- Carbon dioxide missions from energy use in buildings in the United States and Canada has increased by 30% since 1990, 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, with the amount of living area per capita increasing in all three countries of North America.
- 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 30% 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.

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Buildings are responsible for 36% of carbon emissions in North America (2712 Mt $CO_2$ in 2003) (Natural Resources Canada, 2005; SENER México, 2005; U.S. DOE-EIA, 2005a[1]) and 9% in the world (U.S. DOE-EERE, 2005[2]). U.S. buildings alone are responsible for more $CO_2$ emissions than total $CO_2$ emissions of any 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 (U.S. DOE-EERE, 2005.[3]). The discussions in this chapter include an accounting of $CO_2$ emissions from electricity consumed in the buildings sector; however, this accounting 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, such as 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 by at least 60% for offices[4] and 30% for homes.[5] Residential and commercial buildings in the United States and Canada contain 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). 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. 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 consumed by each 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.

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[1]U.S. Sector Emissions from Table 18 in U.S. DOE-EIA (2005a).
[2] See Table 3.1.1 in U.S. DOE-EERE (2005).
[3] See Tables 1.1.6 and 1.1.7 in U.S. DOE-EERE (2005).
[4] Leadership in Energy and Environment Design (LEED) Gold Certification.
[5] U.S. DOE Building America Program.

Currently, the influence of secondary fluxes due to use of materials and consumption of water is more uncertain.

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 10.7 metric tons [6] of carbon dioxide per year in the United States, 6.5 metric tons [7] per year in Canada, and 5.0 metric tons [8] in Mexico (U.S. DOE-EIA, 2005a; Natural Resources Canada, 2005; SENER México, 2005). Energy consumption in a 500-$m^2$ commercial, government, or public-use building in the United States produces 7.1 metric tons of $CO_2$ (U.S. DOE-EIA, 2005a).[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 660 g of $CO_2$ to the atmosphere in 2003 (U.S. DOE-EIA, 2005a).[10] The equivalent amount of energy from natural gas or other fuels contributed about 190 g of $CO_2$ (U.S. DOE-EIA, 2005a).[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 (U.S. DOE-EERE, 2005).[12]

Buildings sector carbon dioxide 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, 9%; and electronics (including televisions, computers, and office equipment), 7.7% (U.S. 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 (U.S. 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.

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

[6] U.S. residential sector emissions of 1213 Mt $CO_2$ divided by 114 million households in 2004.
[7] Canada residential sector emissions of 45.2Mt $CO_2$ divided by 12.2 million households in 2003.
[8] Mexico residential sector emissions of 85.2 Mt $CO_2$ divided by 167 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 U.S. 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; available on-line at http://buildingsdatabook.eere.energy.gov (2003 data).

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 in Mexico, sales of this equipment are now growing significantly, 14% per year over the past 10 years.[15] 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, major appliances are common in middle- and upper-income households, and even the poorest electrified households own a television, a refrigerator, and small appliances.

Many end uses—such as water heating and heating, cooling, and ventilation—occur in most commercial sector buildings. Factors such as climate and building construction influence the carbon emissions from 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 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

[15] *Air conditioner sales 1995–2004 from* Asociacion Nacional de Fabricantes de Aparatos Domesticos (ANFAD).

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.

**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-$ft^2$ (2200-$ft^2$) house requires about 20 Mt of wood and creates 2 to 7 Mt of construction waste (U.S. DOE-EERE, 2005).[16] Building lifetimes are many decades and, especially for commercial buildings, may include several cycles of remodeling and renovation. Water consumption in buildings also impacts the carbon cycle because water supply, treatment, and waste disposal require energy. In California, for example, total water use accounts for 19% of statewide electricity (CEC, 2005). In the United States as a whole, water supplied to residential and commercial customers accounts for about 6% of total national freshwater consumption.

## 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. In general, the trend over the last decade has been toward emissions growth, with emissions from energy use in buildings in the United States and Canada increasing 30% since 1990 (U.S. DOE-EERE, 2005; Natural Resources Canada, 2005),[17] 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 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.[18] On the other hand, a shift from large, extended-family households to nuclear-family and single-occupant households means an increase in

[16] Table 2.1.7. Wood content estimated from lumber content. Construction waste from Table 3.4.1.
[17] U.S.DOE-EERE, 2005 Table 3.1.1.
[18] *Source:* UN Department of Economic and Social Affairs.

the number of households per unit population—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.[19] Per capita energy consumption generally grows with economic development because wealthier people live in larger households and have more appliances. 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 fraction of electricity to total household primary energy has increased. This is significant 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 floorspace and an increase in the size of the average home. The size of an average U.S. single-family home has grown from 160 $m^2$ for a house built in 1980 to 216 $m^2$ in 2003. In the same time, commercial floor space per capita has increased from 20 $m^2$ to 22.6 $m^2$ (U.S. DOE-EERE, 2005).[20] Certain end uses once considered luxuries have now become commonplace. Only 56% of U.S. homes in 1978 used mechanical space-cooling equipment (U.S. DOE-EIA, 2005b). 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**

Although the general trend has been toward growth in per capita emissions, emissions per unit of GDP has 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. Efficiency increases 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.

[19] For example, whether biomass, natural gas or electricity is used for space heating and cooking.
[20] See Table 2.1.6 and 2.2.1 in U.S. DOE-EERE (2005). Residential data are from 1981.

***[SIDEBAR 1 TEXT BOX HERE]***

## 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 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 (S. Wiel and J. E. 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 one or more of these options in a policy portfolio that takes advantage of synergies, avoids unduly burdening certain sectors, and is cost-effective. Major participants include federal agencies, state and local governments, energy and water utilities, private research and development firms, equipment manufacturers and importers, energy services companies (ESCOs), 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 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.

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.

***[ SIDEBAR 2 TEXT BOX HERE]***

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 nongovernment, nonprofit organization. In existence for five years, the LEED program has certified 36 million $m^2$ of commercial and public-sector buildings and has recently implemented a certification system for homes. The LEED program includes a graduated rating system. Typical energy savings achieved by LEED Gold-rated buildings are 50–60% (U.S. GBC, 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. 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 non-carbon-emitting equipment and building systems (including energy demand and supply) 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 roadmap will need to be updated at regular intervals to account for changes in technologies, production practices, and market acceptance of competing solutions.

## REFERENCES

CEC (2005). *California's Water Energy Relationship*. Staff Final Report, California Energy Commission. Sacramento, 2005.

Interlaboratory Working Group, 2000. *Scenarios for a Clean Energy Future* (Oak Ridge, TN; Oak Ridge National Laboratory and Berkeley, CA; Lawrence Berkeley National Laboratory), ORNL/CON-476 and LBNL-44029, November.

Kinsey *et al.* (2002). *The Federal Buildings Research and Development Program: A sharp Tool for Climate Policy*. ACEEE Buildings Summer Study 2002, Pacific Grove.

Natural Resources Canada (2005). *Office of Energy Efficiency National Energy Use Database 2005*. Ottawa http://oee.nrcan.gc.ca/corporate/statistics/neud/dpa/data_e/database_e.cfm

SENER México (2005). Secretaria de Energia—Sistema de Información Energética. México D.F. http://sie.energia.gob.mx/sie/bdiController

U.S. DOE-EERE (2005). *2005 Buildings Energy Data Book*. Office of Energy Efficiency and Renewable Energy. Washington, 2005.

U.S. DOE-EIA (2005a). *Annual Energy Outlook 2005*. EIA-0383(2005). Energy Information Administration. Washington, 2005a.

USDOE-EIA (2005b). *Changes in Energy Usage in Residential Housing Units*. Washington http://www.eia.doe.gov/emeu/recs/recs2001/hc_pdf/ac/hc4-11a_so_region2001.pdf.

USGBC (2005). *LEED for New Construction—Rating System 2.2*, LEED (NC) 2.2, U.S. Green Building Council. Washington, 2005.

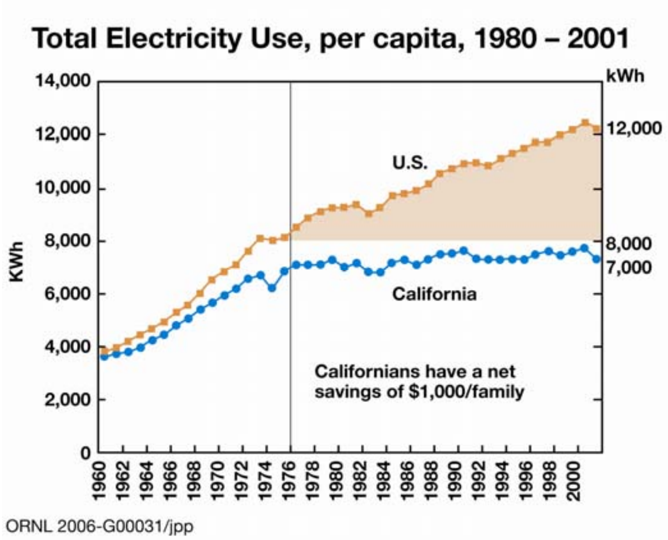
Wiel, S., and J.E. McMahon (2005). *Energy-Efficiency Labels and Standards: A Guidebook for Appliances, Equipment, and Lighting, 2nd Edition*. Washington, Collaborative Labeling and Standards Program.

***[BEGIN SIDEBAR 1]***

Since the mid-1970s, the state of California has pursued an aggressive set of efficiency regulations. As a result, per capita electricity consumption has stabilized in that state even 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

**Per capita electricity consumption in the U.S. and California**




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

***[END SIDEDBAR 1]***

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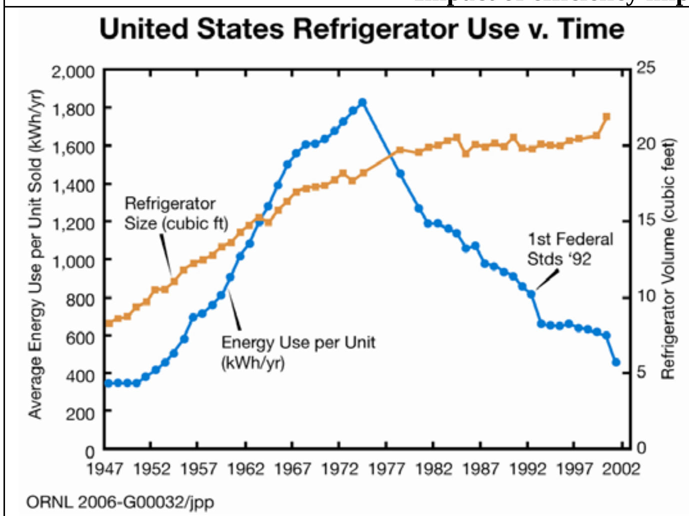
Between 1974 and 2001, the energy consumption of the average refrigerator sold in the United States dropped by 74%, a change driven by market forces and regulations. From 1987 to 2005, the U.S. Congress and DOE promulgated minimum efficiency standards for more than 30 residential and commercial product types. Canada and Mexico also have many product efficiency standards, and a program is under way to harmonize standards throughout North America in connection with North American Free Trade Agreement.

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

**Impact of efficiency improvements**

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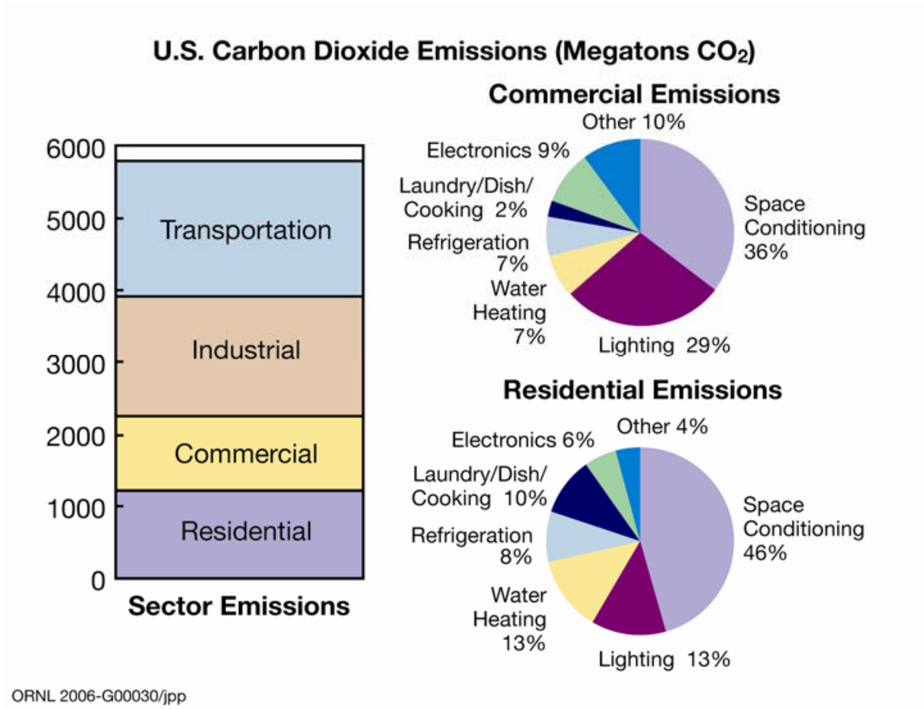


**Fig. 9-1. U.S. carbon emissions by sector and—for commercial and residential buildings—by end use.**

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

| | Greenhouse emission (megatons) $CO_2$ | | | | |
|---|---|---|---|---|---|
| | Electricity | Natural gas | Other fuels | Wood | Total |
| **United States** | **1609.1** | **434.6** | **166.3** | **0.0** | **2210.0** |
| Residential | 823.7 | 265.9 | 105.1 | 0.0 | 1194.7 |
| Commercial | 785.4 | 168.7 | 61.2 | 0.0 | 1015.4 |
| **Canada** | **61.6** | **58.5** | **22.0** | **1.9** | **143.9** |
| Residential | 31.4 | 32.3 | 9.1 | 1.9 | 74.7 |
| Commercial | 30.2 | 26.2 | 12.9 | 0.0 | 69.3 |
| **Mexico** | **52.7** | **1.7** | **16.2** | **2.1** | **72.7** |
| Residential | 40.2 | 1.3 | 15.8 | 2.1 | 59.4 |
| Commercial | 12.5 | 0.4 | 0.4 | 0.0 | 13.3 |
| **Total** | **1723.4** | **494.8** | **204.5** | **4.0** | **2426.6** |

**Table 9-2. Principal drivers of buildings emissions trends**

| Drivers | 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/HH) | 2.5 | –0.6% | 2.6 | –0.9% | 5.3 | –0.1% |
| GDP/Cap (thousand $U.S. 1995) | 31.7 | 2.0% | 23.0 | 1.8% | 3.8 | 1.8% |
| Res. floorspace (billion $m^2$) | 15.7 | 0.0% | 1.5 | 2.4% | NA | NA |
| Comm. floorspace (million $m^2$) | 6.4 | 0.6% | 0.5 | 1.6% | NA | NA |
| Building energy emissions/GDP (g $CO_2$/$U.S.) | 256 | –0.5% | 217 | –0.9% | NA | NA |

# PART III OVERVIEW

## The Carbon Cycle in Land and Water Systems

**Lead Author: R.A. Houghton[1]**

[1]Woods Hole Research Center

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) (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 (~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 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.

## REFERENCES

Bradley, B. A., R. A. Houghton, J. F. Mustard, and S. P. Hamburg. Invasive grass reduces carbon stocks in shrublands of the Western United States *Global Change Biology*, in press.

Caspersen, J. P., S. W. Pacala, J. C. Jenkins, G. C. Hurtt, P. R. Moorcroft, and R. A. Birdsey. 2000. Contributions of land-use history to carbon accumulation in United States forests. *Science* **290**:1148–1151.

Cox, P. M., R. A. Betts, C. D. Jones, S. A. Spall, and I. J. Totterdell. 2000. Acceleration of global warming due to carbon-cycle feedbacks in a coupled climate model. *Nature* **408**:184–187.

Davidson, E. A., and I. A. Janssens. 2006. Temperature sensitivity of soil carbon decomposition and feedbacks to climate change. *Nature* **440**:165–173.

Houghton, R. A., J. L. Hackler, and K. T. Lawrence. 1999. The United States carbon budget: contributions from land-use change. *Science 285*:574–578.

Hurtt, G. C., S. W. Pacala, P. R. Moorcroft, J. Caspersen, E. Shevliakova, R. A. Houghton, and B. Moore III. 2002. Projecting the future of the United States carbon sink. *Proceedings of the National Academy of Sciences* **99**:1389–1394.

Jones, C., C. McConnell, K. Coleman, P. Cox, P. Falloon, D. Jenkinson, and D. Powlson. 2005. Global climate change and soil carbon stocks; predictions from two contrasting models for the turnover of organic carbon in soil. *Global Change Biology* **11**:154–166.

Lal, R. 2001. Fate of eroded soil carbon: emission or sequestration. *In*: Lal, R. (Ed.), *Soil Carbon Sequestration and the Greenhouse Effect*. Soil Science Society of America Special Publication, vol. 57. Madison, Wisconsin, pp. 173-181.

Osterkamp, T. E., and V. E. Romanovsky. 1999: Evidence for warming and thawing of discontinuous permafrost in Alaska. *Permafrost and Periglacial Processes*, 10(1):17–37.

Osterkamp, T. E., L. Viereck, Y. Shur, M. T. Jorgenson, C. Racine, A. Doyle, and R. D. Boone. 2000: Observations of thermokarst and its impact on boreal forests in Alaska, United States. *Arctic, Antarctic and Alpine Research*, **32**:303–315.

Prentice I. C., G. D. Farquhar, M. J. R. Fasham, M. L. Goulden, M. Heimann, V. J. Jaramillo, H. S. Kheshgi, C. Le Quéré, R. J. Scholes, and D. W. R. Wallace. 2001. The carbon cycle and atmospheric carbon dioxide. In: *Climate Change 2001: The Scientific Basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change* [J. T. Houghton, Y. Ding, D. J. Griggs, M. Noguer, P. J. van der Linden, X. Dai, K. Maskell, and C. A. Johnson (eds). Cambridge University Press, Cambridge, United Kingdom and New York, NY, United States, pp. 183–237.

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

Smith, L.C., Y. Sheng, G.M. MacDonald, L.D. Hinzman, 2005: Disappearing Arctic Lakes, *Science*, 308, 1429.

Smith, S. L., M. M. Burgess, and F. M. Nixon. 2001: Response of activelayer and permafrost temperatures to warming during 1998 in the Mackenzie Delta, Northwest Territories and at Canadian Forces Station Alert and Baker Lake, Nunavut. *Geological Survey of Canada Current Research* 2001–E5, 8pp.

Smith, S.V., R.O. Sleezer, W.H. Renwick, and R.W. Buddemeier. 2005. Fates of eroded soil organic carbon: Mississippi Basin case study. *Ecological Applications* **15**:1929-1940.

Stallard, R.F. 1998. Terrestrial sedimentation and the carbon cycle: Coupling weathering and erosion to carbon burial. *Global Biogeochemical Cycles* **12**:231-257.

Wienert, A., and S.P. Hamburg. 2006. Carbon stock changes and greenhouse gas emissions from exurban land development in central New Hampshire. Master's Thesis, Brown University, Providence, Rhode Island.

**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 Lands, Grasslands, Shrublands, and Arid Lands

**Lead Authors: Richard T. Conant[1] and Keith Paustian[1,2]**

**Contributing Authors: Felipe García-Oliva,[3] H. Henry Janzen,[4] Victor J. Jaramillo,[3] Donald E. Johnson,[5] and Suren N. Kulshreshtha[6]**

[1]Natural Resource Ecology Laboratory, Colorado State University, [2]Department of Soil and Crop Science, Colorado State University, [3]Centro de Investigaciones en Ecosistemas, Universidad Nacional Autónoma de México, [4]Environmental Health, Agriculture and Agri-Food Canada, [5]Department of Animal Science, Colorado State University, [6]Department of Agricultural Economics, University of Saskatchewan

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## KEY FINDINGS

- Agricultural lands, including croplands and grazinglands (grasslands, shrublands, and arid lands), occupy 789 million ha (47% of the land area of North America) and contain 78.5±19.5 Gt C (17.4% of North American terrestrial carbon) in the soil alone.
- Agricultural lands in the United States and Canada are currently near neutral with respect to their soil carbon balance (but less so for Mexico because of ongoing land use change). Although agricultural soils are estimated to be sequestering currently 6-15.5 Mt C $yr^{-1}$, the cultivation of organic soils releases 5.3-10.3 Mt C $yr^{-1}$. The emissions of carbon from fossil fuel inputs to agriculture (46.3 Mt C $yr^{-1}$) and the manufacture of 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.
- 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 sequestration.
- 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 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 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.

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# INVENTORY

## Background

Agricultural lands, including croplands and grazing lands—grasslands, 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 17.4% of the terrestrial carbon (Pacala *et al.*, 2006). These lands differ from other types of ecosystems in that most of their carbon is held in soils. Live vegetation in cultivated systems generally contains less than 5% of the total carbon, whereas vegetation in grazing lands contains a greater proportion (5–30%), but still less than that in forested systems (30–65%). These systems in North America contain 78.5±19.5 (±1SE) Gt C in the soil (Table 10-1). Significant increases in vegetation

[1]We refer collectively to these lands as grazing lands since grazing is their primary use, even though not all of these lands are grazed.

carbon stocks in some grazing lands have been observed and, together with soil carbon stocks from croplands and grazing lands, likely contribute significantly to the large North American terrestrial carbon sink (Houghton *et al.*, 1999; Pacala *et al.*, 2001; Eve *et al.*, 2002; Ogle *et al.*, 2003). These lands also emit greenhouse gases: fossil fuel use for on-farm machinery and buildings, for manufacture of agricultural inputs, and for transportation account for 3–5% of total $CO_2$ 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, 2005); and agricultural and grazing lands are the largest anthropogenic source of $N_2O$ emissions (CAST, 2004; see Text Box 1). However, agricultural lands are actively managed and have the capacity to take up more carbon into soil; 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 pools for undisturbed native systems were derived using the intersection of MODIS-IGBP[a] land cover types (Friedl *et al.*, 2002) and mean soil carbon contents to 1m depth from Sombroek *et al.* (1993) spatially arrayed using FAO soil classes (ISRIC, 2002).

## Carbon Dioxide Fluxes from Agricultural and Grazing Land

The basic 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 $CH_4$). In agricultural lands, carbon assimilation is directed towards production of food, fiber, and forage by manipulating species composition and growing conditions. 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 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). Increased 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 land 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 (Figure 10-1; Table 10-2), while still maintaining outputs of food, fiber, and forage. Within 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 improved practices on annual cropland. 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 (Ogle *et al.*, 2003; Matin *et al.*, 2004). 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) fluxes for the years around 2000.** Negative values indicate net flux from the atmosphere to soil and biomass carbon pools. All data are from Canadian (Matin *et al.*, 2004) and U.S. (EPA, 2005) 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 the United States and Canada and 1998 for Mexico. 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.

**Table 10-2. North American agricultural and grazing land carbon fluxes for the years around 2000.** 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, 2005) 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 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.

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 plus $CO_2$ emissions embodied in applied fertilizers and pesticides contribute emissions of 28 Mt C $yr^{-1}$ within the United States (Lal *et al.*, 1998) and 2.8 Mt C $yr^{-1}$ in Canada (Sobool and Kulshreshtha, 2005). 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 $CH_4$ (Figure 10-1; Table 10-2). Methane formation is an anaerobic process and is most significant in waterlogged soils, like those under paddy rice cultivation (U.S.: 0.328 Mt $CH_4$ $yr^{-1}$; Mexico: 0.015 Mt $CH_4$ $yr^{-1}$; Canada: negligible, not reported). Methane is also formed by incomplete biomass combustion of crop residues (U.S.: 0.038 Mt $CH_4$ $yr^{-1}$; Mexico: 0.011 Mt $CH_4$ $yr^{-1}$; Canada: negligible, not reported). 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). 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 that provide energy to microbes and their hosts, 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 (54 Mt $CH_4$ $yr^{-1}$, 21% of total U.S. $CH_4$ emissions), Canada (0.8 Mt $CH_4$ $yr^{-1}$, 22% of total) (Sobool and Kulshreshtha, 2005), and Mexico (2.0 Mt $CH_4$ $yr^{-1}$, 27% of total) with the vast majority of enteric $CH_4$ emissions are from beef (72%) and dairy cattle (23%). Emissions from ruminants are tightly coupled to feed consumption, since $CH_4$ emission per unit of feed energy is consistent, 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 changing environmental conditions, i.e., global warming (NAS, 2001) may make it more difficult to reduce emissions from cropland and grazing lands (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 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 reductions 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).

