

PART III OVERVIEW

The Carbon Cycle in Land and Water Systems

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The six chapters (Chapters 10–15) in Part III consider the current and future carbon balance of terrestrial and aquatic ecosystems in North America. Although the amount of carbon exchanged between these ecosystems and the atmosphere each year through photosynthesis and plant and microbial respiration is large, the net balance for all of the ecosystems, combined, is currently a net sink of 472-592 Mt C yr⁻¹, and offsets only about 25-30% of current fossil fuel emissions from the region (1856 Mt C yr⁻¹ in 2003) (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, CO₂ 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 CO₂ 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 CO₂ 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

1 changes in farming practice may have begun to recover the carbon that was lost decades ago. Grazing
2 lands, although not directly affected by cultivation, were, nevertheless, managed in the United States
3 through fire suppression. The combined effects of grazing and fire suppression are believed to have
4 promoted the invasion of woody vegetation, possibly a carbon sink at present. Wetlands are the second
5 largest net carbon sink (after forests), but the magnitude of the sink was larger in the past than it is today,
6 again, as a result of land-use change (draining of wetlands for agriculture and forestry). The only lands
7 that seem to have escaped management are those lands overlying permafrost, and they are clearly subject
8 to change in the future as a result of global warming. Settled lands, by definition, are managed and are
9 dominated by fossil fuel emissions. Nevertheless, the accumulation of carbon in urban and suburban trees
10 suggests a net sequestration of carbon in the biotic component of long-standing settled lands. Residential
11 lands recently cleared from forests, on the other hand, are sources of carbon (Wienert and Hamburg,
12 2006).

13 From the perspective of carbon and climate, ecosystems are important if (1) they are currently large
14 sources or sinks of carbon or (2) they have the potential to become large sources or sinks of carbon in the
15 future through either management or environmental change, where ‘large’ sources or sinks, in this
16 context, are determined by the product of area (hectares) times flux per unit area (or flux density) (Mg
17 $\text{C ha}^{-1} \text{ yr}^{-1}$).

18 The largest carbon sink in North America (350 Mt C yr^{-1}) is associated with forests (Chapter 11)
19 (Table 1). The sink includes the carbon accumulating in wood products (e.g., in increasing numbers of
20 houses and landfills) as well as in the forests themselves. A sink is believed to exist in wetlands
21 (Chapter 13), including the wetlands overlying permafrost (Chapter 12), although the magnitude of this
22 sink is uncertain. More certain is the fact that the current sink is considerably smaller than it was before
23 wetlands were drained for agriculture and forestry. The other important aspect of wetlands is that they
24 hold nearly two thirds of the carbon in North America. Thus, despite the current net sink in these systems,
25 their potential for future emissions is large.

26

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

29

30 Although management has the potential to increase the carbon sequestered in agricultural (cultivated)
31 lands, these lands today are nearly in balance with respect to carbon (Chapter 10). The carbon lost to the
32 atmosphere from cultivation of organic soils is approximately balanced by the carbon accumulated in
33 mineral soils. In the past, before cultivation, these soils held considerably more carbon than they do today,
34 but about 25% of that carbon was lost soon after the lands were initially cultivated. In large areas of

1 grazing lands, there is the possibility that the invasion and spread of woody vegetation (woody
2 encroachment) is responsible for a significant net carbon sink at present (Chapter 10). The magnitude
3 (and even sign) of this flux is uncertain, however, in part because some ecosystems lose carbon
4 belowground (soils) as they accumulate it aboveground (woody vegetation), and in part because the
5 invasion and spread of exotic grasses into semi-arid lands of the western United States are increasing the
6 frequency of fires, reversing woody encroachment, and releasing carbon (Bradley *et al.*, in press).

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

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

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

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

29 The two factors (flux density and area) demonstrate the level of management required to remove a
30 significant amount of carbon from the atmosphere and keep it on land. Under current conditions,
31 sequestration of 100 Mt C yr^{-1} , for example (~5% of fossil fuel emissions from North America), requires
32 management over hundreds of millions of hectares (e.g., the area presently in agriculture or forests)
33 (Table 1). Enhancement of this terrestrial carbon sink through management would require considerable
34 effort. Nevertheless, the cost (in \$/metric ton CO_2) may be low relative to other options for managing

1 carbon. For example, forestry activities are estimated to have the potential to sequester 100–200 Mt C yr⁻¹
2 in the United States at prices ranging from less than \$10/ton of CO₂ for improved forest management, to
3 \$15/ton for afforestation, to \$30–50/ton for production of biofuels. Somewhat smaller sinks of 10–70 Mt
4 C yr⁻¹ might be sequestered in agricultural soils at low to moderate costs (\$3–30/ton CO₂). The maximum
5 amounts of carbon that might be accumulated in forests and agricultural soils are not known, and thus the
6 number of years these rates of sequestration might be expected to continue is also unknown. It seems
7 unlikely that the amount of carbon currently held in forests and agricultural lands could double. Changes
8 in climate will also affect carbon storage, but the net effect of management and climate is uncertain.

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

13 The effects and cost of managing aquatic systems are less clear. Increasing the area of wetlands, for
14 example, would presumably sequester carbon; but it would also increase emissions of CH₄, countering the
15 desired effect. Fertilization of coastal waters with iron has been proposed for increasing oceanic uptake of
16 CO₂, but neither the amount of carbon that might be sequestered nor the side effects are known
17 (Chapter 15).

18 A few studies have estimated the potential magnitudes of future carbon sinks as a result of
19 management (Chapters 10, 11). However, the contribution of management, as opposed to the
20 environment, in today's sink is unclear (see Chapter 3), and for the future the relative roles of
21 management and environmental change are even less clear. The two drivers might work together to
22 enhance terrestrial carbon sinks, as seems to have been the case during recent decades (Prentice *et al.*,
23 2001) (Chapter 2). On the other hand, they might work in opposing directions. A worst-case scenario,
24 quite possible, is one in which management will become ineffective in the face of large natural sources of
25 carbon not previously experienced in the modern world. In other words, while management is likely to be
26 essential for sequestering carbon, it may not be sufficient to preserve the current terrestrial carbon sink
27 over North America, let alone to offset fossil fuel emissions.

28 At least one other observation about sequestering carbon in terrestrial and aquatic ecosystems should
29 be mentioned. In contrast to the hundreds of millions of hectares that must be managed to sequester
30 100 Mt C annually, a few million hectares of forest fires can release an equivalent amount of carbon in a
31 single year. This disparity in flux densities underscores the fact that a few million hectares are disturbed
32 each year, while hundreds of millions of hectares are recovering from past disturbances. The natural
33 cycling of carbon is large in comparison to net fluxes. The observation is relevant for carbon
34 management, because the cumulative effects of small managed net sinks to mitigate fossil fuel emissions

1 will have to be understood, analyzed, monitored and evaluated in the context of larger, highly variable
2 and uncertain sources and sinks in the natural cycle.

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

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

15 16 **KEY UNCERTAINTIES AND GAPS IN UNDERSTANDING THE CARBON CYCLE OF** 17 **NORTH AMERICA**

- 18 • As mentioned above, the net flux of carbon resulting from woody encroachment and its inverse,
19 woody elimination, is highly uncertain. Even the sign of the flux is in question.
- 20 • Rivers, lakes, dams, and other inland waters are mentioned in Chapter 15 as being a source of carbon,
21 but they are claimed elsewhere to be a sink (Chapter 3). The sign of the net carbon flux attributable to
22 erosion, transport, deposition, accumulation and decomposition is uncertain (e.g., Stallard, 1998; Lal,
23 2001; Smith *et al.*, 2005).
- 24 • Several chapters cite studies that have attempted to quantify potential future carbon sinks in countries
25 in North America, but no reference is made to estimates of future sources of carbon. Clearly, there are
26 modeling studies that project large future carbon emissions, although these studies are largely global
27 in scope (e.g., Cox *et al.*, 2000; Jones *et al.*, 2005). Are there no studies of future carbon sources and
28 sinks for North America? Melting permafrost, in particular, is likely to increase emissions of carbon
29 to the atmosphere, CH₄ as well as CO₂.
- 30 • The sum of land areas reported in these chapters is about 330 million ha larger than the area of North
31 America (Table 1). The reason for this double-counting is unclear, but it implies a double counting of
32 carbon stocks and, perhaps, current sinks, as well.

1 **REFERENCES**

- 2 Bradley, B. A., R. A. Houghton, J. F. Mustard, and S. P. Hamburg. Invasive grass reduces carbon stocks in
3 shrublands of the Western United States *Global Change Biology*, in press.
- 4 Caspersen, J. P., S. W. Pacala, J. C. Jenkins, G. C. Hurtt, P. R. Moorcroft, and R. A. Birdsey. 2000. Contributions of
5 land-use history to carbon accumulation in United States forests. *Science* **290**:1148–1151.
- 6 Cox, P. M., R. A. Betts, C. D. Jones, S. A. Spall, and I. J. Totterdell. 2000. Acceleration of global warming due to
7 carbon-cycle feedbacks in a coupled climate model. *Nature* **408**:184–187.
- 8 Davidson, E. A., and I. A. Janssens. 2006. Temperature sensitivity of soil carbon decomposition and feedbacks to
9 climate change. *Nature* **440**:165–173.
- 10 Houghton, R. A., J. L. Hackler, and K. T. Lawrence. 1999. The United States carbon budget: contributions from land-
11 use change. *Science* **285**:574–578.
- 12 Hurtt, G. C., S. W. Pacala, P. R. Moorcroft, J. Caspersen, E. Shevliakova, R. A. Houghton, and B. Moore III. 2002.
13 Projecting the future of the United States carbon sink. *Proceedings of the National Academy of Sciences*
14 **99**:1389–1394.
- 15 Jones, C., C. McConnell, K. Coleman, P. Cox, P. Falloon, D. Jenkinson, and D. Powlson. 2005. Global climate
16 change and soil carbon stocks; predictions from two contrasting models for the turnover of organic carbon in
17 soil. *Global Change Biology* **11**:154–166.
- 18 Lal, R. 2001. Fate of eroded soil carbon: emission or sequestration. In: Lal, R. (Ed.), *Soil Carbon Sequestration and*
19 *the Greenhouse Effect*. Soil Science Society of America Special Publication, vol. 57. Madison, Wisconsin, pp.
20 173-181.
- 21 Osterkamp, T. E., and V. E. Romanovsky. 1999: Evidence for warming and thawing of discontinuous permafrost in
22 Alaska. *Permafrost and Periglacial Processes*, 10(1):17–37.
- 23 Osterkamp, T. E., L. Viereck, Y. Shur, M. T. Jorgenson, C. Racine, A. Doyle, and R. D. Boone. 2000: Observations
24 of thermokarst and its impact on boreal forests in Alaska, United States. *Arctic, Antarctic and Alpine Research*,
25 **32**:303–315.
- 26 Prentice I. C., G. D. Farquhar, M. J. R. Fasham, M. L. Goulden, M. Heimann, V. J. Jaramillo, H. S. Khashgi, C. Le
27 Quéré, R. J. Scholes, and D. W. R. Wallace. 2001. The carbon cycle and atmospheric carbon dioxide. In:
28 *Climate Change 2001: The Scientific Basis. Contribution of Working Group I to the Third Assessment Report of*
29 *the Intergovernmental Panel on Climate Change* [J. T. Houghton, Y. Ding, D. J. Griggs, M. Noguer, P. J. van
30 der Linden, X. Dai, K. Maskell, and C. A. Johnson (eds). Cambridge University Press, Cambridge, United
31 Kingdom and New York, NY, United States, pp. 183–237.
- 32 Schimel, D., J. Melillo, H. Tian, A. D. McGuire, D. Kicklighter, T. Kittel, N. Rosenbloom, S. Running, P. Thornton,
33 D. Ojima, W. Parton, R. Kelly, M. Sykes, R. Neilson, and B. Rizzo. 2000. Contribution of increasing CO₂ and
34 climate to carbon storage by ecosystems in the United States. *Science* **287**:2004–2006.
- 35 Smith, L.C., Y. Sheng, G.M. MacDonald, L.D. Hinzman, 2005: Disappearing Arctic Lakes, *Science*, 308, 1429.

- 1 Smith, S. L., M. M. Burgess, and F. M. Nixon. 2001: Response of activelayer and permafrost temperatures to
2 warming during 1998 in the Mackenzie Delta, Northwest Territories and at Canadian Forces Station Alert and
3 Baker Lake, Nunavut. *Geological Survey of Canada Current Research 2001–E5*, 8pp.
- 4 Smith, S.V., R.O. Slezzer, W.H. Renwick, and R.W. Buddemeier. 2005. Fates of eroded soil organic carbon:
5 Mississippi Basin case study. *Ecological Applications* **15**:1929-1940.
- 6 Stallard, R.F. 1998. Terrestrial sedimentation and the carbon cycle: Coupling weathering and erosion to carbon burial.
7 *Global Biogeochemical Cycles* **12**:231-257.
- 8 Wienert, A., and S.P. Hamburg. 2006. Carbon stock changes and greenhouse gas emissions from exurban land
9 development in central New Hampshire. Master's Thesis, Brown University, Providence, Rhode Island.
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Table 1. Ecosystems in North America: their areas, net annual fluxes of carbon, and their potential for sources (+) or sinks (-) in the future

Type of ecosystem	Area (10 ⁶ ha)	Current mean flux density (Mg C ha ⁻¹ yr ⁻¹)	Current flux (Mt C yr ⁻¹)	Carbon stocks (Mt C)	Future potential flux (Mt C yr ⁻¹)
Agriculture	231	0.0	0±15 ¹	18,500	-(50 to 100) 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 × 10⁶ ha of permafrost peatlands in Canada are treed (and may be included in forest area as well as permafrost area). Nevertheless, another ~330 × 10⁶ 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 shrublands 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 (10⁶ 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

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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 $6\text{--}15.5$ Mt C yr⁻¹, the cultivation of organic soils releases $5.3\text{--}10.3$ 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 $27\text{--}41$ 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 CO₂ are uncertain.

- 1 • Conservation-oriented management of agricultural lands (e.g., use of conservation tillage, improved
2 cropping and grazing systems, reduced bare fallow, set-asides of fragile lands, and restoration of
3 degraded soils) can significantly increase soil carbon sequestration.
 - 4 • Projections of future trends in agricultural land area and soil carbon stocks are unavailable or highly
5 uncertain because of uncertainty in future land-use change and agricultural management practice.
 - 6 • Annualized prices of \$15/tonne CO₂, would yield mitigation amounts of 168 Mt CO₂ yr⁻¹ through
7 agricultural soil C sequestration and 53 Mt CO₂ yr⁻¹ from fossil fuel use reduction. At lower prices of
8 \$5/tonne CO₂, the corresponding values would be 123 Mt CO₂ yr⁻¹ and 32 Mt CO₂ yr⁻¹, respectively.
 - 9 • Policies designed to suppress emissions of one greenhouse gas need to consider complex
10 interactions to ensure that *net* emissions are reduced. For example, increased use of fertilizer or
11 irrigation may increase crop residues and carbon sequestration, but may stimulate emissions of CH₄
12 or N₂O.
 - 13 • Many of the practices that lead to carbon sequestration and reduced CO₂ and CH₄ emissions from
14 agricultural lands not only increase production efficiencies, but lead to environmental co-benefits, for
15 example, improved soil fertility, reduced erosion and pesticide immobilization.
 - 16 • An expanded network of intensive research sites is needed to better understand the effects of
17 management on carbon cycling and storage in agricultural systems. An extensive national-level
18 network of soil monitoring sites in which changes in carbon stocks are directly measured is needed to
19 reduce the uncertainty in the inventory of agricultural carbon. Better information about the spatial
20 extent of woody encroachment, the amount and growth of woody biomass, and variation in impacts
21 on soil carbon stocks would help reduce the large uncertainty of the carbon impacts of woody
22 encroachment.
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23
24

25

26 INVENTORY

27 Background

28 Agricultural lands, including croplands and grazing lands—grasslands, shrublands, and arid lands¹—
29 occupy 47% of the land area in North America (59% in the United States, 70% in Mexico, and 11% in
30 Canada), and contain 17.4% of the terrestrial carbon (Pacala *et al.*, 2006). These lands differ from other
31 types of ecosystems in that most of their carbon is held in soils. Live vegetation in cultivated systems
32 generally contains less than 5% of the total carbon, whereas vegetation in grazing lands contains a greater
33 proportion (5–30%), but still less than that in forested systems (30–65%). These systems in North
34 America contain 78.5±19.5 (±1SE) Gt C in the soil (Table 10-1). Significant increases in vegetation

¹We refer collectively to these lands as grazing lands since grazing is their primary use, even though not all of these lands are grazed.

1 carbon stocks in some grazing lands have been observed and, together with soil carbon stocks from
2 croplands and grazing lands, likely contribute significantly to the large North American terrestrial carbon
3 sink (Houghton *et al.*, 1999; Pacala *et al.*, 2001; Eve *et al.*, 2002; Ogle *et al.*, 2003). These lands also emit
4 greenhouse gases: fossil fuel use for on-farm machinery and buildings, for manufacture of agricultural
5 inputs, and for transportation account for 3–5% of total CO₂ emissions in developed countries (Enquete
6 Commission, 1995); activities on agricultural and grazing lands, like livestock production, animal waste
7 management, biomass burning, and rice cultivation, emit 35% of global anthropogenic CH₄ (27% of
8 United States, 31% of Mexican, and 27% of Canadian CH₄ emissions) (Mosier *et al.*, 1998b; CISCC,
9 2001; Matin *et al.*, 2004; EPA, 2005); and agricultural and grazing lands are the largest anthropogenic
10 source of N₂O emissions (CAST, 2004; see Text Box 1). However, agricultural lands are actively
11 managed and have the capacity to take up more carbon into soil; thus improving management could lead
12 to substantial reductions in CO₂ and CH₄ emissions and could sequester carbon to offset emissions from
13 other lands or sectors.

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15

16 **Table 10-1. Carbon pools in agricultural and grazing lands in Canada, Mexico, and the United**
17 **States; the area (M ha) for each climatic zone are in parentheses.** Carbon pools for undisturbed native
18 systems were derived using the intersection of MODIS-IGBP^a land cover types (Friedl *et al.*, 2002) and
19 mean soil carbon contents to 1m depth from Sombroek *et al.* (1993) spatially arrayed using FAO soil
20 classes (ISRIC, 2002).

21

22 **Carbon Dioxide Fluxes from Agricultural and Grazing Land**

23 The basic processes governing the carbon balance of agricultural and grazing lands are the same as
24 for other ecosystems: the photosynthetic uptake and assimilation of CO₂ into organic compounds and the
25 release of gaseous carbon through respiration (primarily CO₂ but also CH₄). In agricultural lands, carbon
26 assimilation is directed towards production of food, fiber, and forage by manipulating species
27 composition and growing conditions. Biomass, being predominantly herbaceous (i.e., non-woody), is a
28 small, transient carbon pool (compared to forests) and hence soils constitute the dominant carbon stock.
29 Cropland systems can be among the most productive ecosystems, but restricted growing season length,
30 fallow periods, and grazing-induced shifts in species composition or production can reduce carbon uptake
31 relative to that in other ecosystems. These factors, along with tillage-induced soil disturbances and
32 removal of plant carbon through harvest, have depleted soil carbon stocks by 20–40% or more from pre-
33 cultivated conditions (Davidson and Ackerman, 1993; Houghton and Goodale, 2004). Soil organic carbon
34 stocks in grazing lands (see Text Box 2 for information on inorganic soil carbon stocks) have been

1 depleted to a lesser degree than for cropland (Ogle *et al.*, 2004), and in some regions biomass has
2 increased due to suppression of disturbance and subsequent woody encroachment (see Text Box 3).
3 Woody encroachment is potentially a significant sink for atmospheric CO₂, but the magnitude of the sink
4 is poorly constrained (Houghton *et al.*, 1999; Pacala *et al.*, 2001). Increased decomposition rates of
5 aboveground litter and harvest removal of some (30–50% of forage in grazing systems, 40–50% in grain
6 crops) or all (e.g., corn for silage) of the aboveground biomass, have drastically altered carbon cycling
7 within agricultural lands and thus the sources and sinks of CO₂ to the atmosphere.

8 Much of the carbon lost from agricultural land soil and biomass pools can be recovered with changes
9 in management practices that increase carbon inputs, stabilize carbon within the system, or reduce carbon
10 losses (Figure 10-1; Table 10-2), while still maintaining outputs of food, fiber, and forage. Within Canada
11 and the United States, mineral soils have been sequestering 0.1 and 6.5–16 Mt C yr⁻¹ (Smith *et al.*, 1997;
12 Smith *et al.*, 2001b; Ogle *et al.*, 2003), respectively, largely through improved practices on annual
13 cropland. Conversion of agricultural land to grassland, like under the Conservation Reserve Program in
14 the United States (6 Mt C yr⁻¹ on 14 M ha of land), and afforestation have also sequestered carbon in
15 agricultural and grazing lands. In contrast, cultivation of organic soils (e.g., peat-derived soils) is
16 releasing an estimated 0.1 and 5-10 Mt C yr⁻¹ from soils in Canada and the United States (Ogle *et al.*,
17 2003; Matin *et al.*, 2004). Compared with other systems, the high productivity and management-induced
18 disturbances of agricultural systems promote movement and redistribution (through erosion, runoff and
19 leaching) of organic and inorganic carbon, sequestering potentially large amounts of carbon in sediments
20 and water (Raymond and Cole, 2003; Smith *et al.* 2005; Yoo *et al.*, 2005). However, the net impact of
21 soil erosion on carbon emissions to the atmosphere remains highly uncertain.

22
23 **Figure 10-1. North American agricultural and grazing land CO₂ (left side) and methane (right side)**
24 **fluxes for the years around 2000.** Negative values indicate net flux from the atmosphere to soil and
25 biomass carbon pools. All data are from Canadian (Matin *et al.*, 2004) and U.S. (EPA, 2005) National
26 Inventories and from the second Mexican National Communication (CISCC, 2001), except for Canadian
27 [from Kulshreshtha *et al.* (2000)] and U.S. fossil fuel inputs [from Lal *et al.* (1998)] and woody
28 encroachment [from Houghton *et al.* (1999)]. Values are for 2003 for the United States and Canada and
29 1998 for Mexico. A global warming potential of 23 for methane was used to convert emissions of CH₄ to
30 CO₂ equivalents (IPCC, 2001) and a factor of 12/44 to convert from CO₂ to carbon. Asterisks indicate
31 unavailable data. Data ranges are indicated by error bars where available.

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33 **Table 10-2. North American agricultural and grazing land carbon fluxes for the years around 2000.**
34 Negative numbers (in parentheses) indicate net flux from the atmosphere to soil and biomass carbon pools.
35 Unless otherwise noted, data are from Canadian (Matin *et al.*, 2004) and U.S. (EPA, 2005) National

1 Inventories and from the second Mexican National Communication (CISCC, 2001). Values are for 2003 for
2 United States and Canada and 1998 for Mexico. A global warming potential of 23 for methane was used to
3 convert emissions of CH₄ to CO₂ equivalents (IPCC, 2001) and a factor of 12/44 to convert from CO₂ to
4 carbon.

5
6 Production, delivery, and use of field equipment, fertilizer, seed, pesticides, irrigation water, and
7 maintenance of animal production facilities contribute 3–5% of total fossil fuel CO₂ emissions in
8 developed countries (Enquete Commission, 1995). On-farm fossil fuel emissions plus CO₂ emissions
9 embodied in applied fertilizers and pesticides contribute emissions of 28 Mt C yr⁻¹ within the United
10 States (Lal *et al.*, 1998) and 2.8 Mt C yr⁻¹ in Canada (Sobool and Kulshreshtha, 2005). Energy
11 consumption for heating and cooling high intensity animal production facilities is among the largest CO₂
12 emitters within the agricultural sector (Enquete Commission, 1995).

13 Much of the ammonia production and urea application (U.S.: 4.3 Mt C yr⁻¹; Mexico: 0.4 Mt C yr⁻¹;
14 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;
15 Canada: not reported) are devoted to agricultural uses.

16 17 **Methane Fluxes from Agricultural and Grazing Lands**

18 Cropland and grazing land soils act as both sources and sinks for atmospheric CH₄ (Figure 10-1;
19 Table 10-2). Methane formation is an anaerobic process and is most significant in waterlogged soils, like
20 those under paddy rice cultivation (U.S.: 0.328 Mt CH₄ yr⁻¹; Mexico: 0.015 Mt CH₄ yr⁻¹; Canada:
21 negligible, not reported). Methane is also formed by incomplete biomass combustion of crop residues
22 (U.S.: 0.038 Mt CH₄ yr⁻¹; Mexico: 0.011 Mt CH₄ yr⁻¹; Canada: negligible, not reported). Methane
23 oxidation in soils is a global sink for about 5% of CH₄ produced annually and is mainly limited by CH₄
24 diffusion into the soil. However, intensive cropland management tends to reduce soil methane
25 consumption relative to forests and extensively grazing lands (CAST, 2004). Recent research has shown
26 that live plant biomass and litter produce substantial amounts of CH₄, potentially making plants as large a
27 source of CH₄ as livestock (Keppler *et al.*, 2006). If this is the case, activities that increase plant
28 biomass—and sequester CO₂—may lead to increased CH₄ production (Keppler *et al.*, 2006).

29 30 **Methane Fluxes from Livestock**

31 Enteric fermentation (the process of organic matter breakdown by gut flora within the gastrointestinal
32 tract of animals, particularly ruminants) allows for the digestion of fibrous materials that provide energy
33 to microbes and their hosts, but the extensive fermentation of the ruminant diet requires 5–7% of the
34 dietary gross energy to be belched out as CH₄ to sustain the anaerobic processes (Johnson and Johnson,

1 1995). Methane emissions from livestock contribute significantly to total CH₄ emissions in the United
2 States (54 Mt CH₄ yr⁻¹, 21% of total U.S. CH₄ emissions), Canada (0.8 Mt CH₄ yr⁻¹, 22% of total)
3 (Sobool and Kulshreshtha, 2005), and Mexico (2.0 Mt CH₄ yr⁻¹, 27% of total) with the vast majority of
4 enteric CH₄ emissions are from beef (72%) and dairy cattle (23%). Emissions from ruminants are tightly
5 coupled to feed consumption, since CH₄ emission per unit of feed energy is consistent, except for feedlot
6 cattle with diets high in cereal grain contents, for which the fractional loss falls to one-third to one-half of
7 normal rates (Johnson and Johnson, 1995). Between 1990 and 2002, CH₄ emissions from enteric
8 fermentation fell 2% in the United States but increased by 20% in Canada (EPA, 2000; Matin *et al.*,
9 2004).

10 Methane emissions during manure storage (U.S.: 1.9 Mt CH₄ yr⁻¹; Mexico: 0.06 Mt CH₄ yr⁻¹;
11 Canada: 0.3 Mt CH₄ yr⁻¹) are governed by the amount of degradable organic matter, degree of anoxia,
12 storage temperature, and duration of storage. Unlike enteric CH₄, the major sources of manure CH₄
13 emissions in the United States are from swine (44%) and dairy cattle (39%). Manure CH₄ production is
14 greater for production systems with anoxic lagoons, largely anoxic pits, or manure handled or stored as
15 slurry. Between 1990 and 2002, CH₄ emissions from manure management increased 25% in the United
16 States and 21% in Canada (EPA, 2000; Matin *et al.*, 2004).

17 18 **DRIVERS AND TRENDS**

19 The extent to which agricultural options will contribute to greenhouse gas mitigation will largely
20 depend on government policy decisions, but mitigation opportunities will also be constrained by changing
21 environmental conditions, i.e., global warming (NAS, 2001) may make it more difficult to reduce
22 emissions from cropland and grazing lands (see discussion below). Estimates from national inventories
23 suggest that U.S. and Canadian agricultural soils are currently near neutral or small net sinks for CO₂,
24 which has occurred as a consequence of changing management (e.g., reduced tillage intensity) and
25 government programs designed for purposes other than greenhouse gas mitigation (e.g., soil conservation,
26 commodity regulation). However, to realize the much larger potential for soil carbon sequestration and for
27 significant reductions in CH₄ (and N₂O) emissions, specific policies targeted at greenhouse gas reductions
28 are required. It is generally recognized that farmers (and other economic actors) are, as a group, ‘profit-
29 maximizers,’ which implies that to change from current practices to ones that reduce net emissions,
30 farmers will incur additional costs (termed ‘opportunity cost’). Hence, where the incentives (e.g., carbon
31 offset market payments, government subsidies) to adopt new practices exceed the opportunity costs,
32 farmers will adopt new practices. Crop productivity, production input expenses, marketing costs, etc.
33 (which determine profitability) vary widely within (and between) countries. Thus, the payment needed to
34 achieve a unit of emission reductions will vary, among and within regions. In general, each successive

1 increment of carbon sequestration or emission reduction comes at a progressively higher cost (this
2 relationship is often shown in the form of an upward bending marginal cost curve).

3 Feedbacks between temperature and soil carbon stocks could counteract efforts to reduce greenhouse
4 gases via carbon sequestration within agricultural ecosystems. Increased temperatures tend to increase the
5 rate of biological processes—including plant respiration and organic matter decay and CO₂ release by soil
6 organisms—particularly in temperate climates that prevail across most of North America. Because soil
7 carbon stocks, including those in agricultural lands, contain such large amounts of carbon, small
8 percentage increases in rate of soil organic matter decomposition could lead to substantially increased
9 emissions (Jenkinson *et al.*, 1991; Cox *et al.*, 2000). There is currently a scientific debate about the
10 relative temperature sensitivity of the different constituents making up soil organic matter (e.g., Kätterer
11 *et al.*, 1998; Giardina and Ryan, 2000; Ågren and Bosatta, 2002; Knorr *et al.*, 2005), reflecting
12 uncertainty in the possible degree and magnitude of climate change feedbacks. Despite this uncertainty,
13 the potential for climate feedbacks to influence the carbon balance of agricultural systems by perturbing
14 productivity (and carbon input rates) and organic matter turnover, and potentially soil N₂O and CH₄
15 fluxes, cannot be overlooked.

17 **OPTIONS FOR MANAGEMENT**

18 **Carbon Sequestration**

19 Agricultural and grazing land management practices capable of increasing carbon inputs or
20 decreasing carbon outputs, while still maintaining yields, can be divided into two classes: those that
21 impact carbon inputs, and those that affect carbon release through decomposition and disturbance.
22 Reversion to native vegetation or setting agricultural land aside as grassland, such as in the Canadian
23 Prairie Cover Program and the U.S. Conservation Reserve Program, can increase the proportion of
24 photosynthesized carbon retained in the system and sequester carbon in the soil² (Post and Kwon, 2000;
25 Follett *et al.*, 2001b) (Figure 10-2). In annual cropland, improved crop rotations, yield enhancement
26 measures, organic amendments, cover crops, improved fertilization and irrigation practices, and reduced
27 bare fallow tend to increase productivity and carbon inputs, and thus soil carbon stocks (Lal *et al.*, 1998;
28 Paustian *et al.*, 1998; VandenBygaart *et al.*, 2003) (Figure 10-2). Tillage, traditionally used for soil
29 preparation and weed control, disturbs the soil and stimulates decomposition and loss of soil carbon.
30 Practices that substantially reduce (reduced-till) or eliminate (no-till) tillage-induced disturbances are

²The 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.

1 being increasingly adopted and generally increase soil carbon stocks while maintaining or enhancing
2 productivity levels (Paustian *et al.*, 1997; Ogle *et al.*, 2003) (Figure 10-2). Estimates of the technical
3 potential for annual cropland soil carbon sequestration are on the order of 50–100 Mt C yr⁻¹ in the United
4 States (Lal *et al.*, 2003; Sperow *et al.*, 2003) and approximately 5 Mt C yr⁻¹ in Canada (Boehm *et al.*,
5 2004).

6
7 **Figure 10-2. Relative soil carbon following implementation of new agricultural or grassland**
8 **management practices.** Conventionally tilled, medium-input cultivated land and moderately grazed
9 grasslands with moderate inputs are defaults for agricultural and grazing lands, respectively. Default soil
10 carbon stocks (like those in Table 10-1) can be multiplied by one or more emission factors to estimate
11 carbon sequestration rates. Temperature/precipitation divisions are the same as those described in Table 10-
12 1. Data are from Nabuurs *et al.* (2004) and Ogle *et al.* (2004).

13
14 Within grazing lands, historical overgrazing has substantially reduced productive capacity in many
15 areas, leading to loss of soil carbon stocks (Conant and Paustian, 2002) (Figure 10-2). Conversely,
16 improved grazing management and production inputs—like fertilizer, organic amendments, and
17 irrigation—can increase productivity, carbon inputs, and soil carbon stocks, potentially storing 0.44 Mt C
18 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.*,
19 2001a).

20 21 **Fossil Fuel-Derived Emission Reductions**

22 The efficiency with which on-farm (from tractors and machinery) and off-farm (from production of
23 agricultural input) energy inputs are converted to agricultural products varies several-fold (Lal, 2004).
24 Where more energy-efficient practices can be substituted for less efficient ones, fossil fuel CO₂ emissions
25 can be reduced (Lal, 2004). For example, converting from conventional plowing to no-tillage can reduce
26 on-farm fossil fuel emissions by 25–80% (Frye, 1984; Robertson *et al.*, 2000) and total fossil fuel
27 emissions by 14–25% (West and Marland, 2003). Substitution of legumes for mineral nitrogen can reduce
28 energy input by 15% in cropping systems incorporating legumes (Pimentel *et al.*, 2005). More efficient
29 heating and cooling (e.g., better building insulation) could reduce CO₂ emissions associated with housed
30 animal (e.g., dairy) facilities. Substitution of crop-derived for fossil fuels could decrease net emissions.

31 Energy intensity (energy per unit product) for the U.S. agricultural sector has declined since the 1970s
32 (Paustian *et al.*, 1998). Between 1990 and 2000, fossil fuel emissions on Canadian farms increased by
33 35% (Sobool and Kulshreshtha, 2005).