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 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) (Figure 10-2). In annual cropland, improved crop rotations, yield enhancement measures, organic amendments, cover crops, improved fertilization and irrigation practices, and reduced bare fallow tend to increase productivity and carbon inputs, and thus soil carbon stocks (Lal *et al.*, 1998; Paustian *et al.*, 1998; VandenBygaart *et al.*, 2003) (Figure 10-2). Tillage, traditionally used for soil preparation and weed control, disturbs the soil and stimulates decomposition and loss of soil carbon. Practices that substantially reduce (reduced-till) or eliminate (no-till) tillage-induced disturbances are

[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.

being increasingly adopted and generally increase soil carbon stocks while maintaining or enhancing productivity levels (Paustian *et al*,. 1997; Ogle *et al.*, 2003) (Figure 10-2). Estimates of the technical potential for annual cropland soil carbon sequestration are on the order of 50–100 Mt C $yr^{-1}$ 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.** 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. Temperature/precipitation divisions are the same as those described in Table 10-1. Data are from Nabuurs *et al.* (2004) and Ogle *et al.* (2004).

Within grazing lands, historical overgrazing has substantially reduced productive capacity in many areas, leading to loss of soil carbon stocks (Conant and Paustian, 2002) (Figure 10-2). Conversely, improved grazing management and production inputs—like fertilizer, 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).

## 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, 2004). 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 will limit 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 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/ton sequestered carbon compared with practices where mitigation would be a primary income source, such as land set-asides, even if the latter have a higher biological sequestration potential.
- At higher 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 sequestered) are more economically efficient than per-hectare payments (for adopting specific practices), 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 2005b), 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 equilibrium level after 15–30 years, after which there is no further accumulation of carbon (West and Wali, 2002). 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) can 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 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 information 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.

## REFERENCES

Ågren, G. I., and E. Bosatta, 2002: Reconciling differences in predictions of temperature response of soil organic matter. Soil Biology and Biochemistry, **34**, 129–132.

Agriculture and Agri-Food Canada, 1999: The health of our air. Toward sustainable agriculture in Canada. In: Publication 1981/E. (H. H. Janzen, R. L. Desjardins, J. M. R. Asselin, and B. Grace, eds. Pages 40 Ottowa, Ontario: Agriculture and Agri-Foods Canada.

Antle, J. M., S. Capalbo, S. Mooney, E. T. Elliott, and K. Paustian, 2001: Economic analysis of agricultural soil carbon sequestration: An integrated assessment approach. Journal of Agricultural and Resource Economics, **26**(2), 344–367.

Antle, J. M., S. M. Capalbo, S. Mooney, D. K. Elliott, and K. H. Paustian, 2003: Spatial Heterogeneity, Contract Design, and the Efficiency of Carbon Sequestration Policies for Agriculture. Journal of Environmental Economics and Management, **46**(2), 231–250.

Bellamy, P. H., P. J. Loveland, R. I. Bradley, R. M. Lark, and G. J. D. Kirk, 2005: Carbon losses from all soils across England and Wales 1978–2003. Nature, **437**, 245–248.

Boadi, D., C. Benchaar, J. Chiquette, and D. Masse, 2004: Mitigation strategies to reduce enteric methane emissions from dairy cows: Update review. Canadian Journal of Animal Science, **84**(3), 319–335.

Boehm, M., B. Junkins, R. Desjardins, S. N. Kulshreshtha, and W. Lindwall, 2004: Sink potential of Canadian agricultural soils. Climatic Change, **65**, 297–314.

CAST, 2004: Climate change and greenhouse gas mitigation: Challenges and opportunities for agriculture. Paustian, K., B. A. Babcock, J. Hatvield, C. L. Kling, R. Lal, B. A. McCarl, S. McLaughlin, A. R. Mosier, W. M. Post, C. W. Rice, G. P. Robertson, N. J. Rosenberg, C. Rosenzweig, Cynthia, W. H. Schlesinger, and D. Zilberman (Task Force Members). Ames, IA: Council for Agricultural Science and Technology (CAST).

CISCC, 2001: Second National Communication of Mexico to the UN Framework Convention on Climate Change. http://unfccc.int/resource/docs/natc/mexnc2.pdf: Comité Intersecretarial Sobre Cambio Climático.

Conant, R. T., S. J. Del Grosso, W. J. Parton, and K. Paustian, 2005: Nitrogen pools and fluxes in grassland soils sequestering carbon. Nutrient Cycling in Agroecosystems, **71**(3), 239–248.

Conant, R. T., and K. Paustian, 2002: Potential soil carbon sequestration in overgrazed grassland ecosystems. Global Biogeochemical Cycles, **16**, 1143.

Cox, P. M., R. A. Betts, C. D. Jones, S. A. Spall, and I. J. Totterdell, 2000: Acceleration of global warming due to carbon-cycle feedbacks in a coupled climate model. Nature, **408**, 184–187.

Davidson, E. A., and I. L. Ackerman, 1993: Change in soil carbon inventories following cultivation of previously untilled soils. Biogeochemistry, **20**, 161–193.

Enquete Commission, 1995: Protecting our Green Earth. How to Manage Global Warming Through Environmentally Sound Farming and Preservation of the World's Forests. Bonn: Economica Verlag.

EPA, 2000: Options for reducing methane intermissions internationally. 430-R-90-006, Washington, DC: Environmental Protection Agency.

EPA, 2004: Inventory of U.S. greenhouse gas emissions and sinks: 1990–2002. Washington, DC: U.S. Environmental Protection Agency.

EPA, 2005: Inventory of U.S. greenhouse gas emissions and sinks: 1990–2003. Washington, DC: U.S. Environmental Protection Agency.

EPA, 2006: Inventory of U.S. greenhouse gas emissions and sinks: 1990–2004. Washington, DC: U.S. Environmental Protection Agency.

Eve, M. D., M. Sperow, K. Paustian, and R. F. Follett, 2002: National-scale estimation of changes in soil carbon stocks on agricultural lands. Environmental Pollution, **116**, 431–438.

Follett, R. F., J. M. Kimble, and R. Lal, 2001a: The potential of U.S. grazing lands to sequester carbon and mitigate the greenhouse effect. Chelsea, MI: CRC Press.

Follett, R. F., E. G. Pruessner, S. Samson-Liebig, J. M. Kimble, and S. Waltman, 2001b: Carbon sequestration under the Conservation Reserve Program in the historical grassland soils of the United States of America. In: Soil Management for Enhancing Carbon Sequestration (R. Lal and K. McSweeney, eds. Pages 1–14 Madison, WI: Soil Science Society of America.

Friedl, M. A., A. H. Strahler, X. Zhang, and J. Hodges, 2002: The MODIS land cover product: multi-attribute mapping of global vegetation and land cover properties from time series MODIS data. Proceedings of the International Geoscience and Remote Sensing Symposium, **4**, 3199–3201.

Frye, W. W., 1984: Energy requirements in no-tillage. In: No tillage agricultural principles and practices (R.E. Phillips and S.H. Phillips, eds. Pages 127–151 New York: Van Nostrand Reinhold.

Giardina, C. P., and M. G. Ryan, 2000: Evidence that decomposition rates of organic carbon in mineral soil do not vary with temperature. Nature, **404**, 858–861.

Gregorich, E. G., P. Rochette, A. J. VandenBygaart, and D. A. Angers, 2005: Greenhouse gas contributions of agricultural soils and potential mitigation practices in Eastern Canada. Soil & Tillage Research, **83**(1), 53–72.

Helgason, B. L., H. H. Janzen, M. H. Chantigny, C. F. Drury, B. H. Ellert, E. G. Gregorich, R. L. Lemke, E. Pattey, P. Rochette, and C. Wagner-Riddle, 2005: Toward improved coefficients for predicting direct $N_2O$ emissions from soil in Canadian agroecosystems. Nutrient Cycling in Agroecosystems, **72(1)**, 87–99.

Houghton, R. A., and C. L. Goodale, 2004: Effects of land-use change on the carbon balance of terrestrial ecosystems. Ecosystem and land use change. Geophysical Monograph Series, **53**, 85–96.

Houghton, R. A., J. L. Hackler, and K. T. Lawrence, 1999: The U.S. carbon budget: Contributions from land-use change. Science, **285**, 574–578.

IPCC, 2001: Third Assessment Report. Cambridge, UK: Cambridge University Press.

ISRIC, 2002: FAO Soil Database. CD ROM Rome: International Soil and Reference Information System.

Jackson, R. B., J. L. Banner, E. G. Jobbagy, W. T. Pockman, and D. H. Wall, 2002: Ecosystem carbon loss with woody plant invasion of grasslands. Nature, **418**(6898), 623–626.

Jenkinson, D. S., D. E. Adams, and A. Wild, 1991: Model estimates of $CO_2$ emissions from soil in response to global warming. Nature, **351**, 304–306.

Johnson, K. A., and D. E. Johnson, 1995: Methane emissions from cattle. Journal of Animal Science, **73**, 2483–2492.

Kätterer, T., M. Reichstein, O. Andren, and A. Lomander, 1998: Temperature dependence of organic matter decomposition: a critical review using literature data analyzed with different models. Biology and Fertility of Soils, **27**(3), 258–262.

Keppler, F., J. T. G. Hamilton, M. Brass, and T. Rockmann, 2006: Methane emissions from terrestrial plants under aerobic conditions. Nature, **439**, 187–191.

Knorr, W., I. C. Prentice, J. I. House, and E. A. Holland, 2005: Long-term sensitivity of soil carbon turnover to warming. Nature, **433**, 298–301.

Kulshreshtha, S. N., B. Junkins, and R. Desjardins, 2000: Prioritizing greenhouse gas emission mitigation measures for agriculture. Agricultural Systems, **66**(3), 145–166.

Lal, R., 2002: Why carbon sequestration in agricultural soils? In: Agricultural practices and policies for carbon sequestration in soil (J. Kimble, R. Lal, and R. F. Follett, eds. Pages 21–30 Boca Raton: CRC Press.

Lal, R., 2004: Carbon emission from farm operations. Environment International, **30**(7), 981–990.

Lal, R., R. F. Follett, and J. M. Kimble, 2003: Achieving soil carbon sequestration in the United States: A challenge to policy makers. Soil Science, **168**, 827–845.

Lal, R., J. M. Kimble, R. F. Follett, and C. V. Cole, 1998: The potential of U.S. cropland to sequester carbon and mitigate the greenhouse effect. Chelsea, MI: Ann Arbor Press.

Lewandrowski, J., M. Peters, C. Jones, R. House, M. Sperow, M. D. Eve, and K. Paustian, 2004: Economics of Sequestering Carbon in the U.S. Agricultural Sector. Technical Bulletin No. TB 1909, Washington, DC: Economic Research Service.

Lynch, D. H., R. D. H. Cohen, A. Fredeen, G. Patterson, and R. C. Martin, 2005: Management of Canadian prairie region grazed grasslands: Soil C sequestration, livestock productivity and profitability. Canadian Journal of Soil Science, **85**(2), 183–192.

Matin, A., P. Collas, D. Blain, C. Ha, C. Liang, L. MacDonald, S. McKibbon, C. Palmer, and R. Kerry, 2004: Canada's Greenhouse Gas Inventory: 1990–2002. Greenhouse Gas Division, Environment Canada.

McCarl, B. A., and E. K. Schneider, 2001: The Cost of Greenhouse Gas Mitigation in U.S. Agriculture and Forestry. Science, **294**, 2481–2482.

Mosier, A., C. Kroeze, C. Nevison, O. Oenema, S. Seitzinger, and O. van Cleemput, 1998a: Closing the global N2O budget: nitrous oxide emissions through the agricultural nitrogen cycle - OECD/IPCC/IEA phase II development of IPCC guidelines for national greenhouse gas inventory methodology. Nutrient Cycling in Agroecosystems, **52**(2-3), 225–248.

Mosier, A. R., J. M. Duxbury, J. R. Freney, O. Heinemeyer, K. Minami, and D. E. Johnson, 1998b: Mitigating agricultural emissions of methane. Climatic Change, **40**, 39–80.

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

Nabuurs, G.-J., N. H. Ravindranath, K. Paustian, A. Freibauer, B. Hohenstein, W. Makundi, H. Aalde, A. Y. Abdelgadir, S. A. K. Anwar, J. Barton, K. Bickel, S. Bin-Musa, D. Blain, R. Boer, K. Byrne, C. C. Cerri, L. Ciccarese, D.-C. Choque, E. Duchemin, L. Dja, J. Ford-Robertson, W. Galinski, J. C. Germon, H. Ginzo, M. Gytarsky, L. Heath, D. Loustau, T. Mandouri, J. Mindas, K. Pingoud, J. Raison, V. Savchenko, D. Schone, R. Sievanen, K. Skog, K. A. Smith, D. Xu, M. Bakker, M. Bernoux, J. Bhatti, R. T. Conant, M. E. Harmon, Y. Hirakawa, T. Iehara, M. Ishizuka, E. G. Jobbagy, J. Laine, M. van der Merwe, I. K. Murthy, D. Nowak, S. M. Ogle, P. Sudha, R.J. Scholes, and X. Zhang, 2004: LUCF-sector good practice guidance. In: IPCC Good Practice Guidance for LULUCF (J. Penman, M. Gytarsky, T. Hirishi, T. Krug, and D. Kruger, eds. Hayama, Japan: Institute for Global Environmental Strategies.

NAS, 2001: Climate change science: An analysis of some key questions. Washington, DC: National Academy of Sciences, Committee on the science of climate change, National Research Council.

NRCS, 2005: Anaerobic Digestion Practice Standards. Washington, DC: U.S. Department of Agriculture.

Ogle, S. M., F. J. Breidt, M. D. Eve, and K. Paustian, 2003: Uncertainty in estimating land use and management impacts on soil organic carbon storage for U.S. agricultural lands between 1982 and 1997. Global Change Biology, **9**, 1521–1542.

Ogle, S. M., R. T. Conant, and K. Paustian, 2004: Deriving grassland management factors for a carbon accounting method developed by the intergovernmental panel on climate change. Environmental Management, **33**(4), 474–484.

Pacala, S. W., R. Bridsey, S. Bridgham, J. Caspersen, R. T. Conant, K. Davis, B. Hales, R. Houghton, J. C. Jenkins, M. Johnston, G. Marland, K. Paustian, R. Socolow, R. Tol, and S. C. Wofsy. 2006. The North American carbon budget past and present. *In Prep.*

Pacala, S. W., G. C. Hurtt, D. Baker, P. Peylin, R. A. Houghton, R. A. Birdsey, L. Heath, E. T. Sundquist, R. F. Stallard, P. Ciais, P. Moorcroft, J. P. Casersen, E. Shevliakova, B. Moore, G. Kohlmaier, E. Holland, M. Gloor, M. E. Harmon, S. M. Fan, J. L. Sarmiento, C. L. Goodale, D. Schimel, and C. B. Field, 2001: Consistent land- and atmosphere-based U.S. carbon sink estimates. Science, **292**, 2316–2320.

Paustian, K., O. Andren, H. H. Janzen, R. Lal, P. Smith, G. Tian, H. Tiessen, M. Van Noordwijk, and P. L. Woomer, 1997: Agricultural soils as a sink to mitigate $CO_2$ emissions. Soil Use and Management, **13**, 230–244.

Paustian, K., J. M. Antle, J. Sheehan, and E. A. Paul, 2006: Agriculture's Role in Greenhouse Gas Mitigation. Washington, DC: Pew Center on Global Climate Change.

Paustian, K., C. V. Cole, D. Sauerbeck, and N. Sampson, 1998: $CO_2$ mitigation by agriculture: An overview. Climatic Change, **40**(1), 135–162.

Peoples, M. B., E. W. Boyer, K. W. T. Goulding, P. Heffer, V. A. Ochwoh, B. Vanlauwe, S. Wood, K. Yagi, and O. van Cleemput, 2004: Pathways of nitrogen loss and their impacts on human health and the environment. In: Agriculture and the Nitrogen Cycle (A. R. Mosier, J. K. Syers, and J. R. Freney, eds. Pages 53–69 Washington: Island Press.

Pimentel, D., P. Hepperly, J. Hanson, D. Douds, and R. Seidel, 2005: Environmental, energetic, and economic comparisons of organic and conventional farming systems. Bioscience, **55**(7), 573–582.

Post, W. M., and K. C. Kwon, 2000: Soil carbon sequestration and land-use change: Processes and potential. Global Change Biology, **6**, 317–327.

Raymond, P. A., and J. J. Cole, 2003: Increase in the export of alkalinity from North America's largest river. Science, **301**, 88–91.

Robertson, G. P., and P. R. Grace, 2004: Greenhouse gas fluxes in tropical and temperate agriculture: The need for a full-cost accounting of global warming potentials. Environment, Development and Sustainability, **6**, 51–63.

Robertson, G. P., E. A. Paul, and R. R. Harwood, 2000: Greenhouse gases in intensive agriculture: Contributions of individual gases to the radiative forcing of the atmosphere. Science, **289**, 1922–1925.

Sheinbaum, C., and O. Masera, 2000: Mitigating Carbon Emissions while Advancing National Development Priorities: The Case of Mexico. Climatic Change, **47**(3), 259–282.

Six, J., S. M. Ogle, F. J. Briedt, R. T. Conant, A. R. Mosier, and K. Paustian, 2004: The potential to mitigate global warming with no-tillage management is only realized when practiced in the long term. Global Change Biology, **10**(2), 155–160.

Smith, K. A., and F. Conen, 2004: Impacts of land management on fluxes of trace greenhouse gases. Soil Use and Management, **20**, 255–263.

Smith, P., K. W. Goulding, K. A. Smith, D. S. Powlson, J. U. Smith, P. Falloon, and K. Coleman, 2001a: Enhancing the carbon sink in European agricultural soils: including trace gas flux estimates of carbon mitigation potential. Nutrient Cycling in Agroecosystems, **60**, 237–252.

Smith, S. V., R. O. Sleezer, W. H. Renwick, and R. W. Buddemeier, 2005: Fates of eroded soil organic carbon: Mississippi basin case study. Ecological Applications, **15**(6), 1929–1940.

Smith, W. N., R. L. Desjardins, and B. Grant, 2001b: Estimated changes in soil carbon associated with agricultural practices in Canada. Canadian Journal of Soil Science, **81**(2), 221–227.

Smith, W. N., P. Rochette, C. Monreal, R. L. Desjardins, E. Pattey, and A. Jaques, 1997: The rate of carbon change in agricultural soils in Canada at the landscape level. Canadian Journal of Soil Science, **77**(2), 219–229.

Sobool, D., and S. Kulshreshtha, 2005: Greenhouse Gas Emissions from Agriculture and Agri-Food Systems in Canada. Saskatoon: Department of Agricultural Economics, University of Saskatchewan.

Sombroek, W. G., F. O. Nachtergaele, and A. Hebel, 1993: Amounts, dynamics and sequestering of carbon in tropical and subtropical soils. Ambio, **22**(7), 417–426.

Sperow, M., M. D. Eve, and K. Paustian, 2003: Potential soil C sequestration on U.S. agricultural soils. Climatic Change, **57**, 319–339.

Van Auken, O. W., 2000: Shrub invasions of North American semiarid grasslands. Annual Review of Ecology and Systematics, **31**, 197–205.

VandenBygaart, A. J., E. G. Gregorich, and D. A. Angers, 2003: Influence of agricultural management on soil organic carbon: A compendium and assessment of Canadian studies. Canadian Journal of Soil Science, **83**(4), 363–380.

West, T. O., and G. Marland, 2003: Net carbon flux from agriculture: Carbon emissions, carbon sequestration, crop yield, and land-use change. Biogeochemistry, **63**, 73–83.

West, T. O., and M. K. Wali, 2002: Modeling regional carbon dynamics and soil erosion in disturbed and rehabilitated ecosystems as affected by land use and climate. Water Air and Soil Pollution, **138**(1-4), 141–163.

Yoo, K., R. Amundson, A. M. Heimsath, and W. E. Dietrich, 2005: Erosion of upland hillslope soil organic carbon: Coupling field measurements with a sediment transport model. Global Biogeochemical Cycles, **19**(3), GB3003.

***[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]***

**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 pools for undisturbed native systems were derived 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 stock values were then multiplied by soil carbon loss factors for tillage- and overgrazing-induced losses (Nabuurs *et al.*, 2004; Ogle *et al.*, 2004) to estimate current soil carbon stocks (see Figure 10-2)

| 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.

**Table 10-2. North American agricultural and grazing land carbon fluxes for the years around 2000**
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, 2005) 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 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

| | Canada | Mexico | United States | Total |
|---|---|---|---|---|
| | Mt C yr$^{-1}$ | | | |
| **$CO_2$** | | | | |
| Total production and use of agricultural inputs | 2.8[a] | ND | 28[b] | 46.3 |
| Fertilizer manufacture | 1.7 | ND | 4.7 | 6.4 |
| Mineral soil carbon sequestration | (0.1) | ND | (6.5)–(16) | (6)–(15.5) |
| Organic soil cultivation | 0.1 | ND | 5–10 | 5.3–10.3 |
| Woody encroachment | ND | ND | (120)[c] | (120) |
| Total | 19.8 | ND | (88.8)–(93.3) | |
| | Mt C-equivalents yr$^{-1}$ | | | |
| **$CH_4$** | | | | |
| Rice production | 0 | 0.1 | 1.9 | 2.0 |
| Biomass burning | 0 | 0.1 | 0.2 | 0.3 |
| Livestock | 5.6 | 12.4 | 38.9 | 56.9 |
| Manure | 0.2 | 0.3 | 10.3 | 10.8 |
| Total | 5.8 | 12.9 | 51.3 | 70.0 |

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

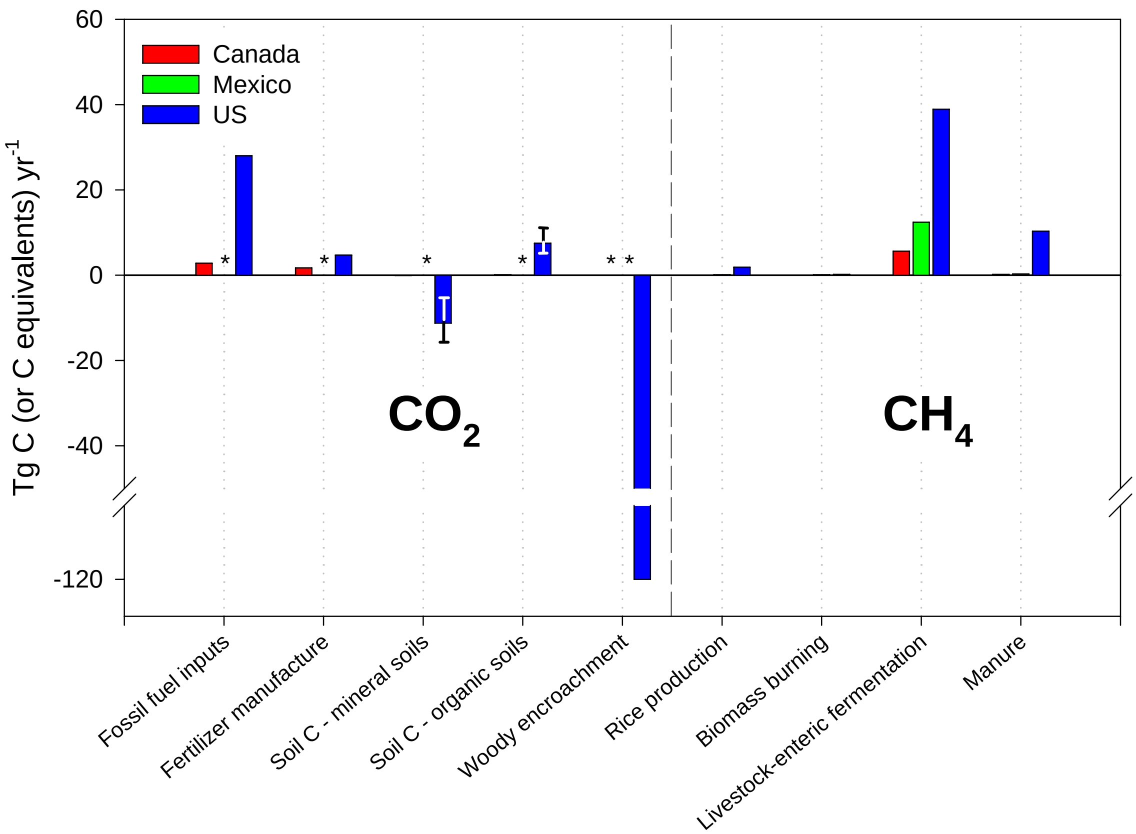


**Figure 10-1. North American agricultural and grazing land $CO_2$ (left side) and methane (right side) fluxes for the years around 2000.** Negative values indicate net flux from the atmosphere to soil and biomass carbon pools. All data are from Canadian (Matin *et al.*, 2004) and U.S. (EPA, 2005) National 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 the United States and Canada and 1998 for Mexico. 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.