34

1 **Methane Emission Reduction**

2 Reducing flood duration and decreasing organic matter additions to paddy rice fields can reduce CH₄
3 emissions. Soil amendments such as ammonium sulfate and calcium carbide inhibit CH₄ formation.
4 Coupled with adoption of new rice cultivars that favor lower CH₄ emissions, these management practices
5 could reduce CH₄ emission from paddy rice systems by as much as 40% (Mosier *et al.*, 1998b).

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

10 Refinement of feed quality, feed rationing, additives, and livestock production efficiency chains can
11 all reduce CH₄ emissions from ruminant livestock with minimal impacts on productivity or profits
12 (CAST, 2004). Boadi *et al.* (2004) review several examples of increases in energy intensity. Wider
13 adoption of more efficient practices could reduce CH₄ production from 5–8% to 2–3% of gross feed
14 energy (Agriculture and Agri-Food Canada, 1999), reducing CH₄ emissions by 20–30% (Mosier *et al.*,
15 1998b).

16 Methane emissions from manure storage are proportional to duration of storage under anoxic
17 conditions. Handling solid rather than liquid manure, storing manure for shorter periods of time, and
18 keeping storage tanks cool will limit emissions from stored manure (CAST, 2004). More important,
19 capture of CH₄ produced during anaerobic decomposition of manure—in covered lagoons or small- or
20 large-scale digesters—can reduce emissions by 70–80% (Mosier *et al.*, 1998b). Use of digester systems is
21 spreading in the United States, with 50 digesters currently in operation and 60 systems in construction or
22 planned (NRCS, 2005). Energy production using CH₄ captured during manure storage will reduce energy
23 demands and associated CO₂ emissions.

24

25 **Environmental Co-benefits from Carbon Sequestration and Emission Reduction**

26 **Activities**

27 Many of the practices that lead to carbon sequestration and reduced CO₂ and CH₄ emissions not only
28 increase production efficiencies but also lead to environmental co-benefits. Practices that sequester
29 carbon in agricultural and grazing land soils improve soil fertility, buffering capacity, and pesticide
30 immobilization (Lal, 2002; CAST, 2004). Increasing soil carbon content makes the soil more easily
31 workable and reduces energy requirements for field operations (CAST, 2004). Decreasing soil
32 disturbance and retaining more surface crop residues enhance water infiltration and prevent wind and
33 water erosion, improving air quality. Increased water retention plus improved fertilizer management
34 reduces nitrogen losses and subsequent NO₃⁻ leaching and downstream eutrophication.

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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 CH₄ and N₂O 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).

1 A recent study commissioned by the U.S. Environmental Protection Agency (EPA 2005b), estimated
2 economic potential for some agricultural mitigation options, assuming constant price scenarios for 2010–
3 2110, where the price represents the incentive required for the mitigation activity. Annualized prices of
4 \$15/ton of CO₂ would yield mitigation amounts of 168 Mt CO₂ per year through agricultural soil carbon
5 sequestration and 53 Mt CO₂ per year from fossil fuel use reduction (compare with estimated U.S.
6 national ecosystem carbon sink of 1760 Mt CO₂ per year). At lower prices of \$5/ton CO₂, the
7 corresponding values would be 123 Mt CO₂ per year (for soil sequestration) and 32 Mt CO₂ per year (for
8 fossil fuel reduction), respectively, reflecting the effect of price on the supply of mitigation activities.
9

10 Other Policy Considerations

11 Agricultural mitigation of CO₂ through carbon sequestration and emission reductions for CH₄ (and
12 N₂O), differ in ways that impact policy design and implementation. Direct emission reductions of CH₄
13 and CO₂ from fossil fuel use are considered ‘permanent’ reductions, while carbon sequestration is a ‘non-
14 permanent’ reduction, in that carbon stored through conservation practices could potentially be re-emitted
15 if management practices revert back to the previous state or otherwise change so that the stored carbon is
16 lost. This *permanence* issue applies to all forms of carbon sinks. In addition, a given change in
17 management (e.g., tillage reduction, pasture improvement, afforestation) will stimulate carbon storage for
18 a finite duration. For many practices, soil carbon storage will tend to level off at a new equilibrium level
19 after 15–30 years, after which there is no further accumulation of carbon (West and Wali, 2002). Thus, to
20 maintain these higher stocks, the management practices will need to be maintained. Key implications for
21 policy are that the value of sequestered carbon will be discounted compared to direct emission reductions
22 to compensate for the possibility of future emissions. Alternatively, long-term contracts will be needed to
23 build and maintain C stocks, which will tend to increase the price per unit of sequestered carbon.
24 However, even temporary storage of carbon has economic value (CAST, 2004), and various proposed
25 concepts of leasing carbon storage or applying discount rates could accommodate carbon sequestration as
26 part of a carbon offset trading system (CAST, 2004). In addition, switching to practices that increase soil
27 carbon (and hence improve soil fertility) can be more profitable to farmers in the long-run, so that
28 additional incentives to maintain the practices once they become well established may not be necessary
29 (Paustian *et al.*, 2006).

30 Another policy issue relating to carbon sequestration is *leakage* (also termed ‘slippage’ in
31 economics), whereby mitigation actions in one area (e.g., geographic region, production system) stimulate
32 additional emissions elsewhere. For forest carbon sequestration, leakage is a major concern—for
33 example, reducing harvest rates in one area (thereby maintaining higher biomass carbon stocks) can
34 stimulate increased cutting and reduction in stored carbon in other areas, as was seen with the reduction in

1 harvesting in the Pacific Northwest during the 1990s (Murray *et al.*, 2004). Preliminary studies suggest
2 that leakage is of minor concern for agricultural carbon sequestration, since most practices would have
3 little or no effect on the supply and demand of agricultural commodities. However, there are uncertain
4 and conflicting views on whether land-set asides—where land is taken out of agricultural production,
5 such as the Conservation Reserve Program in the United States, might be subject to significant leakage.

6 A further question, relevant to policies for carbon sequestration, is how practices for conserving
7 carbon affect emissions of other greenhouse gases. Of particular importance is the interaction of carbon
8 sequestration with N₂O emission, because N₂O is such a potent greenhouse gas (Robertson and Grace,
9 2004; Six *et al.*, 2004; Gregorich *et al.*, 2005). (See Text Box 4). In some environs, carbon-sequestration
10 practices, such as reduced tillage, can stimulate N₂O emissions thereby offsetting part of the benefit;
11 elsewhere, carbon-conserving practices may suppress N₂O emissions, amplifying the net benefit (Smith *et*
12 *al.*, 2001a; Smith and Conen, 2004; Conant *et al.*, 2005; Helgason *et al.*, 2005).

13 Similarly, carbon-sequestration practices might affect emissions of CH₄, if the practice, such as
14 increased use of forages in rotations, leads to higher livestock numbers. These examples demonstrate that
15 policies designed to suppress emission of one greenhouse gas need to also consider complex interactions
16 to ensure that *net* emissions are reduced.

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

24 25 **RESEARCH AND DEVELOPMENT NEEDS**

26 Expanding the network of intensive research sites dedicated to understanding basic processes,
27 coupled with national-level networks of soil monitoring/validation sites could reduce inventory
28 uncertainty and contribute to attributing changes in ecosystem carbon stocks to changes in land
29 management (see Bellamy *et al.*, 2005). Expansion of both networks should be informed by information
30 about how different geographic areas and ecosystems contribute to uncertainty and the likelihood that
31 reducing uncertainty could inform policy decisions. For example, changes in ecosystem carbon stocks due
32 to woody encroachment on grasslands constitute one of the largest, but least certain, aspects of terrestrial
33 carbon cycling in North America (Houghton *et al.*, 1999; Pacala *et al.*, 2001). Better information about
34 the spatial extent of woody encroachment, the amount and growth of woody biomass, and variation in

1 impacts on soil carbon stocks would help reduce that uncertainty. Identifying location, cause, and size of
2 this sink could help identify practices that may promote continued sequestration of carbon and would
3 constrain estimates of carbon storage in other lands, possibly helping identify other policy options.
4 Uncertainty in land use, land use change, soil carbon responses to management (e.g., tillage) on particular
5 soils, and impacts of cultivation on soil carbon stocks (e.g., impacts of erosion) are the largest
6 contributors to uncertainty in the Canadian and U.S. national agricultural greenhouse gas inventories
7 (Ogle *et al.*, 2003; VandenBygaart *et al.*, 2004). Finally, if the goal of a policy instrument is to reduce
8 greenhouse gas emissions, net impacts on CO₂, CH₄, and N₂O emissions, which are not as well
9 understood, should be considered.

11 REFERENCES

- 12 Ågren, G. I., and E. Bosatta, 2002: Reconciling differences in predictions of temperature response of soil organic
13 matter. *Soil Biology and Biochemistry*, **34**, 129–132.
- 14 Agriculture and Agri-Food Canada, 1999: The health of our air. Toward sustainable agriculture in Canada. In:
15 Publication 1981/E. (H. H. Janzen, R. L. Desjardins, J. M. R. Asselin, and B. Grace, eds. Pages 40 Ottawa,
16 Ontario: Agriculture and Agri-Foods Canada.
- 17 Antle, J. M., S. Capalbo, S. Mooney, E. T. Elliott, and K. Paustian, 2001: Economic analysis of agricultural soil
18 carbon sequestration: An integrated assessment approach. *Journal of Agricultural and Resource Economics*,
19 **26**(2), 344–367.
- 20 Antle, J. M., S. M. Capalbo, S. Mooney, D. K. Elliott, and K. H. Paustian, 2003: Spatial Heterogeneity, Contract
21 Design, and the Efficiency of Carbon Sequestration Policies for Agriculture. *Journal of Environmental*
22 *Economics and Management*, **46**(2), 231–250.
- 23 Bellamy, P. H., P. J. Loveland, R. I. Bradley, R. M. Lark, and G. J. D. Kirk, 2005: Carbon losses from all soils
24 across England and Wales 1978–2003. *Nature*, **437**, 245–248.
- 25 Boadi, D., C. Benchaar, J. Chiquette, and D. Masse, 2004: Mitigation strategies to reduce enteric methane emissions
26 from dairy cows: Update review. *Canadian Journal of Animal Science*, **84**(3), 319–335.
- 27 Boehm, M., B. Junkins, R. Desjardins, S. N. Kulshreshtha, and W. Lindwall, 2004: Sink potential of Canadian
28 agricultural soils. *Climatic Change*, **65**, 297–314.
- 29 CAST, 2004: Climate change and greenhouse gas mitigation: Challenges and opportunities for agriculture.
30 Paustian, K., B. A. Babcock, J. Hatfield, C. L. Kling, R. Lal, B. A. McCarl, S. McLaughlin, A. R. Mosier,
31 W. M. Post, C. W. Rice, G. P. Robertson, N. J. Rosenberg, C. Rosenzweig, Cynthia, W. H. Schlesinger, and
32 D. Zilberman (Task Force Members). Ames, IA: Council for Agricultural Science and Technology (CAST).
- 33 CISCC, 2001: Second National Communication of Mexico to the UN Framework Convention on Climate Change.
34 <http://unfccc.int/resource/docs/natc/mexnc2.pdf>: Comité Intersecretarial Sobre Cambio Climático.
- 35 Conant, R. T., S. J. Del Grosso, W. J. Parton, and K. Paustian, 2005: Nitrogen pools and fluxes in grassland soils
36 sequestering carbon. *Nutrient Cycling in Agroecosystems*, **71**(3), 239–248.

- 1 Conant, R. T., and K. Paustian, 2002: Potential soil carbon sequestration in overgrazed grassland ecosystems. *Global*
2 *Biogeochemical Cycles*, **16**, 1143.
- 3 Cox, P. M., R. A. Betts, C. D. Jones, S. A. Spall, and I. J. Totterdell, 2000: Acceleration of global warming due to
4 carbon-cycle feedbacks in a coupled climate model. *Nature*, **408**, 184–187.
- 5 Davidson, E. A., and I. L. Ackerman, 1993: Change in soil carbon inventories following cultivation of previously
6 untilled soils. *Biogeochemistry*, **20**, 161–193.
- 7 Enquete Commission, 1995: *Protecting our Green Earth. How to Manage Global Warming Through*
8 *Environmentally Sound Farming and Preservation of the World's Forests*. Bonn: Economica Verlag.
- 9 EPA, 2000: *Options for reducing methane intermissions internationally*. 430-R-90-006, Washington, DC:
10 Environmental Protection Agency.
- 11 EPA, 2004: *Inventory of U.S. greenhouse gas emissions and sinks: 1990–2002*. Washington, DC: U.S.
12 Environmental Protection Agency.
- 13 EPA, 2005: *Inventory of U.S. greenhouse gas emissions and sinks: 1990–2003*. Washington, DC: U.S.
14 Environmental Protection Agency.
- 15 EPA, 2006: *Inventory of U.S. greenhouse gas emissions and sinks: 1990–2004*. Washington, DC: U.S.
16 Environmental Protection Agency.
- 17 Eve, M. D., M. Sperow, K. Paustian, and R. F. Follett, 2002: National-scale estimation of changes in soil carbon
18 stocks on agricultural lands. *Environmental Pollution*, **116**, 431–438.
- 19 Follett, R. F., J. M. Kimble, and R. Lal, 2001a: *The potential of U.S. grazing lands to sequester carbon and mitigate*
20 *the greenhouse effect*. Chelsea, MI: CRC Press.
- 21 Follett, R. F., E. G. Pruessner, S. Samson-Liebig, J. M. Kimble, and S. Waltman, 2001b: Carbon sequestration under
22 the Conservation Reserve Program in the historical grassland soils of the United States of America. In: *Soil*
23 *Management for Enhancing Carbon Sequestration* (R. Lal and K. McSweeney, eds. Pages 1–14 Madison, WI:
24 Soil Science Society of America.
- 25 Friedl, M. A., A. H. Strahler, X. Zhang, and J. Hodges, 2002: The MODIS land cover product: multi-attribute
26 mapping of global vegetation and land cover properties from time series MODIS data. *Proceedings of the*
27 *International Geoscience and Remote Sensing Symposium*, **4**, 3199–3201.
- 28 Frye, W. W., 1984: Energy requirements in no-tillage. In: *No tillage agricultural principles and practices* (R.E.
29 Phillips and S.H. Phillips, eds. Pages 127–151 New York: Van Nostrand Reinhold.
- 30 Giardina, C. P., and M. G. Ryan, 2000: Evidence that decomposition rates of organic carbon in mineral soil do not
31 vary with temperature. *Nature*, **404**, 858–861.
- 32 Gregorich, E. G., P. Rochette, A. J. VandenBygaart, and D. A. Angers, 2005: Greenhouse gas contributions of
33 agricultural soils and potential mitigation practices in Eastern Canada. *Soil & Tillage Research*, **83**(1), 53–72.
- 34 Helgason, B. L., H. H. Janzen, M. H. Chantigny, C. F. Drury, B. H. Ellert, E. G. Gregorich, R. L. Lemke, E. Pattey,
35 P. Rochette, and C. Wagner-Riddle, 2005: Toward improved coefficients for predicting direct N₂O emissions
36 from soil in Canadian agroecosystems. *Nutrient Cycling in Agroecosystems*, **72**(1), 87–99.

- 1 Houghton, R. A., and C. L. Goodale, 2004: Effects of land-use change on the carbon balance of terrestrial
2 ecosystems. *Ecosystem and land use change. Geophysical Monograph Series*, **53**, 85–96.
- 3 Houghton, R. A., J. L. Hackler, and K. T. Lawrence, 1999: The U.S. carbon budget: Contributions from land-use
4 change. *Science*, **285**, 574–578.
- 5 IPCC, 2001: Third Assessment Report. Cambridge, UK: Cambridge University Press.
- 6 ISRIC, 2002: FAO Soil Database. CD ROM Rome: International Soil and Reference Information System.
- 7 Jackson, R. B., J. L. Banner, E. G. Jobbagy, W. T. Pockman, and D. H. Wall, 2002: Ecosystem carbon loss with
8 woody plant invasion of grasslands. *Nature*, **418**(6898), 623–626.
- 9 Jenkinson, D. S., D. E. Adams, and A. Wild, 1991: Model estimates of CO₂ emissions from soil in response to
10 global warming. *Nature*, **351**, 304–306.
- 11 Johnson, K. A., and D. E. Johnson, 1995: Methane emissions from cattle. *Journal of Animal Science*, **73**, 2483–
12 2492.
- 13 Kätterer, T., M. Reichstein, O. Andren, and A. Lomander, 1998: Temperature dependence of organic matter
14 decomposition: a critical review using literature data analyzed with different models. *Biology and Fertility of*
15 *Soils*, **27**(3), 258–262.
- 16 Keppler, F., J. T. G. Hamilton, M. Brass, and T. Rockmann, 2006: Methane emissions from terrestrial plants under
17 aerobic conditions. *Nature*, **439**, 187–191.
- 18 Knorr, W., I. C. Prentice, J. I. House, and E. A. Holland, 2005: Long-term sensitivity of soil carbon turnover to
19 warming. *Nature*, **433**, 298–301.
- 20 Kulshreshtha, S. N., B. Junkins, and R. Desjardins, 2000: Prioritizing greenhouse gas emission mitigation measures
21 for agriculture. *Agricultural Systems*, **66**(3), 145–166.
- 22 Lal, R., 2002: Why carbon sequestration in agricultural soils? In: *Agricultural practices and policies for carbon*
23 *sequestration in soil* (J. Kimble, R. Lal, and R. F. Follett, eds. Pages 21–30 Boca Raton: CRC Press.
- 24 Lal, R., 2004: Carbon emission from farm operations. *Environment International*, **30**(7), 981–990.
- 25 Lal, R., R. F. Follett, and J. M. Kimble, 2003: Achieving soil carbon sequestration in the United States: A challenge
26 to policy makers. *Soil Science*, **168**, 827–845.
- 27 Lal, R., J. M. Kimble, R. F. Follett, and C. V. Cole, 1998: The potential of U.S. cropland to sequester carbon and
28 mitigate the greenhouse effect. Chelsea, MI: Ann Arbor Press.
- 29 Lewandrowski, J., M. Peters, C. Jones, R. House, M. Sperow, M. D. Eve, and K. Paustian, 2004: Economics of
30 Sequestering Carbon in the U.S. Agricultural Sector. Technical Bulletin No. TB 1909, Washington, DC:
31 Economic Research Service.
- 32 Lynch, D. H., R. D. H. Cohen, A. Fredeen, G. Patterson, and R. C. Martin, 2005: Management of Canadian prairie
33 region grazed grasslands: Soil C sequestration, livestock productivity and profitability. *Canadian Journal of Soil*
34 *Science*, **85**(2), 183–192.
- 35 Matin, A., P. Collas, D. Blain, C. Ha, C. Liang, L. MacDonald, S. McKibbin, C. Palmer, and R. Kerry, 2004:
36 Canada's Greenhouse Gas Inventory: 1990–2002. Greenhouse Gas Division, Environment Canada.

- 1 McCarl, B. A., and E. K. Schneider, 2001: The Cost of Greenhouse Gas Mitigation in U.S. Agriculture and Forestry.
2 Science, **294**, 2481–2482.
- 3 Mosier, A., C. Kroeze, C. Nevison, O. Oenema, S. Seitzinger, and O. van Cleemput, 1998a: Closing the global N₂O
4 budget: nitrous oxide emissions through the agricultural nitrogen cycle - OECD/IPCC/IEA phase II
5 development of IPCC guidelines for national greenhouse gas inventory methodology. Nutrient Cycling in
6 Agroecosystems, **52**(2-3), 225–248.
- 7 Mosier, A. R., J. M. Duxbury, J. R. Freney, O. Heinemeyer, K. Minami, and D. E. Johnson, 1998b: Mitigating
8 agricultural emissions of methane. Climatic Change, **40**, 39–80.
- 9 Murray, B. C., B. A. McCarl, and H. C. Lee, 2004: Estimating leakage from forest carbon sequestration programs.
10 Land Economics, **80**, 109–124.
- 11 Nabuurs, G.-J., N. H. Ravindranath, K. Paustian, A. Freibauer, B. Hohenstein, W. Makundi, H. Aalde, A. Y.
12 Abdelgadir, S. A. K. Anwar, J. Barton, K. Bickel, S. Bin-Musa, D. Blain, R. Boer, K. Byrne, C. C. Cerri,
13 L. Ciccarese, D.-C. Choque, E. Duchemin, L. Dja, J. Ford-Robertson, W. Galinski, J. C. Germon, H. Ginzo,
14 M. Gytarsky, L. Heath, D. Loustau, T. Mandouri, J. Mindas, K. Pingoud, J. Raison, V. Savchenko, D. Schone,
15 R. Sievanen, K. Skog, K. A. Smith, D. Xu, M. Bakker, M. Bernoux, J. Bhatti, R. T. Conant, M. E. Harmon,
16 Y. Hirakawa, T. Iehara, M. Ishizuka, E. G. Jobbagy, J. Laine, M. van der Merwe, I. K. Murthy, D. Nowak,
17 S. M. Ogle, P. Sudha, R.J. Scholes, and X. Zhang, 2004: LUCF-sector good practice guidance. In: IPCC Good
18 Practice Guidance for LULUCF (J. Penman, M. Gytarsky, T. Hirishi, T. Krug, and D. Kruger, eds. Hayama,
19 Japan: Institute for Global Environmental Strategies.
- 20 NAS, 2001: Climate change science: An analysis of some key questions. Washington, DC: National Academy of
21 Sciences, Committee on the science of climate change, National Research Council.
- 22 NRCS, 2005: Anaerobic Digestion Practice Standards. Washington, DC: U.S. Department of Agriculture.
- 23 Ogle, S. M., F. J. Breidt, M. D. Eve, and K. Paustian, 2003: Uncertainty in estimating land use and management
24 impacts on soil organic carbon storage for U.S. agricultural lands between 1982 and 1997. Global Change
25 Biology, **9**, 1521–1542.
- 26 Ogle, S. M., R. T. Conant, and K. Paustian, 2004: Deriving grassland management factors for a carbon accounting
27 method developed by the intergovernmental panel on climate change. Environmental Management, **33**(4), 474–
28 484.
- 29 Pacala, S. W., R. Bridsey, S. Bridgham, J. Caspersen, R. T. Conant, K. Davis, B. Hales, R. Houghton, J. C. Jenkins,
30 M. Johnston, G. Marland, K. Paustian, R. Socolow, R. Tol, and S. C. Wofsy. 2006. The North American carbon
31 budget past and present. *In Prep.*
- 32 Pacala, S. W., G. C. Hurtt, D. Baker, P. Peylin, R. A. Houghton, R. A. Birdsey, L. Heath, E. T. Sundquist, R. F.
33 Stallard, P. Ciais, P. Moorcroft, J. P. Casersen, E. Shevliakova, B. Moore, G. Kohlmaier, E. Holland, M. Gloor,
34 M. E. Harmon, S. M. Fan, J. L. Sarmiento, C. L. Goodale, D. Schimel, and C. B. Field, 2001: Consistent land-
35 and atmosphere-based U.S. carbon sink estimates. Science, **292**, 2316–2320.
- 36 Paustian, K., O. Andren, H. H. Janzen, R. Lal, P. Smith, G. Tian, H. Tiessen, M. Van Noordwijk, and P. L. Woomer,
37 1997: Agricultural soils as a sink to mitigate CO₂ emissions. Soil Use and Management, **13**, 230–244.

- 1 Paustian, K., J. M. Antle, J. Sheehan, and E. A. Paul, 2006: Agriculture's Role in Greenhouse Gas Mitigation.
2 Washington, DC: Pew Center on Global Climate Change.
- 3 Paustian, K., C. V. Cole, D. Sauerbeck, and N. Sampson, 1998: CO₂ mitigation by agriculture: An overview.
4 *Climatic Change*, **40**(1), 135–162.
- 5 Peoples, M. B., E. W. Boyer, K. W. T. Goulding, P. Heffer, V. A. Ochwoh, B. Vanlauwe, S. Wood, K. Yagi, and
6 O. van Cleemput, 2004: Pathways of nitrogen loss and their impacts on human health and the environment. In:
7 *Agriculture and the Nitrogen Cycle* (A. R. Mosier, J. K. Syers, and J. R. Freney, eds. Pages 53–69 Washington:
8 Island Press.
- 9 Pimentel, D., P. Hepperly, J. Hanson, D. Doude, and R. Seidel, 2005: Environmental, energetic, and economic
10 comparisons of organic and conventional farming systems. *Bioscience*, **55**(7), 573–582.
- 11 Post, W. M., and K. C. Kwon, 2000: Soil carbon sequestration and land-use change: Processes and potential. *Global*
12 *Change Biology*, **6**, 317–327.
- 13 Raymond, P. A., and J. J. Cole, 2003: Increase in the export of alkalinity from North America's largest river.
14 *Science*, **301**, 88–91.
- 15 Robertson, G. P., and P. R. Grace, 2004: Greenhouse gas fluxes in tropical and temperate agriculture: The need for a
16 full-cost accounting of global warming potentials. *Environment, Development and Sustainability*, **6**, 51–63.
- 17 Robertson, G. P., E. A. Paul, and R. R. Harwood, 2000: Greenhouse gases in intensive agriculture: Contributions of
18 individual gases to the radiative forcing of the atmosphere. *Science*, **289**, 1922–1925.
- 19 Sheinbaum, C., and O. Masera, 2000: Mitigating Carbon Emissions while Advancing National Development
20 Priorities: The Case of Mexico. *Climatic Change*, **47**(3), 259–282.
- 21 Six, J., S. M. Ogle, F. J. Briedt, R. T. Conant, A. R. Mosier, and K. Paustian, 2004: The potential to mitigate global
22 warming with no-tillage management is only realized when practiced in the long term. *Global Change Biology*,
23 **10**(2), 155–160.
- 24 Smith, K. A., and F. Conen, 2004: Impacts of land management on fluxes of trace greenhouse gases. *Soil Use and*
25 *Management*, **20**, 255–263.
- 26 Smith, P., K. W. Goulding, K. A. Smith, D. S. Powlson, J. U. Smith, P. Falloon, and K. Coleman, 2001a: Enhancing
27 the carbon sink in European agricultural soils: including trace gas flux estimates of carbon mitigation potential.
28 *Nutrient Cycling in Agroecosystems*, **60**, 237–252.
- 29 Smith, S. V., R. O. Slezzer, W. H. Renwick, and R. W. Buddemeier, 2005: Fates of eroded soil organic carbon:
30 Mississippi basin case study. *Ecological Applications*, **15**(6), 1929–1940.
- 31 Smith, W. N., R. L. Desjardins, and B. Grant, 2001b: Estimated changes in soil carbon associated with agricultural
32 practices in Canada. *Canadian Journal of Soil Science*, **81**(2), 221–227.
- 33 Smith, W. N., P. Rochette, C. Monreal, R. L. Desjardins, E. Pattey, and A. Jaques, 1997: The rate of carbon change
34 in agricultural soils in Canada at the landscape level. *Canadian Journal of Soil Science*, **77**(2), 219–229.
- 35 Sobool, D., and S. Kulshreshtha, 2005: Greenhouse Gas Emissions from Agriculture and Agri-Food Systems in
36 Canada. Saskatoon: Department of Agricultural Economics, University of Saskatchewan.

- 1 Sombroek, W. G., F. O. Nachtergaele, and A. Hebel, 1993: Amounts, dynamics and sequestering of carbon in
2 tropical and subtropical soils. *Ambio*, **22**(7), 417–426.
- 3 Sperow, M., M. D. Eve, and K. Paustian, 2003: Potential soil C sequestration on U.S. agricultural soils. *Climatic*
4 *Change*, **57**, 319–339.
- 5 Van Auken, O. W., 2000: Shrub invasions of North American semiarid grasslands. *Annual Review of Ecology and*
6 *Systematics*, **31**, 197–205.
- 7 VandenBygaart, A. J., E. G. Gregorich, and D. A. Angers, 2003: Influence of agricultural management on soil
8 organic carbon: A compendium and assessment of Canadian studies. *Canadian Journal of Soil Science*, **83**(4),
9 363–380.
- 10 West, T. O., and G. Marland, 2003: Net carbon flux from agriculture: Carbon emissions, carbon sequestration, crop
11 yield, and land-use change. *Biogeochemistry*, **63**, 73–83.
- 12 West, T. O., and M. K. Wali, 2002: Modeling regional carbon dynamics and soil erosion in disturbed and
13 rehabilitated ecosystems as affected by land use and climate. *Water Air and Soil Pollution*, **138**(1-4), 141–163.
- 14 Yoo, K., R. Amundson, A. M. Heimsath, and W. E. Dietrich, 2005: Erosion of upland hillslope soil organic carbon:
15 Coupling field measurements with a sediment transport model. *Global Biogeochemical Cycles*, **19**(3), GB3003.
16

1 **[START OF TEXT BOX 1]**

2
3 **Nitrous oxide (N₂O) emissions from agricultural and grazing lands**

4
5 Nitrous oxide (N₂O) is the most potent greenhouse gas in terms of global warming potential, with a radiative
6 forcing 296 times that of CO₂ (IPCC, 2001). Agricultural activities that add mineral or organic nitrogen—
7 fertilization, plant N₂ fixation, manure additions, etc.—augment naturally occurring N₂O emissions from
8 nitrification and denitrification by 0.0125 kg N₂O per kg N applied (Mosier *et al.*, 1998a). Agriculture contributes
9 significantly to total global N₂O fluxes through soil emissions (35% of total global emissions), animal waste
10 handling (12%), nitrate leaching (7%), synthetic fertilizer application (5%), grazing animals (4%), and crop residue
11 management (2%). Agriculture is the largest source of N₂O in the United States (78% of total N₂O emissions),
12 Canada (59%), and Mexico (76%).

13
14 **[END OF TEXT BOX 1]**

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16
17
18
19 **[START OF TEXT BOX 2]**

20
21 **Inorganic soil carbon in agricultural and grazing ecosystems**

22
23 Inorganic carbon in the soil is comprised of primary carbonate minerals, such as calcite (CaCO₃) or dolomite
24 [CaMg(CO₃)₂], or secondary minerals formed when carbonate (CO₃²⁻), derived from soil CO₂, combines with base
25 cations (e.g., Ca²⁺, Mg²⁺) and precipitates within the soil profile in arid and semi-arid ecosystems. Weathering of
26 primary carbonate minerals in humid regions is a source of CO₂, whereas formation of secondary carbonates in drier
27 areas is a sink for CO₂; however, the magnitude of either flux is highly uncertain. Agricultural liming involves
28 addition of primary carbonate minerals to the acid soils to increase the pH. In the United States, about 1 Mt C yr⁻¹ is
29 emitted from liming (EPA, 2006).

30
31 **[END OF TEXT BOX 2]**

1 *[START OF TEXT BOX 3]*

2
3 **Impacts of woody encroachment into grasslands on ecosystem carbon stocks**

4
5 Encroachment of woody species into grasslands—caused by overgrazing-induced reduction in grass biomass
6 and subsequent reduction or elimination of grassland fires—is widespread in the United States and Mexico,
7 decreases forage production, and is unlikely to be reversed without costly mechanical intervention (Van Auken,
8 2000). Encroachment of woody species into grassland tends to increase biomass carbon stocks by $1 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$
9 (Pacala *et al.*, 2001), with estimated net sequestration of $0.12\text{--}0.13 \text{ Gt C yr}^{-1}$ in encroaching woody biomass
10 (Houghton *et al.*, 1999; Pacala *et al.*, 2001). In response to woody encroachment, soil carbon stocks can significantly
11 increase or decrease, thus predicting impacts on soil carbon or ecosystem carbon stocks is very difficult (Jackson *et*
12 *al.*, 2002).

13
14 *[END OF TEXT BOX 3]*

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18
19 *[START OF TEXT BOX 4]*

20
21 **Agricultural and grazing land N₂O emission reductions**

22
23 When mineral soil nitrogen content is increased by nitrogen additions (i.e., fertilizer), a portion of that nitrogen
24 can be transformed to N₂O as a byproduct of two microbiological processes (nitrification and denitrification) and
25 lost to the atmosphere. Coincidental introduction of large amounts of easily decomposable organic matter and NO₃⁻
26 from either a plow down of cover crop or manure addition greatly stimulates denitrification under wet conditions
27 (Peoples *et al.*, 2004). Some practices intended to sequester atmospheric carbon in soil could prompt increases in
28 N₂O fluxes. For example, reducing tillage intensity tends to increase soil moisture, leading to increased N₂O fluxes,
29 particularly in wetter environments (Six *et al.*, 2004). Synchronizing organic amendment applications with plant
30 nitrogen uptake and minimizing manure storage under anoxic conditions can reduce N₂O emissions by 10–25% and
31 will increase nitrogen use efficiency which can decrease indirect emissions (in waterways) by 5–20% (CAST, 2004).

32
33 *[END OF TEXT BOX 4]*

1

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

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)

^aCropland 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

^bTemperate zones are those located above 30° latitude. Tropical zones (<30° latitude) include subtropical regions.

^cDry climates were defined as those where the ratio of mean annual precipitation (MAP) to potential evapotranspiration (PET) is less than 1; in wet areas, MAP/PET >1.

1

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

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 CH₄ to CO₂ equivalents (IPCC, 2001) and a factor of 12/44 to convert from CO₂ to carbon

	Canada	Mexico	United States	Total
	Mt C yr ⁻¹			
CO₂				
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			
CH₄				
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

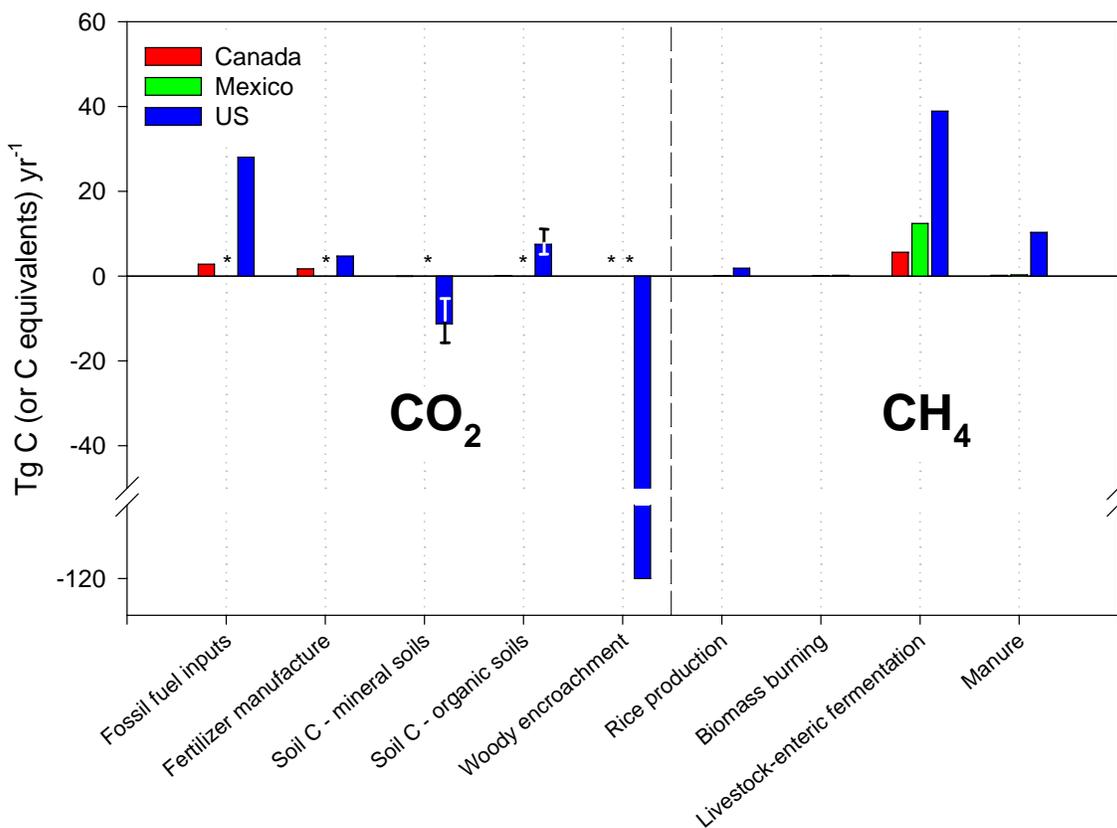
ND = no data reported.

^aFrom Sobool and Kulshreshtha (2005).

^bFrom Lal *et al.* (1998).

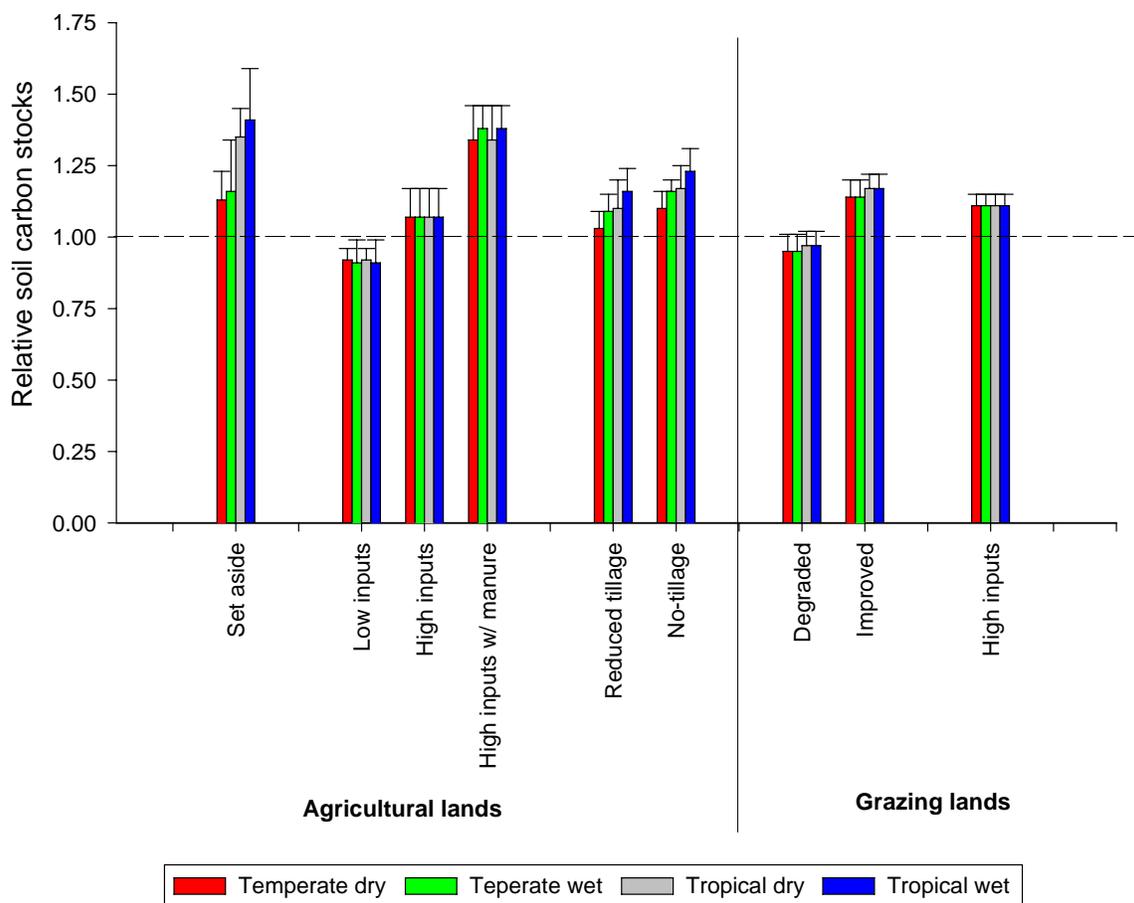
^cFrom Houghton *et al.* (1999).

1



2
 3 **Figure 10-1. North American agricultural and grazing land CO₂ (left side) and methane (right side)**
 4 **fluxes for the years around 2000.** Negative values indicate net flux from the atmosphere to soil and biomass
 5 carbon pools. All data are from Canadian (Matin *et al.*, 2004) and U.S. (EPA, 2005) National Inventories and from
 6 the second Mexican National Communication (CISCC, 2001), except for Canadian [from Kulshreshtha *et al.* (2000)]
 7 and U.S. fossil fuel inputs [from Lal *et al.* (1998)] and woody encroachment [from Houghton *et al.* (1999)]. Values
 8 are for 2003 for the United States and Canada and 1998 for Mexico. A global warming potential of 23 for methane
 9 was used to convert emissions of CH₄ to CO₂ equivalents (IPCC, 2001) and a factor of 12/44 to convert from CO₂ to
 10 carbon. Asterisks indicate unavailable data. Data ranges are indicated by error bars where available.

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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).

Chapter 11. North American Forests

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KEY FINDINGS

- North American forests contain more than 170 Gt of carbon, of which 28% is in live biomass and 72% is in dead organic matter.
- North American forests were a sink of approximately 350 Mt C yr⁻¹ for the decade of the 1990s. This number is highly uncertain.
- There is general understanding that forests of North America were a source of CO₂ 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 CO₂, 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.

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

12 INTRODUCTION

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

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

19
20 North American forests are critical components of the global carbon cycle, exchanging large amounts
21 of CO₂ and other gases with the atmosphere and oceans. Forests and wood products constitute more than
22 60% of the total annual carbon sink on land in North America ($-557 \text{ Mt C yr}^{-1}$; see Chapter 3), including
23 the -23 Mt C yr^{-1} stored in land defined by the census as urban and suburban trees in the United States. In
24 this chapter we present the most recent estimates of the role of forests in the North American carbon
25 balance, describe the main factors that affect forest carbon stocks and fluxes, and discuss management
26 options and research needs.

28 CARBON STOCKS AND FLUXES

29 Ecosystem Carbon Stocks And Pools

30 North American forests contain more than 170 Gt of carbon, of which 28% is in live biomass and
31 72% is in dead organic matter (Table 11-2). Among the three countries, Canada's forests contain the most
32 carbon and Mexico's forests the least.

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

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

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

11 In Mexico, in unmanaged forested areas, temperate forests contain 4,500 Mt C, tropical forests
12 contain 4,100 Mt C, and semiarid forests contain 5,000 Mt C. In forest plantations 800 Mt C are
13 sequestered in long and short rotations, restoration, and bioenergy plantations. Managed temperate and
14 tropical forests store 500 Mt C, and protected forests store 2,000 Mt C. Agroforestry systems harbor
15 100 MtC.

17 **Net North American Forest Carbon Fluxes**

18 According to nearly all published studies, North American lands are a net carbon sink (Pacala *et al.*,
19 2001); however, the magnitude of the Canadian and Mexican forest contribution to the land carbon sink is
20 categorized as highly uncertain (meaning there is 95% certainty that the actual value is within ±100% of
21 the reported estimate). The estimated carbon sink of the United States forests is categorized as uncertain
22 (meaning that there is a 95% certainty that the actual value is within 50% of the reported estimate.) A
23 summary of currently available data from greenhouse gas inventories and other sources suggests that the
24 magnitude of the North American forest carbon sink was approximately –350 Mt C yr⁻¹ for the decade of
25 the 1990s (Table 11-3).

27 **Table 11-3. Change in carbon stocks for forests and wood products by country (Mt C yr⁻¹).**

29 Canadian forests and forest products may be a net sink of about –109 Mt C yr⁻¹ (Table 11-3). These
30 estimates pertain to the area of forest considered to be “managed” under international reporting
31 guidelines, which is 53% of the total area of Canada’s forests. The estimates also include the carbon
32 changes that result from land-use change. Changes in forest soil carbon are not included. High interannual
33 variability is averaged into this estimate—the annual change varied from approximately –190 Mt C in
34 1990 to –70 Mt C in 2003 (Environment Canada, 2005).

1 In the United States, forest ecosystem carbon stocks are estimated to be a net sink of $-236 \text{ Mt C yr}^{-1}$,
2 and for wood products, the estimated sink is -57 Mt C yr^{-1} (Table 11-3). Most of the net sink is in
3 aboveground carbon pools, which account for $-146 \text{ Mt C yr}^{-1}$ (Smith and Heath, 2005). The net sink for
4 the belowground carbon pool is estimated at -90 Mt C (Pacala *et al.*, 2001). The size of the carbon sink in
5 U.S. forest ecosystems appears to have declined slightly over the last decade (Smith and Heath, 2005). In
6 contrast, a steady or increasing supply of timber products now and in the foreseeable future (Haynes,
7 2003) means that the rate of increase in the wood products carbon pool is likely to remain steady.

8 For Mexico, the most comprehensive available estimate for the forest sector suggests a source of
9 $+52 \text{ Mt C}$ per year (Masera *et al.*, 1997). This estimate does not include changes in the wood products
10 carbon pool. The main cause of the estimated source is deforestation, which is offset to a much lesser
11 degree by restoration and recovery of degraded forestland.

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

19 **TRENDS AND DRIVERS**

20 **Overview of Trends and Drivers of Change in Carbon Stocks**

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

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

1 Canada's forests a net carbon sink in the same period (Chen *et al.*, 2003). In the United States between
2 1953 and 1997, carbon stocks in forest ecosystems (excluding soils) increased by about 175 Mt C yr⁻¹,
3 and for the approximate year 2000, the average annual increase in forest ecosystem carbon stocks is
4 146 Mt C yr⁻¹ (Smith and Heath, 2005). This declining trend is based mainly on dynamics of vegetation
5 change following a long history of land-use change and management (Birdsey *et al.*, 2006). Mexico emits
6 52.3 Mt C yr⁻¹ as a consequence of land use change, including deforestation, forest degradation, forest
7 fires and forest regeneration (Masera *et al.* 1997; de Jong *et al.*, 2000). These driving factors are expected
8 to continue influencing forests in the near future.

10 **Effects of Land-Use Change**

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

16 In the last century more than 130 million hectares of land in the conterminous United States were
17 either afforested (62 million ha) or deforested (70 million ha) (Birdsey and Lewis 2003). Even though the
18 net change in the area of forest land was not significant during that time, the magnitude of the shifts in
19 land use caused significant redistribution of carbon stocks among land categories. Over the longer term,
20 Houghton *et al.* (1999) estimated that cumulative changes in forest carbon stocks for the period from
21 1700 to 1990 in the United States were about +25 Gt C, primarily from conversion of forestland to
22 agricultural use and reduction of carbon stocks for wood products.

23 Mexican forests emit +50 to +62 Mt C yr⁻¹ to the atmosphere as a consequence of land use change
24 (Masera *et al.*, 1997). In Mexico, deforestation and forest degradation were responsible for an annual
25 forest loss of 720,000 ha in the late 1980s and early 1990s (Masera *et al.*, 1997). The deforestation rate of
26 unmanaged forests was about 619,000 ha per year in 1990; however, based on total forest cover change
27 between 1993 and 2000, Palacio *et al.* (2000) estimated a deforestation rate of 880,000 ha yr⁻¹.
28 Deforestation is primarily driven by conversion of tropical forest to pastures (73% of deforested tropical
29 evergreen forest, and 61% of deforested tropical deciduous forest, Masera *et al.*, 2001). About 13 to 15%
30 of deforested land gets converted to agricultural land (Masera *et al.*, 2001). The highest deforestation rates
31 occur in the tropical deciduous forests (304,000 ha in 1990) and the lowest in temperate broadleaf forests
32 (59,000 ha in 1990) (Masera *et al.*, 2001). Carbon fluxes in tropical rainforests in La Selva Lacandona
33 resulting from a 31% reduction of closed forest cover between 1976 and 1996 correspond to total

1 emissions of 41.7 ± 12.1 Mt C [95% confidence interval (CI)] with 31.9 ± 7.0 Mt C (95% CI) from
2 vegetation and 9.5 ± 10.4 Mt C (95% CI) from soils (de Jong *et al.*, 2000).

4 **Effects of Forest Management**

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

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

10
11 Forests and other wooded land in Canada occupy about 404 Mha, of which 214 Mha (53%) are under
12 active forest management (Environment Canada 2005). Managed forests are considered to be under the
13 direct influence of human activity and not reserved. Less than 1% of the area under active management is
14 harvested annually. Apps *et al.* (1999) used a carbon budget model to simulate carbon in harvested wood
15 products (HWP) for Canada. Approximately 800 Mt C were stored in the Canadian HWP sector in 1989,
16 of which 50 Mt C were in imported wood products, 550 Mt C in exported products, and 200 Mt C in
17 wood products produced and consumed domestically.

18 Between 1990 and 2000, about 4 Mha yr⁻¹ were harvested in the U.S., two-thirds by means of some
19 form of partial-cut harvest and one-third by a clearcut method (Birdsey and Lewis 2003). Between 1987
20 and 1997, about 1 Mha yr⁻¹ were planted with trees, and about 800,000 ha were treated to improve the
21 quality and/or quantity of timber produced (Birdsey and Lewis 2003). Harvesting in U.S. forests accounts
22 for substantially more tree mortality than natural causes such as wildfire and insect outbreaks (Smith *et*
23 *al.*, 2004). In 2002, about 170 Mt C of tree biomass were removed from forests by harvest, offset by 280
24 Mt C of net primary productivity (which includes growth and mortality from natural causes), making U.S.
25 tree biomass a net sink of -110 Mt C yr⁻¹ (Smith and Heath 2005). The harvested wood resulted in
26 -57 Mt C added to landfills and products in use, and an additional 88 Mt C were emitted from harvested
27 wood burned for energy (Skog and Nicholson 1998).

28 About 80% of the forested area in Mexico is socially owned by communal land grants (*ejidos*) and
29 rural communities. About 95% of timber harvesting occurs in native temperate forests (SEMARNAP
30 1996). Extensive overexploitation (e.g., illegal deforestation and fuelwood extraction) of natural resources
31 from forests have caused dramatic land degradation in forested land (21.4 Mha affected in 1990). It is
32 estimated that illegal wood extraction reaches 13.3 million m³ of wood every year (Torres 2004). Unlike
33 U.S. and Canadian forests, Mexican forests have been affected since pre-Columbian times by the almost
34 ubiquitous influence of a large proportion of the rural population, which controls the carbon fluxes and

1 stocks through fire; wood extraction; legal and illegal logging; shifting agriculture practices; and
2 conversion of land to plantations (e.g., coffee), fields for agricultural crops (e.g., sugar cane), and
3 pastures. Also, the differences in property rights, land ownership, and associated management policies
4 (and lack thereof), which are preeminently important in Mexico, where most of the land is communal,
5 also contribute to different socioeconomic controls over the carbon cycle.

7 **Effects of Climate and Atmospheric Chemistry**

8 Some studies indicate that the combined effects of climate and atmospheric chemistry changes on
9 carbon sequestration are likely to be significantly smaller than the effects of land management and land-
10 use change (Caspersen *et al.*, 2000; Schimel *et al.*, 2000), but existing monitoring and modeling
11 capability is still somewhat inadequate for a definitive assessment of the relative importance of these
12 factors (U.S. Climate Change Science Program 2003). Environmental factors, including climate
13 variability, nitrogen deposition, tropospheric ozone, and elevated CO₂, have been recognized as
14 significant factors affecting the carbon cycle of forests (Aber *et al.*, 2001; Ollinger *et al.*, 2002).
15 Experimental studies have clearly shown that rising atmospheric CO₂ increases photosynthesis in plants.
16 Recent reviews of ecosystem-scale studies known as Free Air CO₂ Exchange (FACE) experiments
17 suggest an increase in net primary productivity (NPP) of 12–23% over all species (Norby *et al.*, 2005;
18 Nowak *et al.*, 2004). However, at the ecosystem scale, it is uncertain whether this effect results in a
19 lasting increase in sequestered carbon or causes a more rapid cycling of carbon between the ecosystem
20 and the atmosphere (Korner *et al.*, 2005; Lichter 2005). Experiments have also shown that the effects of
21 rising CO₂ are significantly moderated by increasing tropospheric ozone (Karnosky *et al.*, 2003; Loya *et al.*,
22 2003). When nitrogen is also considered, reduced soil fertility limits the response to rising CO₂, but
23 nitrogen deposition can increase soil fertility to counteract that effect (Johnson *et al.*, 1998; Oren *et al.*,
24 2001).

26 **Effects of Natural Disturbances**

27 Wildfires were the largest disturbance in the twentieth century in Canada (Weber and Flannigan,
28 1997). In the 1980s and 1990s, the average total burned area was 2.6 Mha yr⁻¹ in Canada's forests, with a
29 maximum 7.6 Mha yr⁻¹ in 1989. Carbon emissions from forest fires are substantial and arise mostly from
30 northern forests (boreal, subarctic). Emissions range from less than +1 Mt C yr⁻¹ in the interior of British
31 Columbia to more than +10 Mt C yr⁻¹ in the western boreal forest. Total emissions from forest land in
32 Canada averaged approximately +27 Mt C yr⁻¹ between 1959 and 1999 (Amiro *et al.*, 2001). Much of the
33 Canadian forest is expected to experience increases in fire severity (Parisien *et al.*, 2005) and burn areas
34 (Flannigan *et al.*, 2005). Outbreaks of forest pests are also likely (Volney and Hirsch, 2005). While some

1 of this disturbance may be reduced through enhanced suppression efforts, a long-term increase in impacts
2 of disturbance is likely in the future, with associated losses of forest carbon stocks.

3 Estimated carbon emissions from four major insect pests in Canadian forests (spruce budworm, jack
4 pine budworm, hemlock looper, and mountain pine beetle) varied from +5 to 10 Mt C yr⁻¹ in the 1970s to
5 less than +2 Mt C yr⁻¹ in the mid-1990s¹. Large emissions occurred in the 1970s and early 1980s as a
6 result of extremely large spruce budworm outbreaks in Ontario and Quebec (18 to 30 Mha in each
7 province). The area of outbreaks and associated carbon emissions has recently increased as a result of the
8 mountain pine beetle outbreak in British Columbia, which affected 3.7 Mha in 2003, when emissions
9 were approximately +4 Mt C yr⁻¹.

10 Natural disturbance is commonplace in U.S. forests, where insects, diseases, and wildfire combined
11 affect more than 30 Mha per decade (Birdsey and Lewis 2003). Damage from weather events (hurricanes,
12 tornados, ice storms) may exceed 20 Mha per decade (Dale *et al.*, 2001). There are few estimates of the
13 impact of selected natural disturbances on carbon pools of temperate forests. McNulty (2002) estimated
14 that large hurricanes in the United States could convert 20 Mt C of live biomass into detrital carbon pools.
15 The impacts of fire are clearly significant. According to one estimate, the average annual carbon
16 emissions from biomass burning in the contemporary United States ranges from 9 to 59 Mt C (Leenhouts
17 1998).

18 Pests and diseases are important natural disturbance agents in temperate forests of Mexico; however,
19 no statistics exist on the extent of the affected land area. The number and area of sites affected by forest
20 fires in Mexico have fluctuated considerably between 1970 and 2002 with a clear tendency of an
21 increasing number of fire events (4,000–7,000 in the 1970s and 1,800–15,000 in the 1990s), and overall,
22 larger areas are being affected (0.08–0.25 Mha in 1970s and 0.05–0.85 Mha in 1990s). During El Nino
23 years, increasing drought increases fire frequencies (Torres 2004). Between 1995 and 2000, an average
24 8,900 fire events occurred per year and affected about 327,000 ha of the forested area. Currently, no
25 estimates are available on the contribution of these fires to CO₂ emissions.

27 **Projections of Future Trends**

28 There have been several continental- to subcontinental-scale assessments of future changes in carbon
29 and vegetation distribution in North America (VEMAP Members, 1995; Pan *et al.*, 1998; Neilson *et al.*,
30 1998; Joyce *et al.*, 2001). For the conterminous United States, the VEMAP study suggested that under
31 most future climate conditions, NPP would respond positively to changing climate (20.8% ± 2.4%) but

¹These 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).

1 that total carbon storage would remain relatively constant ($2.0\% \pm 3.5\%$). Under most climate scenarios
2 the West gets wetter; when coupled with higher CO₂ and longer growing seasons, simulations show
3 woody expansion and increased sequestration of carbon as well as increases in fire (Bachelet *et al.*, 2001).
4 However, recent scenarios from the Hadley model show some drying in the Northwest, which produces
5 some forest decline (Price *et al.* 2004). Many simulations show continued growth in eastern forests
6 through the end of the twenty-first century while others show the opposite, especially in the Southeast.
7 Eastern forests could experience a period of enhanced growth in the early stages of warming, due to
8 elevated CO₂, increased precipitation, and a longer growing season. However, further warming could
9 bring on increasing drought stress, reducing the carrying capacity of the ecosystem and causing carbon
10 losses through drought-induced dieback and increased fire and insect disturbances.

11 Large portions of the Canadian and Alaskan forest are expected to be particularly sensitive to climate
12 change due to its high latitude and interior continental location (Hogg and Bernier, 2005). Climate change
13 effects on forest growth could be positive (e.g., increased rates of photosynthesis and increased water use
14 efficiency) or negative (decreased water availability, higher rates of respiration) (Baldocchi and Amthor,
15 2001). It is difficult to predict the direction of these changes and they will likely vary by species and local
16 conditions of soils and topography (Johnston and Williamson, 2005). Because of the large area of boreal
17 forests and expected high degree of warming, Canada and Alaska require close monitoring over the next
18 few decades as these areas will likely be critical to determining the carbon balance of North America.

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

24

25 **OPTIONS FOR MANAGEMENT**

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

32 Although the science of managing forests specifically for carbon sequestration is not well developed,
33 some management principles are emerging to guide management decisions (Appendix 11.B). The
34 prospective role of forestry in helping to stabilize atmospheric CO₂ depends on harvesting and

1 disturbance rates, expectations of future forest productivity, the fate and longevity of forest products, and
2 the ability to deploy technology and forest practices to increase the retention of sequestered CO₂. Market
3 factors are also important in guiding the behavior of the private sector. The forest sector includes a variety
4 of activities that can contribute to increasing carbon sequestration, including: afforestation, mine land
5 reclamation, forest restoration, agroforestry, forest management, biomass energy, forest preservation,
6 wood products management, and urban forestry (Birdsey *et al.*, 2000).

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

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

21 In a separate modeling study using the CBM-CFS model, Kurz *et al.* (1998) investigated the impacts
22 on landscape-level carbon storage of the transition from natural to managed disturbance regimes. For a
23 boreal landscape in northern Quebec, a simulated fire disturbance interval of 120 yr was replaced by a
24 harvest cycle of 120 yr. The net impact was that the average age of forests in the landscape declined from
25 110 yr to 70 yr, and total carbon storage in forests declined from 16.3 to 14.8 Mt C (including both
26 ecosystem and forest products pools). In this case the disturbance frequencies were the same, so the
27 decline in carbon storage occurred because the harvesting regime preferentially selected older, high-
28 biomass-density stands.

29 Market approaches and incentive programs to manage greenhouse gases, particularly CO₂, are under
30 development in the United States, the European Union, and elsewhere (Totten, 1999). Since forestry
31 activities have highly variable costs because of site productivity and operational variability, most recent
32 studies of forestry potential develop “cost curves,” i.e., estimates of how much carbon will be sequestered
33 by a given activity for various carbon prices (value in a market system) or payments (in an incentive
34 system). There is also a temporal dimension to the analyses because the rate of change in forest carbon

1 stocks is variable over time, with forestry activities tending to have a high initial rate of net carbon
2 sequestration followed by a lower or even a negative rate as forests reach advanced age.

3 Here we address costs of three broad categories of forestry activities: afforestation (conversion of
4 agricultural land to forest), improved management of existing forests, and use of woody biomass for fuel.
5 In general, analyses suggest that improved management of existing forestlands may be attractive to
6 landowners at a carbon prices below \$10 per ton of CO₂, that afforestation requires a moderate price of
7 \$15 per ton of CO₂ or more to induce landowners to participate, and that biofuels become dominant at
8 prices of \$30 to \$50 per ton of CO₂ (Lewandrowski, 2004; Stavins and Richards, 2005; Environmental
9 Protection Agency, 2005). Table 11-5 shows a simple scenario of emissions reduction below baseline,
10 annualized over the time period from 2010 to 2110, for forestry activities as part of a bundle of reduction
11 options for the land base.

12
13 **Table 11-5. Illustrative emissions reduction potential of various forestry activities in the United**
14 **States under a range of prices and sequestration rates.**

15
16 Co-benefits are vitally important for inducing good forest carbon management. For example,
17 conversion of agricultural land to forest will generally have positive effects on water, air, and soil quality
18 and on biodiversity. In practice, some forest carbon sequestration projects have already been initiated
19 even though sequestered carbon has little current value (Winrock International, 2005). In many of the
20 current projects, carbon is a secondary objective that supports other landowner interests, such as
21 restoration of degraded habitat. But co-effects may not all be beneficial. Water quantity may decline
22 because of increased transpiration by trees relative to other vegetation. And taking land out of crop
23 production may affect food prices—at higher carbon prices, nearly 40 million ha may be converted from
24 cropland to forest (Environmental Protection Agency, 2005). Implementation of a forest carbon
25 management policy will need to carefully consider co-effects, both positive and negative.

26 27 **DATA GAPS AND INFORMATION NEEDS FOR DECISION SUPPORT**

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

32

1 Major Data Gaps in Estimates of Carbon Pools and Fluxes

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

9 In the United States, the range of estimates of the size of the land carbon sink is between 0.30 and
10 0.58 Mt C yr⁻¹ (Pacala *et al.*, 2001). Significant data gaps among carbon pools include carbon in wood
11 products, soils, woody debris, and water transport (Birdsey 2004; Pacala *et al.*, 2001). Geographic areas
12 that are poorly represented in the available data sets include much of the Intermountain Western United
13 States and Alaska, where forests of low productivity have not been inventoried as intensively as more
14 productive timberlands (Birdsey 2004). Accurate quantification of the relative magnitude of various
15 causal mechanisms at large spatial scales is not yet possible, given the limitations of our ability to
16 combine various approaches and data sets: large-scale observations, process-based modeling, ecosystem
17 experiments, and laboratory investigations (Foley and Ramankutty, 2004).

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

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

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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.

- 1 Amiro, B.D., A.G. Barr, T.A. Black, H. Iwashita, N. Kljun, J.H. McCaughey, K. Morgenstern, S. Murayama, Z.
2 Nesic, A.L. Orchansky, and N. Saigusa. 2005. Carbon, energy and water fluxes at mature and disturbed forest
3 sites, Saskatchewan, Canada. *Agric. For. Meteorol.* (in press).
- 4 Apps, M.J., W.A. Kurz, S.J. Beukema and J.S. Bhatti. 1999. Carbon Budget of the Canadian Forest Product Sector.
5 *Environmental Science & Policy* 2: 25-41.
- 6 Arain, M.A. and Restrepo-Coupe N. 2005. Net ecosystem production in an eastern white pine plantation in southern
7 Canada. *Agricultural and Forest Meteorology* 128: 223-241.
- 8 Bachelet, Dominique, Ronald P. Neilson, James M. Lenihan, and Raymond J. Drapek. 2001. Climate change effects
9 on vegetation distribution and carbon budget in the United States. *Ecosystems* 4: 164-185.
- 10 Baldocchi, D.D. and J.S. Amthor. 2001. Canopy photosynthesis: history, measurements and models. Pp. 9-31 in
11 Roy, J., B. Saugier and H. Mooney, editors. *Terrestrial Global Productivity*. Academic Press, San Diego, USA.
- 12 Barford CC, Wofsy SC, Goulden ML, Munger JW, Pyle EH, Urbanski SP, Hutryra L, Saleska SR, Fitzjarrald D,
13 Moore K. 2001. Factors controlling long- and short-term sequestration of atmospheric CO₂ in a mid-latitude
14 forest. *Science* 294: 1688-1691.
- 15 Bechtold, William A. and Paul L. Patterson (Eds). 2005. The enhanced forest inventory and analysis program –
16 national sampling design and estimation procedures. Gen. Tech. Rep. SRS-80. Asheville, NC: U.S. Department
17 of Agriculture, Forest Service, Southern Research Station. 85 p.
- 18 Birdsey, R.A. and L.S. Heath. 1995. Carbon changes in U.S. forests. In: *Climate change and the productivity of*
19 *America's forests*, (L.A. Joyce, ed.). U. S. Department of Agriculture, Forest Service, Rocky Mountain Forest
20 and Range Experiment Station General Technical Report, Ft. Collins, CO, p. 56-70.
- 21 Birdsey, Richard A., Ralph Alig, and Darius Adams. 2000. Mitigation activities in the forest sector to reduce
22 emissions and enhance sinks of greenhouse gases. In: L.A. Joyce and R.A. Birdsey (eds.) *The impact of climate*
23 *change on America's forests: a technical document supporting the 2000 USDA Forest Service RPA assessment*.
24 RMRS-GTR-59. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. Fort
25 Collins, CO. 112-131.
- 26 Birdsey, Richard A. 2004. Data gaps for monitoring forest carbon in the United States: an inventory perspective. In:
27 Mickler, Robert A., eds. *Environmental Management*. 33(Supplement 1): S1-S8.
- 28 Birdsey, Richard A., and George M. Lewis. 2003. Current and historical trends in use, management, and disturbance
29 of U.S. forestlands. In: J.M. Kimble, L.S. Heath, and R.A. Birdsey, (eds.) *The potential of U.S. forest soils to*
30 *sequester carbon and mitigate the greenhouse effect*. CRC Press LLC. New York. 15-33.
- 31 Birdsey, Richard, Kurt Pregitzer, and Alan Lucier. 2006. Forest carbon management in the United States, 1600-2100.
32 *Journal of Environmental Quality* [In press].
- 33 Caldeira, Ken, M. Granger Morgan, Dennis Baldocchi, Peter G. Brewer, Chen-Tung Arthur Chen, Gert-Jan
34 Nabuurs, Nebojsa Nakicenovic, and G. Phillip Robertson. 2004. A portfolio of carbon management options. In:
35 Field, Christopher B. and Michael R. Raupach. *The Global Carbon Cycle*. Island Press. Washington, DC.

- 1 Canadian Forest Service (CFS). 2005. State of the Forest Report, 2004-2005. Canadian Forest Service, Natural
2 Resources Canada, Ottawa, ON. Available at: [http://www.nrcan-rncan.gc.ca/cfs-scf/national/what-
4 quoi/sof/latest_e.html](http://www.nrcan-rncan.gc.ca/cfs-scf/national/what-
3 quoi/sof/latest_e.html).
- 4 Caspersen, John P.; Pacala, Stephen W.; Jenkins, Jennifer C.; Hurtt, George C.; Moorcraft, Paul R.; Birdsey,
5 Richard A. 2000. Contributions of land-use history to carbon accumulation in U.S. forests. *Science* 290: 1148-
6 1151.
- 7 Chapin, F. I., G. Woodwell, J. Randerson, G. Lovett, E. Rastetter, D. Baldocchi, D. Clark, M. Harmon, D. Schimel,
8 R. Valentini, C. Wirth, J. Aber, *et al.* in review. Reconciling carbon cycle terminology: A search for consensus.
- 9 Chen JM, Ju W, Cihlar J *et al.* 2003. Spatial distribution of carbon sources and sinks in Canada's forests. *Tellus B*:
10 1-20.
- 11 Clark, K.L., H.L. Gholz, and M.S. Castro. 2004: Carbon dynamics along a chronosequence of slash pine plantations
12 in north Florida. *Ecological Applications* **14**, 1154-1171.
- 13 Dale, Virginia H., Linda A. Joyce, Steve McNulty, Ronald P. Neilson, *et al.* 2001. Climate change and forest
14 disturbances. *Bioscience* 51(9): 723-734.
- 15 De Jong, B.H.J., S. Ochoa-Gaona, M. A. Castillo-Santiago, N. Ramirez-Marcial and M.A. Cairns. 2000. Carbon
16 fluxes and patterns of land-use/land-cover change in the Selva Lacandona, Mexico. *Ambio* 29: 504-511.
- 17 Environment Canada. 2005. Canada's Greenhouse Gas Inventory 1990-2003: Initial Submission. Greenhouse Gas
18 Division, Environment Canada, Ottawa, ON. Available at:
19 http://unfccc.int/national_reports/annex_i_ghg_inventories/national_inventories_submissions/items/2761.php.
- 20 Environmental Protection Agency. 2005. Greenhouse gas mitigation potential in U.S. forestry and agriculture.
21 Washington, DC: U.S. Environmental Protection Agency. 154 p.
- 22 Food and Agriculture Organization 2001. Global forest resources assessment 2000. Main Report. FAO Forestry
23 Paper 140. Rome. 481 p.
- 24 Fitzsimmons, M.J., D.J. Pennock and J. Thorpe. 2004. Effects of deforestation on ecosystem carbon densities in
25 central Saskatchewan, Canada. *Forest Ecology and Management* 188: 349-361.
- 26 Flannigan, M.D., K.A. Logan, B.D. Amiro, W.R. Skinner and B.J. Stocks. 2005. Future area burned in Canada.
27 *Climatic Change* 72: 1-16.
- 28 Foley, Jonathan A., and Navin Ramankutty. A primer on the terrestrial carbon cycle: what we don't know but
29 should. In: Field, Christopher B. and Michael R. Raupach. *The Global Carbon Cycle*. Island Press. Washington,
30 DC.
- 31 Giardina, C.P., D. Binkley, M.G. Ryan, J.H. Fownes, R.S. Senock. 2004. Belowground carbon cycling in a humid
32 tropical forest decreases with fertilization. *Oecologia* 139:545-550.
- 33 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.
34 Kurz, S. Liu, G.-J. Nabuurs, S. Nilsson, and A.Z. Shvidenko. 2002. Forest carbon sinks in the northern
35 hemisphere. *Ecological Applications* 12:891-899.