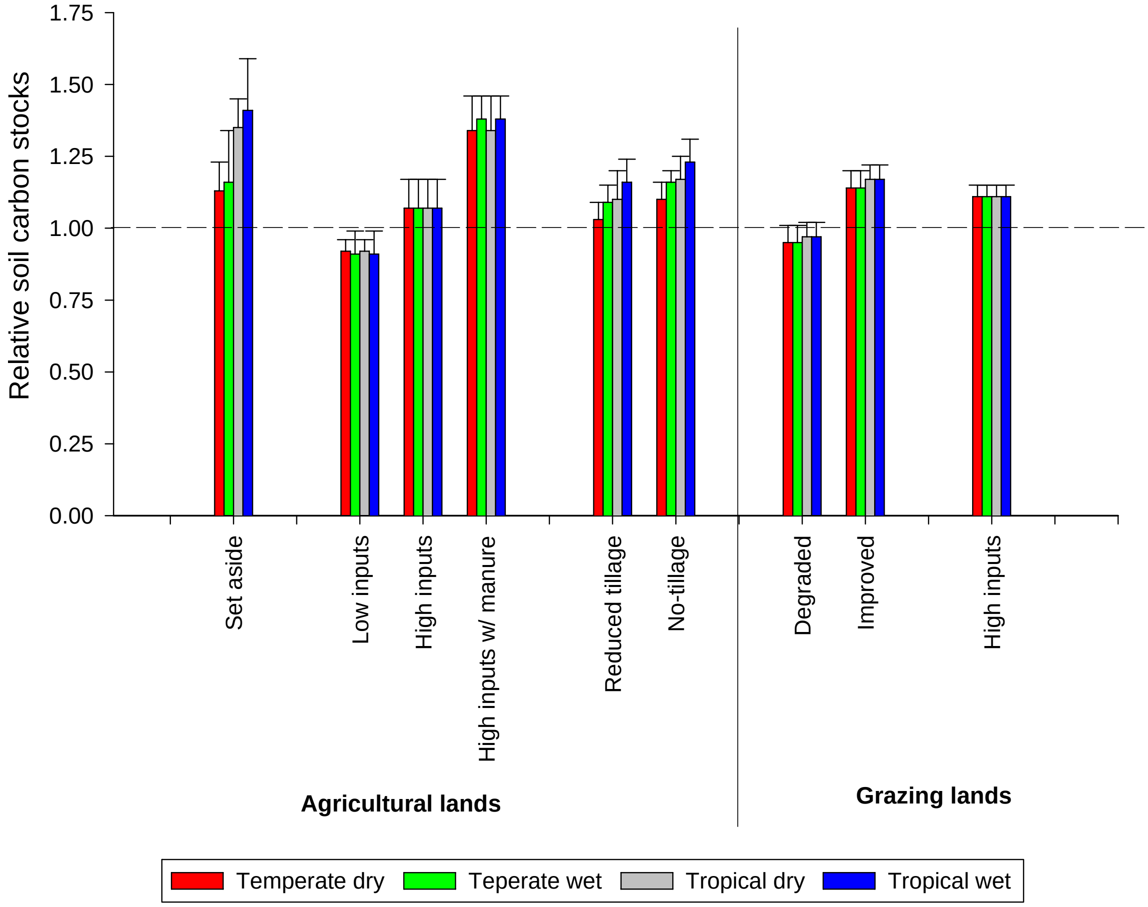


**Figure 10-2. Relative soil carbon following implementation of new agricultural or grassland management practices.** Conventionally tilled, medium-input cultivated land and moderately grazed grasslands with moderate inputs are defaults for agricultural and grazing lands, respectively. Default soil carbon stocks (like those in Table 10-1) can be multiplied by one or more emission factors to estimate carbon sequestration rates. 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

**Lead Authors: Richard A. Birdsey,[1] Jennifer C. Jenkins,[2] Mark Johnston[3] and Elisabeth Huber-Sannwald[4]**

**Contributing Authors: Brian Amiro,[5] Ben de Jong,[6] Jorge D. Etchevers Barra,[7] Nancy French,[8] Felipe García Oliva,[9] Mark Harmon,[10] Linda S. Heath,[1] Victor Jaramillo,[9] Kurt Johnsen,[1] Beverly E. Law,[10] Omar Masera,[9] Ronald Neilson,[1] Yude Pan,[1] Kurt S. Pregitzer,[11] and Erika Marin Spiotta[12]**

[1]USDA Forest Service, [2]University of Vermont, [3]Saskatchewan Research Council, [4]Instituto Potosino de Investigación Científica y Tecnológica, [5]University of Manitoba, [6]ECOSUR, [7]Colegio de Postgraduado, [8]Altarum Institute, [9]Universidad Nacional Autonoma de Mexico, [10]Oregon State University, [11]Michigan Technological University, [12]Stanford University

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## KEY FINDINGS

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

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

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## INTRODUCTION

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

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

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

## CARBON STOCKS AND FLUXES

### Ecosystem Carbon Stocks And Pools

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

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

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

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

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

## Net North American Forest Carbon Fluxes

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

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

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

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

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

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

## TRENDS AND DRIVERS

### Overview of Trends and Drivers of Change in Carbon Stocks

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

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

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

## Effects of Land-Use Change

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

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

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

## Effects of Forest Management

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

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

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

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

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

## Effects of Climate and Atmospheric Chemistry

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

## Effects of Natural Disturbances

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

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

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

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

## Projections of Future Trends

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

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

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

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

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

## OPTIONS FOR MANAGEMENT

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

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

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

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

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

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

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

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

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

## DATA GAPS AND INFORMATION NEEDS FOR DECISION SUPPORT

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

## Major Data Gaps in Estimates of Carbon Pools and Fluxes

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

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

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

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

## Major Data Gaps in Knowledge of Forest Management Effects

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

## Availability Of Decision-Support Tools

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

## REFERENCES

Aber, John, Ronald P. Neilson, Steve McNulty, James M. Lenihan, Dominique Bachelet, and Raymond J. Drapek. 2001. Forest processes and global change: predicting the effects of individual and multiple stressors. BioScience 51(9): 735-751.

Albrecht, A. and S.T. Kandji. 2003. Carbon sequestration in tropical agroforestry systems. Agriculture, Ecosystems and Environments 99: 15’27.

Amiro, B.D., J.B. Todd, B.M. Wotton, K.A. Logan, M.D. Flannigan, B.J. Stocks, J.A. Mason, D.L. Martell and K.G. Hirsch. 2001. Direct carbon emissions from Canadian forest fires, 1959-1999. Can. J. For. Res. 31: 512-525.

Amiro, B.D., A.G. Barr, T.A. Black, H. Iwashita, N. Kljun, J.H. McCaughey, K. Morgenstern, S. Murayama, Z. Nesic, A.L. Orchansky, and N. Saigusa. 2005. Carbon, energy and water fluxes at mature and disturbed forest sites, Saskatchewan, Canada. Agric. For. Meteorol. (in press).

Apps, M.J., W.A. Kurz, S.J. Beukema and J.S. Bhatti. 1999. Carbon Budget of the Canadian Forest Product Sector. Environmental Science & Policy 2: 25-41.

Arain, M.A. and Restrepo-Coupe N. 2005. Net ecosystem production in an eastern white pine plantation in southern Canada. Agricultural and Forest Meteorology 128: 223-241.

Bachelet, Dominique, Ronald P. Neilson, James M. Lenihan, and Raymond J. Drapek. 2001. Climate change effects on vegetation distribution and carbon budget in the United States. Ecosystems 4: 164-185.

Baldocchi, D.D. and J.S. Amthor. 2001. Canopy photosynthesis: history, measurements and models. Pp. 9-31 in Roy, J., B. Saugier and H. Mooney, editors. Terrestrial Global Productivity. Academic Press, San Diego, USA.

Barford CC, Wofsy SC, Goulden ML, Munger JW, Pyle EH, Urbanski SP, Hutyra L, Saleska SR, Fitzjarrald D, Moore K. 2001. Factors controlling long- and short-term sequestration of atmospheric $CO_2$ in a mid-latitude forest. *Science* 294: 1688-1691.

Bechtold, William A. and Paul L. Patterson (Eds). 2005. The enhanced forest inventory and analysis program – national sampling design and estimation procedures. Gen. Tech. Rep. SRS-80. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 85 p.

Birdsey, R.A. and L.S. Heath. 1995. Carbon changes in U.S. forests. In: Climate change and the productivity of America's forests, (L.A. Joyce, ed.). U. S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station General Technical Report, Ft. Collins, CO, p. 56-70.

Birdsey, Richard A., Ralph Alig, and Darius Adams. 2000. Migitation activities in the forest sector to reduce emissions and enhance sinks of greenhouse gases. In: L.A. Joyce and R.A. Birdsey (eds.) The impact of climate change on America's forests: a technical document supporting the 2000 USDA Forest Service RPA assessment. RMRS-GTR-59. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. Fort Collins, CO. 112-131.

Birdsey, Richard A. 2004. Data gaps for monitoring forest carbon in the United States: an inventory perspective. In: Mickler, Robert A., eds. Environmental Management. 33(Supplement 1): S1-S8.

Birdsey, Richard A., and George M. Lewis. 2003. Current and historical trends in use, management, and disturbance of U.S. forestlands. In: J.M. Kimble, L.S. Heath, and R.A. Birdsey, (eds.) The potential of U.S. forest soils to sequester carbon and mitigate the greenhouse effect. CRC Press LLC. New York. 15-33.

Birdsey, Richard, Kurt Pregitzer, and Alan Lucier. 2006. Forest carbon management in the United States,1600-2100. Journal of Environmental Quality [In press].

Caldeira, Ken, M. Granger Morgan, Dennis Baldocchi, Peter G. Brewer, Chen-Tung Arthur Chen, Gert-Jan Nabuurs, Nebojsa Nakicenovic, and G. Phillip Robertson. 2004. A portfolio of carbon management options. In: Field, Christopher B. and Michael R. Raupach. The Global Carbon Cycle. Island Press. Washington, DC.

Canadian Forest Service (CFS). 2005. State of the Forest Report, 2004-2005. Canadian Forest Service, Natural Resources Canada, Ottawa, ON. Available at: http://www.nrcan-rncan.gc.ca/cfs-scf/national/what-quoi/sof/latest_e.html.

Caspersen, John P.; Pacala, Stephen W.; Jenkins, Jennifer C.; Hurtt, George C.; Moorcraft, Paul R.; Birdsey, Richard A. 2000. Contributions of land-use history to carbon accumulation in U.S. forests. Science 290: 1148-1151.

Chapin, F. I., G. Woodwell, J. Randerson, G. Lovett, E. Rastetter, D. Baldocchi, D. Clark, M. Harmon, D. Schimel, R. Valentini, C. Wirth, J. Aber, *et al*. in review. Reconciling carbon cycle terminology: A search for consensus.

Chen JM. Ju W, Cihlar J *et al*. 2003. Spatial distribution of carbon sources and sinks in Canada's forests. Tellus B: 1-20.

Clark, K.L., H.L. Gholz, and M.S. Castro. 2004: Carbon dynamics along a chronosequence of slash pine plantations in north Florida. *Ecological Applications* **14**, 1154-1171.

Dale, Virginia H., Linda A. Joyce, Steve McNulty, Ronald P. Neilson, *et al*. 2001. Climate change and forest disturbances. Bioscience 51(9): 723-734.

De Jong, B.H.J., S. Ochoa-Gaona, M. A. Castillo-Santiago, N. Ramirez-Marcial and M.A. Cairns. 2000. Carbon fluxes and patterns of land-use/land-cover change in the Selva Lacandona, Mexico. Ambio 29: 504-511.

Environment Canada. 2005. Canada's Greenhouse Gas Inventory 1990–2003: Initial Submission. Greenhouse Gas Division, Environment Canada, Ottawa, ON. Available at: http://unfccc.int/national_reports/annex_i_ghg_inventories/national_inventories_submissions/items/2761.php.

Environmental Protection Agency. 2005. Greenhouse gas mitigation potential in U.S. forestry and agriculture. Washington, DC: U.S. Environmental Protection Agency. 154 p.

Food and Agriculture Organization 2001. Global forest resources assessment 2000. Main Report. FAO Forestry Paper 140. Rome. 481 p.

Fitzsimmons, M.J., D.J. Pennock and J. Thorpe. 2004. Effects of deforestation on ecosystem carbon densities in central Saskatchewan, Canada. Forest Ecology and Management 188: 349–361.

Flannigan, M.D., K.A. Logan, B.D. Amiro, W.R. Skinner and B.J. Stocks. 2005. Future area burned in Canada. Climatic Change 72: 1-16.

Foley, Jonathan A., and Navin Ramankutty. A primer on the terrestrial carbon cycle: what we don't know but should. In: Field, Christopher B. and Michael R. Raupach. The Global Carbon Cycle. Island Press. Washington, DC.

Giardina, C.P., D. Binkley, M.G. Ryan, J.H. Fownes, R.S. Senock. 2004. Belowground carbon cycling in a humid tropical forest decreases with fertilization. Oecologia 139:545-550.

Goodale, C.L., M.J. Apps, R.A. Birdsey, C.B. Field, L.S. Heath, R.A. Houghton, J.C. Jenkins, G.H. Kohlmaier,, W. Kurz, S. Liu, G.-J. Nabuurs, S. Nilsson, and A.Z. Shvidenko. 2002. Forest carbon sinks in the northern hemisphere. Ecological Applications 12:891-899.

Griffis, T.J., T.A. Black, K. Morgenstern, A.G. Barr, Z. Nesic, G.B. Drewitt, D. Gaumont-Guay, and J.H. McCaughey, 2003: Ecophysiological controls on the carbon balances of three southern boreal forests, *Agricultural and Forest Meteorology*, **117** (1-2), 53-71.

Harmon, M. E., J. M. Harmon, W. K. Ferrell, and D. Brooks. 1996. Modeling carbon stores in Oregon and Washington forest products: 1900-1992. Climatic Change 33: 521-550.

Harmon, M. 2001. Carbon sequestration in forests - Addressing the scale question. JOURNAL OF FORESTRY **99**:24-29.

Harmon, M. and P. Marks. 2002. Effects of silvicultural practices on carbon stores in Douglas-fir-western hemlock forests in the Pacific Northwest, USA: results from a simulation model. Canadian Journal of Forest Research 32(5): 863-877.

Haynes, Richard W. (ed.). 2003. An analysis of the timber situation in the United States: 1952-2050. Gen. Tech. Rep. PNW-GTR-560. USDA Forest Service. Portland, OR. 254p.

Heath, Linda S. and James E. Smith. 2004. Criterion 5, Indicator 26: Total forest ecosystem biomass and carbon pool, and if appropriate, by forest type, age class and successional change. 14 p. In: Darr, David R., coord. Data report: A supplement to the national report on sustainable forests—2003. FS-766A. Washington, DC: U.S. Department of Agriculture. http://www.fs.fed.us/research/sustain/contents.htm (8 June).

Heath, L.S., and J. E. Smith. 2000. An assessment of uncertainty in forest carbon budget projections. Environmental Science & Policy 3: 73-82.

Hogg, E.H. and P.Y. Bernier. 2005. Climate change impacts on drought-prone forests in western Canada. Forestry Chronicle, in press.

Houghton, R.A., J.L. Hackler, and K.T. Lawrence. 1999. The U.S. carbon budget: contributions from land-use change. Science 285: 574-578.

Humphreys, E.R., Black, T.A., Morgenstern, K., Li, Z., and Nesic, Z., 2005. Net ecosystem production of a Douglas-fir stand for three years following clearcut harvesting. Global Change Biology 11: 450-464.

Janisch, J., and M. Harmon. 2002. Successional changes in live and dead wood carbon stores: implications for net ecosystem productivity. TREE PHYSIOLOGY **22**:77-89.

Jenkins, JC, Chojnacky, DC, Heath, LS, and Birdsey, RA. 2003. National-scale biomass estimators for United States tree species. *Forest Science* 49(1): 12-35.

Johnson, D.W., R.B. Thomas, K.L. Griffen, D.T. Tissue, *et al*. 1998. Effects of carbon dioxide and nitrogen on growth and nitrogen uptake in ponderosa and loblolly pine. J. Environ. Qual. 27:414-425.

Johnston, M. and T. Williamson. 2005. Climate change implications for stand yields and soil expectation values: A northern Saskatchewan case study. Forestry Chronicle 81: 683-690.

Joyce, L., J. Baer, S. McNulty, V. Dale, A. Hansen, L. Irland, R. Neilson, and K. Skog. 2001. Potential consequences of climate variability and change for the forests of the United States. Pages 489-521 in Climate change impacts in the United States, Report for the US Global Change Research Program. Cambridge University Press, Cambridge UK.

Karnosky, D.F., D.R. Zak, K.S. Pregitzer, C.S. Awmack, *et al*. 2003. Tropospheric ozone moderates responses of temperate hardwood forests to elevated CO2: a synthesis of molecular to ecosystem results from the Aspen FACE project. Functional Ecology 17: 289-304.

Korner, C. 2000. Biosphere responses to CO2 enrichment. Ecol. Appl. 10: 1590-1619.

Korner, Christian, Roman Asshof, Olivier Bignucolo, Stephan Haattenschwiler, *et al*. 2005. Carbon flux and growth in mature decidous forest trees exposed to elevated CO2. Science 309: 1360-1362.

Kurz, W.A., Apps, M.J., Webb, T.M., McNamee, P.J. 1992. The carbon budget of the Canadian forest sector: Phase 1. Forestry Canada, Northern Forestry Centre, Information Report NOR-X-326. Edmonton, AB.

Kurz, W.A., S. Beukema and M.J. Apps. 1998. Carbon budget implications of the transition from natural to managed disturbance regimes in forest landscapes. Mitigation and Adaptation Strategies for Global Change 2: 405-421.

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

Kurz, W., M. Apps, E. Banfield and G. Stinson. 2002. Forest carbon accounting at the operational scale. The Forestry Chronicle 78: 672-679.

Law, B.E., E. Falge, D.D. Baldocchi, P. Bakwin, P. Berbigier, K. Davis, A.J. Dolman, M. Falk, J.D. Fuentes, A. Goldstein, A. Granier, A. Grelle, D. Hollinger, I.A. Janssens, P. Jarvis, N.O. Jensen, G. Katul, Y. Mahli, G. Matteucci, R. Monson, W. Munger, W. Oechel, R. Olson, K. Pilegaard, K.T. Paw U, H. Thorgeirsson, R. Valentini, S. Verma, T. Vesala, K. Wilson, and S. Wofsy, 2002: Environmental controls over carbon dioxide and water vapor exchange of terrestrial vegetation. *Agricultural and Forest Meteorology*, **113**, 97-120.

Leenhouts, B. 1998. Assessment of biomass burning in the conterminous United States. Conserv. Ecol. 2(1): 1.

Lewandrowski, Jan, Mark Sperow, Mark Peters, Marlen Eve, Carol Jones, Keith Paustian, and Robert House. 2004. Economics of sequestering carbon in the U.S. agricultural sector. Tech. Bull. 1909. Washington, DC: U.S. Department of Agriculture, Economic Research Service. 61 p.

Lichter, John, Stacy H. Barron, Christine E. Bevacqua, Adrien C. Finzi, Katherine F. Irvine, *et al*. 2005. Soil carbon sequestration and turnover in a pine forest after six years of atmospheric CO2 enrichment. Ecology 86(7): 1835-1847.

Loya, Wendy M., Kurt S. Pregitzer, Noah J. Karberg, John S. King, and Christian P. Giardina. 2003. Reduction of soil carbon formation by tropospheric ozone under increased carbon dioxide levels. Nature 425: 705-707.

Lugo, A. E., J. Figueroa Colón, and F. N. Scatena. 1999. “The Caribbean.” In M.G. Barbour and W.D. Billings, editors. *North American terrestrial vegetation*. Cambridge University Press, England. 530 p.

Masera, O.; Ordóñez, M.J. and Dirzo, R.; 1997a, ‘Carbon Emissions From Mexican Forests: the Current Situation and Long-Term Scenarios’. *Climatic Change*, 35, 265-295.

Masera, O., A. Delia Cerón, A. Ordóñez. 2001. Forestry mitigation options for Mexico: finding synergies between national sustainable development priorities and global concerns. Mitigation and Adaptation Strategies for Global Change 6: 291-312.

McNulty, Steven G. 2002. Hurricane impacts on U.S. forest carbon sequestration. Environmental Pollution 116:S17-S24.

National Forestry Database Program (NFDP). 2005. Compendium of Canadian Forestry Statistics. National Forestry Database Program, Canadian Council of Forest Ministers, Ottawa, ON. Available at: http://nfdp.ccfm.org/compendium/index_e.php.

Natural Resources Canada 2005. The State of Canada's Forests. Canadian Forest Service, Natural Resources Canada, Ottawa, ON. Available at: http://www.nrcan-rncan.gc.ca/cfs-scf/national/what-quoi/sof/latest_e.html

Neilson, R.P., I.C. Prentice, B. Smith, T.G.F. Kittel, and D. Viner. 1998. Simulated changes in vegetation distribution under global warming. Pages 439-456 in R.T. Watson, M.C. Zinyowera, R.H. Moss, and D.J. Dokken, editors. The Regional Impacts of Climate Change: An Assessment of Vulnerability. Cambridge University Press, Cambridge.

Nelson, K.C. and B.H.J. de Jong. 2003. Making global initiatives local realities: carbon mitigation projects in Chiapas, Mexico. Global Environmental Change 13: 19-30.

Norby, Richard J., Evan H. DeLucia, Birget Gielen, *et al*. 2005. Forest response to elevated CO2 is conserved across a broad range of productivity. PNAS 102(50), 18052-18056.

Nowak, R.S., D. S. Ellsworth, S. D. Smith. 2004. Functional responses of plants to elevated atmospheric CO2- do photosynthetic and productivity data from FACE experiments support early predictions? New Phytologist 162:2 253

Ollinger SV, Aber JD, Reich PB, Freuder RJ (2002) Interactive effects of nitrogen deposition, tropospheric ozone, elevated $CO_2$ land use history on the carbon dynamics of northern hardwood forests. *Global Change Biology* **8**, 545-562.

Oren, Ram, David S. Ellworth, Kurt S. Johnsen, Nathan Phillips, *et al.* 2001. Soil fertility limits carbon sequestration by forest ecosystems in a CO2-enriched atmosphere. Nature 411: 469-472.

Osher L. J., P.A. Matson, R.Amundson. 2003. Effect of land use change on soil carbon in Hawaii. Biogeochemistry 65 (2):213-232..

Pacala, S.W.; Hurtt, G.C.; Baker, D.; Peylin, P.; Houghton, R.A.; Birdsey, Richard A.; Heath, L.; Sundquist, E.T.; Stallard, R.F. *et al*. 2001. Consistent land and atmosphere-based U.S. carbon sink estimates. Science. 292: 2316-2320.

Palacio, J.L. *et al*. 2000. *La condición actual de los recursos forestales en México: resultados del Inventario Forestal Nacional 2000*. Technical Note, Investigaciones Geográficas, No. 43 pp. 183-203.

Pan, Y. *et al*. 1998. Modeled responses of terrestrial ecosystems to elevated atmospheric CO2: a comparison of simulations by the biogeochemistry models of the Vegetation/Ecosystem Modeling and Analysis Project (VEMAP). Oecologia 114: 389-404.

Parisien, M.-A., Kafka, V., Flynn, N., Hirsch, K.G., Todd, J.B. and M.D. Flannigan. 2005. Fire behavior potential in central Saskatchewan under predicted climate change. PARC (Prairie Adaptation Research Collaborative), PARC Summary Document 05-01. Regina, SK. 12 p.

Peterson, E.B., Bonnor, G.M., Robinson, G.C. and Peterson, N.M. 1999. Carbon Sequestration Aspects Of An Afforestation Program In Canada's Prairie Provinces. Submitted to Joint Forest Sector Table/Sinks Table, National Climate Change Process, Ottawa, ON. Available at: http://www.nccp.ca/NCCP/national_process/issues/sinks_e.html.

Potter, C., Klooster, S. A., Myneni, R., Genovese, V., Tan, P., Kumar,V. 2003, Continental scale comparisons of terrestrial carbon sinks estimated from satellite data and ecosystem modeling 1982-98. *Global and Planetary Change*, 39, 201-213.

Pregitzer, Kurt S., and Eugenie S. Euskirchen. 2004. Carbon cycling and storage in world forests: biomes patterns related to forest age. Global Change Biology 10: 2052-2077.

Price, D.T., D.H. Halliwell, M.J. Apps, W.A. Kurz, and S.R. Curry. 1997. Comprehensive assessment of carbon stocks and fluxes in a Boreal-Cordilleran forest management unit. Can. J. For. Res. 27: 2005-2016.

Price, D.T., D. W. McKenney, P. Papadopol, T. Logan, and M. F. Hutchinson. 2004. “High resolution future scenario climate data for North America.” *Proc. Amer. Meteor. Soc. 26th Conference on Agricultural and Forest Meteorology,* Vancouver, B.C., 23-26 August, 2004, 13 pp.

Price, D.T., C.H. Peng, M.J. Apps and D.H. Halliwell. 1999. Simulating effects of climate change on boreal ecosystem carbon pools in central Canada. Journal of Biogeography 26: 1237–1248.

Proctor, Patrick, Linda S. Heath, Paul C. Van Deusen, Jeffrey H. Gove, and James E. Smith. 2005. COLE: a web-based tool for interfacing with forest inventory data. In: McRoberts, Ronald E. *et al*. (Eds.), Proceedings of the Fourth Annual Forest Inventory and Analysis Symposium. USDA Forest Service GTR-NC-252.

Ryan, M.G., D. Binkley, and J.H. Fownes. 1997. “Age-related decline in forest productivity: pattern and process.”*Advances in Ecological Research* **27**: 213-262.

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

Schulze, E., J. Lloyd, F. Kelliher, C. Wirth, C. Rebmann, B. Luhker, M. Mund, A. Knohl, I. Milyukova, W. Schulze, W. Ziegler, A. Varlagin, A. Sogachev, R. Valentini, S. Dore, S. Grigoriev, O. Kolle, M. Panfyorov, N. Tchebakova, and N. Vygodskaya. 1999. Productivity of forests in the Eurosiberian boreal region and their potential to act as a carbon sink - a synthesis. *Global Change Biology* **5**:703-722.

Secretaría de Medio Ambiente, Recursos Naturales y Pesca (SEMARNAP); 1996, *Programa Forestal y de Suelo 1995-2000,* Poder ejecutivo Federal, SEMARNAP, México City.

Skog, K.E., and G.A. Nicholson. 1998. Carbon cycling through wood products: the role of wood and paper products in carbon sequestration. Forest Products Journal 48:75-83.
(http://www.fpl.fs.fed.us/documnts/pdf1998/skog98a.pdf )

Smith, W. B., P. D. Miles, J. S. Vissage, and S. A. Pugh. 2004. Forest Resources of the United States, 2002. General Technical Report NC-241, USDA Forest Service, North Central Research Station, St. Paul, MN.

Smith, James E., and Linda S. Heath. 2005. Land use change and forestry and related sections. (Excerpted.) In: U.S. Environmental Protection Agency. Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2003. EPA

430-R-05-003. Available at http://yosemite.epa.gov/oar/globalwarming.nsf/content/ResourceCenterPublicationsGHGEmissionsUSEmissionsInventory2005.html (18 August). [Note that some EPA contractors or employees may have written or edited some text, or formatted tables or redrew some figures for final EPA format.]

Smith, James E. and Linda S. Heath. 2000. Considerations for interpreting probabilistic estimates of uncertainty of forest carbon. P. 102-111. In: Joyce LA and R. Birdsey, eds. The Impact of Climate Change on America's Forests, USDA Forest Service, General Technical Report RMRS-GTR-59. 134 p.

Soto-Pinto, L., G. Jimenez-Ferrer, A. Vargas Guillen, B. de Jong Bergsma, E. Esquivel-Bazan. 2001. Experiencia agroforestral para la captura de carbono en comunidades indigenas de Mexico. Revista Forestal Iberoamericana 1: 44-50

Stainback, G. Andrew, and Janaki R.R. Alavalapati. 2005. Effects of carbon markets on the optimal management of slash pine (Pinus elliottii) plantations. Southern Journal of Applied Forestry 29(1): 27-32.

Stanturf, John A., Robert C. Kellison, F.S. Broerman, and Stephen B. Jones. 2003. Productivity of southern pine plantations – where we are and how did we get here? Journal of Forestry April/May 2003: 26-31.

Stavins, Robert N. and Kenneth R. Richards. 2005. The cost of U.S. forest-based carbon sequestration. Arlington, VA: The Pew Center on Global Climate Change. 40 p. [available at www.pewclimate.org]

Torres R., J.M. 2004. Estudio de tendencias y perspectivas del sector forestal en América Latina al año 2020. Informe Nacional México. FAO. http://www.fao.org/documents/show_cdr.asp?url_file=/docrep/006/j2215s/j2215s11.htm

Totten, Michael. 1999. Getting it right: emerging markets for storing carbon in forests. World Resources Institute. Washington, DC. 49p.

US Climate Change Science Program. 2003. Strategic plan for the climate change science program. Washington, DC: http://www.climatescience.gov/Library/stratplan2003/default.htm

VEMAP Members. 1995. Vegetation/ecosystem modeling and analysis project: comparing biogeography and biogeochemistry models in a continental-scale study of terrestrial ecosystem responses to climate change and CO2 doubling. Global Biochemical Cycles 9: 407-437.

Volney, J.A.W. and K. Hirsch. 2005. Disturbing forest disturbances. Forestry Chronicle 81: 662-668.

Weber, M.G., and M.D. Flannigan, 1997. Canadian boreal forest ecosystem structure and function in a changing climate: impact on fire regimes, Environ. Rev., 5, 145-166.

Winrock International 2005. Ecosystem services. http://www.winrock.org/what/projects.cfm?BU=9086 [Date accessed unknown]

Woodwell, G., and R. Whittaker. 1968. Primary production in terrestrial communities. American Zoologist **8**:19-30.

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

# APPENDIX 11A
# ECOSYSTEM CARBON FLUXES

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

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

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

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

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

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

# APPENDIX 11B

# PRINCIPLES OF FOREST MANAGEMENT FOR ENHANCING CARBON SEQUESTRATION

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

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

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

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

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

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

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

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

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

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

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# Chapter 12. Carbon Cycles in the Permafrost Region of North America

**Lead Author: Charles Tarnocai[1]**

**Contributing Authors: Chien-Lu Ping,[2] and John Kimble[3]**

[1]Agriculture and Agri-Food Canada, [2]University of Alaska, Fairbanks,
[3]USDA Natural Resources Conservation Service (retired)

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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.

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## 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 or no 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. 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.

## REFERENCES


AMAP, 2004: *AMAP Assessment 2002: Persistent Organic Pollutants in the Arctic*. Oslo: Arctic Monitoring and Assessment Programme. xvi+310 p.

Baily, R., and C.T. Cushwa, 1981: Ecoregions of North America. (1:12 million scale map). U.S. Forest Service and U.S. Fish and Wildlife Service.

Bockheim, J.G. and C. Tarnocai, 1998: Recognition of cryoturbation for classifying permafrost-affected soils. *Geoderma*, **81**, 281–293.

Brown, J., O.J. Ferrians, Jr., J.A. Heginbottom, and E.S. Melnikov, 1997: Circum-Arctic Map of Permafrost and Ground Ice Conditions. (1:10 million scale map). International Permafrost Association.

Burgess, M.M., and C. Tarnocai, 1997: Peatlands in the Discontinuous Permafrost Zone along the Norman Wells pipeline, Canada. In: *Proceedings of the International Symposium on Physics, Chemistry, and Ecology of Seasonally Frozen Soils Fairbanks, Alaska, June 10–12, 1997* (I.K. Iskandr, E.A. Wright, J.K. Radke, B.S. Sharratt, P.H. Groenevelt, and L.D. Hinzman, eds.), 417–424. Hanover, U.S.A.: U.S. Army Cold Regions Research and Engineering Laboratory, Special Report 97-10.

Christensen, T., 1991: Arctic and sub-Arctic soil emissions: Possible implications for global climate change. *Polar Record*, **27**, 205–210.

Cryosol Working Group, 2001: Northern and Mid Latitudes Soil Database, Version 1. National Soil Database. Ottawa, ON: Research Branch, Agriculture and Agri-Food Canada.

Davidson, E.A., and I.A. Janssens, 2006: Temperature sensitivity of soil carbon decomposition and feedbacks to climate change. *Nature*, **440**, 165–173.

Driscoll, C.T., J. Holsapple, C.L. Schofield, and R. Munson, 1998: The chemistry and transport of mercury in a small wetland in the Adirondack region of New York, USA. *Biogeochemistry*, 40: 137–146.

Ecoregions Working Group, 1989: *Ecoclimatic Regions of Canada*, First Approximation. Ecoregions Working Group of the Canada Committee on Ecological Land Classification. Ecological Land Classification Series, No. 23, 119 p. and map. Ottawa, ON: Sustainable Development Branch, Canadian Wildlife Service, Conservation and Protection, Environment Canada.

Gorham, E., 1988: Canada's peatlands: their importance for the global carbon cycle and possible effect of "greenhouse" climate warming. Transactions of the Royal Society of Canada, Series V, **3**, 21–23.

Hengeveld, H.G., 2000. Projections for Canada's climate future: A discussion of recent simulations with the Canadian Global Climate Model. *Climate Change Digest*, CCD 00-01, Special Edition, 27 p. Downsview, ON: Meteorological Service of Canada, Environment Canada. (Available from http://www.msc.ec.gc.ca/saib/climate/docs/ccd_00-01.pdf. Last accessed: April 6, 2005.)

Jorgenson, M.T., C.H. Racine, J.C. Walters, and T.E. Osterkamp, 2001: Permafrost degradation and ecological changes associated with a warming climate in central Alaska. *Climate Change*, **48**, 551–579.

Kettles, I.M., and Tarnocai, C., 1999: Development of a model for estimating the sensitivity of Canadian peatlands to climate warming. *Geographie physique et Quaternaire*, **53**, 323–338.

Kokelj, S.V., and C.R. Burn, 2005: Geochemistry of the active layer and near-surface permafrost, Mackenzie delta region, Northwest Territories, Canada. Canadian *Journal of Earth Sciences*, **42**, 37–48.

Kuhry, G.P., 1994: The role of fire in the development of Sphagnum-dominated peatlands in the western boreal Canada. *Journal of Ecology*, **82**, 899–910.

Lacelle, B., C. Tarnocai, S. Waltman, J. Kimble, N. Bliss, B. Worstell, F. Orozco-Chavez, and B. Jakobsen, 2000: North American soil organic carbon map. (1:10 million scale map). Agriculture and Agri-Food Canada, USDA, USGS, INEGI and Institute of Geography, University of Copenhagen.

Liblik, L.K., T.R. Moore, J.L. Bubier, and S.D. Robinson, 1997: Methane emissions from wetlands in the zone of discontinuous permafrost: Fort Simpson, Northwest Territories, Canada. *Global Biogeochemical Cycles*, **11**, 485–494.

Mackay, J.R., 1980: The origin of hummocks, western Arctic coast, Canada. Canadian *Journal of Earth Sciences*, **13**, 889–897.

Melillo, J.M., T.V. Callaghan, F.I. Woodward, E. Salati, and S.K. Sinha, 1990: Effects on Ecosystems. In: *Climate Change: The IPCC Scientific Assessment* (J.T. Houghton, G.J.Jenkins, and J.J. Ephraums, eds.), 283–310 (Chapter 10). Cambridge (UK): Cambridge University Press.

Moore, T.R., 1997: Dissolved organic carbon: sources, sinks, and fluxes and role in the soil carbon cycle. In: *Soil Processes and the Carbon Cycle* (R. Lal, J.M. Kimble, R.F. Follett, and B.A. Stewart, eds.), 281–292 (Chapter 19). Boca Raton: Advances in Soil Science, CRC Press.

Moore, T.R., and N.T. Roulet, 1995: Methane emissions from Canadian peatlands. In: *Soils and Global Change* (R. Lal, J. Kimble, E. Levine, and B.A. Stewart, eds.), 153–164 (Chapter 12). Boca Raton: CRC Lewis Publishers.

National Wetlands Working Group, 1988: *Wetlands of Canada*, Ecological Land Classification Series No. 24, 452 p. Ottawa: Sustainable Development Branch, Environment Canada and Montreal: Polyscience Publications, Ltd.

Oechel, W., and G.L. Vourlitis, 1994: The effect of climate change on land–atmosphere feedbacks in arctic tundra regions. *TREE*, **9**, 324–329.

Peterson, R.A., and W.B. Krantz, 2003: A mechanism for differential frost heave and its implications for patterned-ground formation. *Journal of Glaciology*, **49**(164), 69–80(12).

Reader, R.J., and J.M. Stewart, 1972: The relationship between net primary production and accumulation for a peatland in southeastern Manitoba. *Ecology*, **53**, 1024–1037.

Ritchie, J.C., 1987: *Postglacial Vegetation of Canada*, 178 p. New York, NY: Cambridge University Press.

Robinson, S.D., and T.R. Moore, 1999: Carbon and peat accumulation over the past 1200 years in a landscape with discontinuous permafrost, northwestern Canada. *Global Biogeochemical Cycles*, **13**, 591–601.

Robinson, S.D., and T.R. Moore, 2000: The influence of permafrost and fire upon carbon accumulation in High Boreal peatlands, Northwest Territories, Canada. *Arctic, Antarctic, and Alpine Research*, **32**(2), 155–166.

Robinson, S.D., M.R. Turetsky, I.M. Kettles, and R.K. Wieder, 2003: Permafrost and peatland carbon sink capacity with increasing latitude. In: *Proceedings of the 8$^{th}$ International Conference on Permafrost*, Zurich, Switzerland (M. Phillips, S.M. Springman, and L.U. Arenson, eds.), **2**, 965–970. Lisse, The Netherlands: Balkema Publishers.

Soil Carbon Database Working Group, 1993: Soil carbon for Canadian soils. (digital database). Ottawa, ON: Centre for Land and Biological Resources Research, Research Branch, Agriculture and Agri-Food Canada.

Suchanek, T.H., P.J. Richerson, J.R. Flanders, D.C. Nelson, L.H. Mullen, L.L. Brester, and J.C. Becker, 2000: Monitoring inter-annual variability reveals sources of mercury contamination in Clear Lake, California. *Environmental Monitoring and Assessment*, 64: 299–310.

Tarnocai, C., 1998: The amount of organic carbon in various soil orders and ecological provinces in Canada. In: *Soil Processes and the Carbon Cycle* (R. Lal, J.M. Kimble, R.L.F. Follett, and B.A. Stewart, eds.), 81–92. New York, NY: Advances in Soil Science, CRC Press.

Tarnocai, C., 1999: The effect of climate warming on the carbon balance of Cryosols in Canada. In: *Cryosols and Cryogenic Environments* (C. Tarnocai, R. King, and S. Smith, eds.), special issue of *Permafrost and Periglacial Processes*, **10**(3), 251–263.

Tarnocai, C., 2000: Carbon pools in soils of the Arctic, Subarctic and Boreal regions of Canada. In: *Global Climate Change and Cold Regions Ecosystems* (R. Lal, J.M. Kimble, and B.A. Stewart, eds.), 91–103. Boca Raton, Fla.: Advances in Soil Science, Lewis Publishers.

Tarnocai, C., In press: The effect of climate change on carbon in Canadian peatlands. *Global and Planetary Change*.

Tarnocai, C., I.M. Kettles, and B. Lacelle, 2005: Peatlands of Canada Database. (digital database). Ottawa, ON: Research Branch, Agriculture and Agri-Food Canada.

Trumbore, S.E., and J.W. Harden, 1997: Accumulation and turnover of carbon in organic and mineral soils of the BOREAS northern study area. *Journal of Geophysical Research*, **102**(D24), 28.817–28,830.,

Turetsky, M.R., B.D. Amiro, E. Bosch, and J.S. Bhatti, 2004: Historical burn area in western Canadian peatlands and its relationship to fire weather indices. *Global Biogeochemical Cycles*, **18**, GB4014, doi:10.1029/2004GB002222.

Vandenberghe, J., 1992: Cryoturbations: a sediment structural analysis. *Permafrost and Periglacial Processes*, **4**, 121–135.

van Everdingen, R. (ed.), 1998 revised May 2005: *Multi-language Glossary of Permafrost and Related Ground-ice Terms*. 90 p. Boulder, CO: National Snow and Ice Data Center/World Data Center for Glaciology. (Available from http://nsidc.org/fgdc/glossary)

Van Vliet-Lanoë, B., 1991: Differential frost heave, load casting and convection: converging mechanisms; a discussion of the origin of cryoturbations. *Permafrost and Periglacial Processes*, **2**, 123–139.

Vitt, D.H., L.A. Halsey, I.E. Bauer, and C. Campbell, 2000: Spatial and temporal trends in carbon storage of peatlands of continental western Canada through the Holocene. *Canadian Journal of Earth Sciences*, **37**, 683–693.

Walker, D.A., V.E. Romanovsky, W.B. Krantz, C.L. Ping, R.A. Peterson, M.K. Raynolds, H.E. Epstein, J.G. Jia, and D.C. Wirth, 2002: Biocomplexity of frost boil ecosystem on the Arctic Slope, Alaska. ARCUS 14th Annual Meeting and Arctic Forum 2002, Arlington, Virginia, USA. (Available from http://siempre.arcus.org/4DACTION/wi_pos_displayAbstract/5/391)

Zoltai, S.C., C. Tarnocai, and W.W. Pettapiece, 1978: Age of cryoturbated organic material in earth hummocks from the Canadian arctic. *Proceedings of the Third International Conference on Permafrost*, 325–331. Edmonton, AB.

Zoltai, S.C., S. Taylor, J.K. Jeglum, G.F. Mills, and J.D. Johnson, 1988: Wetlands of Boreal Canada. In: *Wetlands of Canada*, (National Wetlands Working Group) 97–154. Ecological Land Classification Series, No. 24. Ottawa, ON: Sustainable Development Branch, Environment Canada and Montreal, PQ: Polyscience Publications Inc.

**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)

| Type | Peatlands | | Mineral soils | | Total |
|---|---|---|---|---|---|
| | Perennially frozen | Unfrozen | Perennially frozen | Unfrozen | |
| 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).

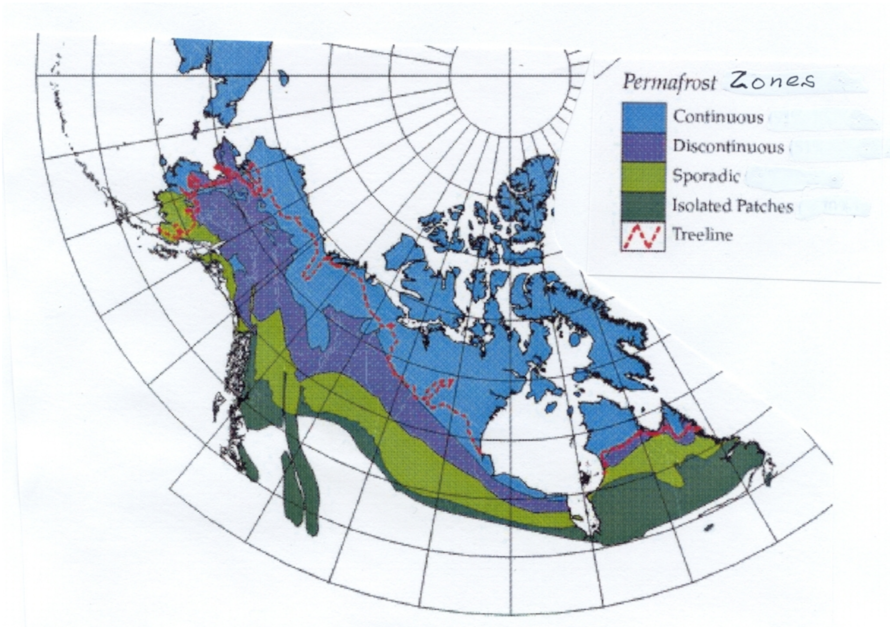



**Fig. 12-1. Permafrost zones in North America (Brown *et al*., 1997).**

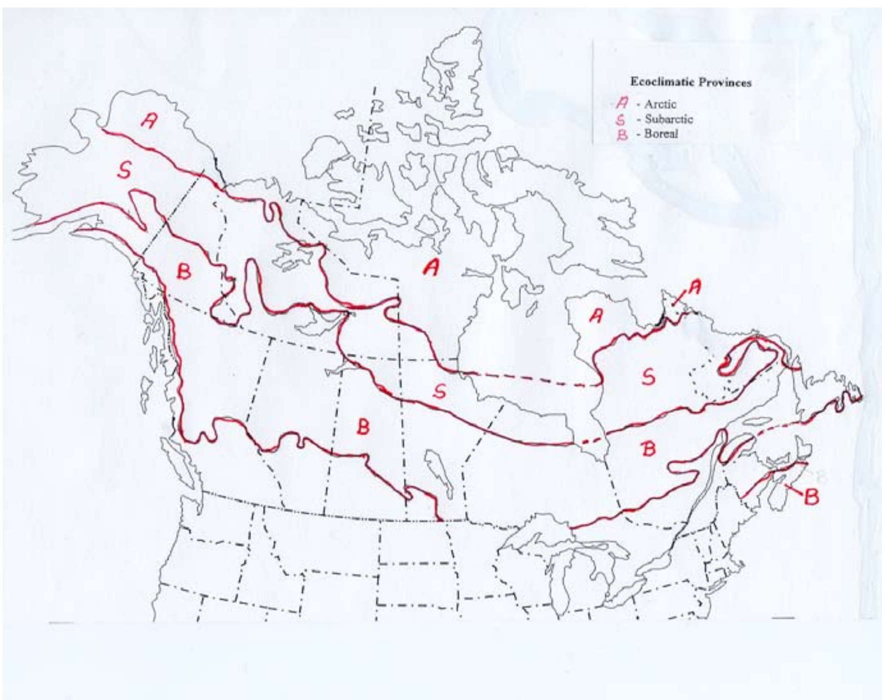
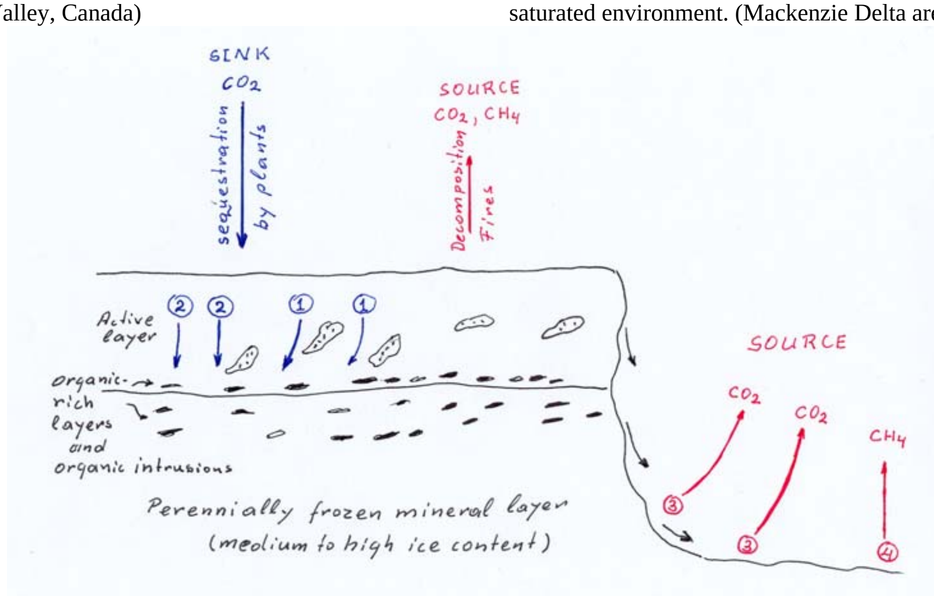



**Fig. 12-2. Arctic, Subarctic, and Boreal ecoclimatic provinces (ecological regions) in North America (Ecoregions Working Group, 1989; Baily and Cushwa, 1981).**

| **Carbon sinks** | **Carbon sources** |
|---|---|
|  |  |
| 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) | 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.**