- 1 Griffis, T.J., T.A. Black, K. Morgenstern, A.G. Barr, Z. Nestic, G.B. Drewitt, D. Gaumont-Guay, and J.H.
2 McCaughey, 2003: Ecophysiological controls on the carbon balances of three southern boreal forests,
3 *Agricultural and Forest Meteorology*, **117** (1-2), 53-71.
- 4 Harmon, M. E., J. M. Harmon, W. K. Ferrell, and D. Brooks. 1996. Modeling carbon stores in Oregon and
5 Washington forest products: 1900-1992. *Climatic Change* 33: 521-550.
- 6 Harmon, M. 2001. Carbon sequestration in forests - Addressing the scale question. *JOURNAL OF FORESTRY*
7 **99**:24-29.
- 8 Harmon, M. and P. Marks. 2002. Effects of silvicultural practices on carbon stores in Douglas-fir-western hemlock
9 forests in the Pacific Northwest, USA: results from a simulation model. *Canadian Journal of Forest Research*
10 **32**(5): 863-877.
- 11 Haynes, Richard W. (ed.). 2003. An analysis of the timber situation in the United States: 1952-2050. Gen. Tech.
12 Rep. PNW-GTR-560. USDA Forest Service. Portland, OR. 254p.
- 13 Heath, Linda S. and James E. Smith. 2004. Criterion 5, Indicator 26: Total forest ecosystem biomass and carbon
14 pool, and if appropriate, by forest type, age class and successional change. 14 p. In: Darr, David R., coord. Data
15 report: A supplement to the national report on sustainable forests—2003. FS-766A. Washington, DC: U.S.
16 Department of Agriculture. <http://www.fs.fed.us/research/sustain/contents.htm> (8 June).
- 17 Heath, L.S., and J. E. Smith. 2000. An assessment of uncertainty in forest carbon budget projections. *Environmental*
18 *Science & Policy* 3: 73-82.
- 19 Hogg, E.H. and P.Y. Bernier. 2005. Climate change impacts on drought-prone forests in western Canada. *Forestry*
20 *Chronicle*, in press.
- 21 Houghton, R.A., J.L. Hackler, and K.T. Lawrence. 1999. The U.S. carbon budget: contributions from land-use
22 change. *Science* 285: 574-578.
- 23 Humphreys, E.R., Black, T.A., Morgenstern, K., Li, Z., and Nestic, Z., 2005. Net ecosystem production of a
24 Douglas-fir stand for three years following clearcut harvesting. *Global Change Biology* 11: 450-464.
- 25 Janisch, J., and M. Harmon. 2002. Successional changes in live and dead wood carbon stores: implications for net
26 ecosystem productivity. *TREE PHYSIOLOGY* **22**:77-89.
- 27 Jenkins, JC, Chojnacky, DC, Heath, LS, and Birdsey, RA. 2003. National-scale biomass estimators for United States
28 tree species. *Forest Science* 49(1): 12-35.
- 29 Johnson, D.W., R.B. Thomas, K.L. Griffen, D.T. Tissue, *et al.* 1998. Effects of carbon dioxide and nitrogen on
30 growth and nitrogen uptake in ponderosa and loblolly pine. *J. Environ. Qual.* 27:414-425.
- 31 Johnston, M. and T. Williamson. 2005. Climate change implications for stand yields and soil expectation values: A
32 northern Saskatchewan case study. *Forestry Chronicle* 81: 683-690.
- 33 Joyce, L., J. Baer, S. McNulty, V. Dale, A. Hansen, L. Irland, R. Neilson, and K. Skog. 2001. Potential
34 consequences of climate variability and change for the forests of the United States. Pages 489-521 in *Climate*
35 *change impacts in the United States, Report for the US Global Change Research Program*. Cambridge
36 University Press, Cambridge UK.

- 1 Karnosky, D.F., D.R. Zak, K.S. Pregitzer, C.S. Awmack, *et al.* 2003. Tropospheric ozone moderates responses of
2 temperate hardwood forests to elevated CO₂: a synthesis of molecular to ecosystem results from the Aspen
3 FACE project. *Functional Ecology* 17: 289-304.
- 4 Korner, C. 2000. Biosphere responses to CO₂ enrichment. *Ecol. Appl.* 10: 1590-1619.
- 5 Korner, Christian, Roman Asshof, Olivier Bignucolo, Stephan Haattenschwiler, *et al.* 2005. Carbon flux and growth
6 in mature deciduous forest trees exposed to elevated CO₂. *Science* 309: 1360-1362.
- 7 Kurz, W.A., Apps, M.J., Webb, T.M., McNamee, P.J. 1992. The carbon budget of the Canadian forest sector: Phase
8 1. Forestry Canada, Northern Forestry Centre, Information Report NOR-X-326. Edmonton, AB.
- 9 Kurz, W.A., S. Beukema and M.J. Apps. 1998. Carbon budget implications of the transition from natural to
10 managed disturbance regimes in forest landscapes. *Mitigation and Adaptation Strategies for Global Change* 2:
11 405-421.
- 12 Kurz, W.A. and M.J. Apps. 1999. A 70-year retrospective analysis of carbon fluxes in the Canadian forest sector.
13 *Ecological Applications* 9: 526-547.
- 14 Kurz, W., M. Apps, E. Banfield and G. Stinson. 2002. Forest carbon accounting at the operational scale. *The*
15 *Forestry Chronicle* 78: 672-679.
- 16 Law, B.E., E. Falge, D.D. Baldocchi, P. Bakwin, P. Berbigier, K. Davis, A.J. Dolman, M. Falk, J.D. Fuentes, A.
17 Goldstein, A. Granier, A. Grelle, D. Hollinger, I.A. Janssens, P. Jarvis, N.O. Jensen, G. Katul, Y. Mahli, G.
18 Matteucci, R. Monson, W. Munger, W. Oechel, R. Olson, K. Pilegaard, K.T. Paw U, H. Thorgeirsson, R.
19 Valentini, S. Verma, T. Vesala, K. Wilson, and S. Wofsy, 2002: Environmental controls over carbon dioxide
20 and water vapor exchange of terrestrial vegetation. *Agricultural and Forest Meteorology*, **113**, 97-120.
- 21 Leenhouts, B. 1998. Assessment of biomass burning in the conterminous United States. *Conserv. Ecol.* 2(1): 1.
- 22 Lewandrowski, Jan, Mark Sperow, Mark Peters, Marlen Eve, Carol Jones, Keith Paustian, and Robert House. 2004.
23 Economics of sequestering carbon in the U.S. agricultural sector. Tech. Bull. 1909. Washington, DC: U.S.
24 Department of Agriculture, Economic Research Service. 61 p.
- 25 Lichter, John, Stacy H. Barron, Christine E. Bevacqua, Adrien C. Finzi, Katherine F. Irvine, *et al.* 2005. Soil carbon
26 sequestration and turnover in a pine forest after six years of atmospheric CO₂ enrichment. *Ecology* 86(7): 1835-
27 1847.
- 28 Loya, Wendy M., Kurt S. Pregitzer, Noah J. Karberg, John S. King, and Christian P. Giardina. 2003. Reduction of
29 soil carbon formation by tropospheric ozone under increased carbon dioxide levels. *Nature* 425: 705-707.
- 30 Lugo, A. E., J. Figueroa Colón, and F. N. Scatena. 1999. "The Caribbean." In M.G. Barbour and W.D. Billings,
31 editors. *North American terrestrial vegetation*. Cambridge University Press, England. 530 p.
- 32 Masera, O.; Ordóñez, M.J. and Dirzo, R.; 1997a, 'Carbon Emissions From Mexican Forests: the Current Situation
33 and Long-Term Scenarios'. *Climatic Change*, 35, 265-295.
- 34 Masera, O., A. Delia Cerón, A. Ordóñez. 2001. Forestry mitigation options for Mexico: finding synergies between
35 national sustainable development priorities and global concerns. *Mitigation and Adaptation Strategies for*
36 *Global Change* 6: 291-312.

- 1 McNulty, Steven G. 2002. Hurricane impacts on U.S. forest carbon sequestration. *Environmental Pollution* 116:S17-
2 S24.
- 3 National Forestry Database Program (NFD). 2005. Compendium of Canadian Forestry Statistics. National Forestry
4 Database Program, Canadian Council of Forest Ministers, Ottawa, ON. Available at:
5 http://nfdp.ccfm.org/compendium/index_e.php.
- 6 Natural Resources Canada 2005. The State of Canada's Forests. Canadian Forest Service, Natural Resources
7 Canada, Ottawa, ON. Available at: http://www.nrca-nrcan.gc.ca/cfs-scf/national/what-quoi/sof/latest_e.html
- 8 Neilson, R.P., I.C. Prentice, B. Smith, T.G.F. Kittel, and D. Viner. 1998. Simulated changes in vegetation
9 distribution under global warming. Pages 439-456 in R.T. Watson, M.C. Zinyowera, R.H. Moss, and D.J.
10 Dokken, editors. *The Regional Impacts of Climate Change: An Assessment of Vulnerability*. Cambridge
11 University Press, Cambridge.
- 12 Nelson, K.C. and B.H.J. de Jong. 2003. Making global initiatives local realities: carbon mitigation projects in
13 Chiapas, Mexico. *Global Environmental Change* 13: 19-30.
- 14 Norby, Richard J., Evan H. DeLucia, Birget Gielen, *et al.* 2005. Forest response to elevated CO₂ is conserved across
15 a broad range of productivity. *PNAS* 102(50), 18052-18056.
- 16 Nowak, R.S., D. S. Ellsworth, S. D. Smith. 2004. Functional responses of plants to elevated atmospheric CO₂- do
17 photosynthetic and productivity data from FACE experiments support early predictions? *New Phytologist* 162:2
18 253
- 19 Ollinger SV, Aber JD, Reich PB, Freuder RJ (2002) Interactive effects of nitrogen deposition, tropospheric ozone,
20 elevated CO₂ land use history on the carbon dynamics of northern hardwood forests. *Global Change Biology* 8,
21 545-562.
- 22 Oren, Ram, David S. Ellworth, Kurt S. Johnsen, Nathan Phillips, *et al.* 2001. Soil fertility limits carbon
23 sequestration by forest ecosystems in a CO₂-enriched atmosphere. *Nature* 411: 469-472.
- 24 Osher L. J., P.A. Matson, R. Amundson. 2003. Effect of land use change on soil carbon in Hawaii. *Biogeochemistry*
25 65 (2):213-232..
- 26 Pacala, S.W.; Hurtt, G.C.; Baker, D.; Peylin, P.; Houghton, R.A.; Birdsey, Richard A.; Heath, L.; Sundquist, E.T.;
27 Stallard, R.F. *et al.* 2001. Consistent land and atmosphere-based U.S. carbon sink estimates. *Science*. 292:
28 2316-2320.
- 29 Palacio, J.L. *et al.* 2000. *La condición actual de los recursos forestales en México: resultados del Inventario*
30 *Forestal Nacional 2000*. Technical Note, Investigaciones Geográficas, No. 43 pp. 183-203.
- 31 Pan, Y. *et al.* 1998. Modeled responses of terrestrial ecosystems to elevated atmospheric CO₂: a comparison of
32 simulations by the biogeochemistry models of the Vegetation/Ecosystem Modeling and Analysis Project
33 (VEMAP). *Oecologia* 114: 389-404.
- 34 Parisien, M.-A., Kafka, V., Flynn, N., Hirsch, K.G., Todd, J.B. and M.D. Flannigan. 2005. Fire behavior potential in
35 central Saskatchewan under predicted climate change. PARC (Prairie Adaptation Research Collaborative),
36 PARC Summary Document 05-01. Regina, SK. 12 p.

- 1 Peterson, E.B., Bonnor, G.M., Robinson, G.C. and Peterson, N.M. 1999. Carbon Sequestration Aspects Of An
2 Afforestation Program In Canada's Prairie Provinces. Submitted to Joint Forest Sector Table/Sinks Table,
3 National Climate Change Process, Ottawa, ON. Available at:
4 http://www.nccp.ca/NCCP/national_process/issues/sinks_e.html.
- 5 Potter, C., Klooster, S. A., Myneni, R., Genovese, V., Tan, P., Kumar, V. 2003, Continental scale comparisons of
6 terrestrial carbon sinks estimated from satellite data and ecosystem modeling 1982-98. *Global and Planetary*
7 *Change*, 39, 201-213.
- 8 Pregitzer, Kurt S., and Eugenie S. Euskirchen. 2004. Carbon cycling and storage in world forests: biomes patterns
9 related to forest age. *Global Change Biology* 10: 2052-2077.
- 10 Price, D.T., D.H. Halliwell, M.J. Apps, W.A. Kurz, and S.R. Curry. 1997. Comprehensive assessment of carbon
11 stocks and fluxes in a Boreal-Cordilleran forest management unit. *Can. J. For. Res.* 27: 2005-2016.
- 12 Price, D.T., D. W. McKenney, P. Papadopol, T. Logan, and M. F. Hutchinson. 2004. "High resolution future
13 scenario climate data for North America." *Proc. Amer. Meteor. Soc. 26th Conference on Agricultural and*
14 *Forest Meteorology*, Vancouver, B.C., 23-26 August, 2004, 13 pp.
- 15 Price, D.T., C.H. Peng, M.J. Apps and D.H. Halliwell. 1999. Simulating effects of climate change on boreal
16 ecosystem carbon pools in central Canada. *Journal of Biogeography* 26: 1237-1248.
- 17 Proctor, Patrick, Linda S. Heath, Paul C. Van Deusen, Jeffrey H. Gove, and James E. Smith. 2005. COLE: a web-
18 based tool for interfacing with forest inventory data. In: McRoberts, Ronald E. *et al.* (Eds.), *Proceedings of the*
19 *Fourth Annual Forest Inventory and Analysis Symposium*. USDA Forest Service GTR-NC-252.
- 20 Ryan, M.G., D. Binkley, and J.H. Fownes. 1997. "Age-related decline in forest productivity: pattern and
21 process." *Advances in Ecological Research* 27: 213-262.
- 22 Schimel, D., J. Melillo, H. Tian, A.D. McGuire, D. Kicklighter, T. Kittel, N. Rosenbloom, S. Running, P. Thornton,
23 D. Ojima, W. Parton, R. Kelly, M. Sykes, R. Neilson, and B. Rizzo. 2000. Contribution of increasing CO₂ and
24 climate to carbon storage by ecosystems in the United States. *Science* 287: 2004-2006.
- 25 Schulze, E., J. Lloyd, F. Kelliher, C. Wirth, C. Rebmann, B. Luhker, M. Mund, A. Knohl, I. Milyukova, W. Schulze,
26 W. Ziegler, A. Varlagin, A. Sogachev, R. Valentini, S. Dore, S. Grigoriev, O. Kolle, M. Panfyorov, N.
27 Tchebakova, and N. Vygodskaya. 1999. Productivity of forests in the Eurosiberian boreal region and their
28 potential to act as a carbon sink - a synthesis. *Global Change Biology* 5:703-722.
- 29 Secretaría de Medio Ambiente, Recursos Naturales y Pesca (SEMARNAP); 1996, *Programa Forestal y de Suelo*
30 *1995-2000*, Poder ejecutivo Federal, SEMARNAP, México City.
- 31 Skog, K.E., and G.A. Nicholson. 1998. Carbon cycling through wood products: the role of wood and paper products
32 in carbon sequestration. *Forest Products Journal* 48:75-83.
33 (<http://www.fpl.fs.fed.us/documnts/pdf1998/skog98a.pdf>)
- 34 Smith, W. B., P. D. Miles, J. S. Vissage, and S. A. Pugh. 2004. *Forest Resources of the United States, 2002*. General
35 Technical Report NC-241, USDA Forest Service, North Central Research Station, St. Paul, MN.
- 36 Smith, James E., and Linda S. Heath. 2005. Land use change and forestry and related sections. (Excerpted.) In: U.S.
37 Environmental Protection Agency. *Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2003*. EPA

- 1 430-R-05-003. Available at
2 <http://yosemite.epa.gov/oar/globalwarming.nsf/content/ResourceCenterPublicationsGHGEmissionsUSEmission>
3 [sInventory2005.html](http://yosemite.epa.gov/oar/globalwarming.nsf/content/ResourceCenterPublicationsGHGEmissionsUSEmission) (18 August). [Note that some EPA contractors or employees may have written or edited
4 some text, or formatted tables or redraw some figures for final EPA format.]
5 Smith, James E. and Linda S. Heath. 2000. Considerations for interpreting probabilistic estimates of uncertainty of
6 forest carbon. P. 102-111. In: Joyce LA and R. Birdsey, eds. *The Impact of Climate Change on America's*
7 *Forests*, USDA Forest Service, General Technical Report RMRS-GTR-59. 134 p.
8 Soto-Pinto, L., G. Jimenez-Ferrer, A. Vargas Guillen, B. de Jong Bergsma, E. Esquivel-Bazan. 2001. Experiencia
9 agroforestal para la captura de carbono en comunidades indigenas de Mexico. *Revista Forestal Iberoamericana*
10 1: 44-50
11 Stainback, G. Andrew, and Janaki R.R. Alavalapati. 2005. Effects of carbon markets on the optimal management of
12 slash pine (*Pinus elliottii*) plantations. *Southern Journal of Applied Forestry* 29(1): 27-32.
13 Stanturf, John A., Robert C. Kellison, F.S. Broerman, and Stephen B. Jones. 2003. Productivity of southern pine
14 plantations – where we are and how did we get here? *Journal of Forestry* April/May 2003: 26-31.
15 Stavins, Robert N. and Kenneth R. Richards. 2005. *The cost of U.S. forest-based carbon sequestration*. Arlington,
16 VA: The Pew Center on Global Climate Change. 40 p. [available at www.pewclimate.org]
17 Torres R., J.M. 2004. *Estudio de tendencias y perspectivas del sector forestal en América Latina al año 2020*.
18 *Informe Nacional México*. FAO.
19 http://www.fao.org/documents/show_cdr.asp?url_file=/docrep/006/j2215s/j2215s11.htm
20 Totten, Michael. 1999. *Getting it right: emerging markets for storing carbon in forests*. World Resources Institute.
21 Washington, DC. 49p.
22 US Climate Change Science Program. 2003. *Strategic plan for the climate change science program*. Washington,
23 DC: <http://www.climatescience.gov/Library/stratplan2003/default.htm>
24 VEMAP Members. 1995. *Vegetation/ecosystem modeling and analysis project: comparing biogeography and*
25 *biogeochemistry models in a continental-scale study of terrestrial ecosystem responses to climate change and*
26 *CO2 doubling*. *Global Biochemical Cycles* 9: 407-437.
27 Volney, J.A.W. and K. Hirsch. 2005. Disturbing forest disturbances. *Forestry Chronicle* 81: 662-668.
28 Weber, M.G., and M.D. Flannigan, 1997. Canadian boreal forest ecosystem structure and function in a changing
29 climate: impact on fire regimes, *Environ. Rev.*, 5, 145-166.
30 Winrock International 2005. *Ecosystem services*. <http://www.winrock.org/what/projects.cfm?BU=9086> [Date
31 accessed unknown]
32 Woodwell, G., and R. Whittaker. 1968. Primary production in terrestrial communities. *American Zoologist* 8:19-30.

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

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

¹There 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).

²Canadian Forest Service, 2005

³Smith *et al.*, 2004

⁴Palacio *et al.*, 2000

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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

¹There is 95% certainty that the actual values are within 25% of those reported in this table (Heath and Smith, 2000; Smith and Heath, 2000).

²Kurz and Apps, 1999

³Heath and Smith, 2004; Birdsey and Heath, 1995

⁴Masera *et al.*, 2001

⁵Includes litter, coarse woody debris, and soil carbon

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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

¹Environment Canada (2005), Goodale *et al.* (2002). There is 95% certainty that the actual values are within 100% of those reported for Canada.

²From 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.

³From Masera, 1997. There is 95% certainty that the actual values are within 100% of those reported for Mexico.

⁴Estimates are not available.

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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

¹From 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).

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Table 11-5. Illustrative emissions reduction potential of various forestry activities in the United States under a range of prices and sequestration rates¹

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

¹Adapted from Environmental Protection Agency (2005). Maximum price analyzed was \$50/t CO₂.

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 $-19.6 \text{ t C ha}^{-1} \text{ yr}^{-1}$ in a white pine plantation in southern Ontario (Arain and Restrepo-Coupe, 2005) to $-3.2 \text{ t C ha}^{-1} \text{ yr}^{-1}$ in a jack pine forest in (Amiro *et al.*, 2005; Griffis *et al.*, 2003). In recently disturbed forests, NEP ranges from $+58.0 \text{ t C ha}^{-1} \text{ yr}^{-1}$ in a harvested Douglas-fir forest (Humphreys *et al.*, 2005) to $+5.7 \text{ t C ha}^{-1} \text{ yr}^{-1}$ 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 $+12.7 \text{ t C ha}^{-1} \text{ yr}^{-1}$ to a sink of $-5.9 \text{ t C ha}^{-1} \text{ yr}^{-1}$. 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 $-2.5 \text{ t C ha}^{-1} \text{ yr}^{-1}$, 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 $-5.9 \text{ t C ha}^{-1} \text{ yr}^{-1}$ (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 $+13 \text{ t C ha}^{-1} \text{ yr}^{-1}$ 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 $-11.0 \text{ t C ha}^{-1} \text{ yr}^{-1}$ (Lugo *et al.*, 1999). Belowground NPP measurements exist for only one site with $-19.5 \text{ t C ha}^{-1} \text{ yr}^{-1}$ (Lugo *et al.*, 1999). In Hawaii, aboveground

1 and belowground NPP of native forests dominated by *Metrosideros polymorpha* vary depending on
 2 substrate age and precipitation regime. Aboveground NPP ranges between -4.0 to -14.0 t C ha⁻¹ yr⁻¹,
 3 while belowground NPP ranges between -5.2 and -9.0 t C ha⁻¹ yr⁻¹ (Giardina *et al.*, 2004). Soil carbon
 4 emissions along the substrate age gradient range from $+2.2$ to $+3.3$ t C ha⁻¹ yr⁻¹, and along the
 5 precipitation gradient from $+4.0$ to $+9.7$ t C ha⁻¹ yr⁻¹ (Osher *et al.*, 2003). NEP estimates are not available
 6 for these tropical forests, so their net impact on atmospheric carbon stocks cannot be calculated.

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Table 11A-1. Comparison of net ecosystem exchange (NEE) for different types and ages of temperate forests. Positive NEE means the forest is a sink for atmospheric CO₂. 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)

9

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

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

24

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

27

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

31 **Practices aimed at reducing R_h** (i.e., minimizing the time forests are a source to the atmosphere
32 following disturbance): low impact harvesting (that does not promote soil respiration); utilization of
33 logging residues (biomass energy and fuels); incorporation of logging residue into soil during site

1 prep (but note that this could also speed up decomposition); thinning to capture mortality;
2 fertilization.

3
4 Since NECB changes with time as forests age, if a landscape is composed of stands with different
5 ages then carbon gains in one stand can be offset by losses from another stand. The net result of these
6 stand-level changes determines overall landscape-level carbon stores. Note that disturbance-induced Rh
7 losses are typically larger than annual gains, such that a landscape where forest area is increasing might
8 still be neutral with respect to carbon stocks overall. Thus, at the landscape level practices designed to
9 enhance carbon sequestration must, on balance, replace lower-C-density systems with higher-C-density
10 systems. Examples of these practices include: reducing fire losses; emphasizing very long-lived forest
11 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

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KEY FINDINGS

- Much of northern North America (more than 6 million km²) 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 CO₂ in response to an already detectable polar warming.
- The soils of the permafrost region of North America contain 213 Gt of organic carbon, approximately 61% of the carbon in all soils of North America.
- The soils of the permafrost region of North America are currently a net sink of approximately 11 Mt C yr⁻¹.
- The soils of the permafrost region of North America have been slowly accumulating carbon for the last 5-8 thousand years. More recently, increased human activity in the region has resulted in permafrost degradation and at least localized loss of soil carbon.
- Patterns of climate, especially the region's cool and cold temperatures and their interaction with soil hydrology to produce wet and frozen soils, are primarily responsible for the historical accumulation of carbon in the region. Non-climatic drivers of carbon change include human activities, including flooding associated with hydroelectric development, that degrade permafrost and lead to carbon loss. Fires, increasingly common in the region, also lead to carbon loss.
- Projections of future warming of the polar regions of North America lead to projections of carbon loss from the soils of the permafrost region, with upwards of 78% (34 Gt) and 41% (40 Gt) of carbon stored in soils of the Subarctic and Boreal regions, respectively, being severely or extremely severely affected by future climate change.
- Options for management of carbon in the permafrost region of North America, including construction methods that cause as little disturbance of the permafrost and surface as possible, are primarily those which avoid permafrost degradation and subsequent carbon losses.

- Most research needs for the permafrost region are focused on reducing uncertainties in knowing how much carbon is vulnerable to a warming climate and how sensitive that carbon loss is to climate change. Development and adoption of measures that reduce or avoid the negative impact of human activities on permafrost are also needed.
-

INTRODUCTION

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

Permafrost is defined, on the basis of temperature, as soils or rocks that remain below 0°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

1 wetlands characterized by more than 40 cm of peat accumulation) cover large areas in the Boreal,
2 Subarctic, and southern part of the Arctic ecoclimatic provinces.

3
4 **Figure 12-2. Arctic, Subarctic, and Boreal ecoclimatic provinces (ecological regions) in North**
5 **America (Ecoregions Working Group, 1989; Baily and Cushwa, 1981).**

6
7 Although northern ecosystems (Arctic, Subarctic, and Boreal) in North America cover
8 approximately 14% of the global land area, they contain approximately 25% of the world's total soil
9 organic carbon (Oechel and Vourlitis, 1994). In addition, Oechel and Vourlitis (1994) indicate that the
10 tundra (Arctic) ecosystems alone contain approximately 12% of the global soil carbon pool, even though
11 they account for only 6% of the total global land area. The soils of the permafrost region of North
12 America are currently a carbon sink and are unique because they are able to actively sequester carbon and
13 store it for thousands of years.

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

17 18 **PROCESSES AFFECTING THE CARBON CYCLE IN A PERMAFROST** 19 **ENVIRONMENT**

20 **Soils of the Permafrost Region**

21 Soils cover approximately 6,211,340 km² of the area of the North American permafrost region
22 (Tables 12-1 and 12-2), with approximately 58% of the soil area being occupied by permafrost-affected
23 (perennially frozen) soils (Cryosols/Gelisols) and the remainder by non-permafrost soils. Approximately
24 17% of this area is associated with organic soils (peatlands), the remainder with mineral soils. It is
25 important to distinguish between mineral soils and organic soils in the region because different processes
26 are responsible for the carbon cycle in these two types of soils.

27
28 **Table 12-1. Areas of mineral soils in the various permafrost zones.**

29
30 **Table 12-2. Areas of peatlands (organic soils) in the various permafrost zones.**

31 32 **Mineral Soils**

33 The schematic diagram in Fig. 12-3 provides general information about the carbon sinks and sources
34 in mineral soils. Most of the permafrost-affected mineral soils are carbon sinks because of the process of

1 cryoturbation, which moves organic matter into the deeper soil layers. Other processes, such as
2 decomposition, wildfires, and thermal degradation, release carbon into the atmosphere and, thus, act as
3 carbon sources.

4
5 **Figure 12-3. Carbon cycle in permafrost-affected upland (mineral) soils, showing below-ground**
6 **organic carbon sinks and sources.**

7
8 For unfrozen soils and noncryoturbated frozen soils in the permafrost region, the carbon cycle is
9 similar to that in soils occurring in temperate regions. In these soils, organic matter is deposited on the
10 soil surface. Some soluble organic matter may move downward, but because these soils are not affected
11 by cryoturbation, they have no mechanism for moving organic matter from the surface into the deeper soil
12 layers and preserving it from decomposition and wildfires. Most of their below-ground carbon originates
13 from roots and its residence time is relatively short.

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

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

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

1 becomes deeper and the organic material is able to move even deeper into the soil (translocation).
2 However, if such factors cause thawing of the soil and melting of the ground ice, some or all of the
3 organic materials locked in the system could be exposed to the atmosphere. This change in soil
4 environment gives rise to both aerobic and anaerobic decomposition, releasing carbon into the atmosphere
5 as carbon dioxide and methane, respectively (Fig. 12-3). At this stage, the soil can become a major carbon
6 source.

7 If, however, the permafrost table rises (and the active layer becomes shallower) because of
8 reestablishment of the vegetation or buildup of the surface organic layer, this deep organic material
9 becomes part of the permafrost and is, thus, more securely preserved. This is the main reason that
10 permafrost-affected soils contain high amounts of organic carbon not only in the upper (0–100 cm) layer,
11 but also in the deeper layers. These cryoturbated, permafrost-affected soils are effective carbon sinks.
12

13 **Peatlands (Organic Soils)**

14 The schematic diagram in Fig. 12-4 provides general information about the processes driving the
15 carbon sinks and sources in peatland soils. The water-saturated conditions, low soil temperatures, and
16 acidic conditions of northern peatlands provide an environment in which very little or no decomposition
17 occurs; hence, the litter is converted to peat and preserved. This gradual buildup process has been ongoing
18 in peatlands during the last 5,000–8,000 years, resulting in peat deposits that are an average of 2–3 m
19 thick and, in some cases, up to 10 m thick. At this stage, peatlands can act as very effective carbon sinks
20 for many thousands of years (Fig. 12-4).
21

22 **Figure 12-4. Carbon cycle in permafrost peatlands, showing below-ground organic carbon sinks and**
23 **sources.**
24

25 **Carbon dynamics:** Data for carbon accumulation in various peatland types in the permafrost regions
26 are given in Table 12-3. Although some values for the rate of peat accumulation are higher (associated
27 with unfrozen peatlands), the values for frozen peatlands, which are more widespread, generally range
28 around $13 \text{ g C m}^{-2} \text{ yr}^{-1}$. Peat accumulations in the various ecological regions were calculated on the basis
29 of the thickness of the deposit and the date of the basal peat. The rate of peat accumulation is generally
30 highest in the Boreal region and decreases northward (Table 12-3). Note, however, that if the surface of
31 the peat deposit has eroded, the calculated rate of accumulation (based on the age of the basal peat and a
32 decreased deposit thickness) will appear to be higher than it should be. This is probably the reason for
33 some of the high rates of peat accumulation found for the Arctic region, which likely experienced a rapid
34 rate of accumulation during the Hypsithermal Maximum with subsequent erosion of the surface of some

1 of the deposits reducing their thicknesses. Wildfires, decomposition, and leaching of soluble organic
2 compounds release approximately one-third of the carbon input, causing most of the carbon loss in these
3 peatlands.

4
5 **Table 12-3. Organic carbon accumulation and loss in various Canadian peatlands.** Positive values
6 indicate net flux into the atmosphere (source); negative values indicate carbon sequestration (land sinks).

8 **BELOW-GROUND CARBON STOCKS**

9 The carbon content of mineral soils to a 1-m depth is 49–61 kg m⁻² for permafrost-affected soils and
10 12–17 kg m⁻² for unfrozen soils (Tables 12-4 and 12-5). The carbon content of organic soils (peatlands)
11 for the total depth of the deposit is 81–129 kg m⁻² for permafrost-affected soils and 43–144 kg m⁻² for
12 unfrozen soils (Tables 12-4 and 12-5) (Tarnocai, 1998 and 2000).

13
14 **Table 12-4. Soil carbon pools and fluxes for the permafrost areas of Canada.** Positive flux numbers
15 indicate net flux into the atmosphere (source); negative values indicate carbon sequestration (land sinks).

16
17 **Table 12-5. Average organic carbon content for soils in the various ecological regions (Tarnocai 1998**
18 **and 2000).**

19
20 Soils in the permafrost region of North America contain 213 Gt of organic carbon (Tables 12-6 and
21 12-7), which is approximately 61% of the organic carbon in all soils on this continent (Lacelle *et al.*,
22 2000). Mineral soils contain approximately 99 Gt of organic carbon in the 0- to 100-cm depth
23 (Table 12-6). Although peatlands (organic soils) cover a smaller area than mineral soils (17% vs 83%),
24 they contain approximately 114 Gt of organic carbon in the total depth of the deposit, or more than half
25 (54%) of the soil organic carbon of the region (Table 12-7).

26
27 **Table 12-6. Organic carbon mass in mineral soils in the various permafrost zones.**

28
29 **Table 12-7. Organic carbon mass in peatlands (organic soils) in the various permafrost zones.**

31 **CARBON FLUXES**

32 **Mineral Soils**

33 Very little information is available about carbon fluxes in both unfrozen and perennially frozen
34 mineral soils in the permafrost regions. For unfrozen upland mineral soils, Trumbore and Harden (1997)

1 report a carbon accumulation of 60–100 g C m⁻² yr⁻¹ (Table 12-4). They further indicate that the slow
2 decomposition results in rapid organic matter accumulation, but the turnover time due to wildfires (every
3 500–1000 years) eliminates the accumulated carbon except for the deep carbon derived from roots in the
4 subsoil. The turnover time for this deep carbon is 100–1600 years. Therefore, the carbon stocks in these
5 unfrozen soils are low, and the turnover time of this carbon is 100 to 1000 years.

6 As with unfrozen mineral soils, very little information has been published on the carbon cycle in
7 perennially frozen mineral soils. The carbon cycle in these soils differs from that in unfrozen soils in that,
8 because of cryogenic activities, these soils are able to move the organic matter deposited on the soil
9 surface into the deeper soil layers. Assuming that cryoturbation was active in these soils during the last
10 six thousand years (Zoltai *et al.*, 1978), an average of 9 Mt C have been added annually to these soils.
11 Most of this carbon has been cryoturbated into the deeper soil layers, but some of the carbon in the
12 surface organic layer is released by decomposition and, periodically, by wildfires. The schematic diagram
13 in Fig. 12-5 shows the carbon cycle in these soils.

14
15 **Figure 12-5. Carbon cycle in perennially frozen mineral soils in the permafrost region.**

16 17 **Peatlands (Organic Soils)**

18 Peatland vegetation deposits various amounts of organic material (litter) annually on the peatland
19 surface. Reader and Stewart (1972) found that the amount of litter (dry biomass) deposited annually on
20 the bog surface in Boreal peatlands in Manitoba, Canada was 489–1750 g m⁻². Approximately 25% of the
21 original litter fall was found to have decomposed during the following year. In the course of the study,
22 they found that the average annual accumulation rate was 10% of the annual net primary production.
23 Robinson *et al.* (2003) found that, in the Sporadic Permafrost Zone, mean carbon accumulation rates over
24 the past 100 years for unfrozen bogs and frost mounds were 88.6 and 78.5 g m⁻² yr⁻¹, respectively. They
25 also found that, in the Discontinuous Permafrost Zone, the mean carbon accumulation rate during the past
26 1200 years in frozen peat plateaus was 13.31 g m⁻² yr⁻¹, while in unfrozen fens and bogs the comparable
27 rates were 20.34 and 21.81 g m⁻² yr⁻¹, respectively.

28 Because peatlands cover large areas in the permafrost region of North America, their contribution to
29 the carbon stocks is significant (Table 12-5). Zoltai *et al.* (1988) estimated that the annual carbon
30 accumulation capacity of Boreal peatlands is approximately 9.8 Mt. Gorham (1988), in contrast,
31 estimated that Canadian peatlands accumulate approximately 30 Mt of carbon annually.

32 Currently, wildfires are probably the greatest natural force in converting peatlands to a carbon source.
33 Ritchie (1987) found that the western Canadian Boreal forests have a fire return interval of 50–100 years,
34 while Kuhry (1994) indicated that, for wetter Sphagnum bogs, the interval is 400–1700 years. For peat

1 plateau bogs, each fire resulted in an average decrease in carbon mass of 1.46 kg m^{-2} and an average
2 decrease in height of 2.74 cm, which represents about 150 years of peat accumulation (Robinson and
3 Moore, 2000). In recent years, the number of these wildfires has increased, as has the area burned,
4 releasing increasing amounts of carbon into the atmosphere.

5 The schematic diagram presented in Fig. 12-6 summarizes the carbon cycle in peatlands in the
6 permafrost region. Based on average values for the rate of peat accumulation, approximately 17 g C m^{-2}
7 yr^{-1} , or 18 Mt C, is added annually to peatlands in this region of North America. Approximately 1.46 kg
8 C m^{-2} is released to the atmosphere every 600 years by wildfires in the northern boreal peatlands. In
9 addition, decomposition of unfrozen peatlands releases approximately $2.0 \text{ g C m}^{-2} \text{ yr}^{-1}$, and a further 2.0 g
10 $\text{C m}^{-2} \text{ yr}^{-1}$ is released by leaching of dissolved organic carbon (DOC), leading to a carbon decrease of
11 approximately 4 Mt annually, not including that released by wildfires (Fig. 12-6). Note that these values
12 are based on current measurements. However, rates of peat accumulation have varied during the past
13 6000–8000 years, with periods during which the rate of peat accumulation was much higher than at
14 present.

15
16 **Figure 12-6. Carbon cycle in peatlands in the permafrost region.**

17 18 **Total Flux**

19 Based on the limited data available for this vast, and largely inaccessible, area of the continent,
20 approximately 27 Mt C yr^{-1} is deposited on the surface of mineral soils and peatlands (organic soils) in
21 the permafrost region of North America. Approximately 8 Mt yr^{-1} of surface carbon (excluding
22 vegetation) is released by decomposition and wildfires, and by leaching into the water systems. Thus, the
23 soils in the permafrost region of North America currently act as a sink for approximately 19 Mt C yr^{-1} and
24 as a source for approximately 8 Mt C yr^{-1} and are, therefore, a net carbon sink (Figs. 12-5 and 12-6).

25 26 **POSSIBLE EFFECTS OF GLOBAL CLIMATE CHANGE**

27 The permafrost region is unique because the soils in this vast area contain large amounts of organic
28 materials and much of the carbon has been actively sequestered by peat accumulation (organic soils) and
29 cryoturbation (mineral soils) and stored in the permafrost for many thousands of years. Historical patterns
30 of climate are responsible for the large amount of carbon found in the soils of the region today, but
31 cryoturbation is a consequence of the region's current cool to cold climate and the effects of that climate
32 on soil hydrology. As a result, patterns of climate and climate change are dominant drivers of carbon
33 cycling in the region. Future climate change will determine the fate of that carbon and whether the region

1 will remain a slow but significant carbon sink, or whether it will reverse and become a source, rapidly
2 releasing large amounts of CO₂ and methane to the atmosphere.

4 **Peatlands**

5 A model for estimating the sensitivity of peatlands to global climate change was developed using
6 current climate (1x CO₂), vegetation, and permafrost data together with the changes in these variables
7 expected in a 2x CO₂ environment (Kettles and Tarnocai, 1999). The data generated by this model were
8 used to produce a peatland sensitivity map. Using GIS techniques, this map was overlaid on the peatland
9 map of Canada to determine both the sensitivity ratings of the various peatland areas and the associated
10 organic carbon masses. The sensitivity ratings, or classes, used are no change, very slight, slight,
11 moderate, severe, and extremely severe. Because global climate change is expected to have the greatest
12 impact on the ecological processes and permafrost distribution in peatlands in the severe and extremely
13 severe categories (Kettles and Tarnocai, 1999), the areas and carbon masses of peatlands in these two
14 sensitivity classes are considered to be most vulnerable to climate change. The sensitivity ratings are
15 determined by the degree of change in the ecological zonation combined with the degree of change in the
16 permafrost zonation, with the greater the change, the more severe the sensitivity rating. For example, if a
17 portion of the Subarctic becomes Boreal in ecology and the associated sporadic permafrost disappears (no
18 permafrost remains in the region), the sensitivity of this region is rated as extremely severe. If however, a
19 portion of the Boreal remains Boreal in ecology, but the discontinuous permafrost disappears (no
20 permafrost remains in the region), the sensitivity of this region is rated as severe.

21 The peatland sensitivity model indicates that the greatest effect of global climate change will occur in
22 the Subarctic region, where about 85% (314,270 km²) of the peatland area and 78% (33.96 Gt) of the
23 organic carbon mass will be severely or extremely severely affected by climate change, with 66% of the
24 area and 57% of the organic carbon mass being extremely severely affected (Fig. 12-7) (Tarnocai, in
25 press). The second largest effect will occur in the Boreal region, where about 49% (353,100 km²) of the
26 peatland area and 41% (40.20 Gt) of the organic carbon mass will be severely or extremely severely
27 affected, with 10% of both the area and organic carbon mass being extremely severely affected. These
28 two regions contain almost all (99%) of the Canadian peatland area and organic carbon mass that is
29 predicted to be severely or extremely severely affected (Fig. 12-7) (Tarnocai, in press).

30
31 **Figure 12-7. The organic carbon mass in the various sensitivity classes for the Subarctic and Boreal**
32 **Ecoclimatic Provinces (ecological regions) (Tarnocai, in press).**
33

1 In the Subarctic region and the northern part of the Boreal region, where most of the perennially
2 frozen peatlands occur, the increased temperatures are expected to cause increased thawing of the
3 perennially frozen peat. Thawing of the ice-rich peat and the underlying mineral soil will initially result in
4 water-saturated conditions. These water-saturated conditions, together with the higher temperatures, result
5 in anaerobic decomposition, leading to the production of CH₄.

6 In the southern part of the Boreal region, where the peatlands are generally unfrozen, the main impact
7 is expected to be drought conditions resulting from higher summer temperatures and higher
8 evapotranspiration. Under such conditions, peatlands become a net source of CO₂ because the oxygenated
9 conditions lead to aerobic decomposition (Melillo *et al.*, 1990; Christensen, 1991). These dry conditions
10 will likely also increase wildfires and, eventually, burning of peat, leading to the release of CO₂ to the
11 atmosphere.

12

13 **Permafrost-Affected Mineral Soils**

14 The same model described above was used to determine the effect of climate change on mineral
15 permafrost-affected soils. The model suggests that approximately 21% (11.9 Gt) of the total organic
16 carbon in these soils could be severely or extremely severely affected by climate warming (Tarnocai,
17 1999). The model also suggests that the permafrost will probably disappear from the soils (the soils will
18 become unfrozen) in the Sporadic and Isolated Patches permafrost zones. The main reason for the high
19 sensitivity of mineral soils in these zones is that soil temperatures at both the 100- and 150-cm depths are
20 only slightly below freezing (-0.3°C). The slightest disturbance or climate warming could initiate rapid
21 thawing in these soils, with resultant loss of carbon (Tarnocai, 1999).

22

23 **NON-CLIMATIC DRIVERS**

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

31 Increased human activity as a result of the construction of pipelines, roads, airstrips, and mines,
32 expansion of agriculture, and development and expansion of town sites has disturbed the natural soil
33 cover and exposed the organic-rich soil layers, leading to increased soil temperatures and, hence,
34 decomposition of the exposed organic materials. Burgess and Tarnocai (1997), studying the Norman

1 Wells Pipeline, provide some examples of the effect of pipeline construction on frozen peatlands and
2 permafrost in Canada.

3 Shoreline erosion along rivers, lakes, and oceans and thermal erosion (thermokarst) are also common
4 processes in the permafrost region, exposing the carbon-rich frozen soil layers to the atmosphere and
5 making the organic materials available for decomposition. As a result, carbon is released into the
6 atmosphere as either CO₂ or methane, or it enters the water system as dissolved organic carbon.

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

22

23 **OPTIONS FOR MANAGEMENT OF CARBON IN THE PERMAFROST REGION**

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

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

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

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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.

1 **REFERENCES**

- 2 AMAP, 2004: *AMAP Assessment 2002: Persistent Organic Pollutants in the Arctic*. Oslo: Arctic Monitoring and
3 Assessment Programme. xvi+310 p.
- 4 Baily, R., and C.T. Cushwa, 1981: *Ecoregions of North America*. (1:12 million scale map). U.S. Forest Service and
5 U.S. Fish and Wildlife Service.
- 6 Bockheim, J.G. and C. Tarnocai, 1998: Recognition of cryoturbation for classifying permafrost-affected soils.
7 *Geoderma*, **81**, 281–293.
- 8 Brown, J., O.J. Ferrians, Jr., J.A. Heginbottom, and E.S. Melnikov, 1997: *Circum-Arctic Map of Permafrost and*
9 *Ground Ice Conditions*. (1:10 million scale map). International Permafrost Association.
- 10 Burgess, M.M., and C. Tarnocai, 1997: Peatlands in the Discontinuous Permafrost Zone along the Norman Wells
11 pipeline, Canada. In: *Proceedings of the International Symposium on Physics, Chemistry, and Ecology of*
12 *Seasonally Frozen Soils Fairbanks, Alaska, June 10–12, 1997* (I.K. Iskandr, E.A. Wright, J.K. Radke, B.S.
13 Sharratt, P.H. Groenevelt, and L.D. Hinzman, eds.), 417–424. Hanover, U.S.A.: U.S. Army Cold Regions
14 Research and Engineering Laboratory, Special Report 97-10.
- 15 Christensen, T., 1991: Arctic and sub-Arctic soil emissions: Possible implications for global climate change. *Polar*
16 *Record*, **27**, 205–210.
- 17 Cryosol Working Group, 2001: *Northern and Mid Latitudes Soil Database, Version 1*. National Soil Database.
18 Ottawa, ON: Research Branch, Agriculture and Agri-Food Canada.
- 19 Davidson, E.A., and I.A. Janssens, 2006: Temperature sensitivity of soil carbon decomposition and feedbacks to
20 climate change. *Nature*, **440**, 165–173.
- 21 Driscoll, C.T., J. Holsapple, C.L. Schofield, and R. Munson, 1998: The chemistry and transport of mercury in a
22 small wetland in the Adirondack region of New York, USA. *Biogeochemistry*, **40**: 137–146.
- 23 Ecoregions Working Group, 1989: *Ecoclimatic Regions of Canada*, First Approximation. Ecoregions Working
24 Group of the Canada Committee on Ecological Land Classification. Ecological Land Classification Series, No.
25 23, 119 p. and map. Ottawa, ON: Sustainable Development Branch, Canadian Wildlife Service, Conservation
26 and Protection, Environment Canada.
- 27 Gorham, E., 1988: Canada's peatlands: their importance for the global carbon cycle and possible effect of
28 "greenhouse" climate warming. *Transactions of the Royal Society of Canada, Series V*, **3**, 21–23.
- 29 Hengeveld, H.G., 2000. Projections for Canada's climate future: A discussion of recent simulations with the
30 Canadian Global Climate Model. *Climate Change Digest*, CCD 00-01, Special Edition, 27 p. Downsview, ON:
31 Meteorological Service of Canada, Environment Canada. (Available from
32 http://www.msc.ec.gc.ca/saib/climate/docs/ccd_00-01.pdf. Last accessed: April 6, 2005.)
- 33 Jorgenson, M.T., C.H. Racine, J.C. Walters, and T.E. Osterkamp, 2001: Permafrost degradation and ecological changes
34 associated with a warming climate in central Alaska. *Climate Change*, **48**, 551–579.
- 35 Kettles, I.M., and Tarnocai, C., 1999: Development of a model for estimating the sensitivity of Canadian peatlands to
36 climate warming. *Geographie physique et Quaternaire*, **53**, 323–338.

- 1 Kokelj, S.V., and C.R. Burn, 2005: Geochemistry of the active layer and near-surface permafrost, Mackenzie delta
2 region, Northwest Territories, Canada. *Canadian Journal of Earth Sciences*, **42**, 37–48.
- 3 Kuhry, G.P., 1994: The role of fire in the development of Sphagnum-dominated peatlands in the western boreal
4 Canada. *Journal of Ecology*, **82**, 899–910.
- 5 Lacelle, B., C. Tarnocai, S. Waltman, J. Kimble, N. Bliss, B. Worstell, F. Orozco-Chavez, and B. Jakobsen, 2000:
6 North American soil organic carbon map. (1:10 million scale map). Agriculture and Agri-Food Canada, USDA,
7 USGS, INEGI and Institute of Geography, University of Copenhagen.
- 8 Liblik, L.K., T.R. Moore, J.L. Bubier, and S.D. Robinson, 1997: Methane emissions from wetlands in the zone of
9 discontinuous permafrost: Fort Simpson, Northwest Territories, Canada. *Global Biogeochemical Cycles*, **11**,
10 485–494.
- 11 Mackay, J.R., 1980: The origin of hummocks, western Arctic coast, Canada. *Canadian Journal of Earth Sciences*,
12 **13**, 889–897.
- 13 Melillo, J.M., T.V. Callaghan, F.I. Woodward, E. Salati, and S.K. Sinha, 1990: Effects on Ecosystems. In: *Climate*
14 *Change: The IPCC Scientific Assessment* (J.T. Houghton, G.J. Jenkins, and J.J. Ephraums, eds.), 283–310
15 (Chapter 10). Cambridge (UK): Cambridge University Press.
- 16 Moore, T.R., 1997: Dissolved organic carbon: sources, sinks, and fluxes and role in the soil carbon cycle. In: *Soil*
17 *Processes and the Carbon Cycle* (R. Lal, J.M. Kimble, R.F. Follett, and B.A. Stewart, eds.), 281–292 (Chapter
18 19). Boca Raton: Advances in Soil Science, CRC Press.
- 19 Moore, T.R., and N.T. Roulet, 1995: Methane emissions from Canadian peatlands. In: *Soils and Global Change* (R.
20 Lal, J. Kimble, E. Levine, and B.A. Stewart, eds.), 153–164 (Chapter 12). Boca Raton: CRC Lewis Publishers.
- 21 National Wetlands Working Group, 1988: *Wetlands of Canada*, Ecological Land Classification Series No. 24, 452 p.
22 Ottawa: Sustainable Development Branch, Environment Canada and Montreal: Polyscience Publications, Ltd.
- 23 Oechel, W., and G.L. Vourlitis, 1994: The effect of climate change on land–atmosphere feedbacks in arctic tundra
24 regions. *TREE*, **9**, 324–329.
- 25 Peterson, R.A., and W.B. Krantz, 2003: A mechanism for differential frost heave and its implications for patterned-
26 ground formation. *Journal of Glaciology*, **49**(164), 69–80(12).
- 27 Reader, R.J., and J.M. Stewart, 1972: The relationship between net primary production and accumulation for a
28 peatland in southeastern Manitoba. *Ecology*, **53**, 1024–1037.
- 29 Ritchie, J.C., 1987: *Postglacial Vegetation of Canada*, 178 p. New York, NY: Cambridge University Press.
- 30 Robinson, S.D., and T.R. Moore, 1999: Carbon and peat accumulation over the past 1200 years in a landscape with
31 discontinuous permafrost, northwestern Canada. *Global Biogeochemical Cycles*, **13**, 591–601.
- 32 Robinson, S.D., and T.R. Moore, 2000: The influence of permafrost and fire upon carbon accumulation in High
33 Boreal peatlands, Northwest Territories, Canada. *Arctic, Antarctic, and Alpine Research*, **32**(2), 155–166.
- 34 Robinson, S.D., M.R. Turetsky, I.M. Kettles, and R.K. Wieder, 2003: Permafrost and peatland carbon sink capacity
35 with increasing latitude. In: *Proceedings of the 8th International Conference on Permafrost*, Zurich, Switzerland
36 (M. Phillips, S.M. Springman, and L.U. Arenson, eds.), **2**, 965–970. Lisse, The Netherlands: Balkema
37 Publishers.

- 1 Soil Carbon Database Working Group, 1993: Soil carbon for Canadian soils. (digital database). Ottawa, ON: Centre
2 for Land and Biological Resources Research, Research Branch, Agriculture and Agri-Food Canada.
- 3 Suchanek, T.H., P.J. Richerson, J.R. Flanders, D.C. Nelson, L.H. Mullen, L.L. Brester, and J.C. Becker, 2000:
4 Monitoring inter-annual variability reveals sources of mercury contamination in Clear Lake, California.
5 *Environmental Monitoring and Assessment*, 64: 299–310.
- 6 Tarnocai, C., 1998: The amount of organic carbon in various soil orders and ecological provinces in Canada. In: *Soil*
7 *Processes and the Carbon Cycle* (R. Lal, J.M. Kimble, R.L.F. Follett, and B.A. Stewart, eds.), 81–92. New
8 York, NY: Advances in Soil Science, CRC Press.
- 9 Tarnocai, C., 1999: The effect of climate warming on the carbon balance of Cryosols in Canada. In: *Cryosols and*
10 *Cryogenic Environments* (C. Tarnocai, R. King, and S. Smith, eds.), special issue of *Permafrost and Periglacial*
11 *Processes*, **10**(3), 251–263.
- 12 Tarnocai, C., 2000: Carbon pools in soils of the Arctic, Subarctic and Boreal regions of Canada. In: *Global Climate*
13 *Change and Cold Regions Ecosystems* (R. Lal, J.M. Kimble, and B.A. Stewart, eds.), 91–103. Boca Raton, Fla.:
14 Advances in Soil Science, Lewis Publishers.
- 15 Tarnocai, C., In press: The effect of climate change on carbon in Canadian peatlands. *Global and Planetary Change*.
- 16 Tarnocai, C., I.M. Kettles, and B. Lacelle, 2005: Peatlands of Canada Database. (digital database). Ottawa, ON:
17 Research Branch, Agriculture and Agri-Food Canada.
- 18 Trumbore, S.E., and J.W. Harden, 1997: Accumulation and turnover of carbon in organic and mineral soils of the
19 BOREAS northern study area. *Journal of Geophysical Research*, **102**(D24), 28,817–28,830.,
- 20 Turetsky, M.R., B.D. Amiro, E. Bosch, and J.S. Bhatti, 2004: Historical burn area in western Canadian peatlands
21 and its relationship to fire weather indices. *Global Biogeochemical Cycles*, **18**, GB4014,
22 doi:10.1029/2004GB002222.
- 23 Vandenberghe, J., 1992: Cryoturbations: a sediment structural analysis. *Permafrost and Periglacial Processes*, **4**,
24 121–135.
- 25 van Everdingen, R. (ed.), 1998 revised May 2005: *Multi-language Glossary of Permafrost and Related Ground-ice*
26 *Terms*. 90 p. Boulder, CO: National Snow and Ice Data Center/World Data Center for Glaciology. (Available
27 from <http://nsidc.org/fgdc/glossary>)
- 28 Van Vliet-Lanoë, B., 1991: Differential frost heave, load casting and convection: converging mechanisms; a
29 discussion of the origin of cryoturbations. *Permafrost and Periglacial Processes*, **2**, 123–139.
- 30 Vitt, D.H., L.A. Halsey, I.E. Bauer, and C. Campbell, 2000: Spatial and temporal trends in carbon storage of
31 peatlands of continental western Canada through the Holocene. *Canadian Journal of Earth Sciences*, **37**, 683–
32 693.
- 33 Walker, D.A., V.E. Romanovsky, W.B. Krantz, C.L. Ping, R.A. Peterson, M.K. Raynolds, H.E. Epstein, J.G. Jia,
34 and D.C. Wirth, 2002: Biocomplexity of frost boil ecosystem on the Arctic Slope, Alaska. ARCUS 14th Annual
35 Meeting and Arctic Forum 2002, Arlington, Virginia, USA. (Available from
36 http://siempre.arcus.org/4DACTION/wi_pos_displayAbstract/5/391)

1 Zoltai, S.C., C. Tarnocai, and W.W. Pettapiece, 1978: Age of cryoturbated organic material in earth hummocks from
2 the Canadian arctic. *Proceedings of the Third International Conference on Permafrost*, 325–331. Edmonton,
3 AB.

4 Zoltai, S.C., S. Taylor, J.K. Jeglum, G.F. Mills, and J.D. Johnson, 1988: Wetlands of Boreal Canada. In: *Wetlands of*
5 *Canada*, (National Wetlands Working Group) 97–154. Ecological Land Classification Series, No. 24. Ottawa,
6 ON: Sustainable Development Branch, Environment Canada and Montreal, PQ: Polyscience Publications Inc.

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Table 12-1. Areas of mineral soils in the various permafrost zones

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

^aCalculated using the Soil Carbon of Canada Database (Soil Carbon Database Working Group, 1993).

^bCalculated using the Northern and Mid Latitudes Soil Database (Cryosol Working Group, 2001).

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Table 12-2. Areas of peatlands (organic soils) in the various permafrost zones

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

^aCalculated using the Peatlands of Canada Database (Tarnocai *et al.*, 2005).

^bCalculated using the Northern and Mid Latitudes Soil Database (Cryosol Working Group, 2001).

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1 **Table 12-3. Organic carbon accumulation and loss in various Canadian peatlands.** Positive
 2 values indicate net flux into the atmosphere (source); negative values
 3 indicate carbon sequestration (land sinks)

Peatlands	Amount of carbon
Boreal peatlands	-9.8 Mt yr ^{-1a}
All Canadian peatlands	-30 Mt yr ^{-1b}
All mineral and organic soils	-18 mg m ⁻² yr ^{-1c}
Rich fens	-13.58 g m ⁻² yr ^{-1d}
Poor fens (unfrozen, Discontinuous Permafrost Zone)	-20.34 g m ⁻² yr ^{-1d}
Peat plateaus (frozen, Discontinuous Permafrost Zone)	-13.31 g m ⁻² yr ^{-1d}
Collapse fens	-13.54 g m ⁻² yr ^{-1d}
Bogs (unfrozen, Discontinuous Permafrost Zone)	-21.81 g m ⁻² yr ^{-1d}
Dissolved organic carbon (DOC)	+2 g m ⁻² yr ^{-1e}
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 ^{-2g}
Carbon release by fires in all terrain	+27 Mt yr ^{-1h}
Carbon release by fires in Western Canadian peatlands	+5.9 Mt yr ^{-1h}

4 ^aZoltai *et al.*, 1988.

5 ^bGorham, 1988.

6 ^cLiblik *et al.*, 1997.

7 ^dRobinson and Moore, 1999.

8 ^eMoore, 1997.

9 ^fCalculated based on the thickness of the deposit and the date of the basal peat (National Wetlands
 10 Working Group, 1988).

11 ^gRobinson and Moore, 2000.

12 ^hTuretsky *et al.*, 2004.

1 **Table 12-4. Soil carbon pools and fluxes for the permafrost areas of Canada.** Positive flux numbers indicate net
 2 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			

3 ^aCalculated using the Peatlands of Canada Database (Tarnocai *et al.*, 2005).

4 ^bCalculated using the Soil Carbon of Canada Database (Soil Carbon Database Working Group, 1993).

5 ^cTarnocai, 1998.

6 ^dUsing C accumulation rate of 0.13 mg ha⁻¹ yr⁻¹ (this report).

7 ^eUsing C accumulation rate of 0.194 mg ha⁻¹ yr⁻¹ (Vitt *et al.*, 2000).

8 ^fRobinson and Moore, 1999.

9 ^gTrumbore and Harden, 1997.

10 ^hFires recur every 150–190 years (Kuhry, 1994; Robinson and Moore, 2000).

11 ⁱRobinson and Moore, 2000.

12 ^jMoore and Roulet, 1995.

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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

^aFor the 1-m depth.

^bFor the total depth of the peat deposit.

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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

^aCalculated for the 0–100 cm depth.

^bCalculated using the Soil Carbon of Canada Database (Soil Carbon Database Working Group, 1993).

^cCalculated using the Northern and Mid Latitudes Soil Database (Cryosol Working Group, 2001).

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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

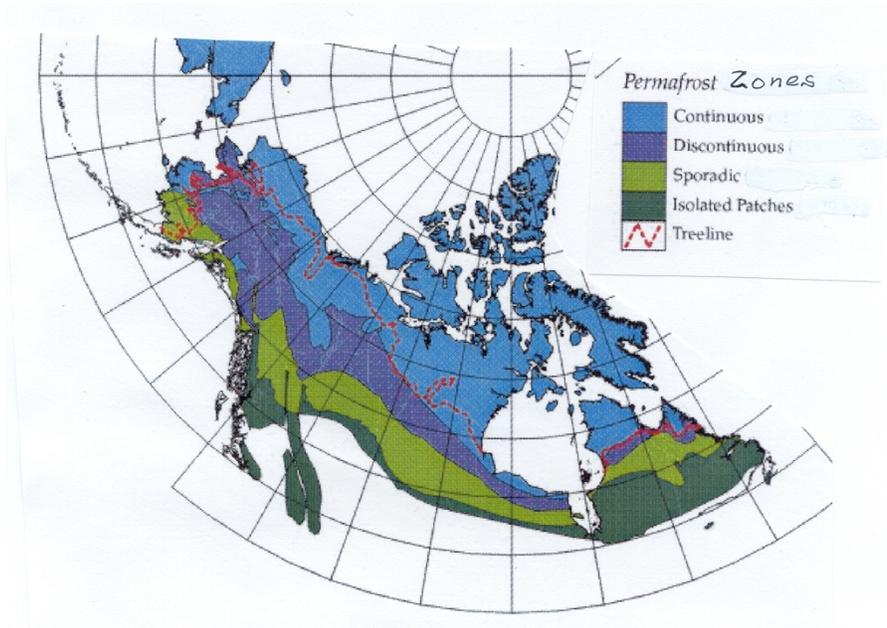
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^aCalculated for the total depth of the peat deposit.

^bCalculated using the Peatlands of Canada Database (Tarnocai *et al.*, 2005).

^cCalculated using the Northern and Mid Latitudes Soil Database (Cryosol Working Group, 2001).

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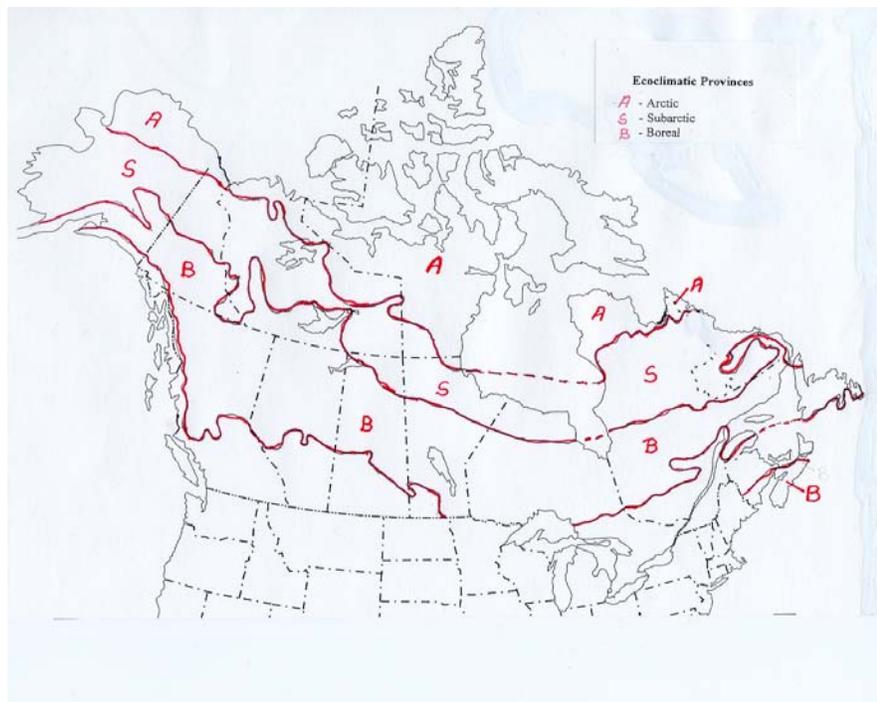


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Fig. 12-1. Permafrost zones in North America (Brown *et al.*, 1997).

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Fig. 12-2. Arctic, Subarctic, and Boreal ecoclimatic provinces (ecological regions) in North America

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(Ecoregions Working Group, 1989; Baily and Cushwa, 1981).

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Carbon sinks

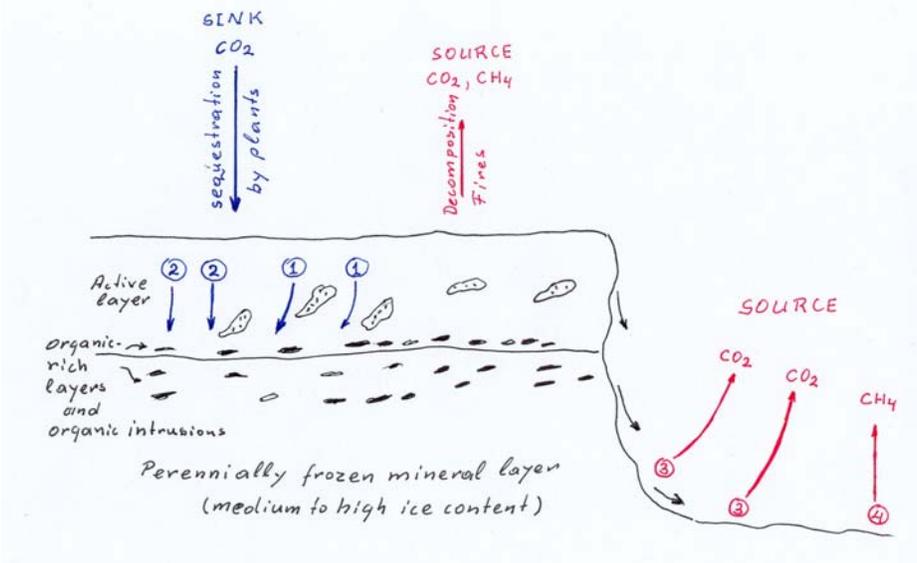


Permafrost-affected soil with a thick surface organic layer, dark-colored organic intrusions in the brown soil layer, and an underlying frozen, high-ice-content layer. The organic intrusions were translocated from the surface by cryoturbation. (Mackenzie Valley, Canada)

Carbon sources



Eroding high-ice-content permafrost soil composed of a dark frozen soil layer with an almost pure ice layer below. The thawing process generated a flow slide in which high-organic-content soil materials slumped into the water-saturated environment. (Mackenzie Delta area, Canada)

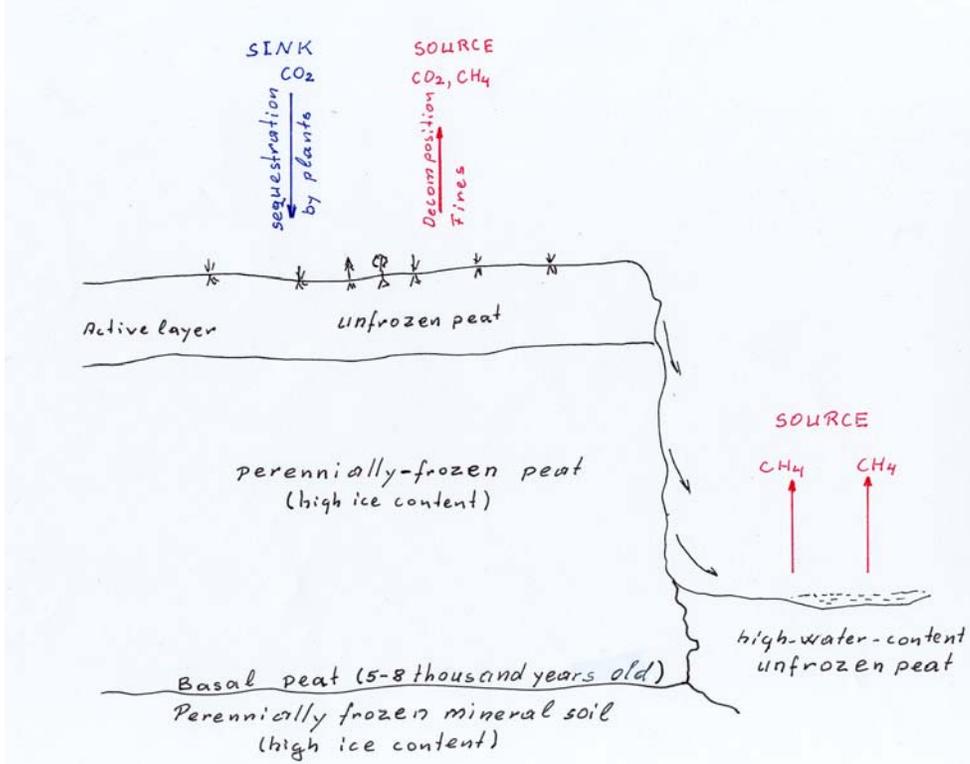


Perennially frozen deposit composed of an active layer that freezes and thaws annually and an underlying perennially frozen layer that has a high ice content.