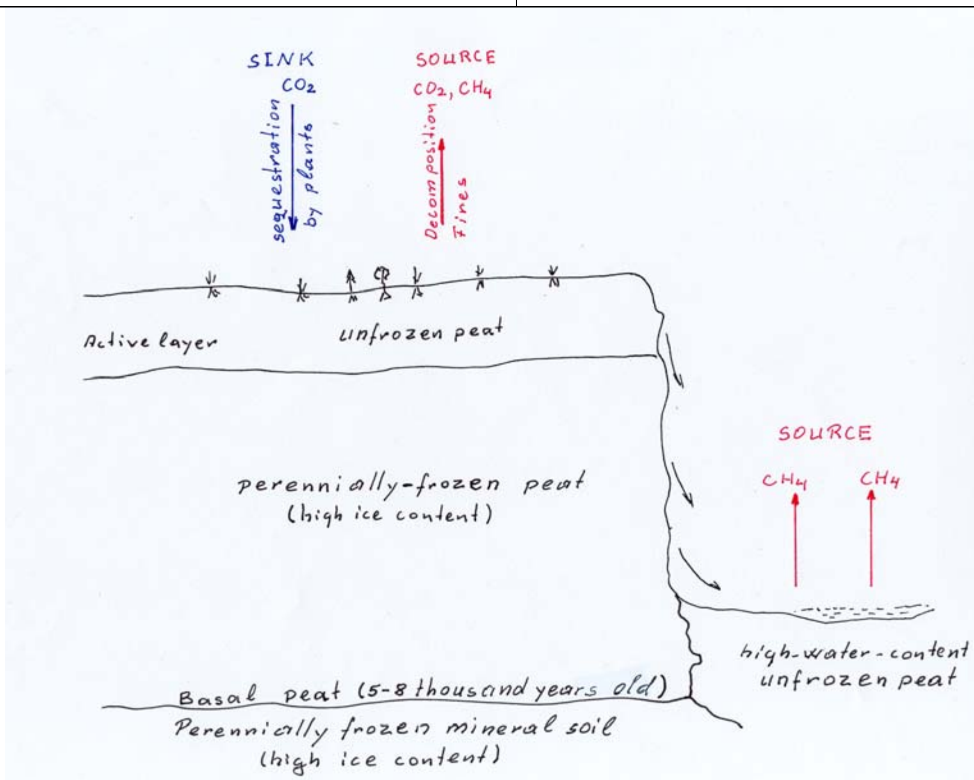
<table>
<tr><th>Carbon sinks</th><th>Carbon sources</th></tr>
<tr><td><br>Perennially frozen peat deposit with multiple dark-colored peat layers. (Mackenzie River Delta area, Canada)</td><td><br>Eroding perennially frozen peat deposit, showing the large blocks of peat slumping into the water- saturated collapsed area. (Fort Simpson area, Canada)</td></tr>
<tr><td colspan="2">
</td></tr>
<tr><td>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.<br><br>Organic material is deposited annually on the peatland surface. Although a large portion ($\geq$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.</td><td>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.</td></tr>
</table>

**Fig. 12-4. Carbon cycle in permafrost peatlands, showing below-ground organic carbon sinks and sources.**

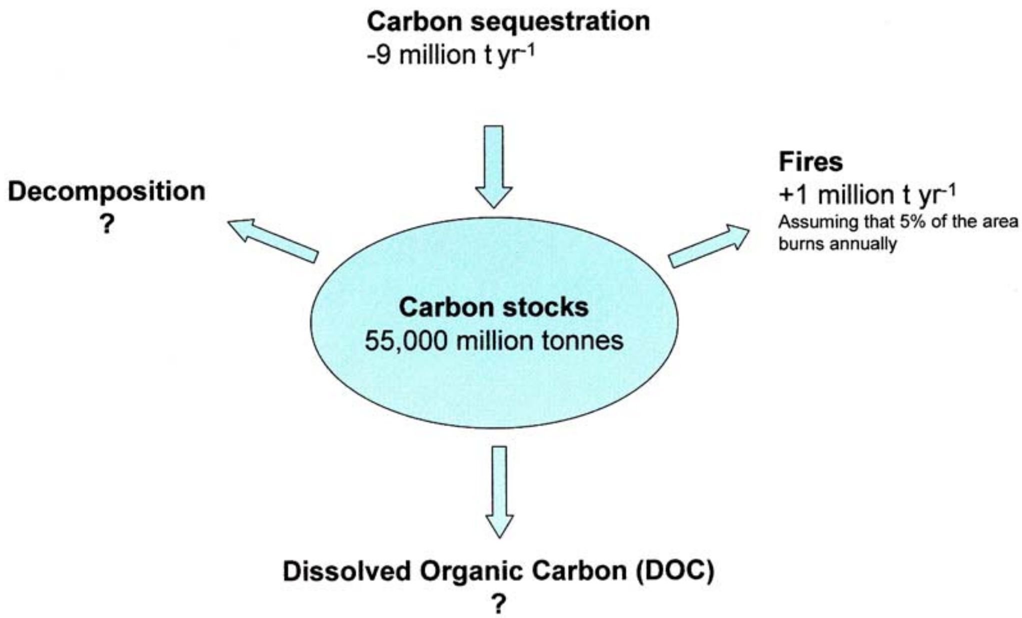



**Fig. 12-5. Carbon cycle in perennially frozen mineral soils in the permafrost region.**

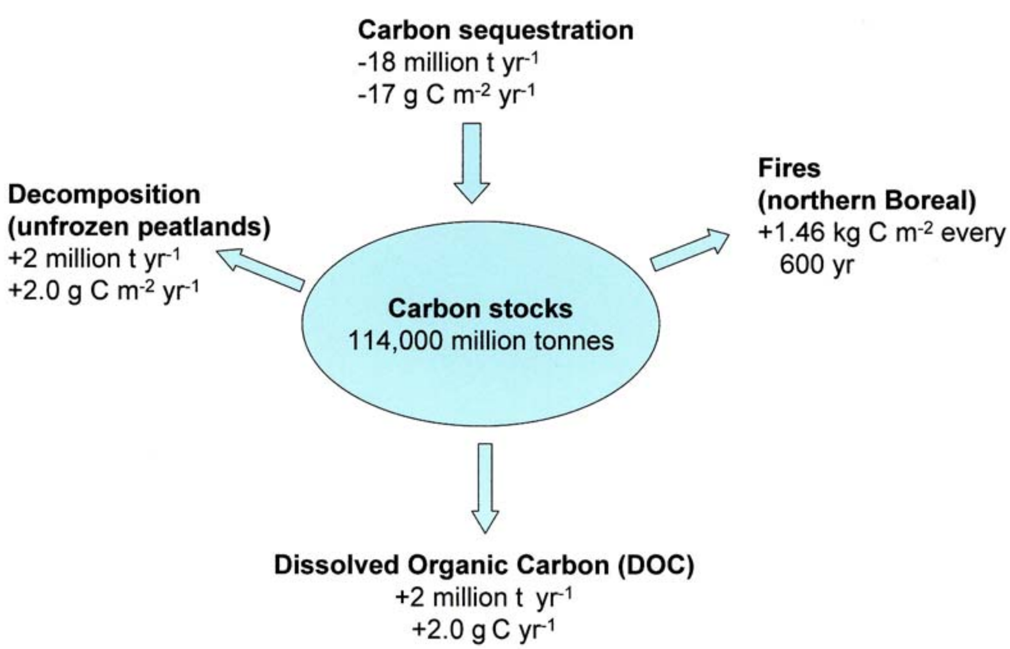
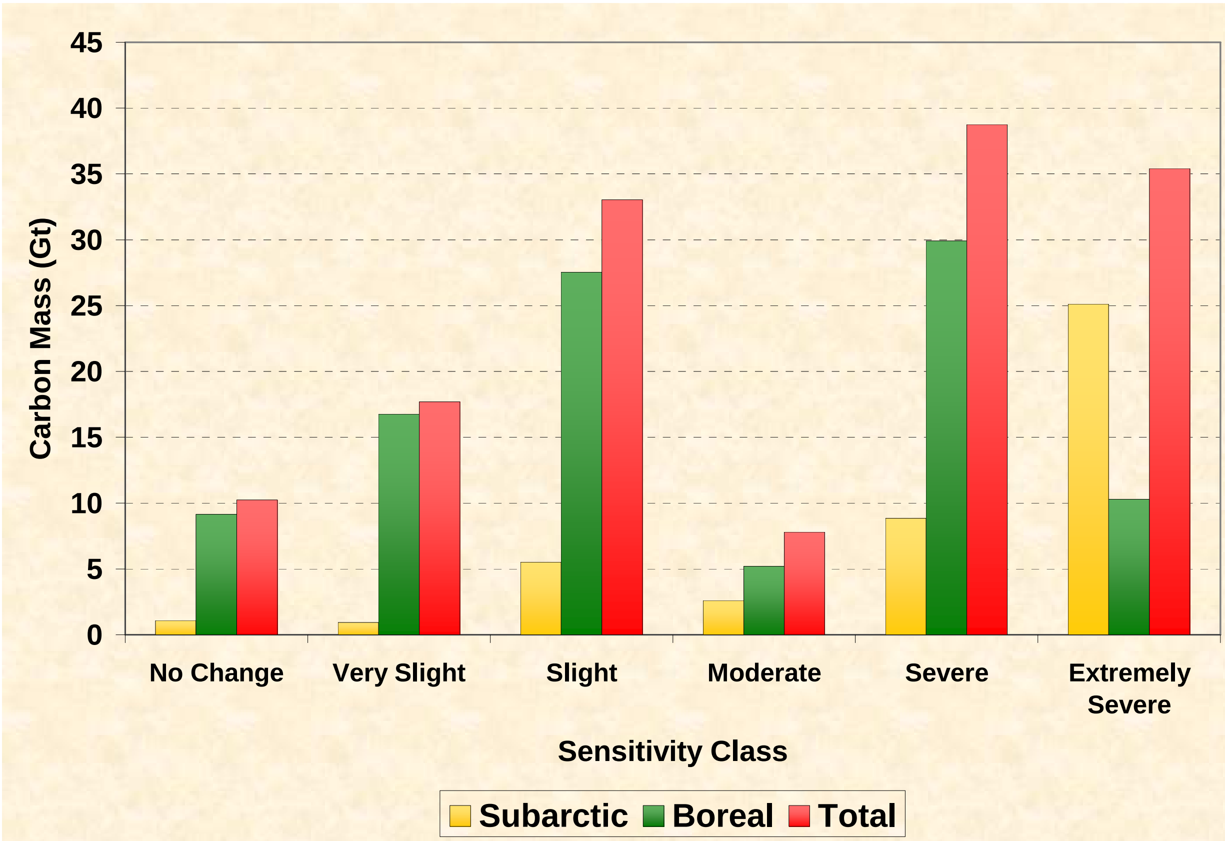



**Fig. 12-6. Carbon cycle in peatlands in the permafrost region.**

1




2 **Fig. 12-7. The organic carbon mass in the various sensitivity classes for the Subarctic and**
3 **Boreal Ecoclimatic Provinces (ecological regions) (Tarnocai, in press).**

4

# Chapter 13. Wetlands

**Lead Author: Scott D. Bridgham[1]**

**Contributing Authors: J. Patrick Megonigal,[2] Jason K. Keller,[2] Carl Trettin,[3] and Norman B. Bliss[4]**

[1]Center for Ecology and Evolutionary Biology, University of Oregon, [2]Smithsonian Environmental Research Center, [3]Center for Forested Wetland Research, USDA Forest Service, [4]SAIC, USGS Center for Earth Resources Observation and Science

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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.

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## 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 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). This peat has formed over thousands of years, and therefore the potential emissions from the large pool of soil carbon are likely more significant to the global carbon budget than the current soil carbon sequestration rate. 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 ($CH_4$) $yr^{-1}$, 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.

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 the emissions of greenhouse gases from unperturbed wetlands is important only in establishing a baseline condition. Thus, we consider changes from historical (~1800) fluxes and present and future perturbations 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**. 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.

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 $6.0 \times 10^6$ $km^2$ (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 21 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 extremely unlikely that the actual sequestration rate is this high, as the data are probably strongly biased by researchers choosing wetlands with high sediment deposition to study this process. More fundamentally, carbon in sediments that are simply redistributed in the landscape due to erosion from a terrestrial source to a wetland sink does not represent carbon sequestration except to the extent that decomposition rates are lower in wetlands. Much of this sediment-associated carbon is probably relatively stable in upland soils, so FWMS wetlands may not represent a substantial sediment carbon sink at the landscape scale. There are no data to our knowledge to evaluate this important caveat. Based upon this reasoning, we somewhat arbitrarily reduced our calculated FWMS wetland sediment carbon sequestration rate by 75% to -34 Mt C $yr^{-1}$ (Table 13A-2 in Appendix 13A). This is still a substantial sink and an important unknown in carbon budgets. 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 1.4 Mt C $yr^{-1}$ due to loss of area (Table 13-1). 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 divided into allochthonous carbon (sediment-derived carbon from outside the wetland) and autochthonous carbon (derived from rates of plant productivity being greater than decomposition within the wetland). As with FWMS wetlands, soil carbon sequestration in estuarine wetlands is overestimated to the extent that allochthonous carbon simply represents redistribution of carbon in the landscape. There is also large uncertainty in the area of mud flats.

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. With 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). Our synthesis 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.*, 2004a).

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 (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).

The global warming potential (GWP) of a gas depends on its instantaneous radiative forcing and its lifetime in the atmosphere, with methane having 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]. Thus, depending upon the time frame and within the large confidence limits of many of our estimates in

[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].

Table 13-1, 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 (but note caution in the *Introduction* in converting this into radiative forcing). 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).

## 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 and many peatlands 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

Historically, the destruction of wetlands through land-use changes has had the largest effect on the carbon fluxes and the GWPs 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 moderate decrease in radiative forcing) (Table 13A-1 and Appendix 13A). 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). 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, in press).

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 Keddy, 1989; Roulet, 2000; Joosten and Clarke, 2002 and references therein). 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 predicted 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). Another methane-process model suggested that net methane emissions from northern wetlands have increased by 0.08 Mt $CH_4$ $yr^{-1}$ during the twentieth century and by 1.0 Mt $CH_4$ $yr^{-1}$ during the 1980s (Zhuang *et al.*, 2004). Inverse modeling also shows that atmospheric anomalies in methane during the 1990s may be partially explained by interannual climate effects on wetland emissions (Fletcher *et al.*, 2004b; Wang *et al.*, 2004). Thus, the above-mentioned studies suggest that past climate change has already had an effect on wetland methane emissions and that this will only be exacerbated in the future.

Other important anthropogenic forcing factors that will affect future methane emissions include atmospheric sulfate deposition (Vile *et al.*, 2003; Gauci *et al.*, 2004), atmospheric carbon dioxide concentrations (Megonigal and Schlesinger, 1997; Vann and Megonigal, 2003), 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.

The effects of global change on carbon sequestration in peatlands is 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. Northern peatlands may also emit more methane with warmer temperatures, depending on changes in water table levels. The effects of global change on estuarine wetlands is of concern because sequestration rates are rapid, and they can be expected to increase with the rate of sea level rise provided the estuarine wetland area does not decline. It remains to be determined whether rising atmospheric carbon dioxide, temperature, nitrogen deposition, and shoreline construction will permit the area of estuarine wetlands to remain stable.

## 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, in press). 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 evaluate this. Roulet (2000) came to a similar conclusion concerning the restoration of Canadian wetlands. 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). 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), although they did not account 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), with the largest unknown being the role of carbon sequestration by 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 their protection. 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 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.

Global change effects on the carbon pools and fluxes of North American wetlands are the largest future unknown. 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 wetlands.

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## REFERENCES

Armentano, T. B., and E. S. Menges, 1986: Patterns of change in the carbon balance of organic soil-wetlands of the temperate zone. *Journal of Ecology*, **74**, 755–774.

Aselmann, I., and P. J. Crutzen, 1989: Global distribution of natural freshwater wetlands and rice paddies, their net primary productivity, seasonality and possible methane emissions. *Journal of Atmospheric Chemistry*, **8**, 307–359.

Barker, J. R., G. A. Baumgardner, D. P. Turner, and J. J. Lee, 1996: Carbon dynamics of the conservation and wetland reserve program. *Journal of Soil and Water Cons.*, **51**, 340–346.

Bartlett, K. B., and R. C. Harriss, 1993: Review and assessment of methane emissions from wetlands. *Chemosphere*, **26**, 261–320.

Bridgham, S. D., C.-L. Ping, J. L. Richardson, and K. Updegraff, 2000: Soils of northern peatlands: Histosols and Gelisols. In: *Wetland Soils: Genesis, Hydrology, Landscapes, and Classification* (J. L. Richardson, and M. J. Vepraskas, eds.), 343–370. Boca Raton, FL: CRC Press.

Bridgham, S. D., K. Updegraff, and J. Pastor, 1998: Carbon, nitrogen, and phosphorus mineralization in northern wetlands. *Ecology*, **79**, 1545–1561.

Cao, M., K. Gregson, and S. Marshall, 1998: Global methane emission from wetlands and its sensitivity to climate change. *Atmospheric Environment*, **32**, 3293–3299.

Cleary, J., N. T. Roulet, and T. R. Moore, 2005: Greenhouse gas emissions from Canadian peat extraction, 1990–2000: A life-cycle analysis. *Ambio*, **34**, 456–461.

Cowardin, L. M., V. Carter, F. C. Golet, and E. T. LaRoe, 1979: Classification of wetlands and deepwater habitats of the United States. FWS/OBS-79/31, Washington, D.C.: Fish and Wildlife Service, U.S. Department of the Interior.

Dahl, T. E., 2000: Status and Trends of Wetlands in the Conterminous United States, 1986 to 1997.Washington, D.C.: Fish and Wildlife Service, U.S. Department of the Interior.

Dugan, P., ed., 1993: Wetlands in Danger—A World Conservation Atlas. New York: Oxford University Press.

Ehhalt, D., M. Prather, F. Dentener, E. Dlugokencky, E. Holland, I. Isaksen, J. Katima, V. Kirchhoff, P. Matson, P. Midgley, and M. Wang, 2001: Atmospheric chemistry and greenhouse gases. In: *Climate Change 2001: The Scientific Basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change* (J. T. Houghton, Y. Ding, D. J. Griggs, M. Noguer, P. J. van der Linden, X. Dai, K. Maskell, and C. A. Johnson, eds.), 239–287. Cambridge: Cambridge University Press.

Finlayson, C. M., N. C. Davidson, A. G. Spiers, and N. J. Stevenson, 1999: Global wetland inventory—current status and future priorities. *Marine Freshwater Research*, **50**, 717–727.

Fletcher, S. E. M., P. P. Tans, L. M. Bruhwiler, J. B. Miller, and M. Heimann, 2004a: $CH_4$ sources estimated from atmospheric observations of $CH_4$ and its $^{13}C/^{12}C$ isotopic ratios: 2. Inverse modeling of $CH_4$ fluxes from geographical regions. *Global Biogeochemical Cycles*, **18**, doi:10.1029/2004GB002224.

Fletcher, S. E. M., P. P. Tans, L. M. Bruhwiler, J. B. Miller, and M. Heimann, 2004b: $CH_4$ sources estimated from atmospheric observations of $CH_4$ and its $^{13}C/^{12}C$ isotopic ratios: 1. Inverse modeling of source processes. *Global Biogeochemical Cycles*, **18**, doi:10.1029/2004GB002223.

Frolking, S., N. Roulet, and J. Fuglestvedt, 2006: How northern peatlands influence the earth's radiative budget: Sustained methane emission versus sustained carbon sequestration. *JGR-Biogeosciences*, **111**, G01008, doi:01010.01029/02005JG000091.

Frolking, S., N. T. Roulet, T. R. Moore, P. M. Lafleur, J. L. Bubier, and P. M. Crill, 2002: Modeling seasonal to annual carbon balance of Mer Bleue Bog, Ontario, Canada. *Global Biogeochemical Cycles*, **16**, 10.1029.2001GB00147, 02002.

Gauci, V., E. Matthews, N. Dise, B. Walter, D. Koch, G. Granberg, and M. Vile, 2004: Sulfur pollution suppression of the wetland methane source in the 20th and 21st centuries. *Proceeding of the National Academy of Sciences, USA*, **101**, 12583–12587.

Gedney, N., P. M. Cox, and C. Huntingford, 2004: Climate feedbacks from methane emissions. *Geophysical Research Letters*, **31**, L20503, doi:20510.21029/22004GL020919.

Gorham, E., 1991: Northern peatlands: Role in the carbon cycle and probable responses to climatic warming. *Ecological Applications*, **1**, 182–195.

Hines, M. E., and K. N. Duddleston, 2001: Carbon flow to acetate and $C_1$ compounds in northern wetlands. *Geophysical Research Letters*, **28**, 4251–4254.

Johnston, C. A., 1991: Sediment and nutrient retention by freshwater wetlands: effects on surface water quality. *Critical Reviews in Environmental Control*, **21**, 491–565.

Joosten, H., and D. Clarke, 2002: Wise Use of Mires and Peatlands - Background Principles including a Framework for Decision-Making. Saarijärvi, Finland: International Mire Conservation Group and International Peat Society.

Keller, J. K., S. D. Bridgham, C. T. Chapin, and C. M. Iversen, 2005: Limited effects of six years of fertilization on carbon mineralization dynamics in a Minnesota fen. *Soil Biology and Biochemistry*, **37**, 1197–1204.

Lynch-Stewart, P., I. Kessel-Taylor, and C. Rubec, 1999: Wetlands and Government: Policy and Legislation for Wetland Conservation in Canada, No. 1999-1: North American Wetlands Conservation Council (Canada).

Maltby, E., and P. Immirzi, 1993: Carbon dynamics in peatlands and other wetland soils, regional and global perspectives. *Chemosphere*, **27**, 999–1023.

Matthews, E., and I. Fung, 1987: Methane emission from natural wetlands: Global distribution, area, and environmental characteristics of sources. *Global Biogeochemical Cycles*, **1**, 61–86.

Megonigal, J. P., and W. H. Schlesinger, 1997: Enhanced $CH_4$ emissions from a wetland soil exposed to elevated $CO_2$. *Biogeochemistry*, **37**, 77–88.

Mitra, S., R. Wassmann, and P. L. G. Vlek, 2005: An appraisal of global wetland area and its organic carbon stock. *Current Science*, **88**, 25–35.

Moore, D. R. J., and P. A. Keddy, 1989: The relationship between species richness and standing crop in wetlands: the importance of scale. *Vegetatio*, **79**, 99–106.

Moore, T. R., and N. T. Roulet, 1995: Methane emissions from Canadian peatlands. In: *Soils and Global Change* (R. Lal, J. Kimble, E. Levine, and B. A. Stewart, eds.), 153–164. Boca Raton, FL: Lewis Publishers.

Moore, T. R., N. T. Roulet, and J. M. Waddington, 1998: Uncertainty in predicting the effect of climatic change on the carbon cycling of Canadian peatlands. *Climatic Change*, **40**, 229–245.

Moser, M., C. Prentice, and S. Frazier, 1996, A global overview of wetland loss and degradation: Ramsar 6th Meeting of the Conference of the Contracting Parties in Brisbane, Australia.

National Research Council, 1995: Wetlands: Characteristics and Boundaries. Washington, D.C.: National Academy Press.

National Research Council, 2001: Compensating for wetland losses under the clean water act. Washington, D.C.: National Academy Press.

National Wetlands Working Group, 1997: The Canadian Wetland Classification System. Waterloo, Ontario, Canada: Wetlands Research Centre, University of Waterloo.

OECD, 1996: Guidelines for aid agencies for improved conservation and sustainable use of tropical and subtropical wetlands. Paris: Organization for Economic Co-operation and Development.

Ramaswamy, V., O. Boucher, J. Haigh, D. Hauglustaine, J. Haywood, G. Myhre, T. Nakajima, G. Y. Shi, and S. Solomon, 2001: Radiative forcing of climate change. In: *Climate Change 2001: The Scientific Basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change* (J. T. Houghton, Y. Ding, D. J. Griggs, M. Noguer, P. J. van der Linden, X. Dai, K. Maskell, and C. A. Johnson, eds.), 349–416. Cambridge: Cambridge University Press.

Roulet, N. T., 2000: Peatlands, carbon storage, greenhouse gases, and the Kyoto Protocol: prospects and significance for Canada. *Wetlands*, **20**, 605–615.

Rubec, C., 1996: The status of peatland resources in Canada. In: *Global Peat Resources*. E. Lappalainen, ed., 243–252. Jyskä, Finland: International Peat Society and Geological Survey of Finland.

Stallard, R. F., 1998: Terrestrial sedimentation and the carbon cycle: Coupling weathering and erosion to carbon burial. *Global Biogeochemical Cycles*, **12**, 231–257.

Strack, M., J. M. Waddington, and E.-S. Tuittila, 2004: Effect of water table drawdown on northern peatland methane dynamics: Implications for climate change. *Global Biogeochemical Cycles*, **18**, GB4003, doi:4010.1029/2003GB002209, 002004.

Turetsky, M. R., B. D. Amiro, E. Bosch, and J. S. Bhatti, 2004: Historical burn area in western Canadian peatlands and its relationship to fire weather indices. *Global Biogeochemical Cycles*, **18**, GB4014, doi:1029/2004GB002222, 002004.

Updegraff, K., S. D. Bridgham, J. Pastor, P. Weishampel, and C. Harth, 2001: Response of $CO_2$ and $CH_4$ emissions in peatlands to warming and water-table manipulation. *Ecological Applications*, **11**, 311–326.

Vann, C. D., and J. P. Megonigal, 2003: Elevated $CO_2$ and water depth regulation of methane emissions: Comparison of woody and non-woody wetland plant species. *Biogeochemistry*, **63**, 117–134.

Vile, M. A., S. D. Bridgham, R. K. Wieder, and M. Novák, 2003: Atmospheric sulfur deposition alters pathways of gaseous carbon production in peatlands. *Global Biogeochemical Cycles*, **17**, 1058–1064.

Wang, J. S., J. A. Logan, M. B. McElroy, B. N. Duncan, I. A. Megretskaia, and R. M. Yantosca, 2004: A 3-D model analysis of the slowdown and interannual variability in the methane growth rate from 1988 to 1997. *Global Biogeochemical Cycles*, **18**, GB3011, doi:101029/102003GB002180.

Watson, R. T., I. R. Noble, B. Bolin, N. H. Ravindranath, D. J. Verardo, and D. J. Dokken, 2000: IPCC Special Report on Land Use, Land-Use Change and Forestry. Cambridge, UK: Cambridge University Press.

Whiting, G. J., and J. P. Chanton, 1993: Primary production control of methane emissions from wetlands. *Nature*, **364**, 794–795.

Wylynko, D., ed., 1999: Prairie wetlands and carbon sequestration: assessing sinks under the Kyoto Protocol. Winnipeg, Manitoba, Canada: Institute for Sustainable Development, Ducks Unlimited Canada, and Wetlands International.

Zedler, J. B., and S. Kercher, in press: Wetland resources: status, trends, ecosystem services, and restorability. *Annual Review of Environmental Resources*.

Zhuang, Q., J. M. Melillo, D. W. Kicklighter, R. G. Prin, A. D. McGuire, P. A. Steudler, B. S. Felzer, and S. Hu, 2004: Methane fluxes between terrestrial ecosystems and the atmosphere at northern high latitudes during the past century: A restrospective analysis with a process-based biogeochemistry model. *Global Biogeochemical Cycles*, **18**, GB 3010, doi:3010.1029/2004GB002239.

**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 was limited by the level of classification used to define wetland categories with 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 (×10$^3$ km$^2$).** Historical refers to approximately 1800, unless otherwise specified

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).

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 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). 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), 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 salt marshes and mangroves 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). There are no reliable estimates of mud flat area for Mexico.

## 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 $ha^{-1}$ ($CV^2$ = 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 found soil carbon densities of 162 Mg C $ha^{-1}$ for FWMS wetlands in the conterminous United States and Mexico, which was used in this analysis.

However, some caution is necessary regarding the use of Gleysol or wet mineral soil carbon densities, as 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. 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

[2] CV is the "coefficient of variation," or 100 times the standard deviation divided by the mean.

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.** "Sequestration in current wetlands" refers to carbon sequestration in wetlands that currently exist; "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

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 Chimura *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 Alaska, and country-specific marsh or mangrove estimates were used for mudflats. Although Chimura *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. Chimura *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 (Twilley *et al*., 1992). Biomass was estimated from an empirical equation for aboveground biomass as a function of latitude (Twilley *et al.* 1992). 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$^o$). 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.** Positive flux numbers indicate a net flux into the atmosphere, whereas negative numbers indicate a net flux into the ecosystem

## 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 lower rates occuring 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 $m^{-2}$ $yr^{-1}$. However, we believe that these values cannot be used directly as estimates of carbon sequestration rates for of two reasons. First, it is likely that researchers preferentially choose wetlands with high sedimentation rates to study this process. Secondly, and more fundamentally, at a landscape scale a redistribution of sediments from uplands to wetlands represents no net carbon sequestration if the decomposition rate of carbon is the same in both environments. The carbon associated with sediments is likely relatively recalcitrant and often physically protected from decomposers by association with mineral soils. 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. Because of this reasoning, we somewhat arbitrarily reduced our calculated rates of carbon sequestration in FWMS wetlands by 75% to -34 g C $m^{-2}$ $yr^{-1}$, which still represents a substantial carbon sink.

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). 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 Chimura *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.

## 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 methane fluxes 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.** 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 used annual flux was the measured annual flux if that was available; otherwise, it was the calculated annual flux.

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

We have limited our focus on plant carbon fluxes to those processes that would result in the accumulation of plant carbon biomass on an interannual basis. 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 have used this lower value to estimate that U.S. wetland forests currently sequester -10.3 Mt C $yr^{-1}$.

We have assumed that the largely undisturbed forested wetlands of Alaska and Canada are at an approximate steady state in terms of biomass, with no interannual plant carbon accumulation. It is likely that plant carbon sequestration occurs largely as woody biomass, so we also assumed that non-forested wetlands have no interannual plant carbon accumulation.

## REFERENCES

Alford, D. P., R. D. Delaune, and C. W. Lindau, 1997: Methane flux from Mississippi River deltaic plain wetlands. *Biogeochemistry*, **37**, 227-236.

Armentano, T. B., and E. S. Menges, 1986: Patterns of change in the carbon balance of organic soil- wetlands of the temperate zone. *Journal of Ecology*, **74**, 755-774.

Aselmann, I., and P. J. Crutzen, 1989: Global distribution of natural freshwater wetlands and rice paddies, their net primary productivity, seasonality and possible methane emissions. *Journal of Atmospheric Chemistry*, **8**, 307-359.

Bartlett, K. B., and R. C. Harriss, 1993: Review and assessment of methane emissions from wetlands. *Chemosphere*, **26**, 261-320.

Batjes, N. H., 1996: Total carbon and nitrogen in the soils of the world. *European Journal of Soil Science*, **47**, 151-163.

Birdsey, R. A., 1992: Carbon storage and accumulation in United States forest ecosystems.General Technical Report WO-59 Washington, DC: USDA Forest Service.

Bridgham, S. D., C.-L. Ping, J. L. Richardson, and K. Updegraff, 2000: Soils of northern peatlands: Histosols and Gelisols. In: *Wetland Soils: Genesis, Hydrology, Landscapes, and Classification* (J. L. Richardson, and M. J. Vepraskas, eds.), 343-370. Boca Raton, FL: CRC Press.

Brown, M. J., G. M. Smith, and J. McCollum, 2001: Wetland forest statistics for the south Atlantic states.RB-SRS-062 Asheville, North Carolina: Southern Research Station, U.S. Forest Service.

Chmura, G. L., S. C. Anisfeld, D. R. Cahoon, and J. C. Lynch, 2003: Global carbon sequestration in tidal, saline wetland soils. *Global Biogeochemical Cycles*, **17**, 1111.

Cleary, J., N. T. Roulet, and T. R. Moore, 2005: Greenhouse gas emissions from Canadian peat extraction, 1990-2000: A life-cycle analysis. *Ambio*, **34**, 456-461.

Clymo, R. S., J. Turunen, and K. Tolonen, 1998: Carbon accumulation in peatland. *Oikos*, **81**, 368-388.

Cowardin, L. M., V. Carter, F. C. Golet, and E. T. LaRoe, 1979: Classification of wetlands and deepwater habitats of the United States.FWS/OBS-79/31 Washington, DC: Fish and Wildlife Service, US Department of the Interior.

Dahl, T. E., 1990: Wetland losses in the United States 1970's to 1980's.Washington, DC: Fish and Wildlife Service, U.S. Department of the Interior.

Dahl, T. E., 2000: Status and Trends of Wetlands in the Conterminous United States, 1986 to 1997.Washington, DC: Fish and Wildlife Service, U.S. Department of the Interior.

Dahl, T. E., and C. E. Johnson, 1991: Status and Trends of Wetlands in the Conterminous United States, Mid-1970's to Mid-1980's.have Washington, D.C.: U.S. Department of the Interior, Fish and Wildlife Service.

Davidson, I., R. Vanderkam, and M. Padilla, 1999: Review of wetland inventory information in North America.Canberra, Australia.

Ehhalt, D., M. Prather, F. Dentener, E. Dlugokencky, E. Holland, I. Isaksen, J. Katima, V. Kirchhoff, P. Matson, P. Midgley, and M. Wang, 2001: "Atmospheric chemistry and greenhouse gases." In *Climate Change 2001: The Scientific Basis*. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change (J. T. Houghton, Y. Ding, D. J. Griggs, M. Noguer, P. J. van der Linden, X. Dai, K. Maskell, and C. A. Johnson, eds.), 239-287. Cambridge: Cambridge University Press.

Eswaran, H., E. Van Den Berg, and J. Kimble, 1995: Global soil carbon resources. In: *Soils and Global Change* (R. Lal, J. Kimble, E. Levine, and B. A. Stewart, eds.), 27-43. Boca Raton, Florida: Lewis Publishers.

FAO, 1991: The Digitized Soil Map of the World.World Soil Resource Report, 64 Rome: Food and Agriculture Organization.

FAO-UNESCO, 1974: *Soil Map of the World (1:5,000,000)*. Paris: UNESCO.

Frayer, W. E., T. J. Monahan, D. C. Bowden, and F. A. Graybill, 1983: Status and Trends of Wetlands and Deepwater Habitats in the Conterminous United States, 1950s to 1970s. Fort Collins, Colorado: Dept. of Forest and Wood Sciences, Colorado State University.

Gorham, E., 1991: Northern peatlands: Role in the carbon cycle and probable responses to climatic warming. *Ecological Applications*, **1**, 182-195.

Group, N. W. W., 1988: Wetlands of Canada: Sustainable Development Branch, Environment Canada, Ontario, and Polyscience Publications, Montreal, Quebec.

Hall, J. V., W. E. Frayer, and B. O. Wilen, 1994: Status of Alaska Wetlands. Anchorage, Alaska: U.S. Fish and Wildlife Service.

Halsey, L. A., D. H. Vitt, and L. D. Gignac, 2000: *Sphagnum*-dominated peatlands in North America since the last glacial maximum: their occurence and extent. *The Bryologist*, **103**, 334-352.

Hanson, A. R., and L. Calkins, 1996: Wetlands of the Maritime Provinces: Revised Documentation for the Wetlands Inventory.Technical Report No. 267 Sackville, New Brunswick: Canadian Wildlife Service, Atlantic Region.

Johnston, C. A., 1991: Sediment and nutrient retention by freshwater wetlands: effects on surface water quality. *Critical Reviews in Environmental Control*, **21**, 491-565.

Joosten, H., and D. Clarke, 2002: Wise Use of Mires and Peatlands - Background Principles including a Framework for Decision-Making. Saarijärvi, Finland: International Mire Conservation Group and International Peat Society.

Kelly, C. A., J. W. M. Rudd, R. A. Bodaly, N. T. Roulet, V. L. St. Louis, A. Heyes, T. R. Moore, S. Schiff, R. Aravena, K. J. Scott, B. Dyck, R. Harris, B. Warner, and G. Edwards, 1997: Increase in fluxes of greenhouse gases and methyl mercury following flooding of an experimental reservoir. *Environmental Science & Technology*, **31**, 1334-1344.

Kivinen, E., and P. Pakarinen, 1981: Geographical distribution of peat resources and major peatland complex types in the world. *Annales Academiae Scientiarum Fennicae,* **Series A. III. 132**, 1-28.

Lappalainen, E., 1996: General review on world peatland and peat resources. In: *Global Peat Resources* (E. Lappalainen, ed., 53-56. Jyskä, Finland: International Peat Society and Geological Survey of Finland.

Maltby, E., and P. Immirzi, 1993: Carbon dynamics in peatlands and other wetland soils, regional and global perspectives. *Chemosphere*, **27**, 999-1023.

Malterer, T. J., 1996: Peat resources of the United States. In: *Global Peat Resources* (E. Lappalainen, ed., 253-260. Jyska, Finland.

Matthews, E., and I. Fung, 1987: Methane emission from natural wetlands: Global distribution, area, and environmental characteristics of sources. *Global Biogeochemical Cycles*, **1**, 61-86.

Meade, R. H., T. R. Yuzyk, and T. J. Day, 1990: Movement and storage of sediments in rivers of the United States and Canada. In: *Surface Water Hydrology, Geol. of N. Am., 0-1* (M. G. Wolman, and H. C. Riggs, eds.), 255-280. Boulder, CO: Geological Society of American.

Mendelssohn, I. A., and K. L. McKee, 2000: Saltmarshes and mangroves. In: *North American Terrestrial Vegetation* (M. G. Barbour, and W. D. Billings, eds.), 501-536. Cambridge, UK: Cambridge University Press.

Mitsch, W. J., and J. G. Gosselink, 1993: WetlandsNew York: Van Nostrand Reinhold.

Moore, T. R., and N. T. Roulet, 1995: Methane emissions from Canadian peatlands. In: *Soils and Global Change* (R. Lal, J. Kimble, E. Levine, and B. A. Stewart, eds.), 153-164. Boca Raton, FL: Lewis Publishers.

Moore, T. R., N. T. Roulet, and J. M. Waddington, 1998: Uncertainty in predicting the effect of climatic change on the carbon cycling of Canadian peatlands. *Climatic Change*, **40**, 229-245.

National Wetlands Working Group, 1988: *Wetlands of Canada*, Sustainable Development Branch, Environment Canada, Ontario, and Polyscience Publications, Montreal, Quebec.

NRCS, 1999: Soil Taxonomy: A Basic System of Soil Classification for Making and Interpreting Soil Surveys. Washington, DC: Natural Resources Conservation Service, U.S. Department of Agriculture.

Ovenden, L., 1990: Peat accumulation in northern wetlands. *Quaternary Research*, **33**, 377-386.

Rieger, S., D. B. Schoephoster, and C. E. Furbush, 1979: Exploratory Soil Survey of Alaska. Anchorage, Alaska: USDA Soil Conservation Service.

Robinson, S. D., and T. R. Moore, 1999: Carbon and peat accumulation over the past 1200 years in a landscape with discontinuous permafrost, northwestern Canada. *Global Biogeochemical Cycles*, **13**, 591-602.

Rubec, C., 1996: The status of peatland resources in Canada. In: *Global Peat Resources* (E. Lappalainen, ed., 243-252. Jyskä, Finland: International Peat Society and Geological Survey of Finland.

Spalding, M., F. Blasco, and C. Field, eds., 1997: World Mangrove Atlas. Okinawa, Japan: The International Society for Mangrove Ecosystems.

Spiers, A. G., 1999: Review of international/continental wetland resources. In: *Global Review of Wetland Resources and Priorities* (C. M. Finlayson, and A. G. Spiers, eds.).

Stallard, R. F., 1998: Terrestrial sedimentation and the carbon cycle: Coupling weathering and erosion to carbon burial. *Global Biogeochemical Cycles*, **12**, 231–257.

Tarnocai, C., 1998: The amount of organic carbon in various soil orders and ecological provinces in Canada. In: *Soil Processes and the Carbon Cycle* (R. Lal, J. M. Kimble, R. F. Follett, and B. A. Stewart, eds.), 81-92. Boca Raton, Florida: CRC Press.

Tarnocai, C., I. M. Kettles, and B. Lacelle, 2005: Peatlands of Canada. Ottawa.Ottawa, Canada: Agriculture and Agri-Food Canada, Research Branch.

Tolonen, K., and J. Turunen, 1996: Accumulation rates of carbon in mires in Finland and implications for climactic change. *Holocene*, **6**, 171-178.

Trumbore, S. E., and J. W. Harden, 1997: Accumulation and turnover of carbon in organic and mineral soils of the BOREAS northern study area. *Journal of Geophysical Research*, **102**, 28, 817-828, 830.

Turetsky, M. R., R. K. Wieder, L. A. Halsey, and D. Vitt, 2002: Current distribution and diminishing peatland carbon sink. *Geophysical Research Letters*, **29**, 10.1029/2001GL014000, 012002.

Turunen, J., N. T. Roulet, and T. R. Moore, 2004: Nitrogen deposition and increased carbon accumulation in ombrotrophic peatlands in eastern Canada. *Global Biogeochemical Cycles*, **18**, GB3002, doi:3010.1029/2003GB002154.

Twilley, R. R., R. H. Chen, and T. Hargis, 1992: Carbon sinks in mangroves and their implications to carbon budget of tropical coastal ecosystems. *Water, Air and Soil Pollution*, **64**, 265-288.

Vitt, D. H., L. A. Halsey, I. E. Bauer, and C. Campbell, 2000: Spatial and temporal trends in carbon storage of peatlands of continental western Canada through the Holocene. *Canadian Journal of Earth Sciences*, **37**, 683-693.

Vitt, D. H., L. A. Halsey, and S. C. Zoltai, 1994: The bog landforms of continental western Canada in relation to climate and permafrost patterns. *Arctic and Alpine Research*, **26**, 1-13.

Webb, R. S., and T. Webb III, 1988: Rates of sediment accumulation in pollen cores from small lakes and mires of eastern North America. *Quaternary Research*, **30**, 284-297.

WEC, 2001: Survey of Energy Resources. http://www.worldenergy.org/wec-geis/publications/reports/ser/peat/peat.asp

Werner, C., K. Davis, P. Bakwin, C. Yi, D. Hurst, and L. Lock, 2003: Regional-scale measurements of CH4 exchange from a tall tower over a mixed temperate/boreal lowland and wetland forest. *Global Change Biology*, **9**, 1251-1261.

West, A. E., P. D. Brooks, M. C. Fisk, L. K. Smith, E. A. Holland, C. H. Jaeger III, S. Babcock, R. S. Lai, and S. K. Schmidt, 1999: Landscape patterns of $CH_4$ fluxes in an alpine tundra ecosystem. *Biogeochemistry*, **45**, 243-264.

Wickland, K. P., R. G. Striegl, S. K. Schmidt, and M. A. Mast, 1999: Methane flux in subalpine wetland and unsaturated soils in the southern Rocky Mountains. *Global Biogeochemical Cycles*, **13**, 101-113.

Wilson, J. O., P. M. Crill, K. B. Bartlett, D. I. Sebacher, R. C. Harriss, and R. L. Sass, 1989: Seasonal variation of methane emissions from a temperate swamp. *Biogeochemistry*, **8**, 55-71.

Yavitt, J. B., 1997: Methane and carbon dioxide dynamics in *Typha latifolia* (L.) wetlands in central New York state. *Wetlands*, **17**, 394-406.

Yavitt, J. B., G. E. Lang, and A. J. Sexstone, 1990: Methane fluxes in wetland and forest soils, beaver ponds, and low-order streams of a temperate forest ecosystem. *Journal of Geophysical Research*, **95**, 22463-22474.

Yavitt, J. B., R. K. Wieder, and G. E. Lang, 1993: $CO_2$ and $CH_4$ dynamcis of a *Sphagnum*-dominated peatland in West Virginia. *Global Biogeochemical Cycles*, **7**, 259-274.

**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 | 9b | 698 |
| 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[q] | 0 | 5[c] | ND[r] | 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[s] | 1,706[s] | 23 | 11 | 19 | 3,166 |
| Global | | | | | | | |
| Current | 3,443[t] | | 2,289 to 2,341[u] | 22[v] | 181[w] | ND | ~6,000 |
| Historical | 3,880-4,086[x] | | 5,000[y] | ND | ND | ND | ~9,000[y] |

[a]Tarnocai *et al.* (2005).

[b]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 *et al.*, 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).

[o]Total historical wetland area from Dahl (1990) minus historical peatland area minus historical estuarine area.

[p]Davidson *et al.* (1999).

[q]Spiers (1999).

[r]ND indicates that no data are available.

[s]Assuming that historical proportion of peatlands to total wetlands in Mexico was the same as today.

[t]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).

[u]Matthews and Fung (1987); Aselmann and Crutzen (1989). For subsequent calculations, used the average of 2,315,000 $km^2$.

[v]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.

[w]Spalding *et al.* (1997).

[x]Maltby and Immirzi (1993). For subsequent calculations, used 4,000,000 $km^2$.

[y]Approximately 50% loss from Moser *et al.* (1996). For subsequent calculations, used an original global mineral-soil wetland area of 5,000,000 $km^2$.

**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 wetlands that currently exist; "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[f] | -5.1[g] | -0.1 | 0.0 | -1.2[d] | -25.5 |
| Oxidation in Former Wetlands | 0.2[h] | | 0.0[i] | 0.0[j] | 0.0 | 0.0 | 0.2 |
| Historical Loss in Sequestration Capacity | 0.0e | 0.2[f] | 6.5[g] | 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[k] | 6.2[k] | 26.0[L] | 0.0 | 0.0 | 0.1 | 41.7 |
| Sequestration in Current Wetlands | -1.1[e] | -0.8[f] | -18.0[g] | -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[m] | 5.1[L] | 0.4 | 0.1 | 0.1 | 19.7 |
| Sequestration in Current Wetlands | 0 | -11.6[n] | -10.1[g] | -3.9 | -0.5 | -0.5 | -26.6 |
| Oxidation in Former Wetlands | 0 | 18.0[o] | 0.1[i] | 0.0 | 0.0 | 0.0 | 18.1 |
| Historical Loss in Sequestration Capacity | 0 | 2.1[n] | 14.5[g] | 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[m] | 0.3[L] | 0.0 | 0.1 | ND* | 1.9 |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| Sequestration in Current Wetlands | 0 | -1.6[p] | -0.7[g] | 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 | | 0.1 | 0.0 | 0.0 | 0.0 | 18.3 |
| 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 | 234 to 679[q] | | 46[r] | 0.4[s] | 5.0[s] | ND | 286 to 730 |
| Sequestration in Current Wetlands | -40 to -70[t] | | -75[g] | -4.6[s] | -38.0[s] | ND | -158 to -188 |
| Oxidation in Former Wetlands | 160 to 250[u] | | ND | 0 | 0 | 0 | 160 to 250 |
| Historical Loss in Sequestration Capacity | 16[u] | | 87[g] | 0.5[v] | 12.7[w] | ND | 116 |
| Change in Flux From Wetland Conversions | 176 to 266[u] | | > 87[x] | 0.5 | 12.7 | ND | 276 to 366 |

*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]Soil carbon accumulation rate of 0.13 Mg C $ha^{-1}$ $yr^{-1}$ (see Chapter 12 in this report).

[f]Carbon accumulation rate of 0.19 Mg C $ha^{-1}$ $yr^{-1}$. This is an average value of the reported range of long-term apparent accumulation rate of 0.05–0.35 (Ovenden, 1990, Maltby and Immirzi, 1993; Trumbore and Harden, 1997; Vitt *et al.*, 2000; Turunen *et al.*, 2004).

[g]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). It was assumed that the actual rate was 25% of the potential rate because of bias in choosing sampling sites and considerations of the redistribution of sediment due to erosion without a change in the sequestration rate on a landscape scale.

[h]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).

[i]Assumed that the oxidized soil C is lost over 50 yr.

[j]Assumed that conversion of tidal systems is caused by fill and results in burial and preservation of SOM define SOM rather than oxidation.

[k]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).

[L]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.

[m]Assumed soil carbon density of 1,500 Mg C $ha^{-1}$.

[n]Webb and Webb (1988).

[o]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.

[p]Using peat accumulation rate of 1.6 Mg C $ha^{-1}$ (range 1.0–2.25) (Maltby and Immirzi, 1993).

[q]Gorham (1991), Maltby and Immirzi (1993), Eswaran *et al.* (1995), Batjes (1996), Lappalainen (1996), Joosten and Clarke (2002).

[r]Soil carbon density of 199 Mg C $ha^{-1}$ (Batjes, 1996).

[s]Chmura *et al.* (2003).

[t]Joosten and Clarke (2002). Using the peatland estimate in Table 13A-1 and a carbon accumulation rate of 0.19 Mg C ha-1 yr-1, we calculate a global flux of –65 Mt C $yr^{-1}$ in peatlands.

[u]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) 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.

[v]Assumed that global rates approximate the North America rate because most salt marshes inventoried are in North America.

[w]Assumed 25% loss globally since the late 1800s.