Organic material deposited annually on the soil surface builds up as an organic soil layer. Some of this surface organic material is translocated into the deeper soil layers by cryoturbation (1). In addition, soluble organic matter is translocated into the deeper soil layers by movement of water to the freezing front and by gravity (2). Because these deeper soil layers have low temperatures (0 to -15°C), the organic material decomposes very slowly. Thus more organic material accumulates as long as the soil is frozen. In this state, the permafrost soil acts as a carbon sink.

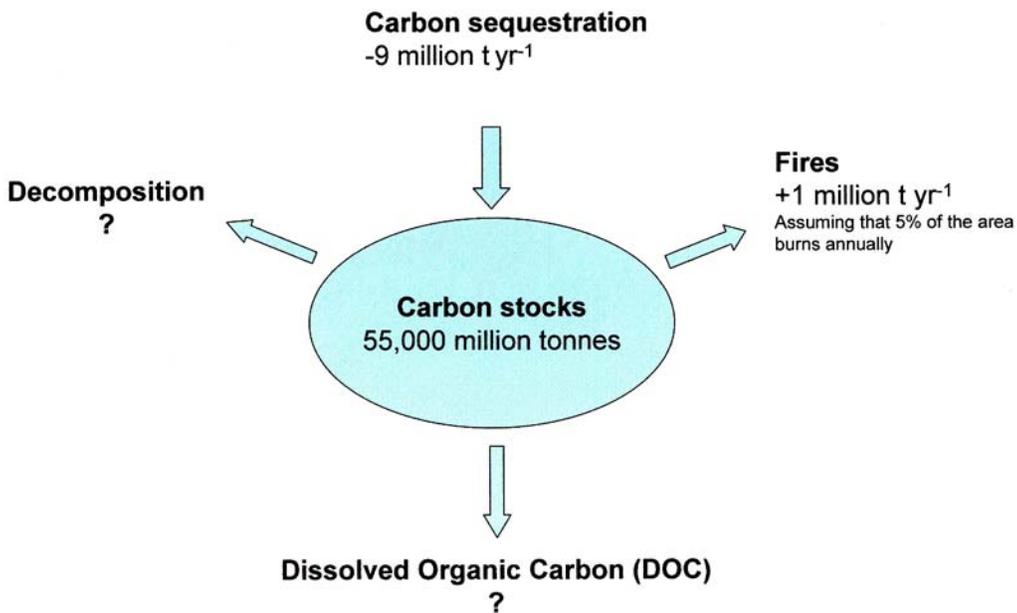
Thermal erosion initiated by climate warming, wildfires or human activity causes the high-ice-content mineral soils to thaw, releasing the organic materials locked in the system. In this environment aerobic (3) and anaerobic (4) decomposition occurs releasing carbon dioxide and methane. In this state, the soil is a source of carbon.

2 **Fig. 12-3. Carbon cycle in permafrost-affected upland (mineral) soils, showing below-ground organic**
 3 **carbon sinks and sources.**
 4

Carbon sinks	Carbon sources
 <p data-bbox="207 604 748 653">Perennially frozen peat deposit with multiple dark-colored peat layers. (Mackenzie River Delta area, Canada)</p>	 <p data-bbox="818 604 1321 674">Eroding perennially frozen peat deposit, showing the large blocks of peat slumping into the water-saturated collapsed area. (Fort Simpson area, Canada)</p>
	
<p data-bbox="207 1478 789 1549">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.</p> <p data-bbox="207 1562 789 1751">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.</p>	<p data-bbox="818 1478 1341 1646">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.</p>

1 Fig. 12-4. Carbon cycle in permafrost peatlands, showing below-ground organic carbon sinks and
 2 sources.

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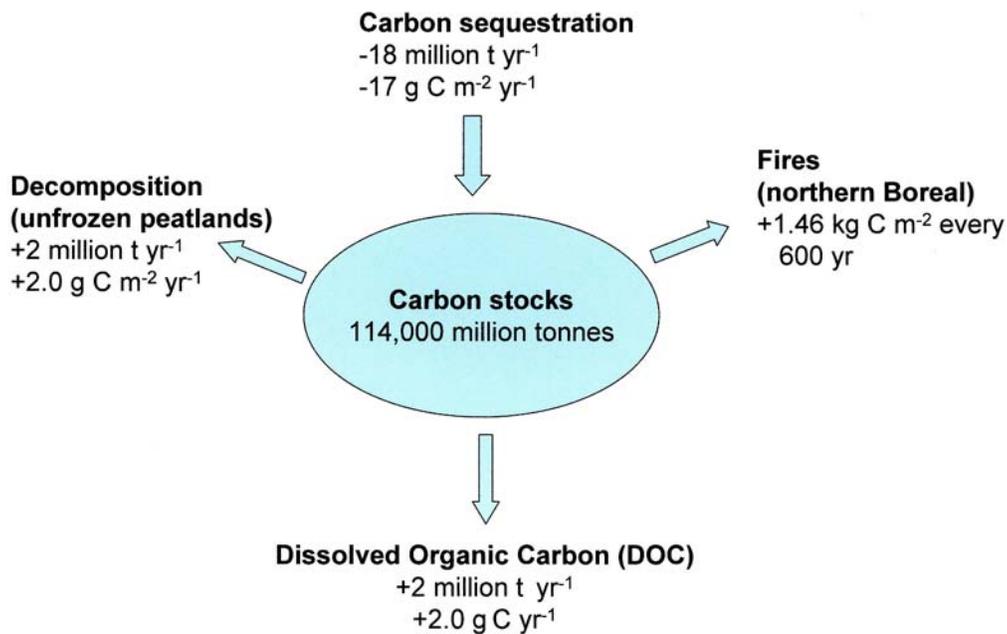
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Fig. 12-5. Carbon cycle in perennially frozen mineral soils in the permafrost region.

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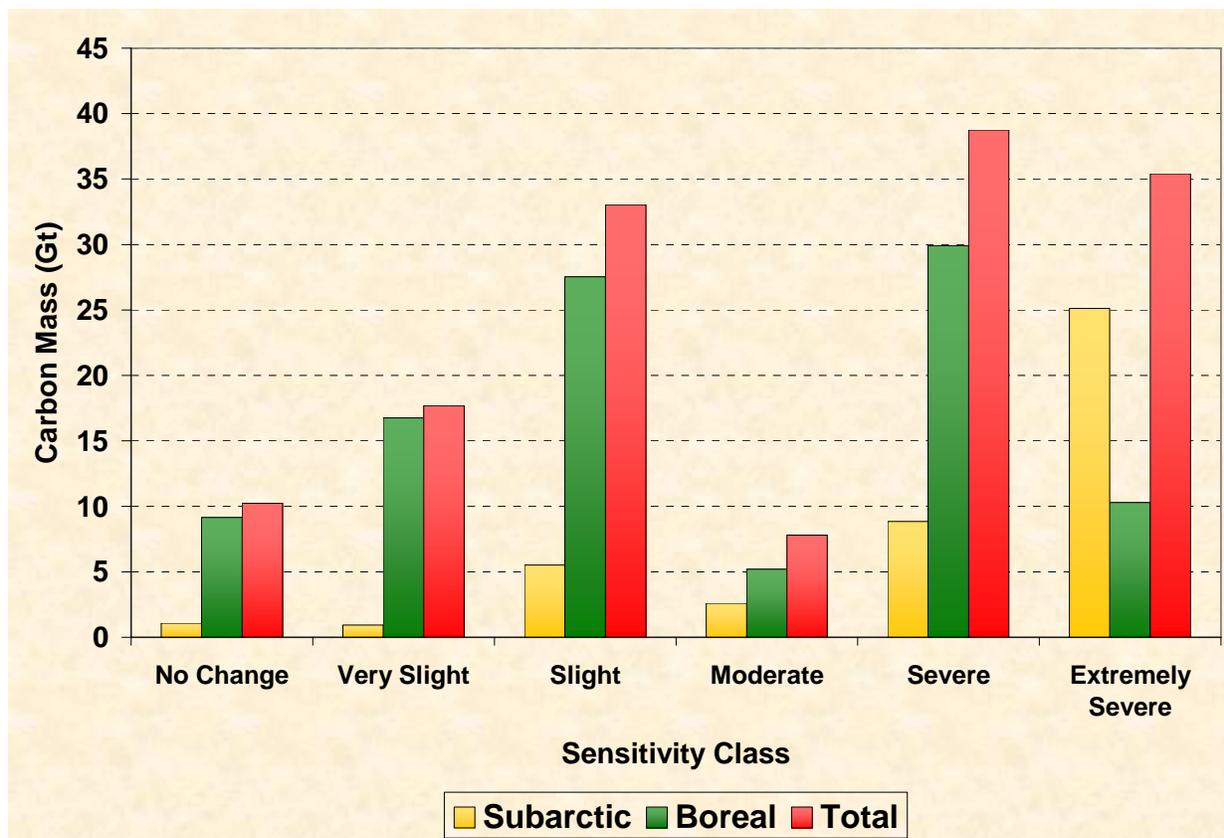
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Fig. 12-6. Carbon cycle in peatlands in the permafrost region.

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Fig. 12-7. The organic carbon mass in the various sensitivity classes for the Subarctic and Boreal Ecoclimatic Provinces (ecological regions) (Tarnocai, in press).

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4

Chapter 13. Wetlands

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KEY FINDINGS

- North America is home to approximately 41% of the global wetland area, encompassing about 2.5 million km² with a carbon pool of approximately 220 Gt, mostly in peatland soils.
 - North American wetlands currently are a CO₂ 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 CO₂ 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.
-

1 INTRODUCTION

2 While there are a variety of legal and scientific definitions of a wetland (National Research Council,
3 1995; National Wetlands Working Group, 1997), most emphasize the presence of waterlogged conditions
4 in the upper soil profile during at least part of the growing season, and plant species and soil conditions
5 that reflect these hydrologic conditions. Waterlogging tends to suppress microbial decomposition more
6 than plant productivity, so wetlands are known for their ability to accumulate large amounts of carbon,
7 most spectacularly seen in large peat deposits that are often many meters deep. Thus, when examining
8 carbon dynamics, it is important to distinguish between freshwater wetlands with surface soil organic
9 matter deposits >40 cm thick (i.e., peatlands) and those with lesser amounts of soil organic matter (i.e.,
10 freshwater mineral-soil wetlands, FWMS). Some wetlands have permafrost; fluxes and pools in wetlands
11 with and without permafrost are discussed separately in Appendix 13A. We also differentiate between
12 freshwater wetlands and estuarine wetlands (salt marshes, mangroves, and mud flats) with marine-derived
13 salinity.

14 Peatlands occupy about 3% of the terrestrial global surface, yet they contain 16–33% of the total soil
15 carbon pool (Gorham, 1991; Maltby and Immirzi, 1993). Most peatlands occur between 50 and 70° N,
16 although significant areas occur at lower latitudes (Matthews and Fung, 1987; Aselmann and Crutzen,
17 1989; Maltby and Immirzi, 1993). Large areas of peatlands exist in Alaska, Canada, and in the northern
18 midwestern, northeastern, and southeastern United States (Bridgman *et al.*, 2000). This peat has formed
19 over thousands of years, and therefore the potential emissions from the large pool of soil carbon are likely
20 more significant to the global carbon budget than the current soil carbon sequestration rate. Large areas of
21 wetlands have been converted to other land uses globally and in North America (Dugan, 1993; OECD,
22 1996), which may have resulted in a net flux of carbon to the atmosphere (Armentano and Menges, 1986;
23 Maltby and Immirzi, 1993). Additionally, wetlands emit 92–237 Mt methane (CH₄) yr⁻¹, a large fraction
24 of the total annual global flux of about 600 Mt CH₄ yr⁻¹ (Ehhalt *et al.*, 2001). This is important because
25 methane is a potent greenhouse gas, second in importance to only carbon dioxide (Ehhalt *et al.*, 2001).

26 A number of previous studies have examined the role of peatlands in the global carbon balance
27 (reviewed in Mitra *et al.*, 2005). Roulet (2000) focused on the role of Canadian peatlands in the Kyoto
28 process. Here we augment these previous studies by considering all types of wetlands (not just peatlands)
29 and integrate new data to examine the carbon balance in the wetlands of Canada, the United States, and
30 Mexico.

31 Given that many undisturbed wetlands are a natural sink for carbon dioxide and a source of methane,
32 a note of caution in interpretation of our data is important. Using the International Panel on Climate
33 Change (IPCC) terminology, a radiative forcing denotes “an externally imposed perturbation in the
34 radiative energy budget of the Earth’s climate system” (Ramaswamy *et al.*, 2001). Thus, it is the change

1 from a baseline condition in greenhouse gas fluxes in wetlands that constitute a radiative forcing that will
2 impact climate change, and the emissions of greenhouse gases from unperturbed wetlands is important
3 only in establishing a baseline condition. Thus, we consider changes from historical (~1800) fluxes and
4 present and future perturbations of greenhouse gas fluxes in North American wetlands.

6 INVENTORIES

7 Current Wetland Area and Rates of Loss

8 The current and historical wetland area and rates of loss are the basis for all further estimates of pools
9 and fluxes in this chapter. The loss of wetlands has caused the oxidation of their soil carbon, particularly
10 in peatlands; reduced their ability to sequester carbon; and reduced their emissions of methane. The
11 strengths and weakness of the wetland inventories of Canada, the United States, and Mexico are discussed
12 in Appendix 13A.

13 The conterminous United States has 312,000 km² of FWMS wetlands, 93,000 km² of peatlands, and
14 23,000 km² of estuarine wetlands, which encompass 5.5% of the land area (Table 13-1). This represents
15 just 48% of the original wetland area in the conterminous United States (Table 13A-1 in Appendix 13A).
16 However, wetland losses in the United States have declined from 1,855 km² yr⁻¹ in the 1950s–1970s to
17 237 km² yr⁻¹ in the 1980s–1990s (Dahl, 2000). Such data mask large differences in loss rates among
18 wetland classes and conversion of wetlands to other classes, with potentially large effects on carbon
19 stocks and fluxes (Dahl, 2000). For example, the majority of wetland losses in the United States have
20 occurred in FWMS wetlands. As of the early 1980s, 84% of U.S. peatlands were unaltered (Armentano
21 and Menges, 1986; Maltby and Immirzi, 1993; Rubec, 1996), and, given the current regulatory
22 environment in the United States, recent rates of loss are likely small.

23
24 **Table 13-1. The area, carbon pool, net carbon balance, and methane flux from wetlands in North**
25 **America and the world.** Positive fluxes indicate net fluxes to the atmosphere, whereas negative fluxes
26 indicate net fluxes into an ecosystem. Citations and assumptions in calculations are in the text and in
27 Appendix 13A.

28
29 Canada has 1,301,000 km² of wetlands, covering 14% of its land area, of which 87% are peatlands
30 (Table 13-1). Canada has lost about 14% of its wetlands, mainly due to agricultural development of
31 FWMS wetlands (Rubec, 1996), although the ability to estimate wetland losses in Canada is limited by
32 the lack of a regular wetland inventory.

1 The wetland area in Mexico is estimated at 36,000 km² (Table 13-1), with an estimated historical loss
2 of 16,000 km² (Table 13A-1 in Appendix 13A). However, given the lack of a nationwide wetland
3 inventory and a general paucity of data, this number is highly uncertain.

4 Problems with inadequate wetland inventories are even more prevalent in lesser developed countries
5 (Finlayson *et al.*, 1999). We estimate a global wetland area of 6.0×10^6 km² (Table 13-1); thus, North
6 America currently has about 43% of the global wetland area. It has been estimated that about 50% of the
7 world's historical wetlands have been converted to other uses (Moser *et al.*, 1996).

8 9 **Carbon Pools**

10 We estimate that North American wetlands have a current soil and plant carbon pool of 220 Gt, of
11 which approximately 98% is in the soil (Table 13-1). The majority of this carbon is in peatlands, with
12 FWMS wetlands contributing about 18% of the carbon pool. The large amount of soil carbon (27 Gt) in
13 Alaskan FWMS wetlands had not been identified in previous studies (see Appendix 13A).

14 15 **Soil Carbon Fluxes**

16 North American peatlands currently have a net carbon balance of about -18 Mt C yr⁻¹ (Table 13-1),
17 but several large fluxes are incorporated into this estimate. (**Negative numbers indicate net fluxes into
18 the ecosystem, whereas positive numbers indicate net fluxes into the atmosphere.**) Peatlands
19 sequester -34 Mt C yr⁻¹ (Table 13A-2 in Appendix 13A), but peatlands in the conterminous United States
20 that have been drained for agriculture and forestry had a net oxidative flux of 18 Mt C yr⁻¹ as of the early
21 1980s (Armentano and Menges, 1986). Despite a substantial reduction in the rate of wetland loss since the
22 1980s (Dahl 2000), drained organic soils continue to lose carbon over many decades, so the actual flux to
23 the atmosphere is probably close to the 1980s estimate. There has also been a loss in sequestration
24 capacity in drained peatlands of 2.4 Mt C yr⁻¹ (Table 13-1), so the overall soil carbon sink of North
25 American peatlands is about 21 Mt C yr⁻¹ smaller than it would have been in the absence of disturbance.

26 Very little attention has been given to the role of FWMS wetlands in North American or global
27 carbon balance estimates, with the exception of methane emissions. Carbon sequestration associated with
28 sediment deposition is a potentially large, but poorly quantified, flux in wetlands (Stallard, 1998). Using a
29 review by Johnston (1991), we calculate a substantial carbon accumulation rate in sedimentation in
30 FWMS wetlands of -129 g C m⁻² yr⁻¹ (see Appendix 13A). However, it is extremely unlikely that the
31 actual sequestration rate is this high, as the data are probably strongly biased by researchers choosing
32 wetlands with high sediment deposition to study this process. More fundamentally, carbon in sediments
33 that are simply redistributed in the landscape due to erosion from a terrestrial source to a wetland sink
34 does not represent carbon sequestration except to the extent that decomposition rates are lower in

1 wetlands. Much of this sediment-associated carbon is probably relatively stable in upland soils, so FWMS
2 wetlands may not represent a substantial sediment carbon sink at the landscape scale. There are no data to
3 our knowledge to evaluate this important caveat. Based upon this reasoning, we somewhat arbitrarily
4 reduced our calculated FWMS wetland sediment carbon sequestration rate by 75% to -34 Mt C yr^{-1} (Table
5 13A-2 in Appendix 13A). This is still a substantial sink and an important unknown in carbon budgets. For
6 example, Stallard (1998) estimated that global wetlands are a large sediment sink, with a flux on the order
7 of -1 Gt C yr^{-1} . However, this analysis was based on many assumptions and was acknowledged by the
8 author to be a first guess at best.

9 Decomposition of soil carbon in FWMS wetlands that have been converted to other land uses appears
10 to be responsible for only a negligible loss of soil carbon currently (Table 13A-2 in Appendix 13A).
11 However, due to the historical loss of FWMS wetland area, we estimate that they currently sequester
12 21 Mt C yr^{-1} less than they did prior to disturbance (Table 13-1). This estimate has the same unknowns
13 described in the previous paragraph on current sediment carbon sequestration in FWMS wetlands.

14 We estimate that estuarine wetlands currently sequester $-9.7 \text{ Mt C yr}^{-1}$, with a historical reduction in
15 sequestration capacity of 1.4 Mt C yr^{-1} due to loss of area (Table 13-1). Despite the relatively small area
16 of estuarine wetlands, they currently contribute about 26% of total wetland carbon sequestration in the
17 conterminous United States and about 14% of the North American total. Estuarine wetlands sequester
18 carbon at a rate about 10 times higher on an area basis than other wetland ecosystems due to high
19 sedimentation rates, high soil carbon content, and constant burial due to sea level rise. Estimates of
20 sediment deposition rates in estuarine wetlands are robust, but it is unknown to what extent soil carbon
21 sequestration is divided into allochthonous carbon (sediment-derived carbon from outside the wetland)
22 and autochthonous carbon (derived from rates of plant productivity being greater than decomposition
23 within the wetland). As with FWMS wetlands, soil carbon sequestration in estuarine wetlands is
24 overestimated to the extent that allochthonous carbon simply represents redistribution of carbon in the
25 landscape. There is also large uncertainty in the area of mud flats.

26 Overall, North American wetland soils appear to be a substantial carbon sink with a net flux of
27 -70 Mt C yr^{-1} (with very large error bounds because of FWMS wetlands) (Table 13-1). The large-scale
28 conversion of wetlands to upland uses has led to a reduction in the wetland soil carbon sequestration
29 capacity of 25 Mt C yr^{-1} from the likely historical rate (Table 13-1), but this estimate is driven by large
30 losses of FWMS wetlands with their highly uncertain sedimentation carbon sink. With the current net
31 oxidative flux of 18 Mt C yr^{-1} from conterminous U.S. peatlands, we estimate that North American
32 wetlands currently sequester 43 Mt C yr^{-1} less than they did historically (Table 13A-2 in Appendix 13A).
33 Furthermore, North American peatlands and FWMS wetlands have lost 2.6 Gt and 4.9 Gt of soil carbon,
34 respectively, and collectively they have lost 2.4 Gt of plant carbon since approximately 1800. Very little

1 data exist to estimate carbon fluxes for freshwater Mexican wetlands, but because of their small area, they
2 will not likely have a large impact on the overall North American estimates.

3 The global wetland soil carbon balance has only been examined in peatlands. The current change in
4 soil carbon flux in peatlands is about 176 to 266 Mt C yr⁻¹ (Table 13A-2 in Appendix 13A), largely due to
5 the oxidation of peat drained for agriculture and forestry and secondarily due to peat combustion for fuel
6 (Armentano and Menges, 1986; Maltby and Immerzi 1993). Thus, globally peatlands are a moderate
7 atmospheric source of carbon. The cumulative historical shift in soil carbon stocks has been estimated to
8 be 5.5 to 7.1 Gt C (Maltby and Immerzi, 1993).

10 Methane and Nitrous Oxide Emissions

11 We estimate that North American wetlands emit 26 Mt CH₄ yr⁻¹ (Table 13-1). Our synthesis is
12 substantially higher than the previous estimate by Bartlett and Harriss (1993) (see Appendix 13A). A
13 mechanistic methane model yielded similar rates of 3.8 and 7.1 Mt CH₄ yr⁻¹ for Alaska and Canada,
14 respectively (Zhuang et al., 2004). For comparison, a regional inverse atmospheric modeling approach
15 estimated total methane emissions (from all sources) of 16 and 54 Mt CH₄ yr⁻¹ for boreal and temperate
16 North America, respectively (Fletcher *et al.*, 2004a).

17 Methane emissions are currently about 24 Mt CH₄ yr⁻¹ less than they were historically in North
18 American wetlands (see Table 13A-4 in Appendix 13A) because of the loss of wetland area. We do not
19 consider the effects of conversion of wetlands from one type to another (Dahl 2000), which may have a
20 significant impact on methane emissions. Similarly, we estimate that global methane emissions from
21 natural wetlands are only about half of what they were historically (Table 13A-4 in Appendix 13A).
22 However, this may be an overestimate because wetland losses have been higher in more developed
23 countries than less developed countries (Moser *et al.*, 1996), and wetlands at lower latitudes have higher
24 emissions on average (Bartlett and Harriss, 1993).

25 When we multiplied the very low published estimates of nitrous oxide emissions from natural and
26 disturbed wetlands (Joosten and Clarke, 2002) by North American wetland area, the flux was insignificant
27 (data not shown).

28 The global warming potential (GWP) of a gas depends on its instantaneous radiative forcing and its
29 lifetime in the atmosphere, with methane having GWPs of 1.9, 6.3, and 16.9 CO₂-carbon equivalents on a
30 mass basis across 500-year, 100-year, and 20-year time frames, respectively (Ramaswamy *et al.*, 2001).¹
31 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 CO₂-mass equivalents. We have converted them into CO₂-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].

1 Table 13-1, North American wetlands as a whole currently are in a range between approximately neutral
2 and a large source of net CO₂-carbon equivalents to the atmosphere (but note caution in the *Introduction*
3 in converting this into radiative forcing). It is likely that FWMS wetlands, with their high methane
4 emissions, are a net source of CO₂-carbon equivalents to the atmosphere. In contrast, estuarine wetlands
5 are a net sink for CO₂-carbon equivalents because they support both rapid rates of carbon sequestration
6 and low methane emissions. However, caution should be exercised in using GWPs to draw conclusions
7 about changes in the net flux of CO₂-carbon equivalents because GWPs are based upon a pulse of a gas
8 into the atmosphere, whereas carbon sequestration is more or less continuous. For example, if one
9 considers continuous methane emissions and carbon sequestration in peat over time, most peatlands are a
10 net sink for CO₂-carbon equivalents because of the long lifetime of carbon dioxide sequestered as peat
11 (Frolking *et al.*, 2006).

12 13 **Plant Carbon Fluxes**

14 We estimate that wetland forests in the conterminous United States currently sequester
15 -10.3 Mt C yr⁻¹ as increased plant biomass (see Table 13A-3 in Appendix 13A). Sequestration in plants in
16 undisturbed wetland forests in Alaska and many peatlands is probably minimal, although there may be
17 substantial logging of Canadian forested peatlands that we do not have the data to account for.

18 19 **TRENDS AND DRIVERS OF WETLAND CARBON FLUXES**

20 Historically, the destruction of wetlands through land-use changes has had the largest effect on the
21 carbon fluxes and the GWPs of North American wetlands. The primary effects have been a reduction in
22 their ability to sequester carbon (a small to moderate increase in radiative forcing depending on carbon
23 sequestration by sedimentation in FWMS and estuarine wetlands), oxidation of their soil carbon reserves
24 upon drainage (a small increase in radiative forcing), and a reduction in the emission of methane to the
25 atmosphere (a moderate decrease in radiative forcing) (Table 13A-1 and Appendix 13A). While extensive
26 research has been done on carbon cycling and pools in North American wetlands, to our knowledge, this
27 is the first attempt at an overall carbon budget for all of the wetlands of North America, although others
28 have examined the carbon budget for North American peatlands as part of global assessments (Armentano
29 and Menges, 1986; Maltby and Immirzi, 1993; Joosten and Clarke, 2002). Globally, the disturbance of
30 peatlands appears to have shifted them into a net source of carbon to the atmosphere. Any positive effect
31 of wetland loss due to a reduction in their methane emissions, and hence radiative forcing, will be more
32 than negated by the loss of the many ecosystem services they provide such as havens for biodiversity,
33 recharge of groundwater, reduction in flooding, fish nurseries, etc. (Zedler and Kercher, in press).

1 A majority of the effort in examining future global change impacts on wetlands has focused on
2 northern peatlands because of their large soil carbon reserves, although under current climate conditions
3 they have modest methane emissions (Moore and Keddy, 1989; Roulet, 2000; Joosten and Clarke, 2002
4 and references therein). Data (Bartlett and Harriss, 1993; Moore *et al.*, 1998; Updegraff *et al.*, 2001) and
5 modeling (Gedney *et al.*, 2004; Zhuang *et al.*, 2004) strongly support the contention that water table
6 position and temperature are the primary environmental controls over methane emissions. How this
7 generalization plays out with future climate change is, however, more complex. For example, most
8 climate models predicted much of Canada will be warmer and drier in the future. Based upon this
9 prediction, Moore *et al.* (1998) proposed a variety of responses to climate change in the carbon fluxes
10 from different types of Canadian peatlands. Methane emissions may increase in collapsed former-
11 permafrost bogs (which will be warmer and wetter) but decrease in fens and other types of bogs (warmer
12 and drier). A methane-process model predicted that modest warming will increase global wetland
13 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
14 have increased by 0.08 Mt CH₄ yr⁻¹ during the twentieth century and by 1.0 Mt CH₄ yr⁻¹ during the 1980s
15 (Zhuang *et al.*, 2004). Inverse modeling also shows that atmospheric anomalies in methane during the
16 1990s may be partially explained by interannual climate effects on wetland emissions (Fletcher *et al.*,
17 2004b; Wang *et al.*, 2004). Thus, the above-mentioned studies suggest that past climate change has
18 already had an effect on wetland methane emissions and that this will only be exacerbated in the future.

19
20 Other important anthropogenic forcing factors that will affect future methane emissions include
21 atmospheric sulfate deposition (Vile *et al.*, 2003; Gauci *et al.*, 2004), atmospheric carbon dioxide
22 concentrations (Megonigal and Schlesinger, 1997; Vann and Megonigal, 2003), and nutrient additions
23 (Keller *et al.*, 2005). These external forcing factors in turn will interact with internal ecosystem
24 constraints such as pH and carbon quality (Moore and Roulet, 1995; Bridgham *et al.*, 1998), anaerobic
25 carbon flow (Hines and Duddleston, 2001), and net ecosystem productivity and plant community
26 composition (Whiting and Chanton, 1993; Updegraff *et al.*, 2001; Strack *et al.*, 2004) to determine the
27 actual response.

28 The effects of global change on carbon sequestration in peatlands is probably of minor importance as
29 a global flux because of the relatively low rate of peat accumulation. However, losses of soil carbon
30 stocks in peatlands drained for agriculture and forestry (Table 13A-2 in Appendix 13A) attest to the
31 possibility of large losses from the massive soil carbon deposits in northern peatlands if they become
32 substantially drier in a future climate. Furthermore, Turetsky *et al.* (2004) estimated that up to
33 5.9 Mt C yr⁻¹ are released from western Canadian peatlands by fire and predicted that increases in fire
34 frequency may cause these systems to become net atmospheric carbon sources. Northern peatlands may

1 also emit more methane with warmer temperatures, depending on changes in water table levels. The
2 effects of global change on estuarine wetlands is of concern because sequestration rates are rapid, and
3 they can be expected to increase with the rate of sea level rise provided the estuarine wetland area does
4 not decline. It remains to be determined whether rising atmospheric carbon dioxide, temperature, nitrogen
5 deposition, and shoreline construction will permit the area of estuarine wetlands to remain stable.
6

7 **OPTIONS AND MEASURES**

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

16 There has been strong interest expressed in using carbon sequestration as a rationale for wetland
17 restoration and creation in the United States, Canada, and elsewhere (Wylynko, 1999; Watson *et al.*,
18 2000). However, high methane emissions from conterminous U.S. wetlands suggest that creating and
19 restoring wetlands may increase net radiative forcing, although adequate data do not exist to evaluate this.
20 Roulet (2000) came to a similar conclusion concerning the restoration of Canadian wetlands. The
21 possibility of increasing radiative forcing by creating or restoring wetlands does not apply to estuarine
22 wetlands, which emit relatively little methane compared to the carbon they sequester. Restoration of
23 drained peatlands may stop the rapid loss of their soil carbon, which may compensate for increased
24 methane emissions. However, Canadian peatlands restored from peat extraction operations increased their
25 net emissions of carbon because of straw addition during the restoration process, although it was assumed
26 that they would eventually become a net sink (Cleary *et al.*, 2005).

27 Regardless of their internal carbon balance, the area of restored wetlands is currently too small to
28 form a significant carbon sink at the continental scale. Between 1986 and 1997, only 4,157 km² of
29 uplands were converted into wetlands in the conterminous United States (Dahl, 2000). However, larger
30 areas of wetland restoration may have a significant impact on carbon sequestration. A simulation model
31 of planting 20,000 km² into bottomland hardwood trees as part of the Wetland Reserve Program in the
32 United States showed a sequestration of 4 Mt C yr⁻¹ through 2045 (Barker *et al.*, 1996), although they did
33 not account for the GWP of increased methane emissions.

1 Potentially more significant is the conversion of wetlands from one type to another; for example,
2 8.7% (37,200 km²) of the wetlands in the conterminous United States in 1997 were in a previous wetland
3 category in 1986 (Dahl, 2000). The net effect of these conversions on wetland carbon fluxes is unknown.
4 Similarly, Roulet (2000) argued that too many uncertainties exist to include Canadian wetlands in the
5 Kyoto Protocol.

6 In summary, North American wetlands form a very large carbon pool because of storage as peat and
7 are a small-to-moderate carbon sink (excluding methane effects), with the largest unknown being the role
8 of carbon sequestration by sedimentation in FWMS wetlands. With the exception of estuarine wetlands,
9 methane emissions from wetlands may largely offset any positive benefits of carbon sequestration in soils
10 and plants. Given these conclusions, it is probably unwarranted to use carbon sequestration as a rationale
11 for the protection and restoration of FWMS wetlands, although the many other ecosystem services that
12 they provide justify their protection. However, protecting and restoring peatlands will stop the loss of
13 their soil carbon (at least over the long term), and estuarine wetlands are an important carbon sink given
14 their limited areal extent and low methane emissions. The most important areas for further scientific
15 research in terms of current carbon fluxes in the United States are to establish an unbiased, landscape-
16 level sampling scheme to determine sediment carbon sequestration in FWMS and estuarine wetlands and
17 to take additional measurements of annual methane emissions to better constrain these important fluxes. It
18 would also be beneficial if the approximately decadal National Wetland Inventory (NWI) status and
19 trends data were collected in sufficient detail with respect to the Cowardin *et al.* (1979) classification
20 scheme to determine changes among mineral-soil wetlands and peatlands.