[x]> 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.

| | Permafrost peatlands | Non-perma-frost peatlands | Mineral-soil freshwater | Salt marsh | Mangrove | **Total** |
|---|---|---|---|---|---|---|
| Canada | | | | | | |
| Pool Size in Current Wetlands | 1.4[a] | | 0.3[b] | 0.0[c] | 0.0 | 1.7 |
| Sequestration in Current Wetlands | 0.0 | ND* | | 0.0 | 0.0 | 0.0 |
| Alaska | | | | | | |
| Pool Size in Current Wetlands | 0.4[a] | | 1.1[d] | 0.0 | 0.0 | 1.5 |
| Sequestration in Current Wetlands | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| Conterminous United States | | | | | | |
| Pool Size in Current Wetlands | 0.0 | 1.5[d] | | 0.0 | 0.0 | 1.5 |
| Sequestration in Current Wetlands | 0.0 | -10.3[e] | | 0.0 | 0.0 | -10.3 |
| Mexico | | | | | | |
| Pool Size in Current Wetlands | 0.0 | 0.0[b] | 0.0[b] | 0.0 | 0.1 | 0.1 |
| Sequestration in Current Wetlands | 0.0 | ND | ND | 0.0 | ND | 0.0 |
| North America | | | | | | |
| Pool Size in Current Wetlands | 4.8 | | | 0.0 | 0.1 | 4.9 |
| Sequestration in Current Wetlands | 0.0 | -10.3 | | 0.0 | ND | -10.3 |
| Global | | | | | | |
| Pool Size in Current Wetlands | 6.9[b] | | 4.6[b] | 0.0[f] | 4.0[g] | 15.5 |
| Sequestration in Current Wetlands | 0.0 | ND | ND | 0.0 | ND | ND |

*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 Dahl (2000).

[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[a,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 | 92–237[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).

[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 used annual flux was the measured annual flux if that was available; otherwise, it was the calculated annual flux

| 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 | Kelley *et al.* (1995) |
| Banks Tidal | NC | C, E | FW | 20.1 | | | 7.3 | 7.3 | Kelley 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

**Lead Author: Diane E. Pataki[1]**

**Contributing Authors: Alan S. Fung,[2] David J. Nowak,[3] E. Gregory McPherson,[3] Richard V. Pouyat,[3] Nancy Golubiewski,[4] Christopher Kennedy,[5] Patricia Romero Lankao,[6] and Ralph Alig[3]**

[1]University of California, Irvine; [2]Dalhousie University; [3]USDA Forest Service; [4]Landcare Research; [5]University of Toronto; [6]UAM-Xochimilco

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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.

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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 20% 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. 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 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.

## REFERENCES

Agarwal C, Green GM, Grove JM, Evans TP, Schweik CM (2000) A review and assessment of land-use change models: Dynamics of space, time and human choice. CIPEC Collaborative Report Series No. 1, Center for the Study of Institutions, Populations, and Environmental Change, Indiana University and the USDA Forest Service.

Akbari H (2002) Shade trees reduce building energy use and $CO_2$ emissions from power plants. Environmental Pollution 116:S119–S126.

Akbari H, Konopacki S (2005) Calculating energy-saving potentials of heat-island reduction strategies. Energy Policy 33:721–756.

Akbari H, Kurn DM, Bretz SE, Hanford JW (1997) Peak power and cooling energy savings of shade trees. Energy and Buildings 25:139–148.

Akbari H, Taha H (1992). The impact of trees and white surfaces on residential heating and cooling energy use in four Canadian cities. Energy 17:141–149.

Alig RJ, Kline JD, Lichtenstein M (2004) Urbanization on the U.S. landscape: Looking ahead in the 21st century. Landscape and Urban Planning 69:219–234.

Alig RJ, Plantinga A, Ahn S, Kline JD (2003) Land use changes involving forestry for the United States: 1952 to 1997, with projections to 2050. USDA Forest Service, Pacific Northwest Research Station, Portland, OR. Gen. Tech. Report 587.

Anderson WP, Kanaroglou PS, Miller EJ (1996). Urban form, energy and the environment: A review of issues, evidence and policy. Urban Studies 33:7–35.

Betsill MM (2001) Mitigating climate change in U.S. cities: opportunities and obstacles. Local Environment 6:393–406.

CEC (2001). The North American Mosaic: A State of the Environment Report. Commission for Environmental Cooperation, Montreal, Canada.

Center for International Earth Science Network (CIESIN) CU, International Food Policy Research Institute (IPFRI), the World Bank, Centro Internacional de Agricultura Tropical (CIAT) (2004) Global Rural-Urban Mapping Project (GRUMP): Urban extents., http://sedac.ciesin.columbia.edu/gpw, last accessed 3 Dec 2005.

Cifuentes L, Borja-Aburto VH, Gouveia N, Thurston G, Davis DL (2001) Assessing health benefits of urban air pollution reductions associated with climate change mitigation (2000–2020): Santiago, Sao Paulo, Mexico City, and New York City. Environmental Health Perspectives 109:419–425.

Decker EH, Elliot S, Smith FA, Blake DR, Rowland FS (2000) Energy and material flow through the urban ecosystem. Annual Review of Energy and the Environment 25:685–740.

Easterling WE, Polsky C, Goodin DG, Mayfield MW, Muraco WA, Yarnal B (2003) Changing places and changing emissions: comparing local, state, and United States emissions. In: Team AoAGGR (ed) Global change and local places: Estimating, understanding and reducing greenhouse gases. Cambridge University Press, Cambridge, pp 143–157.

EPA (2000) Projecting land-use change: A summary of models for assessing the effects of community growth and change on land-use patterns. U.S. Environmental Protection Agency, Washington, D.C. EPA/600/R-00/098.

Ewing R, Pendall R, Chen D (2003) Measuring sprawl and its transportation impacts. Transportation Research Record 1831:175–183.

Folke C, Jansson A, Larsson J, Costanza R (1997) Ecosystem appropriation by cites. Ambio 26:167–172.

Golubiewski NE (2006) Urbanization transforms prairie carbon pools: Effects of landscaping in Colorado's Front Range. Ecological Applications 16(2): 555-51.

Gomez-Ibanez JA (1991) A global view of automobile dependence: review of "'Cities and automobile dependence: An international sourcebook" by P.W.G. Newman and J.R. Kenworthy". Journal of the American Planning Association 57:376–379.

Gonzalez GA (2005) Urban sprawl, global warming and the limits of ecological modernisation. Environmental Politics 14:344–362.

Gordon P, Richardson HW (1989) Gasoline consumption and cities: A reply. Journal of the American Planning Association 55:342–346.

Grimm NB, Grove JM, Pickett STA, Redman CL (2000) Integrated approaches to long-term studies of urban ecological systems. Bioscience 50:571–584.

Grimmond CSB, King TS, Cropley FD, Nowak DJ, Souch C (2002) Local-scale fluxes of carbon dioxide in urban environments: methodological challenges and results from Chicago. Environmental Pollution 116:S243–S254.

Grimmond CSB, Salmond JA, Oke TR, Offerle B, Lemonsu A (2004) Flux and turbulence measurements at a densely built-up site in Marseille: heat, mass (water and carbon dioxide), and momentum. Journal of Geophysical Research–Atmospheres 109:DOI: 10.1029/2004JD004936.

Huang YJ, Akbari H, Taha H, Rosenfeld H (1987) The potential of vegetation in reducing summer cooling loads in residential buildings. Journal of Climate and Applied Meteorology 26:1103–1116.

Hunt JD, Kriger DS, Miller EJ (2005) Current operation urban land-use-transport modelling frameworks: A review. Transport Reviews 25:329–376.

ICLEI (1993) Cities for climate protection: An international campaign to reduce urban emissions of greenhouse gases. http://www.iclei.org/index.php?id=1651, last accessed 30 Mar 2006.

ICLEI (2000) Best practices for climate protection: A local government guide. ICLEI, Berkeley, CA.

Imhoff ML, Bounoua L, DeFries RS, Lawrence WT, Stutzer D, Compton JT, Ricketts T (2004) The consequences of urban land transformations on net primary productivity in the United States. Remote Sensing of the Environment 89:434–443.

Ironmonger DS, Aitken CK, Erbas B (1995) Economies of scale in energy use in adult-only households. Energy Economics 17:301–310.

Jaccard M, Failing L, Berry T (1997) From equipment to infrastructure: community energy management and greenhouse gas emission reduction. Energy Policy 25:1065–1074.

Kaye JP, Burke IC, Mosier AR, Guerschman JP (2004) Methane and nitrous oxide fluxes from urban soils to the atmosphere. Ecological Applications 14:975–981.

Kaye JP, McCulley RL, Burke IC (2005) Carbon fluxes, nitrogen cycling, and soil microbial communities in adjacent urban, native and agricultural ecosystems. Global Change Biology 11:575–587.

Kenworthy JR, Newman PWG (1990) Cities and transport energy: lessons from a global survey. Ekistics 34:258–268.

Koerner B, Klopatek J (2002) Anthropogenic and natural $CO_2$ emission sources in an arid urban environment. Environmental Pollution 116:S45–S51.

Kousky C, Schneider SH (2003) Global climate policy: will cities lead the way? Climate Policy 3:359–372.

Lin JC, Gerbig C, Wofsy SC, Andrews AE, Daube BC, Grainger BC, Stephens BB, Bakwin PS, Hollinger DY (2004) Measuring fluxes of trace gases at regional scales by Lagrangian observations: Application to the $CO_2$ budget and rectification airborne (COBRA study). Journal of Geophysical Research–Atmospheres 109:DOI:10.1029/2004JD004754.

Liu J, Daily GC, Ehrich PR, Luck GW (2003) Effects of household dynamics on resource consumption and biodiversity. Nature 421:530–533.

MacKellar FL, Lutz W, Prinz C, Goujon A (1995) Population, households, and $CO_2$ emissions. Population and Development Review 21:849–865.

McPherson EG, Simpson JR, Peper PF, Maco SE, Xiao Q (2005) Municipal forest benefits and costs in five U.S. cities. Journal of Forestry In press.

Mindali O, Raveh A, Saloman I (2004) Urban density and energy consumption: A new look at old statistics. Transportation Research Record 38A:143–162.

NAHB (2005) Housing facts, figures and trends. National Association of Home Builders, Washington, DC.

Nemitz E, Hargreaves K, McDonald AG, Dorsey JR, Fowler D (2002) Mictrometeorological measurements of the urban heat budget and $CO_2$ emissions on a city scale. Environmental Science and Technology 36:3139–3146.

Newman PWG (1999) Sustainability and cities: extending the metabolism model. Landscape and Urban Planning 44:219–226.

Nowak DJ, Crane DE (2002) Carbon storage and sequestration by urban trees in the USA. Environmental Pollution 116:381–389.

Nowak DJ, Walton JT, Dwyer JF, Kaya LG, Myeong S (2005) The increasing influence of urban environments on U.S. forest management. Journal of Forestry In press.

Oke TR (1989) The micrometeorology of the urban forest. Phil. Trans. R. Soc. Lond. B 324:335–349.

Pataki DE, Alig RJ, Fung AS, Golubiewski NE, Kennedy CA, McPherson EG, Nowak DJ, Pouyat RV, Romero Lankao P (2006a) Urban ecosystems and the North American carbon cycle. Global Change Biology In press.

Pataki DE, Bowling DR, Ehleringer JR (2003) The seasonal cycle of carbon dioxide and its isotopic composition in an urban atmosphere: anthropogenic and biogenic effects. Journal of Geophysical Research 108:4735.

Pataki DE, Bowling DR, Ehleringer JR, Zobitz JM (2006b) High resolution monitoring of urban carbon dioxide sources. Geophysical Research Letters 33, L03813, doi:10.1029/2005GL024822.

Pickett STA, Cadenasso ML, Grove JM, Nilon CH, Pouyat RV, Zipperer WC, Costanza R (2001) Urban ecological systems: linking terrestrial ecological, physical, and socioeconomic components of metropolitan areas. Annual Review of Ecology and Systematics 32:127–157.

Pouyat R, Groffman P, Yesilonis I, Hernandez L (2002) Soil carbon pools and fluxes in urban ecosystems. Environmental Pollution 116:S107–S118.

Pouyat RV, Yesilonis I, Nowak DJ (2006) Carbon storage by urban soils in the USA. Journal of Environmental Quality In press.

Qian Y, Follet RF (2002) Assessing soil carbon sequestration in turfgrass systems using long-term soil testing data. Agronomy Journal 94:930–935.

Romero Lankao P, Lopez H, Rosas A, Gunther G, Correa Z (2004) Can cities reduce global warming? Urban development and the carbon cycle in Latin America. IAI, UAM-X, IHDP, GCP, Mexico.

Sahely HR, Dudding S, Kennedy CA (2003) Estimating the urban metabolism of Canadian cities: Greater Toronto Area case study. Canadian Journal of Civil Engineering 30:468–483.

Soegaard H, Moller-Jensen L (2003) Toward a spatial $CO_2$ budget of metropolitan region based on textural image classification and flux measurements. Remote Sensing of the Environment 87:283–294.

Taha H (1997) Urban climates and heat islands: albedo, evapotranspiration, and anthropogenic heat. Energy and Buildings 25:99–103.

United Nations (2002) Demographic Yearbook, http://unstats.un.org/unsd/demographic/products/dyb/default.htm.

United Nations (2004) World urbanization prospects: The 2003 Revision. U.N. Dept. of Economic and Social Affairs, Population Division, New York. E.04.XIII.6.

United Nations Habitat (2003) Global Observatory Database, http://www.unchs.org/programmes/guo, last accessed 10 Nov 2005.

Warren-Rhodes K, Koenig A (2001) Ecosystem appropriation by Hong Kong and its implications for sustainable development. Ecological Economics 39:347–359.

West JJ, Osnaya P, Laguna I, Martinez J, Fernandez A (2004) Co-control of urban air pollutants and greenhouse gases in Mexico City. Environmental Science and Technology 38:3474–3481.

**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, Lakes and Rivers

**Lead Authors: Francisco P. Chavez[1] and Taro Takahashi[2]**

**Contributing Authors: Wei-Jun Cai,[3] Gernot Friederich,[1] Burke Hales,[4] Rik Wanninkhof,[5] and Richard A. Feely[6]**

[1]Monterey Bay Aquarium Research Institute, [2]Lamont-Doherty Earth Observatory of Columbia University, [3]University of Georgia, [4]Oregon State University, [5]Atlantic Oceanographic and Meteorological Laboratory, NOAA, [6]Pacific Marine Environmental Laboratory, NOAA

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## KEY FINDINGS

- The global oceans currently take up between 1.3 and 2.3 Gt C $yr^{-1}$ from the atmosphere.
- The carbon budgets of ocean margins (coastal regions) are not 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 carbon stored in the coastal regions of North America.
- New observations reveal that on average, nearshore waters surrounding North America are neither a source nor a sink 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 $yr^{-1}$.
- With the exception of one or two time-series sites, almost nothing is known about historical trends in sea-air 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.
- There are no existing projections of whether North America's coastal oceans will remain a source of $CO_2$ in the future or become a sink.
- 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.

- Highly variable sea-air $CO_2$ 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.

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## INVENTORIES (STOCKS AND FLUXES, QUANTIFICATION)

This chapter focuses on the role that aquatic systems play in modulating atmospheric carbon dioxide ($CO_2$). The chapter quantifies water-atmosphere $CO_2$ fluxes and considers how the underlying stocks and rate processes affect them. 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 transformations in aquatic systems, the limited data on the transformations, and space considerations have led to the focus on water-atmosphere $CO_2$ fluxes. Aquatic systems, primarily the oceans, act as a huge reservoir for inorganic carbon, containing about 50 times as much $CO_2$ as the atmosphere, and atmospheric concentration of $CO_2$ would be much higher in the absence of current ocean processes.

The ocean's biological pump converts inorganic carbon in the upper ocean to organic particulate carbon by photosynthesis, transports the organic carbon from the surface by sinking, and therefore plays a critical role in removing atmospheric $CO_2$ (Gruber and Sarmiento, 2002) in combination with physical and chemical processes. The net sea-air $CO_2$ flux over the global ocean appears to be well constrained to be 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 atmosphere is well mixed and nearly homogenous. The large spatial variability in sea-air $CO_2$ 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 sea-air flux of $CO_2$ in coastal waters surrounding the North American continent.

**Table 15-1. Climatological mean distribution of the net sea-air $CO_2$ 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 (wind speed)$^2$ dependence for sea-air 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 $yr^{-1}$

**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).

## 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 sea-air $CO_2$ 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 into the California Current and into Monterey Bay. Phytoplankton growing in the upwelled water use $CO_2$ as a carbon source, and $CO_2$ is drawn to low levels in those areas. Phytoplankton carbon eventually sinks or is subducted below the euphotic zone, where it decays, elevating the $CO_2$ levels of subsurface waters. Where the level of surface $CO_2$ is higher than the level of atmospheric $CO_2$, diffusion drives $CO_2$ into the atmosphere. Conversely, where the level of surface $CO_2$ is lower than that of atmospheric$CO_2$, diffusion drives $CO_2$ into the ocean. The net 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 sea-air $CO_2$ 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 sea-air $CO_2$ flux depends on quantities that are not well constrained, the mass balance provides widely varying results.

## 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) 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 (Cai *et al.*, 2003). 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 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).**

## Synthesis of Available North American Sea-Air 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. $pCO_2$ 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 sea-air $pCO_2$ difference (delta $pCO_2$) 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 sea-air $CO_2$ 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 sea-air $CO_2$ 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 sea-air $pCO_2$ 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 sea-air $CO_2$ 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 sea-air gas transfer rate by Wanninkhof (1992). The ± uncertainties represent one standard deviation.

**Table 15-3. Climatological mean annual sea-air $CO_2$ 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.

The offshore patterns follow the same general trend found in the global 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 sea-air $CO_2$ 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 sea-air $pCO_2$ 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 sea-air $CO_2$ 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 sea-air $CO_2$ 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 sea-air $CO_2$ 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. Sea-air $CO_2$ 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 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 sea-air $CO_2$ 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 $CO_2$. The open oceans become more intense $CO_2$ sinks toward higher latitudes, especially north of 35°N (Figure 15-3B). Between 15°N and 45°N, the open ocean waters are a $CO_2$ sink (Takahashi *et al.*, 2002), whereas the nearshore waters are a $CO_2$ source. Accordingly, in contrast to the Pacific coast, the latitude where Atlantic coastal waters become a $CO_2$ sink is located further north than that for the open ocean fluxes. 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. These waters have salinities around 20 and 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 the 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 draw-down 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 (sea-air $pCO_2$ differences ranging from –80 to –180 μatm) with little changes between the coastal and open ocean areas. The sea-air $CO_2$ flux during winter months is not known.

## 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 sea-air 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 results do not include some portions of the Arctic Sea, Bering Sea, Gulf of Alaska, Gulf of Mexico, or Caribbean Sea because of insufficient data. 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 sea-air $CO_2$ 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.

## 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 sea-air $CO_2$ 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 sea-air fluxes of $CO_2$.

The sea-air 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.*, in press) 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) but so is photosynthetic uptake of $CO_2$ (Chavez *et al*. 2002) so the net effect of climate variability and change on sea-air fluxes remains uncertain. What is certain is that the biological, chemical and physical processes controlling the sea-air fluxes of $CO_2$ are strongly affected by natural and anthropogenic change and that efforts to track them need to be considered in global carbon management plans.

## 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. Iron fertilization would be an economical alternative, but 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 sea-air $CO_2$ 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 sea-air $CO_2$ flux events may introduce errors in continental $CO_2$ fluxes calculated by atmospheric inversion methods. Achieving a comprehensive understanding of the carbon cycle in waters surrounding the North American continent will require development of advanced technologies and sustained 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.

## REFERENCES

Bakker, D. C. E., Watson, A. J. and Law, C. S. (2001). Southern Ocean iron enrichment promotes inorganic carbon drawdown. Dee-Sea Res. II, 48, 2483–2507.

Battle, M., Bender, M. L., Tans, P. P., White, J. W. C., Ellis, J. T., Conway, T. and Francey, R. J. (2000). Global carbon sinks and their variability inferred from atmospheric $O_2$ and $\delta^{13}C$. *Science*, 287, 2467–2470.

Bender, M. L., Ho, D. T., Hendricks, M. B., Mika, R., Bazttle, M. O., Tans, P. P., Conway, T. J., Sturtevant, B. and Cassar, N. (2005). Atmospheric $O_2/N_2$ changes, 1993–2002: Implications for the partitioning of fossil fuel $CO_2$ sequestration. Glob. Biogeochem. Cycles, 19, GB4017, doi:10.1029/2004GB002410.

Benner, R. and Opsahl, S. (2001). Molecular indicators of the sources and transformations of dissolved organic matter in the Mississippi River plume. Organic Geochem., 32, 597–611.

Boehme, S.E., Sabine, C.L. and Reimers, C.E. (1998) $CO_2$ fluxes from a coastal transect: a time-series approach. Marine Chemistry, 63, 49–67.

Borges, A.V. (2005) Do we have enough pieces of the jigsaw to integrate $CO_2$ fluxes in the Coastal Ocean?, *Estuaries*, *28*, 3–27.

Borges, A.V., B. Delille, and M. Frankignoulle (in press) Budgeting sinks and sources of $CO_2$ in the coastal ocean: Diversity of ecosystems counts.

Boyd, P. W., *et al.* (2000). A mesoscale phytoplankton bloom in the polar Southern Ocean stimulated by iron fertilization. Nature, 407, 695–702.

Brewer, P.G. (2003) Direct Injection of Carbon Dioxide into the Oceans. In: "The Carbon Dioxide Dilemma: Promising Technologies and Policies." National Academies Press, pp. 43–51

Cai, W.-J. (2003). Riverine inorganic carbon flux and rate of biological uptake in the Mississippi River plume. Geophys. Res. Lett., 30, 1032.

Cai, W.-J., Wang, Z. A. and Wang, Y.-C. (2003). The role of marsh-dominated heterotrophic continental margins in transport of $CO_2$ between the atmosphere, the land-sea interface and the oceans. Geophys. Res. Letters, 30, No. 16, 1849, doi"10.1029/2003GL017633, 2003.

Cai, W. J., and M. Dai (2004), Comment on "Enhanced open ocean storage of $CO_2$ from shelf sea pumping", Science, *306*, 1477c.

Chavez, F. P., Strutton, P. G., Friederich, G. E., Feely, R. A., Feldman, G. C., Foley, D. G. and McPhaden, M. J. (1999). Biological and chemical response of the equatorial Pacific Ocean to 1997–98 El Niño. Science, 286, 2126–2131.

Chavez, F. P., Pennington, J. T., Castro, C. G., Ryan, J. P., Michisaki, R. M., Schlining, B., Walz, P., Buck, K. R., McFayden, A., and Collins, C. A. (2002). Biological and chemical consequences of the 1997–98 El Niño in central California waters. *Progress in Oceanography*, *54*, 205–232.

Chavez, F. P., Ryan, J., Lluch-Cota, S. and Miguel, N. C. (2003). From anchovies to sardines and back: Multidecadal change in the Pacific Ocean. *Science, 299,* 217–221.

Chisholm, S.W., P.G. Falkowski and J. Cullen. (2001). Discrediting ocean fertilization, Science, 294, 309–310.

Codispoti, L. A. and Friederich, G. E. (1986) Variability in the inorganic carbon system over the southeastern Bering Sea shelf during the spring of 1980 and spring-summer 1981. Continental Shelf Research, 5, 133–160.

Coale, K. H. *et al.* (2004). Southern Ocean iron enrichment experiment: Carbon cycling in high- and low-Si waters. Science, 304, 408–414.

DaSilva, A., Young, C., and Levitus, S. (1994). *Atlas of Marine Surface Data 1994*, NOAA Atlas NESDIS 6, U.S. Department of Commerce, Washington, D.C.

DeGrandpre, M.D., Hammar, T.R., Wallace, D.W.R. and Wirick, C.D. (1997) Simultaneous mooring-based measurements of seawater $CO_2$ and $O_2$ off Cape Hatteras, North Carolina. Limnology and Oceanography, 42, 21–28.

DeGrandpre, M.D., Olbu., G. J., Beatty, C.M., Hammar, T. R. (2002) Air-sea $CO_2$ fluxes on the U.S. Middle Atlantic Bight. Deep-Sea Research II, 49, 4355–4367.

Doney, S. C., Anderson, R., Bishop, J., Caldeira, K., Carlson, C., Carr, M.-E., Feely, R., Hood, M., Hopkinson, C., Jahnke, R., Karl, D., Kleypas, J., Lee, C., Letelier, R., McClain, C., Sabine, C., Sarmiento, J., Stephens, B., and Weller, R. (2004). *Ocean Carbon and Climate Change (OCCC): An Implementation Strategy for U.S. Ocean Carbon Cycle Science*, UCAR, Boulder, CO, 108 pp.

Ducklow, H. W., McCallister, S. L. (2004). The biogeochemistry of carbon dioxide in the coastal oceans. In "THE SEA" Vol.. 13, A. R. Robinson, and K. H. Brink editors, vol. 13, 269–315, John Wiley & Sons, New York.

Feely, R. A., Boutin, J., Coasca, C. E., Dandonneau, Y., Etcheto, J., Inoue, H., Ishii, M., LeQuere, C., Mackey, D. J., McPhaden, M., Metzl, N., Poisson, A. and Wanninkhof, R. (2002). Seasonal and interannual variability of $CO_2$ in the equatorial Pacific. Deep-Sea Res., II 49, 2443–2469.