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

28 Global change effects on the carbon pools and fluxes of North American wetlands are the largest
29 future unknown. We will not be able to accurately predict the role of North American wetlands as
30 potential positive or negative feedbacks to anthropogenic climate change without knowing the integrative
31 effects of changes in temperature, precipitation, atmospheric carbon dioxide concentrations, and
32 atmospheric deposition of nitrogen and sulfur within the context of internal ecosystem drivers of
33 wetlands. To our knowledge, no manipulative experiment has simultaneously measured more than two of
34 these perturbations in any North American wetland, and few have been done at any site. Modeling

1 expertise of the carbon dynamics of wetlands has rapidly improved in the last few years (Frolking *et al.*,
2 2002; Zhuang *et al.*, 2004 and references therein), but this needs even further development in the future,
3 including for FWMS wetlands.

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15 REFERENCES

- 16 Armentano, T. B., and E. S. Menges, 1986: Patterns of change in the carbon balance of organic soil-wetlands of the
17 temperate zone. *Journal of Ecology*, **74**, 755–774.
- 18 Aselmann, I., and P. J. Crutzen, 1989: Global distribution of natural freshwater wetlands and rice paddies, their net
19 primary productivity, seasonality and possible methane emissions. *Journal of Atmospheric Chemistry*, **8**, 307–
20 359.
- 21 Barker, J. R., G. A. Baumgardner, D. P. Turner, and J. J. Lee, 1996: Carbon dynamics of the conservation and
22 wetland reserve program. *Journal of Soil and Water Cons.*, **51**, 340–346.
- 23 Bartlett, K. B., and R. C. Harriss, 1993: Review and assessment of methane emissions from wetlands. *Chemosphere*,
24 **26**, 261–320.
- 25 Bridgham, S. D., C.-L. Ping, J. L. Richardson, and K. Updegraff, 2000: Soils of northern peatlands: Histosols and
26 Gelisols. In: *Wetland Soils: Genesis, Hydrology, Landscapes, and Classification* (J. L. Richardson, and M. J.
27 Vepraskas, eds.), 343–370. Boca Raton, FL: CRC Press.
- 28 Bridgham, S. D., K. Updegraff, and J. Pastor, 1998: Carbon, nitrogen, and phosphorus mineralization in northern
29 wetlands. *Ecology*, **79**, 1545–1561.
- 30 Cao, M., K. Gregson, and S. Marshall, 1998: Global methane emission from wetlands and its sensitivity to climate
31 change. *Atmospheric Environment*, **32**, 3293–3299.
- 32 Cleary, J., N. T. Roulet, and T. R. Moore, 2005: Greenhouse gas emissions from Canadian peat extraction, 1990–
33 2000: A life-cycle analysis. *Ambio*, **34**, 456–461.
- 34 Cowardin, L. M., V. Carter, F. C. Golet, and E. T. LaRoe, 1979: Classification of wetlands and deepwater habitats
35 of the United States. FWS/OBS-79/31, Washington, D.C.: Fish and Wildlife Service, U.S. Department of the
36 Interior.

- 1 Dahl, T. E., 2000: Status and Trends of Wetlands in the Conterminous United States, 1986 to 1997. Washington,
2 D.C.: Fish and Wildlife Service, U.S. Department of the Interior.
- 3 Dugan, P., ed., 1993: Wetlands in Danger—A World Conservation Atlas. New York: Oxford University Press.
- 4 Ehhalt, D., M. Prather, F. Dentener, E. Dlugokencky, E. Holland, I. Isaksen, J. Katima, V. Kirchhoff, P. Matson, P.
5 Midgley, and M. Wang, 2001: Atmospheric chemistry and greenhouse gases. In: *Climate Change 2001: The
6 Scientific Basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental
7 Panel on Climate Change* (J. T. Houghton, Y. Ding, D. J. Griggs, M. Noguer, P. J. van der Linden, X. Dai, K.
8 Maskell, and C. A. Johnson, eds.), 239–287. Cambridge: Cambridge University Press.
- 9 Finlayson, C. M., N. C. Davidson, A. G. Spiers, and N. J. Stevenson, 1999: Global wetland inventory—current
10 status and future priorities. *Marine Freshwater Research*, **50**, 717–727.
- 11 Fletcher, S. E. M., P. P. Tans, L. M. Bruhwiler, J. B. Miller, and M. Heimann, 2004a: CH₄ sources estimated from
12 atmospheric observations of CH₄ and its ¹³C/¹²C isotopic ratios: 2. Inverse modeling of CH₄ fluxes from
13 geographical regions. *Global Biogeochemical Cycles*, **18**, doi:10.1029/2004GB002224.
- 14 Fletcher, S. E. M., P. P. Tans, L. M. Bruhwiler, J. B. Miller, and M. Heimann, 2004b: CH₄ sources estimated from
15 atmospheric observations of CH₄ and its ¹³C/¹²C isotopic ratios: 1. Inverse modeling of source processes. *Global
16 Biogeochemical Cycles*, **18**, doi:10.1029/2004GB002223.
- 17 Frolking, S., N. Roulet, and J. Fuglestedt, 2006: How northern peatlands influence the earth's radiative budget:
18 Sustained methane emission versus sustained carbon sequestration. *JGR-Biogeosciences*, **111**, G01008,
19 doi:01010.01029/02005JG000091.
- 20 Frolking, S., N. T. Roulet, T. R. Moore, P. M. Lafleur, J. L. Bubier, and P. M. Crill, 2002: Modeling seasonal to
21 annual carbon balance of Mer Bleue Bog, Ontario, Canada. *Global Biogeochemical Cycles*, **16**,
22 10.1029.2001GB00147, 02002.
- 23 Gauci, V., E. Matthews, N. Dise, B. Walter, D. Koch, G. Granberg, and M. Vile, 2004: Sulfur pollution suppression
24 of the wetland methane source in the 20th and 21st centuries. *Proceeding of the National Academy of Sciences,
25 USA*, **101**, 12583–12587.
- 26 Gedney, N., P. M. Cox, and C. Huntingford, 2004: Climate feedbacks from methane emissions. *Geophysical
27 Research Letters*, **31**, L20503, doi:20510.21029/22004GL020919.
- 28 Gorham, E., 1991: Northern peatlands: Role in the carbon cycle and probable responses to climatic warming.
29 *Ecological Applications*, **1**, 182–195.
- 30 Hines, M. E., and K. N. Duddleston, 2001: Carbon flow to acetate and C₁ compounds in northern wetlands.
31 *Geophysical Research Letters*, **28**, 4251–4254.
- 32 Johnston, C. A., 1991: Sediment and nutrient retention by freshwater wetlands: effects on surface water quality.
33 *Critical Reviews in Environmental Control*, **21**, 491–565.
- 34 Joosten, H., and D. Clarke, 2002: Wise Use of Mires and Peatlands - Background Principles including a Framework
35 for Decision-Making. Saarijärvi, Finland: International Mire Conservation Group and International Peat
36 Society.

- 1 Keller, J. K., S. D. Bridgham, C. T. Chapin, and C. M. Iversen, 2005: Limited effects of six years of fertilization on
2 carbon mineralization dynamics in a Minnesota fen. *Soil Biology and Biochemistry*, **37**, 1197–1204.
- 3 Lynch-Stewart, P., I. Kessel-Taylor, and C. Rubec, 1999: Wetlands and Government: Policy and Legislation for
4 Wetland Conservation in Canada, No. 1999-1: North American Wetlands Conservation Council (Canada).
- 5 Maltby, E., and P. Immirzi, 1993: Carbon dynamics in peatlands and other wetland soils, regional and global
6 perspectives. *Chemosphere*, **27**, 999–1023.
- 7 Matthews, E., and I. Fung, 1987: Methane emission from natural wetlands: Global distribution, area, and
8 environmental characteristics of sources. *Global Biogeochemical Cycles*, **1**, 61–86.
- 9 Megonigal, J. P., and W. H. Schlesinger, 1997: Enhanced CH₄ emissions from a wetland soil exposed to elevated
10 CO₂. *Biogeochemistry*, **37**, 77–88.
- 11 Mitra, S., R. Wassmann, and P. L. G. Vlek, 2005: An appraisal of global wetland area and its organic carbon stock.
12 *Current Science*, **88**, 25–35.
- 13 Moore, D. R. J., and P. A. Keddy, 1989: The relationship between species richness and standing crop in wetlands:
14 the importance of scale. *Vegetatio*, **79**, 99–106.
- 15 Moore, T. R., and N. T. Roulet, 1995: Methane emissions from Canadian peatlands. In: *Soils and Global Change*
16 (R. Lal, J. Kimble, E. Levine, and B. A. Stewart, eds.), 153–164. Boca Raton, FL: Lewis Publishers.
- 17 Moore, T. R., N. T. Roulet, and J. M. Waddington, 1998: Uncertainty in predicting the effect of climatic change on
18 the carbon cycling of Canadian peatlands. *Climatic Change*, **40**, 229–245.
- 19 Moser, M., C. Prentice, and S. Frazier, 1996, A global overview of wetland loss and degradation: Ramsar 6th
20 Meeting of the Conference of the Contracting Parties in Brisbane, Australia.
- 21 National Research Council, 1995: *Wetlands: Characteristics and Boundaries*. Washington, D.C.: National Academy
22 Press.
- 23 National Research Council, 2001: *Compensating for wetland losses under the clean water act*. Washington, D.C.:
24 National Academy Press.
- 25 National Wetlands Working Group, 1997: *The Canadian Wetland Classification System*. Waterloo, Ontario,
26 Canada: Wetlands Research Centre, University of Waterloo.
- 27 OECD, 1996: *Guidelines for aid agencies for improved conservation and sustainable use of tropical and subtropical*
28 *wetlands*. Paris: Organization for Economic Co-operation and Development.
- 29 Ramaswamy, V., O. Boucher, J. Haigh, D. Hauglustaine, J. Haywood, G. Myhre, T. Nakajima, G. Y. Shi, and
30 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*
31 *Change* (J. T. Houghton, Y. Ding, D. J. Griggs, M. Noguer, P. J. van der Linden, X. Dai, K. Maskell, and C. A.
32 Johnson, eds.), 349–416. Cambridge: Cambridge University Press.
- 33
34 Roulet, N. T., 2000: Peatlands, carbon storage, greenhouse gases, and the Kyoto Protocol: prospects and
35 significance for Canada. *Wetlands*, **20**, 605–615.
- 36 Rubec, C., 1996: The status of peatland resources in Canada. In: *Global Peat Resources*. E. Lappalainen, ed., 243–
37 252. Jyskä, Finland: International Peat Society and Geological Survey of Finland.

- 1 Stallard, R. F., 1998: Terrestrial sedimentation and the carbon cycle: Coupling weathering and erosion to carbon
2 burial. *Global Biogeochemical Cycles*, **12**, 231–257.
- 3 Strack, M., J. M. Waddington, and E.-S. Tuittila, 2004: Effect of water table drawdown on northern peatland
4 methane dynamics: Implications for climate change. *Global Biogeochemical Cycles*, **18**, GB4003,
5 doi:4010.1029/2003GB002209, 002004.
- 6 Turetsky, M. R., B. D. Amiro, E. Bosch, and J. S. Bhatti, 2004: Historical burn area in western Canadian peatlands
7 and its relationship to fire weather indices. *Global Biogeochemical Cycles*, **18**, GB4014,
8 doi:1029/2004GB002222, 002004.
- 9 Updegraff, K., S. D. Bridgman, J. Pastor, P. Weishampel, and C. Harth, 2001: Response of CO₂ and CH₄ emissions
10 in peatlands to warming and water-table manipulation. *Ecological Applications*, **11**, 311–326.
- 11 Vann, C. D., and J. P. Megonigal, 2003: Elevated CO₂ and water depth regulation of methane emissions:
12 Comparison of woody and non-woody wetland plant species. *Biogeochemistry*, **63**, 117–134.
- 13 Vile, M. A., S. D. Bridgman, R. K. Wieder, and M. Novák, 2003: Atmospheric sulfur deposition alters pathways of
14 gaseous carbon production in peatlands. *Global Biogeochemical Cycles*, **17**, 1058–1064.
- 15 Wang, J. S., J. A. Logan, M. B. McElroy, B. N. Duncan, I. A. Megretskaia, and R. M. Yantosca, 2004: A 3-D model
16 analysis of the slowdown and interannual variability in the methane growth rate from 1988 to 1997. *Global
17 Biogeochemical Cycles*, **18**, GB3011, doi:101029/102003GB002180.
- 18 Watson, R. T., I. R. Noble, B. Bolin, N. H. Ravindranath, D. J. Verardo, and D. J. Dokken, 2000: IPCC Special
19 Report on Land Use, Land-Use Change and Forestry. Cambridge, UK: Cambridge University Press.
- 20 Whiting, G. J., and J. P. Chanton, 1993: Primary production control of methane emissions from wetlands. *Nature*,
21 **364**, 794–795.
- 22 Wylynko, D., ed., 1999: Prairie wetlands and carbon sequestration: assessing sinks under the Kyoto Protocol.
23 Winnipeg, Manitoba, Canada: Institute for Sustainable Development, Ducks Unlimited Canada, and Wetlands
24 International.
- 25 Zedler, J. B., and S. Kercher, in press: Wetland resources: status, trends, ecosystem services, and restorability.
26 *Annual Review of Environmental Resources*.
- 27 Zhuang, Q., J. M. Melillo, D. W. Kicklighter, R. G. Prin, A. D. McGuire, P. A. Steudler, B. S. Felzer, and S. Hu,
28 2004: Methane fluxes between terrestrial ecosystems and the atmosphere at northern high latitudes during the
29 past century: A retrospective analysis with a process-based biogeochemistry model. *Global Biogeochemical
30 Cycles*, **18**, GB 3010, doi:3010.1029/2004GB002239.

1 **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
 2 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
 3 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	**

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^aEstuarine includes salt marsh, mangrove, and mudflat, except for Mexico and global for which no mudflat estimates were available.

^bIncludes soil C and plant C, but overall soil C is 98% of the total pool.

^cIncludes 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).

^dNo 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

1 peatland estimate of 132,000 km² (Table 13A-1) is 22% of the often cited value by Kivinen and Pakarinen
2 (1981) of 596,000 km².

3
4 **Table 13A-1. Current and historical area of wetlands in North America and the world (×10³ km²).**
5 Historical refers to approximately 1800, unless otherwise specified
6

7 Kivinen and Pakarinen also used NRCS soils data (Rieger *et al.*, 1979) for their peatland estimates, but
8 they defined a peatland as having a minimum organic layer thickness of 30 cm, whereas the current U.S.
9 and Canadian soil taxonomies require a 40-cm thickness. The original 1979 Alaska soil inventory has
10 been reclassified with current U.S. soil taxonomy (J. Moore, Alaska State Soil Scientist, personal comm.).
11 Using the reclassified soil inventory, Alaska has 417,000 km² of wetlands with a histic modifier that are
12 not Histosols or Histels, indicating significant carbon accumulation in the surface horizons of FWMS
13 wetlands. Thus, we conclude that Kivinen and Pakarinen's Alaska peatland area estimate is higher
14 because many Alaskan wetlands have a thin organic horizon that is not deep enough to qualify as a
15 peatland under current soil taxonomy. Our smaller peatland area significantly lowers our estimate of
16 carbon pools and fluxes in Alaskan peatlands compared to earlier studies (see *Carbon Pools* below).

17 A regular national inventory of Canada's wetlands has not been undertaken, although wetland area
18 has been mapped by ecoregion (National Wetlands Working Group, 1988). Extensive recent effort has
19 gone into mapping Canadian peatlands (Tarnocai, 1998; Tarnocai *et al.*, 2005). We calculated mineral-
20 soil wetlands as the difference between total wetland area and peatland area in National Wetland Working
21 Group (1988). Historical FWMS wetland area was obtained from Rubec (1996). There are no reliable
22 country-wide estimates of mud flat area for Canada, but a highly uncertain extrapolation from a limited
23 number of regional estimates was possible.

24 No national wetland inventories have been done for Mexico. Current freshwater wetland estimates for
25 Mexico were taken from Davidson *et al.* (1999), who used inventories of discrete wetland regions
26 performed by a variety of organizations. Thus, freshwater wetland area estimates for Mexico are highly
27 unreliable and are possibly a large underestimate. For salt marshes and mangroves area in Mexico, we
28 used the estimates compiled by Mendelsohn and McKee (2000), which are similar to estimates reported
29 in Davidson *et al.* (1999) and Spalding *et al.* (1997). There are no reliable estimates of mud flat area for
30 Mexico.

31 **CARBON POOLS**

32 **Freshwater Mineral-Soil (Gleysol) Carbon Pools**

34 Gleysol is a soil classification used by the Food and Agriculture Organization (FAO) and many
35 countries that denotes mineral soils formed under waterlogged conditions (FAO-UNESCO, 1974).

1 Tarnocai (1998) reported a soil carbon density of 200 Mg C ha⁻¹ for Canadian Gleysols but did not
2 indicate to what depth this extended. Batjes (1996) determined soil carbon content globally from the *Soil*
3 *Map of the World* (FAO, 1991) and a large database of soil pedons. He gave a very similar average value
4 for soil carbon density of 199 Mg C ha⁻¹ (CV² = 212%, n = 14 pedons) for Gleysols of the world to 2-m
5 depth; to 1-m depth, he reported a soil carbon density of 131 Mg C ha⁻¹ (CV = 109%, n = 142 pedons).

6 Gleysols are not part of the U.S. soil taxonomy scheme, and mineral soils with attributes reflecting
7 waterlogged conditions are distributed among numerous soil groups. We used the NRCS State Soil
8 Geographic (STATSGO) soils database to query for soil carbon density in “wet” mineral soils of the
9 conterminous United States (all soils that had a surface texture described as peat, muck, or mucky peat, or
10 appeared on the 1993 list of hydric soils, which were not classified as Histosols) (N. Bliss, query of
11 NRCS STATSGO database, Dec. 2005). We found soil carbon densities of 162 Mg C ha⁻¹ for FWMS
12 wetlands in the conterminous United States and Mexico, which was used in this analysis.

13 However, some caution is necessary regarding the use of Gleysol or wet mineral soil carbon densities,
14 as apparently they include large areas of seasonally wet soils that are not considered wetlands by the more
15 conservative definition of wetlands used by the United States and many other countries and organizations.
16 For example, Eswaran *et al.* (1995) estimated that global wet mineral-soil area was 8,808,000 km², which
17 is substantially higher than the commonly accepted mineral-soil wetland area estimated by Matthews and
18 Fung (1987) of 2,289,000 km² and Aselmann and Crutzen (1989) of 2,341,000 km², even accounting for
19 substantial global wetland loss. In our query of the NRCS STATSGO database for the United States, we
20 found 1,258,000 km² of wet soils in the conterminous United States versus our estimate of 312,000 km²
21 of FWMS wetlands currently and 762,000 km² historically (Table 13A-1). We assume that including
22 these wet-but-not-wetland soils will decrease the estimated soil carbon density, but to what degree we do
23 not know. However, just considering the differences in area will give large differences in the soil carbon
24 pool. For example, Eswaran *et al.* (1995) estimated that wet mineral soils globally contain 108 Gt C to
25 1-m depth, whereas our estimate is 46 Gt C to 2-m depth (Table 13A-2).

26 For Alaska, many soil investigations have been conducted since the STATSGO soil data was coded.
27 We updated STATSGO by calculating soil carbon densities from data obtained from the NRCS on
28 479 pedons collected in Alaska, and then we used this data for both FWMS wetlands and peatlands. For
29 some of the Histosols, missing bulk densities were calculated using averages of measured bulk densities
30 for the closest matching class in the USDA Soil Taxonomy (NRCS, 1999). A matching procedure was
31 developed for relating sets of pedons to sets of STATSGO components. If there were multiple
32 components for each map unit in STATSGO, the percentage of the component was used to scale area and
33 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.

1 U.S. Soil Taxonomy: Orders, Suborders, Great Groups, and Subgroups. For example, the soil carbon for
2 all pedons having the same soil order were averaged, and the carbon content was applied to all of the soil
3 components of the same order (e.g., Histosol pedons are used to characterize Histosol components). At
4 the Order level, all components were matched with pedon data. At the suborder level, pedon data were not
5 available to match approximately 20,000 km² (compared to the nearly 1,500,000-km² area of soil in the
6 state), but the soil characteristics were more closely associated with the appropriate land areas than at the
7 Order level. At the Great Group and Subgroup levels, pedon data were unavailable for much larger areas,
8 even though the quality of the data when available became better. For this study, we used the Suborder-
9 level matching. The resulting soil carbon density for Alaskan FWMS wetlands was 469 Mg C ha⁻¹,
10 reflecting large areas of wetlands with a histic epipedon as noted above.

11 12 **Peatland Soil Carbon Pools**

13 The carbon pool of permafrost and non-permafrost peatlands in Canada had been previously
14 estimated by Tarnocai *et al.* (2005) based upon an extensive database. Good soil-carbon density data are
15 unavailable for peatlands in the United States, as the NRCS soil pedon information typically only goes to
16 a maximum depth of between 1.5 to 2 m, and many peatlands are deeper than this. Therefore, we used the
17 carbon density estimates of Tarnocai *et al.* (2005) of 1,441 Mg C ha⁻¹ for Histosols and 1,048 Mg C ha⁻¹
18 for Histels to estimate the soil carbon pool in Alaskan peatlands.

19 The importance of our using a smaller area of Alaskan peatlands becomes obvious here. Using the
20 larger area from Kivinen and Pakarinen (1981), Halsey *et al.* (2000) estimated that Alaskan peatlands
21 have a soil carbon pool of 71.5 Gt, almost 5-fold higher than our estimate. However, some of the
22 difference in soil carbon between the two estimates can be accounted for by the 26 Gt C that we
23 calculated resides in Alaskan FWMS wetlands (Table 13A-2).

24 25 **Table 13A-2. Soil carbon pools (Gt) and fluxes (Mt yr⁻¹) of wetlands in North America and the world.**

26 “Sequestration in current wetlands” refers to carbon sequestration in wetlands that currently exist;
27 “oxidation in former wetlands” refers to emissions from wetlands that have been converted to non-wetland
28 uses or conversion among wetland types due to human influence; “historical loss in sequestration capacity”
29 refers to the loss in the carbon sequestration function of wetlands that have been converted to non-wetland
30 uses; “change in flux from wetland conversions” is the sum of the two previous fluxes. Positive flux
31 numbers indicate a net flux into the atmosphere, whereas negative numbers indicate a net flux into the
32 ecosystem

33
34 The peatlands of the conterminous United States are different in texture, and probably depth, from those
35 in Canada and Alaska, so it is probably inappropriate to use the soil carbon densities for Canadian

1 peatlands for those in the conterminous United States. For example, we compared the relative percentage
2 of the Histosol suborders (excluding the small area of Folists, as they are predominantly upland soils) for
3 Canada (Tarnocai, 1998), Alaska (updated STATSGO data, J. Moore, personal comm.), and the
4 conterminous U.S. (NRCS, 1999). The relative percentage of Fibrists, Hemists, and Saprists, respectively,
5 in Canada are 37%, 62%, and 1%, in Alaska are 53%, 27%, and 20%, and in the conterminous United
6 States are 1%, 19%, and 80%. Using the STATSGO database (N. Bliss, query of NRCS STATSGO
7 database, December 2005), the average soil carbon density for Histosols in the conterminous United
8 States is 1,089 Mg C ha⁻¹, but this is an underestimate as many peatlands were not sampled to their
9 maximum depth. Armentano and Menges (1986) reported average carbon density of conterminous U.S.
10 peatlands to 1-m depth of 1,147 to 1,125 Mg C ha⁻¹. Malterer (1996) gave soil carbon densities of
11 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
12 C ha⁻¹ for Saprists, but it is unclear how he derived these estimates. Batjes (1996) and Eswaran *et al.*
13 (1995) gave average soil carbon densities to 1-m depth for global peatlands of 776 and 2,235 Mg C ha⁻¹,
14 respectively. We chose to use an average carbon density of 1,500 Mg C ha⁻¹, which is in the middle of the
15 reported range.

16

17 **Estuarine Soil Carbon Pools**

18 Tidal wetland soil carbon density was based on a country-specific analysis of data reported in an
19 extensive compilation by Chimura *et al.* (2003). There were more observations for the United States
20 (n = 75) than Canada (n = 34) or Mexico (n = 4), and consequently there were more observations of
21 marshes than mangroves. The Canadian salt marsh estimate was used for Alaska, and country-specific
22 marsh or mangrove estimates were used for mudflats. Although Chimura *et al.* (2003) reported some
23 significant correlations between soil carbon density and mean annual temperature, scatter plots suggest
24 the relationships are weak or driven by a few sites. Thus, we did not separate the data by region or latitude
25 and used mean values for scaling. Chimura *et al.* (2003) assumed a 50-cm-deep profile for the soil carbon
26 pool, which may be an underestimate.

27

28 **Plant Carbon Pools**

29 While extensive data on plant biomass in individual wetlands have been published, no systematic
30 inventory of wetland plant biomass has been undertaken in North America. Nationally, the forest carbon
31 biomass pool (including aboveground and belowground biomass) has been estimated to be 5.49 kg C m⁻²
32 (Birdsey, 1992), which we used for forested wetlands in the United States and Canada. This approach
33 assumes that wetland forests do not have substantially different biomass carbon densities from upland
34 forests. There is one regional assessment of forested wetlands in the southeastern United States, which

1 comprise approximately 35% of the total forested wetland area in the conterminous United States. We
2 utilized the southeastern U.S. regional inventory to evaluate this assumption; aboveground tree biomass
3 averaged $125.2 \text{ m}^3 \text{ ha}^{-1}$ for softwood stands and $116.1 \text{ m}^3 \text{ ha}^{-1}$ for hardwood stands. Using an average
4 wood density and carbon content, the carbon density for these forests would be 3.3 kg C m^{-2} for softwood
5 stands and 4.2 kg C m^{-2} for hardwood stands. However, these estimates do not include understory
6 vegetation, belowground biomass, or dead trees, which account for 49% of the total forest biomass
7 (Birdsey, 1992). Using that factor to make an adjustment for total forest biomass, the range would be 4.9
8 to 6.6 kg C m^{-2} for the softwood and hardwood stands, respectively. Accordingly, the assumption of using
9 5.49 kg C m^{-2} seems reasonable for a national-level estimate.

10 The area of forested wetlands in Canada came from Tarnocai *et al.* (2005), for Alaska from Hall *et al.*
11 (1994), and for the conterminous United States from Dahl (2000).

12 Since Tarnocai *et al.* (2005) divided Canadian peatland area into bog and fen, we used aboveground
13 biomass for each community type from Vitt *et al.* (2000), and assumed that 50% of biomass is
14 belowground. We used the average bog and fen plant biomass from Vitt *et al.* (2000) for Alaskan
15 peatlands. For other wetland areas, we used an average value of $2,000 \text{ g C m}^{-2}$ for non-forested wetland
16 biomass carbon density (Gorham, 1991).

17 Tidal marsh root and shoot biomass data were estimated from a compilation in Table 8-7 in Mitsch
18 and Gosselink, (1993). There was no clear latitudinal or regional pattern in biomass, so we used mean
19 values for each. Mangrove biomass has been shown to vary with latitude (Twilley *et al.*, 1992). Biomass
20 was estimated from an empirical equation for aboveground biomass as a function of latitude (Twilley *et*
21 *al.* 1992). We made a simple estimate using a single latitude that visually bisected the distribution of
22 mangroves either in the United States (26.9°) or Mexico (23.5°). Total biomass was estimated using a
23 root-to-shoot ratio of 0.82 and a carbon-mass-to-biomass ratio of 0.45, both from Twilley *et al.* (1992).

24 Plant biomass carbon data are presented in Table 13A-3.

25
26 **Table 13A-3. Plant carbon pools (Gt) and fluxes (Mt yr^{-1}) of wetlands in North America and the**
27 **world.** Positive flux numbers indicate a net flux into the atmosphere, whereas negative numbers indicate a
28 net flux into the ecosystem
29

30 CARBON FLUXES

31 Peatland Soil Carbon Accumulation Rates

32 Most studies report the long-term apparent rate of carbon accumulation (LORCA) in peatlands based
33 upon basal peat dates, but this assumes a linear accumulation rate through time. However, due to the slow
34 decay of the accumulated peat, the true rate of carbon accumulation will always be less than the LORCA

1 (Clymo *et al.*, 1998), so most reported rates are inherently biased upwards. Tolonen and Turunen (1996)
2 found that the true rate of peat accumulation was about 67% of the LORCA.

3 For estimates of soil carbon sequestration in conterminous U.S. peatlands, we used the data from 82
4 sites and 215 cores throughout eastern North America (Webb and Webb III, 1988). They reported a
5 median accumulation rate of 0.066 cm yr⁻¹ (mean = 0.092, sd = 0.085). We converted this value into a
6 carbon accumulation rate of -1.2 Mg C ha⁻¹ yr⁻¹ by assuming 58% C (see NRCS Soil Survey Laboratory
7 Information Manual, available on-line at <http://soils.usda.gov/survey/nscd/lim/>), a bulk density of 0.59 g
8 cm⁻³, and an organic matter content of 55%. (Positive carbon fluxes indicate net fluxes to the atmosphere,
9 whereas negative carbon fluxes indicate net fluxes into an ecosystem.) The bulk density and organic
10 matter content were the average from all Histosol soil map units greater than 202.5 ha (n = 5,483) in the
11 conterminous United States from the National Soil Information System (NASIS) data base provided by S.
12 Campbell (USDA NRCS, Portland, OR). For comparison, Armentano and Menges (1986) used soil
13 carbon accumulation rates that ranged from -0.48 Mg C ha⁻¹ yr⁻¹ in northern conterminous U.S. peatlands
14 to -2.25 Mg C ha⁻¹ yr⁻¹ in Florida peatlands.

15 Peatlands accumulate lesser amounts of soil carbon at higher latitudes, with especially lower rates
16 occurring in permafrost peatlands (Ovenden, 1990, Robinson and Moore, 1999). The rates used in this
17 report reflect this gradient, going from -0.13 to -0.19 to -1.2 Mg C ha⁻¹ yr⁻¹ in permafrost peatlands, non-
18 permafrost Canadian and Alaskan peatlands, and peatlands in the conterminous United States and
19 Mexico, respectively (Table 13A-2).

21 **Freshwater Mineral-Soil Wetland Carbon Accumulation Rates**

22 Many studies have estimated sediment deposition rates in FWMS wetlands, with an average rate of
23 1,680 g m⁻² yr⁻¹ (range 0 to 7,840) in a review by Johnston (1991). Assuming 7.7% carbon for FWMS
24 wetlands (Batjes, 1996), this gives a substantial accumulation rate of -129 g C m⁻² yr⁻¹. Johnston (1991)
25 found many more studies that just reported vertical sediment accumulation rates, with an average of
26 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,
27 1996), this converts into an impressive accumulation rate of -733 g C m⁻² yr⁻¹. However, we believe that
28 these values cannot be used directly as estimates of carbon sequestration rates for of two reasons. First, it
29 is likely that researchers preferentially choose wetlands with high sedimentation rates to study this
30 process. Secondly, and more fundamentally, at a landscape scale a redistribution of sediments from
31 uplands to wetlands represents no net carbon sequestration if the decomposition rate of carbon is the same
32 in both environments. The carbon associated with sediments is likely relatively recalcitrant and often
33 physically protected from decomposers by association with mineral soils. Thus, despite the anaerobic
34 conditions in wetlands, decomposition rates in deposited sediments may not be substantially lower than in

1 the uplands from which those sediments were eroded. Because of this reasoning, we somewhat arbitrarily
2 reduced our calculated rates of carbon sequestration in FWMS wetlands by 75% to $-34 \text{ g C m}^{-2} \text{ yr}^{-1}$, which
3 still represents a substantial carbon sink.

4 Agriculture typically increases sedimentation rates by 10- to 100-fold, and 90% of sediments are
5 stored within the watershed, or about 3 Gt yr^{-1} in the United States (Meade *et al.*, 1990, as cited in
6 Stallard, 1998). Converting this to 1.5% C equates to -45 Mt C yr^{-1} , part of which will be stored in
7 wetlands and is well within our estimated storage rate in FWMS wetlands (Table 13A-2).

9 **Estuarine Carbon Accumulation Rates**

10 Carbon accumulation in tidal wetlands was assumed to be entirely in the soil pool. This should
11 provide a reasonable estimate because marshes are primarily herbaceous, and mangrove biomass should
12 be in steady state unless the site was converted to another use. An important difference between soil
13 carbon sequestration in tidal and non-tidal systems is that tidal sequestration occurs primarily through
14 burial driven by sea level rise. For this reason, carbon accumulation rates can be estimated well with data
15 on changes in soil surface elevation and carbon density. Rates of soil carbon accumulation were
16 calculated from Chimura *et al.* (2003) as described for the soil carbon pool (above). These estimates are
17 based on a variety of methods, such as ^{210}Pb dating and soil elevation tables, which integrate vertical soil
18 accumulation rates over periods of time ranging from 1–100 yr.

20 **Extractive Uses of Peat**

21 Use of peat for energy production is, and always has been, negligible in North America, as opposed to
22 other parts of the world (WEC, 2001). However, Canada produces a greater volume of horticultural and
23 agricultural peat than any other country in the world (WEC, 2001). Currently, 124 km^2 of Canadian
24 peatlands have been under extraction now or in the past (Cleary *et al.*, 2005). A life-cycle analysis by
25 these authors estimated that as of 1990 Canada emitted 0.9 Mt yr^{-1} of $\text{CO}_2\text{-C}$ equivalents through peat
26 extraction. The U.S. production of horticultural peat is about 19% of Canada's (Joosten and Clarke,
27 2002), which assuming a similar life-cycle as for Canada, suggests that the United States produces 0.2 Mt
28 of $\text{CO}_2\text{-C}$ equivalents through peat extraction.