Feely, R. A., Takahashi, T., Wanninkhof, R., McPhaden, M. J., Cosca, C. E., Sutherland, S. C. and Carr, M-E. (in press). Decadal variability of the air-sea CO2 fluxes in the equatorial Pacific Ocean. Journal of Geophysical Research.

Friederich, G.E., P.G. Brewer, R. Herlein, and F.P. Chavez (1995). Measurement of sea surface partial pressure of $CO_2$ from a moored buoy. Deep-Sea Research, 42, 1175–1186.

Friederich, G., P. Walz, M. Burczynski and F.P. Chavez (2002) Inorganic Carbon in the Central California Upwelling System During the 1997–1999 El Niño -La Niña Event. Progress in Oceanography, **54**, 185–204.

Gattuso, J. M., Frankignoulle, M. and Wollast, R. (1998). Carbon and carbonate metabolism in coastal aquatic ecosystem. Annual Review of Ecology and Systematics, 29, 405–434.

Gervais, F., U. Riebesell and M.Y. Gorbunov (2002) Changes in primary productivity and chlorophyll a in response to iron fertilization in the Southern Polar Frontal Zone. Limnol. Oceanogr., 47, 1324.

Gruber, N. and Sarmiento, J. L. (2002). Large –scale biogeochemical-physical interactions in elemental cycles. In *The Sea*, Vol. 12, A. R. Robinson, J. McCarthy and B. J. Rothschild editors, 337–399. John Wiley & Sons, New York.

Hales, B. and Takahashi, T. (2004). High-resolution biogeochemical investigation of the Ross Sea, Antarctica, during the AESOPS (U. S.JGOFS) Program . Global Biogeochem. Cycles, Vol. 18, No. 3, GB3006, doi. 10.1029/2003GB002165

Hales, B., Takahashi, T. and Bandstra, L. (2005). Atmospheric $CO_2$ uptake by a coastal upwelling system. Global Biogeochem. Cycles, 19. doi.10.1029/2004GB002295.sinks. Global Biogeocehm. Cycles, 18, GB1010, doi:10.1029/2003GB002111.

Hare, S. R. and Mantua, N. J. (2000). Empirical evidence for North Pacific regime shifts in 1977 and 1989. Progress in Oceanogr., 47, 103–145.

Hedges, J. I., R. G. Keil, and R. Benner (1997) What happens to terrestrial organic matter in the ocean?, Organic Geochemistry, *27*, 195–212.

Keeling, R. F. and Garcia, H. The change in oceanic $O_2$ inventory associated with recent global warming. Proc. U. S. National Acad. Sci., 99, 7848–7853, 2002.

Körtzinger, A. (2003) A significant CO2 sink in the tropical Atlantic Ocean associated with the Amazon river plume, Geophysical Research Letters, *30*, 2287, doi:10.1029/2003GL018841.

Liu, K. K., Iseki, K., Chao, S.-Y. (2000). Continental margin carbon fluxes. In “The Changing Ocean Carbon cycle”. R. Hansen, H. W. Ducklow and J. G. field editors, Cambridge University Press, Cambridge, UK, pp. 187–239.

Lohrenz, S. E., Dagg, M. J. and Whitledge, T. E. (1999). Nutrients, irradiance, and mixing as factors regulating primary production in coastal water impacted by the Mississippi River plume. Continental Shelf Res., 19, 1113–1141.

Martin, J.H. (1990) Glacial-interglacial $CO_2$ change: The iron hypothesis. Paleoceanography, 5, 1–13.

Millero, F. J., Hiscock, W.T., Huang, F., Roche, M., and Zhang, J-Z. (2001) Seasonal variation of the carbonate system in Florida Bay. Bulletin of Marine Science 68, 101–123.

Park, P.K., Gordon, L.I. and Alvarez-Borrego, S. (1974) The carbon dioxide system of the Bering Sea, In Oceanography of the Bering Sea, D.W. Hood editor, Occasional Publication No. 2, Institute of Marine Science, University of Alaska, Fairbanks, AK.

Patra, P. K., Maksyutov, S., Ishizawa, M., Nakazawa, T., Takahashi, T. and Ukita, J. (2005). Interannual and decadal changes in the sea-air $CO_2$ flux from atmospheric $CO_2$ inverse modeling. Glob. Biogeochem. Cycles, 19, GB4013, doi:10.1029/2004GB002257, 2005

Pennington, J.T., Castro, C.G., Collins, C.A., Evans, W.W. IV, Friederich, G.E., Michisaki, R.P., Chavez, F.P. (in press) A carbon budget for the northern and central California coastal upwelling system. Continental Margins Task Team, The Synthesis Book. Chapter 2.2, California Current System. 32 mss. pp.

Quay, P., Sommerup, R., Westby, T., Sutsman, J. and McNichol, A. (2003). Changes in the $^{13}C/^{12}C$ of dissolved inorganic carbon in the ocean as a tracer of anthropogenic $CO_2$ uptake. Glob. Biogeochem. Cycles, 17, No. 1, DOI 10.1029/2001GB001817.

Sabine, C. L., Feely R. A., Gruber, N., Key, R. M., Lee, K., Bullister, J. L., Wanninkhof, R., Wong, C. S., Wallace, D. W. R., Tilbrook, B., Peng, T-H, Kozyr, A., Ono, T. and Rios, A. F. (2004). The oceanic sink for anthropogenic $CO_2$. Science, 305, 367–371.

Sarmiento, J. L. and Sundquist, E. T. (1992). Revised budget for the oceanic uptake of anthropogenic carbon dioxide. Nature, 356, 589–593.

Sarmiento, J. L., Monfray, P., Maier-Reimer, E., Aumont, O., Murnane, R. J. and Orr, J. C. (2000). Sea-air $CO_2$ fluxes and carbon transport: A comparison of three ocean general circulation models. Glob. Biogeochem. Cycles, 14, 1267–1281.

Simpson, J.J. (1985) Air-sea exchange of carbon dioxide and oxygen induced by phytoplankton: Methods and interpretation, in Mapping Strategies in Chemical Oceanography, A. Zirino, editor, American Chemical Society, Washington, D.C., 409–450.

Smith, S. V. and Hollinbaugh (1993). Coastal metabolism and the oceanic organic carbon balance. Review of Geophys., 31, 75–89.

Takahashi, T., Sutherland, S. C., Sweeney, C., Poisson, A., Metzl, N., Tillbrook, B., Bates, N., Wanninkhof, R., Feely, R. A., Sabine, C., Olafsson, J. and Nojiri, Y. (2002). Global sea-air $CO_2$ flux based on climatological surface ocean $pCO_2$, and seasonal biological and temperature effects, Deep-Sea Res. II, 49, 1601–1622.

Takahashi, T., Sutherland, S. C., Feely, R. A. and Cosca, C. (2003). Decadal variation of the surface water $pCO_2$ in the western and central Equatorial Pacific. Science, 302, 852–856.

Thomas, H., Y. Bozec, K. Elkalay, and H. J. W. De Baar (2004), Enhanced open ocean storage of $CO_2$ from shelf sea pumping, Science, *304*, 1005–1008.

Tsunogai, S., S. Watanabe, and T. Sato (1999) Is there a "continental shelf pump" for the absorption of atmospheric $CO_2$?, Tellus *B*, *5*, 701–712.

van Geen, A. Takesue, R. K., Goddard, J., Takahashi, T., Barth, J.A., and Smith, R.L. (2000) Carbon and nutrient dynamics during upwelling off Cape, Blanco, Oregon. Deep-Sea Research II, 49, 4369–4385.

Wanninkhof, R. (1992). Relationship between wind speed and gas exchange. Jour. Geophys. Res., 97, 7373–7382.

Ware, D. M. and Thomson, R. D. (2005) Bottom-Up Ecosystem Trophic Dynamics Determine Fish Production in the Northeast Pacific. Science, 308, 1280 – 1284.

**Table 15-1. Climatological mean distribution of the net sea-air $CO_2$ 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 (wind speed)$^2$ dependence for sea-air 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<br>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 sea-air $CO_2$ 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 (km²) and mean annual sea-air $CO_2$ 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 (km²) | | | | | Mean sea-air $CO_2$ 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).

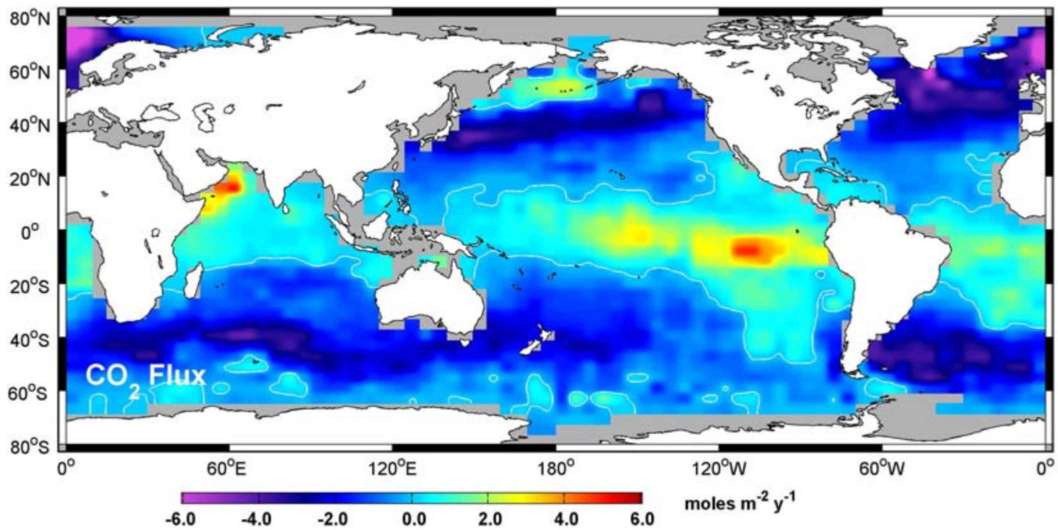


**Figure 15-2. In the top panel, mean air/sea $CO_2$ flux is calculated from shipboard measurements on a line perpendicular to the central California coast.** Flux within Monterey Bay (~0–20 km offshore) is into the ocean, flux across the active upwelling region (~20–75 km offshore) is from the ocean, 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 atmospheric$CO_2$, diffusion drives $CO_2$ 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).

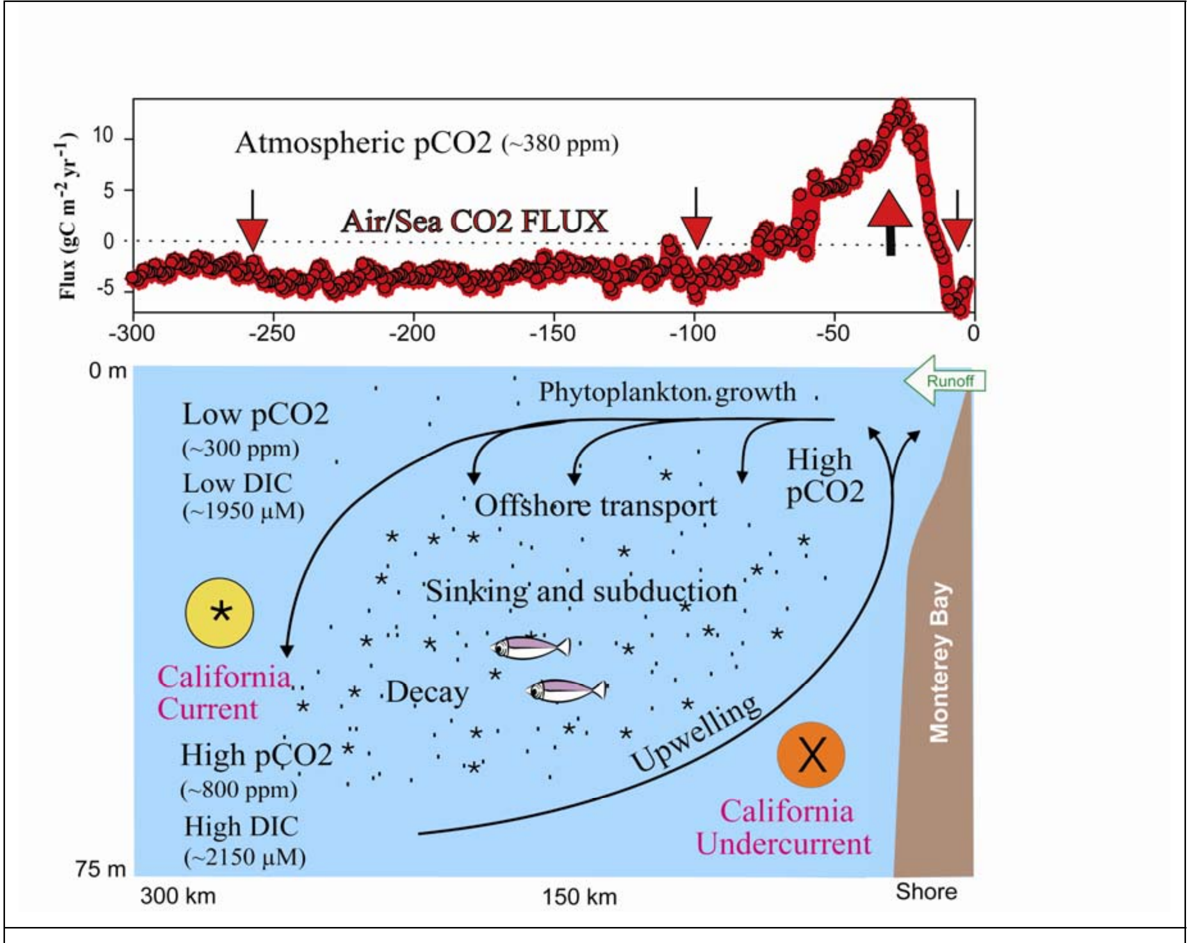


**Figure 15-3. (A). Distribution of coastal $CO_2$ partial pressure measurements made between 1979 and 2004. (B). The distribution of the net sea-air $CO_2$ flux over 1° × 1° pixel areas (N-S 100 km, E-W 80 km) around North America.** The flux (grams of carbon per square meter per year) represents the 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).

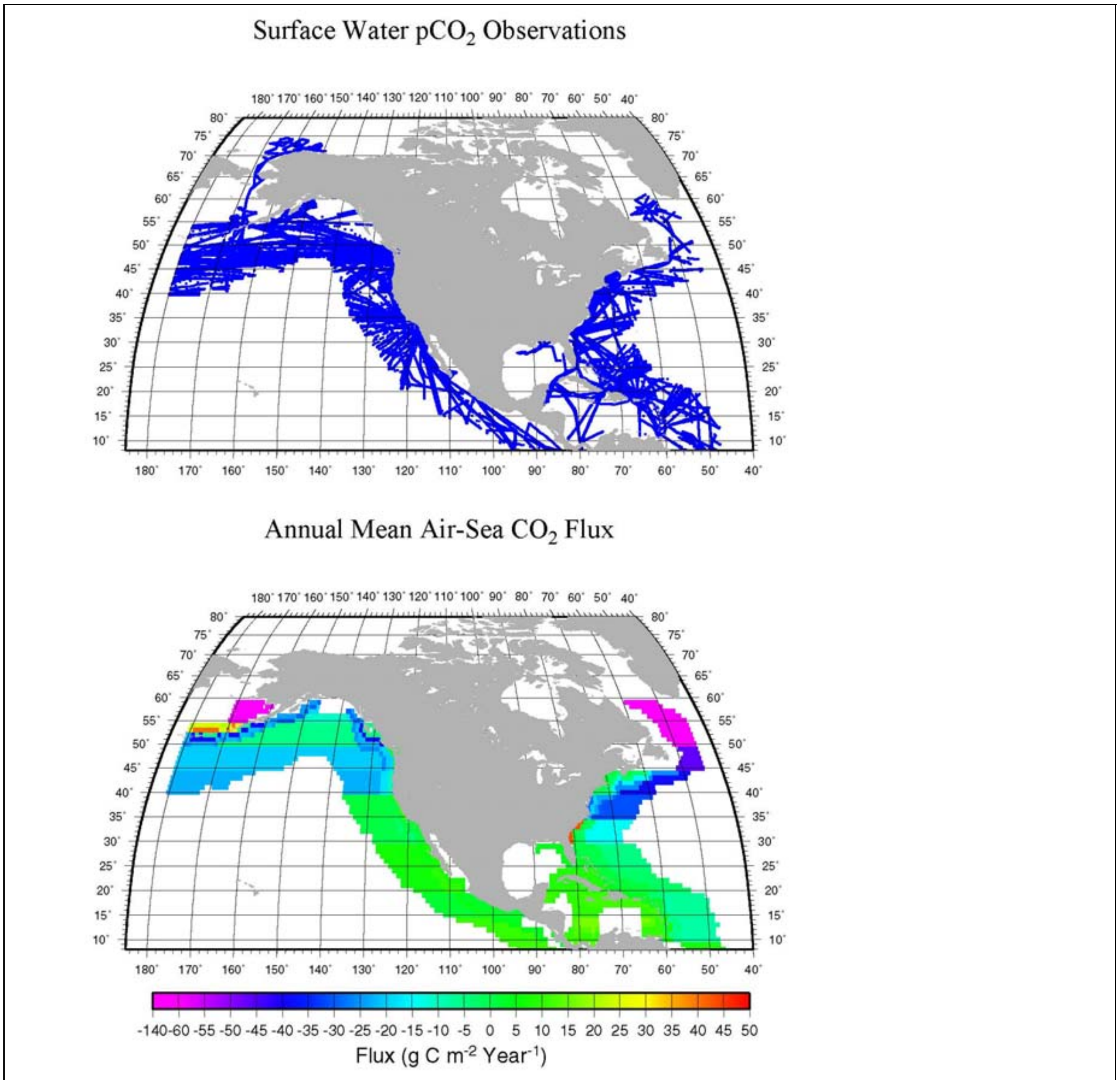


**Figure 15-4. Estimated sea-air $CO_2$ fluxes (grams of carbon per square meter per year) from 550,000 seawater $CO_2$ partial pressure ($pCO_2$) observations made from 1979 to 2004 in ocean waters surrounding the North American continent.** (A) Waters within one degree (about 80 km) of the coast and (B) open ocean waters between 300 and 900 km from the shore (see Figure 15-3B). The annual mean sea-air $pCO_2$ 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 sea-air $CO_2$ 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 sea-air gas transfer rate by Wanninkhof (1992). The ± uncertainties represent one standard deviation.

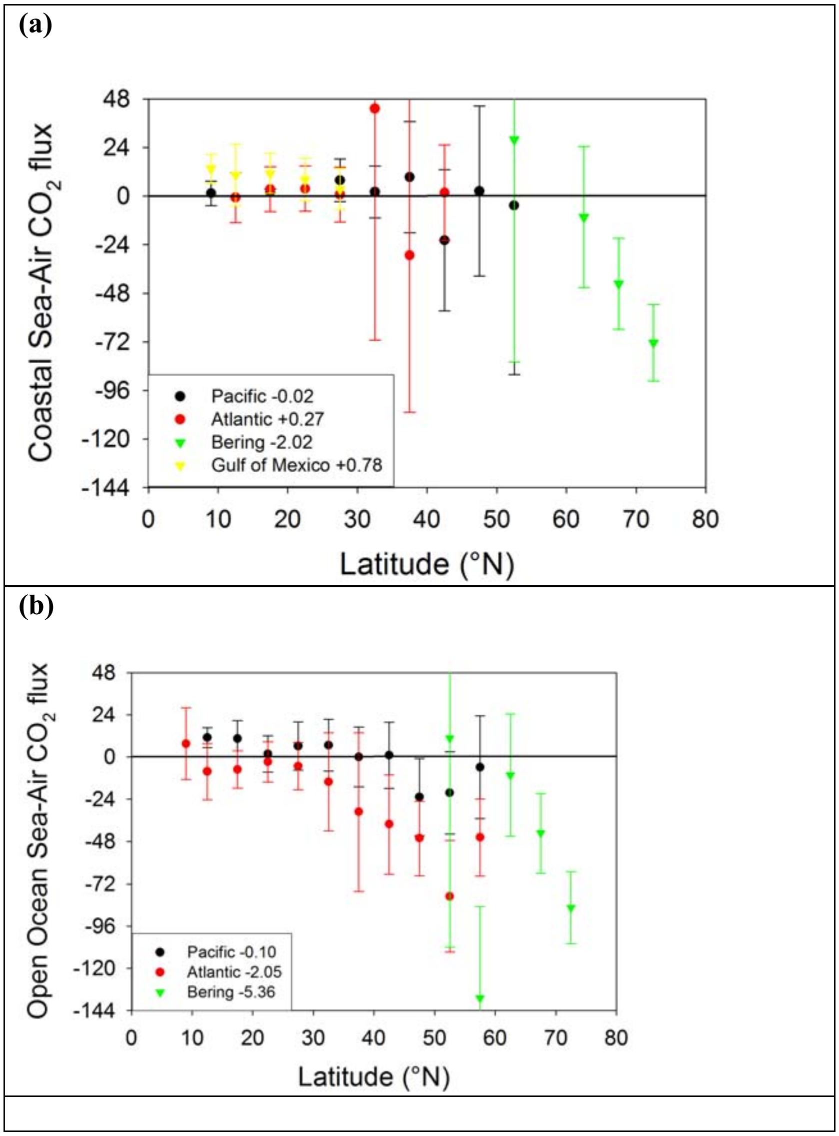
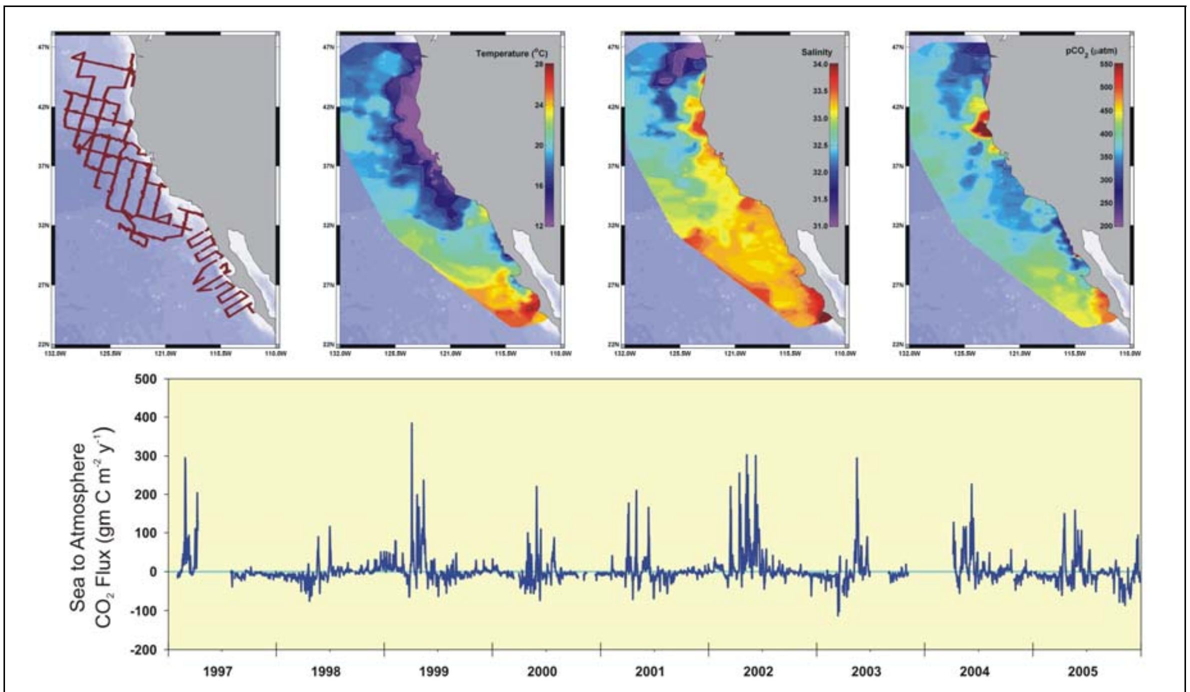


**Figure 15-5. Time-space variability of coastal waters off the west coast of North America.** (A) Quasi-synoptic distribution of the temperature, salinity, and $pCO_2$ in surface waters during July–September 2005. The Columbia River plume (~46°N) and the upwelling of deep waters off the Cape Mendocino (~40°N) are clearly seen. (B) 1997–2005 time-series data for sea-air $CO_2$ 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 sea-air $CO_2$ 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. Sea-air $CO_2$ 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 sea-air $pCO_2$ difference, which represents the thermodynamic driving potential for sea-air $CO_2$ gas transfer, was estimated using the atmospheric $CO_2$ concentration data. Finally, the net sea-air $CO_2$ 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 sea-air 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.

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