30 **Methane Fluxes**

31 Moore *et al.* (1995) reported a range of methane fluxes from 0 to $130 \text{ g CH}_4 \text{ m}^{-2} \text{ yr}^{-1}$ from 120
32 peatland sites in Canada, with the majority $<10 \text{ g CH}_4 \text{ m}^{-2} \text{ yr}^{-1}$. They estimated a low average flux rate of
33 2 to $3 \text{ g CH}_4 \text{ m}^{-2} \text{ yr}^{-1}$, which equaled an emission of 2–3 $\text{Mt CH}_4 \text{ yr}^{-1}$ from Canadian peatlands. We used
34 an estimate of $2.5 \text{ g CH}_4 \text{ m}^{-2} \text{ yr}^{-1}$ for Canadian peatlands and Alaskan freshwater wetlands (Table 13A-4).

1
2 **Table 13A-4. Methane fluxes (Mt yr^{-1}) from wetlands in North America and the world.**

3
4 To our knowledge, the last synthesis of methane fluxes was done by Bartlett and Harriss (1993). We
5 supplemented their analysis with all other published field studies (using chamber or eddy covariance
6 techniques) we could find that reported annual or average daily methane fluxes in the conterminous
7 United States (Table 13A-5). We excluded a few studies that used cores or estimated diffusive fluxes.

8
9 **Table 13A-5. Methane fluxes measured in the conterminous United States.** The conversion factor is the
10 ratio of the daily average flux to the measured annual flux $\times 10^3$. The calculated annual flux was
11 determined based upon the average conversion factor for freshwater (FW) and saltwater wetlands (SW).
12 The used annual flux was the measured annual flux if that was available; otherwise, it was the calculated
13 annual flux.

14
15 In cases where multiple years from the same site were presented, we took the average of those years.
16 Similarly, when multiple sites of the same type were presented in the same paper, we took the average.
17 Studies were separated into freshwater and estuarine systems.

18 In cases where papers presented both an annual flux and a mean daily flux, we calculated a
19 conversion factor [annual flux/(average daily flux $\times 10^3$)] to quantify the relationship between those two
20 numbers (Table 13A-5). When we looked at all studies ($n = 30$), this conversion factor was 0.36,
21 suggesting that there is a 360-day emission season. There was surprisingly little variation in this ratio, and
22 it was similar in freshwater (0.36) and estuarine (0.34) wetlands. In contrast, previous syntheses used a
23 150-day emission season for temperate wetlands (Matthews and Fung, 1987, Bartlett and Harriss, 1993).
24 While substantial winter methane emissions have been found in some studies, it is likely that flux data
25 from most studies have a non-normal distribution with occasional periods of high flux rates that are better
26 captured with annual measurements.

27 Using the conversion factors for freshwater and estuarine wetlands, we estimated average annual
28 fluxes from the average daily fluxes. For freshwater wetlands, the calculated average annual flux rate was
29 $38.6 \text{ g CH}_4 \text{ m}^{-2} \text{ yr}^{-1}$ ($n = 74$), which is slightly larger than the average actual measured flux rate of
30 $32.1 \text{ g CH}_4 \text{ m}^{-2} \text{ yr}^{-1}$ ($n = 32$). For estuarine wetlands, the average calculated annual flux rate was
31 $9.8 \text{ g CH}_4 \text{ m}^{-2} \text{ yr}^{-1}$ ($n = 25$), which is smaller than the average measured flux rate of $16.9 \text{ g CH}_4 \text{ m}^{-2} \text{ yr}^{-1}$
32 ($n = 13$). However, if we remove one outlier, the average measured flux rate is $10.2 \text{ g CH}_4 \text{ m}^{-2} \text{ yr}^{-1}$.

33 Finally, we combined both approaches. In cases where a paper presented an annual value, we used
34 that number. In cases where only an average daily number was presented, we used that value corrected

1 with the appropriate conversion factor. For conterminous U.S. wetlands, FWMS Canadian wetlands, and
2 Mexican wetlands, we used an average flux of $36 \text{ g CH}_4 \text{ m}^{-2} \text{ yr}^{-1}$, and for estuarine wetlands, we used an
3 average flux of $10.3 \text{ g CH}_4 \text{ m}^{-2} \text{ yr}^{-1}$.

5 Plant Carbon Fluxes

6 We have limited our focus on plant carbon fluxes to those processes that would result in the
7 accumulation of plant carbon biomass on an interannual basis. Tree biomass carbon sequestration
8 averages $-140 \text{ g C m}^{-2} \text{ yr}^{-1}$ in U.S. forests across all forest types (Birdsey, 1992). Using the tree growth
9 estimates from the southeastern U.S. regional assessment of wetland forests (Brown *et al.*, 2001) yields an
10 even lower estimate of sequestration in aboveground tree biomass (approx. $-50.2 \text{ g C m}^{-2} \text{ yr}^{-1}$). We have
11 used this lower value to estimate that U.S. wetland forests currently sequester $-10.3 \text{ Mt C yr}^{-1}$.

12 We have assumed that the largely undisturbed forested wetlands of Alaska and Canada are at an
13 approximate steady state in terms of biomass, with no interannual plant carbon accumulation. It is likely
14 that plant carbon sequestration occurs largely as woody biomass, so we also assumed that non-forested
15 wetlands have no interannual plant carbon accumulation.

18 REFERENCES

- 19 Alford, D. P., R. D. Delaune, and C. W. Lindau, 1997: Methane flux from Mississippi River deltaic plain wetlands.
20 *Biogeochemistry*, **37**, 227-236.
- 21 Armentano, T. B., and E. S. Menges, 1986: Patterns of change in the carbon balance of organic soil- wetlands of the
22 temperate zone. *Journal of Ecology*, **74**, 755-774.
- 23 Aselmann, I., and P. J. Crutzen, 1989: Global distribution of natural freshwater wetlands and rice paddies, their net
24 primary productivity, seasonality and possible methane emissions. *Journal of Atmospheric Chemistry*, **8**, 307-
25 359.
- 26 Bartlett, K. B., and R. C. Harriss, 1993: Review and assessment of methane emissions from wetlands. *Chemosphere*,
27 **26**, 261-320.
- 28 Batjes, N. H., 1996: Total carbon and nitrogen in the soils of the world. *European Journal of Soil Science*, **47**, 151-
29 163.
- 30 Birdsey, R. A., 1992: Carbon storage and accumulation in United States forest ecosystems. General Technical Report
31 WO-59 Washington, DC: USDA Forest Service.
- 32 Bridgham, S. D., C.-L. Ping, J. L. Richardson, and K. Updegraff, 2000: Soils of northern peatlands: Histosols and
33 Gelisols. In: *Wetland Soils: Genesis, Hydrology, Landscapes, and Classification* (J. L. Richardson, and M. J.
34 Vepraskas, eds.), 343-370. Boca Raton, FL: CRC Press.
- 35 Brown, M. J., G. M. Smith, and J. McCollum, 2001: Wetland forest statistics for the south Atlantic states. RB-SRS-
36 062 Asheville, North Carolina: Southern Research Station, U.S. Forest Service.

- 1 Chmura, G. L., S. C. Anisfeld, D. R. Cahoon, and J. C. Lynch, 2003: Global carbon sequestration in tidal, saline
2 wetland soils. *Global Biogeochemical Cycles*, **17**, 1111.
- 3 Cleary, J., N. T. Roulet, and T. R. Moore, 2005: Greenhouse gas emissions from Canadian peat extraction, 1990-
4 2000: A life-cycle analysis. *Ambio*, **34**, 456-461.
- 5 Clymo, R. S., J. Turunen, and K. Tolonen, 1998: Carbon accumulation in peatland. *Oikos*, **81**, 368-388.
- 6 Cowardin, L. M., V. Carter, F. C. Golet, and E. T. LaRoe, 1979: Classification of wetlands and deepwater habitats
7 of the United States. FWS/OBS-79/31 Washington, DC: Fish and Wildlife Service, US Department of the
8 Interior.
- 9 Dahl, T. E., 1990: Wetland losses in the United States 1970's to 1980's. Washington, DC: Fish and Wildlife Service,
10 U.S. Department of the Interior.
- 11 Dahl, T. E., 2000: Status and Trends of Wetlands in the Conterminous United States, 1986 to 1997. Washington, DC:
12 Fish and Wildlife Service, U.S. Department of the Interior.
- 13 Dahl, T. E., and C. E. Johnson, 1991: Status and Trends of Wetlands in the Conterminous United States, Mid-1970's
14 to Mid-1980's. Washington, D.C.: U.S. Department of the Interior, Fish and Wildlife Service.
- 15 Davidson, I., R. Vanderkam, and M. Padilla, 1999: Review of wetland inventory information in North
16 America. Canberra, Australia.
- 17 Ehhalt, D., M. Prather, F. Dentener, E. Dlugokencky, E. Holland, I. Isaksen, J. Katima, V. Kirchhoff, P. Matson, P.
18 Midgley, and M. Wang, 2001: "Atmospheric chemistry and greenhouse gases." In *Climate Change 2001: The
19 Scientific Basis*. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental
20 Panel on Climate Change (J. T. Houghton, Y. Ding, D. J. Griggs, M. Noguer, P. J. van der Linden, X. Dai, K.
21 Maskell, and C. A. Johnson, eds.), 239-287. Cambridge: Cambridge University Press.
- 22 Eswaran, H., E. Van Den Berg, and J. Kimble, 1995: Global soil carbon resources. In: *Soils and Global Change* (R.
23 Lal, J. Kimble, E. Levine, and B. A. Stewart, eds.), 27-43. Boca Raton, Florida: Lewis Publishers.
- 24 FAO, 1991: The Digitized Soil Map of the World. World Soil Resource Report, 64 Rome: Food and Agriculture
25 Organization.
- 26 FAO-UNESCO, 1974: *Soil Map of the World (1:5,000,000)*. Paris: UNESCO.
- 27 Frayer, W. E., T. J. Monahan, D. C. Bowden, and F. A. Graybill, 1983: Status and Trends of Wetlands and
28 Deepwater Habitats in the Conterminous United States, 1950s to 1970s. Fort Collins, Colorado: Dept. of Forest
29 and Wood Sciences, Colorado State University.
- 30 Gorham, E., 1991: Northern peatlands: Role in the carbon cycle and probable responses to climatic warming.
31 *Ecological Applications*, **1**, 182-195.
- 32 Group, N. W. W., 1988: Wetlands of Canada: Sustainable Development Branch, Environment Canada, Ontario, and
33 Polyscience Publications, Montreal, Quebec.
- 34 Hall, J. V., W. E. Frayer, and B. O. Wilen, 1994: Status of Alaska Wetlands. Anchorage, Alaska: U.S. Fish and
35 Wildlife Service.
- 36 Halsey, L. A., D. H. Vitt, and L. D. Gignac, 2000: *Sphagnum*-dominated peatlands in North America since the last
37 glacial maximum: their occurrence and extent. *The Bryologist*, **103**, 334-352.

- 1 Hanson, A. R., and L. Calkins, 1996: Wetlands of the Maritime Provinces: Revised Documentation for the Wetlands
2 Inventory. Technical Report No. 267 Sackville, New Brunswick: Canadian Wildlife Service, Atlantic Region.
- 3 Johnston, C. A., 1991: Sediment and nutrient retention by freshwater wetlands: effects on surface water quality.
4 *Critical Reviews in Environmental Control*, **21**, 491-565.
- 5 Joosten, H., and D. Clarke, 2002: Wise Use of Mires and Peatlands - Background Principles including a Framework
6 for Decision-Making. Saarijärvi, Finland: International Mire Conservation Group and International Peat
7 Society.
- 8 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.
9 Aravena, K. J. Scott, B. Dyck, R. Harris, B. Warner, and G. Edwards, 1997: Increase in fluxes of greenhouse
10 gases and methyl mercury following flooding of an experimental reservoir. *Environmental Science &*
11 *Technology*, **31**, 1334-1344.
- 12 Kivinen, E., and P. Pakarinen, 1981: Geographical distribution of peat resources and major peatland complex types
13 in the world. *Annales Academiae Scientiarum Fennicae, Series A. III.* **132**, 1-28.
- 14 Lappalainen, E., 1996: General review on world peatland and peat resources. In: *Global Peat Resources* (E.
15 Lappalainen, ed.), 53-56. Jyskä, Finland: International Peat Society and Geological Survey of Finland.
- 16 Maltby, E., and P. Immirzi, 1993: Carbon dynamics in peatlands and other wetland soils, regional and global
17 perspectives. *Chemosphere*, **27**, 999-1023.
- 18 Malterer, T. J., 1996: Peat resources of the United States. In: *Global Peat Resources* (E. Lappalainen, ed.), 253-260.
19 Jyska, Finland.
- 20 Matthews, E., and I. Fung, 1987: Methane emission from natural wetlands: Global distribution, area, and
21 environmental characteristics of sources. *Global Biogeochemical Cycles*, **1**, 61-86.
- 22 Meade, R. H., T. R. Yuzyk, and T. J. Day, 1990: Movement and storage of sediments in rivers of the United States
23 and Canada. In: *Surface Water Hydrology, Geol. of N. Am., 0-1* (M. G. Wolman, and H. C. Riggs, eds.), 255-
24 280. Boulder, CO: Geological Society of American.
- 25 Mendelsohn, I. A., and K. L. McKee, 2000: Saltmarshes and mangroves. In: *North American Terrestrial Vegetation*
26 (M. G. Barbour, and W. D. Billings, eds.), 501-536. Cambridge, UK: Cambridge University Press.
- 27 Mitsch, W. J., and J. G. Gosselink, 1993: Wetlands New York: Van Nostrand Reinhold.
- 28 Moore, T. R., and N. T. Roulet, 1995: Methane emissions from Canadian peatlands. In: *Soils and Global Change* (R.
29 Lal, J. Kimble, E. Levine, and B. A. Stewart, eds.), 153-164. Boca Raton, FL: Lewis Publishers.
- 30 Moore, T. R., N. T. Roulet, and J. M. Waddington, 1998: Uncertainty in predicting the effect of climatic change on
31 the carbon cycling of Canadian peatlands. *Climatic Change*, **40**, 229-245.
- 32 National Wetlands Working Group, 1988: *Wetlands of Canada*, Sustainable Development Branch, Environment
33 Canada, Ontario, and Polyscience Publications, Montreal, Quebec.
- 34 NRCS, 1999: Soil Taxonomy: A Basic System of Soil Classification for Making and Interpreting Soil Surveys.
35 Washington, DC: Natural Resources Conservation Service, U.S. Department of Agriculture.
- 36 Ovenden, L., 1990: Peat accumulation in northern wetlands. *Quaternary Research*, **33**, 377-386.

- 1 Rieger, S., D. B. Schoepfoster, and C. E. Furbush, 1979: Exploratory Soil Survey of Alaska. Anchorage, Alaska:
2 USDA Soil Conservation Service.
- 3 Robinson, S. D., and T. R. Moore, 1999: Carbon and peat accumulation over the past 1200 years in a landscape with
4 discontinuous permafrost, northwestern Canada. *Global Biogeochemical Cycles*, **13**, 591-602.
- 5 Rubec, C., 1996: The status of peatland resources in Canada. In: *Global Peat Resources* (E. Lappalainen, ed., 243-
6 252. Jyskä, Finland: International Peat Society and Geological Survey of Finland.
- 7 Spalding, M., F. Blasco, and C. Field, eds., 1997: World Mangrove Atlas. Okinawa, Japan: The International
8 Society for Mangrove Ecosystems.
- 9 Spiers, A. G., 1999: Review of international/continental wetland resources. In: *Global Review of Wetland Resources*
10 *and Priorities* (C. M. Finlayson, and A. G. Spiers, eds.).
- 11 Stallard, R. F., 1998: Terrestrial sedimentation and the carbon cycle: Coupling weathering and erosion to carbon
12 burial. *Global Biogeochemical Cycles*, **12**, 231-257.
- 13 Tarnocai, C., 1998: The amount of organic carbon in various soil orders and ecological provinces in Canada. In: *Soil*
14 *Processes and the Carbon Cycle* (R. Lal, J. M. Kimble, R. F. Follett, and B. A. Stewart, eds.), 81-92. Boca
15 Raton, Florida: CRC Press.
- 16 Tarnocai, C., I. M. Kettles, and B. Lacelle, 2005: Peatlands of Canada. Ottawa.Ottawa, Canada: Agriculture and
17 Agri-Food Canada, Research Branch.
- 18 Tolonen, K., and J. Turunen, 1996: Accumulation rates of carbon in mires in Finland and implications for climactic
19 change. *Holocene*, **6**, 171-178.
- 20 Trumbore, S. E., and J. W. Harden, 1997: Accumulation and turnover of carbon in organic and mineral soils of the
21 BOREAS northern study area. *Journal of Geophysical Research*, **102**, 28, 817-828, 830.
- 22 Turetsky, M. R., R. K. Wieder, L. A. Halsey, and D. Vitt, 2002: Current distribution and diminishing peatland
23 carbon sink. *Geophysical Research Letters*, **29**, 10.1029/2001GL014000, 012002.
- 24 Turunen, J., N. T. Roulet, and T. R. Moore, 2004: Nitrogen deposition and increased carbon accumulation in
25 ombrotrophic peatlands in eastern Canada. *Global Biogeochemical Cycles*, **18**, GB3002,
26 doi:3010.1029/2003GB002154.
- 27 Twilley, R. R., R. H. Chen, and T. Hargis, 1992: Carbon sinks in mangroves and their implications to carbon budget
28 of tropical coastal ecosystems. *Water, Air and Soil Pollution*, **64**, 265-288.
- 29 Vitt, D. H., L. A. Halsey, I. E. Bauer, and C. Campbell, 2000: Spatial and temporal trends in carbon storage of
30 peatlands of continental western Canada through the Holocene. *Canadian Journal of Earth Sciences*, **37**, 683-
31 693.
- 32 Vitt, D. H., L. A. Halsey, and S. C. Zoltai, 1994: The bog landforms of continental western Canada in relation to
33 climate and permafrost patterns. *Arctic and Alpine Research*, **26**, 1-13.
- 34 Webb, R. S., and T. Webb III, 1988: Rates of sediment accumulation in pollen cores from small lakes and mires of
35 eastern North America. *Quaternary Research*, **30**, 284-297.
- 36 WEC, 2001: Survey of Energy Resources. [http://www.worldenergy.org/wec-](http://www.worldenergy.org/wec-geis/publications/reports/ser/peat/peat.asp)
37 [geis/publications/reports/ser/peat/peat.asp](http://www.worldenergy.org/wec-geis/publications/reports/ser/peat/peat.asp)

- 1 Werner, C., K. Davis, P. Bakwin, C. Yi, D. Hurst, and L. Lock, 2003: Regional-scale measurements of CH₄
2 exchange from a tall tower over a mixed temperate/boreal lowland and wetland forest. *Global Change Biology*,
3 **9**, 1251-1261.
- 4 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.
5 Schmidt, 1999: Landscape patterns of CH₄ fluxes in an alpine tundra ecosystem. *Biogeochemistry*, **45**, 243-264.
- 6 Wickland, K. P., R. G. Striegl, S. K. Schmidt, and M. A. Mast, 1999: Methane flux in subalpine wetland and
7 unsaturated soils in the southern Rocky Mountains. *Global Biogeochemical Cycles*, **13**, 101-113.
- 8 Wilson, J. O., P. M. Crill, K. B. Bartlett, D. I. Sebacher, R. C. Harriss, and R. L. Sass, 1989: Seasonal variation of
9 methane emissions from a temperate swamp. *Biogeochemistry*, **8**, 55-71.
- 10 Yavitt, J. B., 1997: Methane and carbon dioxide dynamics in *Typha latifolia* (L.) wetlands in central New York
11 state. *Wetlands*, **17**, 394-406.
- 12 Yavitt, J. B., G. E. Lang, and A. J. Sexstone, 1990: Methane fluxes in wetland and forest soils, beaver ponds, and
13 low-order streams of a temperate forest ecosystem. *Journal of Geophysical Research*, **95**, 22463-22474.
- 14 Yavitt, J. B., R. K. Wieder, and G. E. Lang, 1993: CO₂ and CH₄ dynamics of a *Sphagnum*-dominated peatland in
15 West Virginia. *Global Biogeochemical Cycles*, **7**, 259-274.

1 **Table 13A-1. Current and historical area of wetlands in North America and the world ($\times 10^3$ km²).** Historical refers to approximately 1800, unless otherwise
 2 specified.

	Permafrost peatlands	Non-permafrost peatlands	Mineral-soil freshwater	Salt marsh	Mangrove	Mudflat	Total
<u>Canada</u>							
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
<u>Alaska</u>							
Current	89 ⁱ	43 ⁱ	556 ^j	1.4 ^c	0	7 ^k	696
Historical	89	43	556	1.4	0	9b	698
<u>Conterminous United States</u>							
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
<u>Mexico</u>							
Current	0	10 ^p	21 ^q	0	5 ^c	ND ^r	36
Historical	0		45 ^p	0	7 ^h	ND	52
<u>North America</u>							
Current	511	861	1,047	20	8	15	2,461
Historical	513	894 ^s	1,706 ^s	23	11	19	3,166
<u>Global</u>							
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

3
 4 ^aTarnocai *et al.* (2005).

5 ^bNational Wetlands Working Group (1988).

6 ^cMendelssohn and McKee (2000).

7 ^dEstimated from the area of Canadian salt marshes and the ratio of mudflat to salt marsh area reported by Hanson and Calkins (1996).

8 ^eAccounting for losses due to permafrost melting in western Canada (Vitt *et al.*, 1994). This is an underestimate, as similar, but undocumented, losses have
 9 probably also occurred in eastern Canada and Alaska.

10 ^f9000 km² lost to reservoir flooding (Rubec, 1996), 250 km² to forestry drainage (Rubec, 1996), 124 km² to peat harvesting for horticulture (Cleary *et al.*,
 11 2005), and 16 km² to oil sands mining (Turetsky *et al.*, 2002). See note e for permafrost melting estimate.

- 1 ^gRubec (1996).
- 2 ^hAssumed same loss rate as the conterminous United States since 1954 (Dahl, 2000).
- 3 ⁱHistorical area from NRCS soil inventory (Bridgham *et al.*, 2000), except Alaska inventory updated by N. Bliss from a February 2006 query of the
4 STATSGO database. Less than 1% wetland losses have occurred in Alaska (Dahl, 1990).
- 5 ^jTotal freshwater wetland area from Hall *et al.* (1994) minus peatland area.
- 6 ^kHall *et al.*, 1994.
- 7 ^LHistorical area from Bridgham *et al.* (2000) minus losses in Armentano and Menges (1986).
- 8 ^mOverall freshwater wetland area from Dahl (2000) minus peatland area.
- 9 ⁿDahl (2000).
- 10 ^oTotal historical wetland area from Dahl (1990) minus historical peatland area minus historical estuarine area.
- 11 ^pDavidson *et al.* (1999).
- 12 ^qSpiers (1999).
- 13 ^rND indicates that no data are available.
- 14 ^sAssuming that historical proportion of peatlands to total wetlands in Mexico was the same as today.
- 15 ^tBridgham *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
16 2,974,000–3,985,000 km² (Matthews and Fung, 1987; Aselmann and Crutzen, 1989; Maltby and Immerzi, 1993; Bridgham *et al.*, 2000; Joosten and Clarke,
17 2002).
- 18 ^uMatthews and Fung (1987); Aselmann and Crutzen (1989). For subsequent calculations, used the average of 2,315,000 km².
- 19 ^vChmura *et al.* (2003). Underestimated because no inventories were available for the continents Asia, South America and Australia which are mangrove-
20 dominated but also support salt marsh.
- 21 ^wSpalding *et al.* (1997).
- 22 ^xMaltby and Immerzi (1993). For subsequent calculations, used 4,000,000 km².
- 23 ^yApproximately 50% loss from Moser *et al.* (1996). For subsequent calculations, used an original global mineral-soil wetland area of 5,000,000 km².

1 **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
 2 sequestration in wetlands that currently exist; “oxidation in former wetlands” refers to emissions from wetlands that have been converted to non-wetland uses or
 3 conversion among wetland types due to human influence; “historical loss in sequestration capacity” refers to the loss in the carbon sequestration function of
 4 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
 5 indicate a net flux into the atmosphere, whereas negative numbers indicate a net flux into the ecosystem
 6

	Permafrost peatlands	Non-permafrost peatlands	Mineral-soil freshwater	Salt marsh	Mangrove	Mudflat	Total
<u>Canada</u>							
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	0.0 ^e	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
<u>Alaska</u>							
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
<u>Conterminous United States</u>							
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
<u>Mexico</u>							
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

- 1
- 2 *ND indicates that no data are available.
- 3 ^aTarnocai *et al.* (2005).
- 4 ^bTarnocai (1998).
- 5 ^cRates calculated from Chimura *et al.* (2003); areas from Mendelssohn and McKee (2000).
- 6 ^dAssumed the same carbon density and accumulation rates as the adjacent vegetated wetland ecosystem (mangrove data for Mexico and salt marsh data
- 7 elsewhere).
- 8 ^eSoil carbon accumulation rate of 0.13 Mg C ha⁻¹ yr⁻¹ (see Chapter 12 in this report).
- 9 ^fCarbon 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
- 10 (Ovenden, 1990, Maltby and Immirzi, 1993; Trumbore and Harden, 1997; Vitt *et al.*, 2000; Turunen *et al.*, 2004).

- 1 ^gPotential 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,
2 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
3 sediment due to erosion without a change in the sequestration rate on a landscape scale.
- 4 ^hSum 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
5 (Turetsky *et al.*, 2002).
- 6 ⁱAssumed that the oxidized soil C is lost over 50 yr.
- 7 ^jAssumed that conversion of tidal systems is caused by fill and results in burial and preservation of SOM define SOM rather than oxidation.
- 8 ^kSoil carbon densities of 1,441 Mg C ha⁻¹ for Histosols and 1,048 Mg C ha⁻¹ for Histels (Tarnocai *et al.*, 2005).
- 9 ^lSoil 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
10 and soil pedon information.
- 11 ^mAssumed soil carbon density of 1,500 Mg C ha⁻¹.
- 12 ⁿWebb and Webb (1988).
- 13 ^oEstimated loss rate as of early 1980s (Armentano and Menges,1986). Overall wetlands losses in the United States have declined dramatically since then
14 (Dahl, 2000) and probably even more so for Histosols, so this number may still be representative.
- 15 ^pUsing peat accumulation rate of 1.6 Mg C ha⁻¹ (range 1.0–2.25) (Maltby and Immirzi, 1993).
- 16 ^qGorham (1991), Maltby and Immirzi (1993), Eswaran *et al.* (1995), Batjes (1996), Lappalainen (1996), Joosten and Clarke (2002).
- 17 ^rSoil carbon density of 199 Mg C ha⁻¹ (Batjes, 1996).
- 18 ^sChmura *et al.* (2003).
- 19 ^tJoosten 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
20 –65 Mt C yr⁻¹ in peatlands.
- 21 ^uCurrent 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
22 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
23 northern peatlands, and from Maltby and Immirzi (1993) for tropical peatlands.
- 24 ^vAssumed that global rates approximate the North America rate because most salt marshes inventoried are in North America.
- 25 ^wAssumed 25% loss globally since the late 1800s.
- 26 ^x> sign indicates that this a minimal loss estimate.

1 **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
 2 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
<u>Canada</u>						
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
<u>Alaska</u>						
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
<u>Conterminous United States</u>						
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
<u>Mexico</u>						
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
<u>North America</u>						
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
<u>Global</u>						
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

3 *ND indicates that no data are available.

4 ^aBiomass for non-forested peatlands from Vitt *et al.* (2000), assuming 50% of biomass is belowground. Forest biomass density from
 5 Birdsey (1992) and forested area from Tarnocai *et al.* (2005) for Canada and from Hall *et al.* (1994) for Alaska.

6 ^bAssumed 2000 g C m⁻² in aboveground and belowground plant biomass (Gorham, 1991).

7 ^cBiomass data from Mitsch and Gosselink (1993).

8 ^dBiomass for non-forested wetlands from Gorham (1991). Forest biomass density from Birdsey (1992), and forested area from Dahl (2000).

9 ^e50 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).

10 ^fAssumed that global pools approximate those from North America because most salt marshes inventoried are in North America.

11 ^gTwilley *et al.* (1992).

1 **Table 13A-4. Methane fluxes (Mt yr⁻¹) from wetlands in North America and the world.**

	Permafrost peatlands	Non-permafrost peatlands	Mineral-soil freshwater	Salt marsh	Mangrove	Mudflat	Total
<u>Canada</u>							
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
<u>Alaska</u>							
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
<u>Conterminous United States</u>							
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
<u>Mexico</u>							
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
<u>North America</u>							
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
<u>Global</u>							
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

2 *ND indicates that no data are available.

3 ^aUsed CH₄ 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 CH₄ flux of
 4 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
 5 published CH₄ fluxes for the United States (see Table 13A-5).

6 ^bIncludes a 17-fold increase in CH₄ flux (Kelly *et al.*, 1997) in the 9000 km² of reservoirs that have been formed on peatlands (Rubec, 1996) and an estimated CH₄ flux of 15 g
 7 m⁻² yr⁻¹ (Moore *et al.*, 1998) from 2,630 km² of melted permafrost peatlands (Vitt *et al.*, 1994).

8 ^cAssumed trace gas fluxes from unvegetated estuarine wetlands (i.e., mudflats) was the same as adjacent wetlands.

9 ^dBartlett and Harriss (1993).

10 ^eAssumed that global rates approximate the North America rate because most salt marshes area is in North America.

11 ^fEhhalt *et al.* (2001).

12 ^gAssumed a conservative 25% loss since the late 1800s.

1 **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
 2 flux $\times 10^3$. The calculated annual flux was determined based upon the average conversion factor for freshwater (FW) and saltwater wetlands (SW). The used
 3 annual flux was the measured annual flux if that was available; otherwise, it was the calculated annual flux
 4

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 <i>et al.</i> (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 <i>et al.</i> (1999)
Tundra - Acomastylis Meadow	CO	C	FW	-0.5			-0.2	-0.2	West <i>et al.</i> (1999)
Tundra - Kobresia Meadow	CO	C	FW	-0.8			-0.3	-0.3	West <i>et al.</i> (1999)
Moist Grassy	CO	C	FW	6.1	1.9	0.32	2.2	1.9	Wickland <i>et al.</i> (1999)
Moist Mossy	CO	C	FW	1.5	0.5	0.33	0.5	0.5	Wickland <i>et al.</i> (1999)
Wetland	CO	C	FW		41.7			41.7	Wickland <i>et al.</i> (1999)
Hardwood Hammock	FL	C	FW	0.0			0.0	0.0	Bartlett <i>et al.</i> (1989)
Dwarf Cypress / Sawgrass	FL	C	FW	7.5			2.7	2.7	Bartlett <i>et al.</i> (1989)
Spikerush	FL	C	FW	29.4			10.7	10.7	Bartlett <i>et al.</i> (1989)
Sawgrass < 1m	FL	C	FW	38.8			14.1	14.1	Bartlett <i>et al.</i> (1989)
Sawgrass/Spkerush/Periphyton	FL	C	FW	45.1			16.4	16.4	Bartlett <i>et al.</i> (1989)
Swamp Forest	FL	C	FW	68.9			25.1	25.1	Bartlett <i>et al.</i> (1989)
Sawgrass > 1m	FL	C	FW	71.9			26.2	26.2	Bartlett <i>et al.</i> (1989)
Sawgrass	FL	C	FW	107.0			38.9	38.9	Burke <i>et al.</i> (1988)
Pond Open Water	FL	C	FW	624.0			227.1	227.1	Burke <i>et al.</i> (1988)
Everglades - Cladium	FL	C	FW	45.4			16.5	16.5	Chanton <i>et al.</i> (1993)
Everglades - Typha	FL	C	FW	142.9			52.0	52.0	Chanton <i>et al.</i> (1993)
Wet Prairie (Marl)	FL	C	FW	87.0			31.6	31.6	Happell <i>et al.</i> (1993)
Wet Prairie (Marl)	FL	C	FW	27.4			10.0	10.0	Happell <i>et al.</i> (1993)
Marsh (Marl)	FL	C	FW	30.0			10.9	10.9	Happell <i>et al.</i> (1993)
Marsh (Marl)	FL	C	FW	49.6			18.0	18.0	Happell <i>et al.</i> (1993)

Marsh (Peat)	FL	C	FW	45.4			16.5	16.5	Happell <i>et al.</i> (1993)
Marsh (Peat)	FL	C	FW	13.0			4.7	4.7	Happell <i>et al.</i> (1993)
Marsh (Peat)	FL	C	FW	163.6			59.6	59.6	Happell <i>et al.</i> (1993)
Marsh (Peat)	FL	C	FW	20.4			7.4	7.4	Happell <i>et al.</i> (1993)
Wet Prairie / Sawgrass	FL	C	FW	61.0			22.2	22.2	Harriss <i>et al.</i> (1988)
Wetland Forest	FL	C	FW	59.0			21.5	21.5	Harriss <i>et al.</i> (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 <i>et al.</i> (1997)
Freshwater Marsh	LA	C	FW	251.0			91.4	91.4	Alford <i>et al.</i> (1997)
Fresh	LA	C	FW	587.0	213.0	0.36	213.6	213.0	DeLaune <i>et al.</i> (1983)
Fresh	LA	C	FW	49.0	18.7	0.38	17.8	18.7	DeLaune <i>et al.</i> (1983)
Sphagnum Bog	MD	C	FW	-1.1			-0.4	-0.4	Yavitt <i>et al.</i> (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 <i>et al.</i> (1995)
Open Bogs	MN	C	FW		0.0			0.0	Bridgham <i>et al.</i> (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 <i>et al.</i> (1991)
Submergent	MN	C	FW		14.4			14.4	Naiman <i>et al.</i> (1991)
Deep Water	MN	C	FW		0.5			0.5	Naiman <i>et al.</i> (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 <i>et al.</i> (1995)
Banks Tidal	NC	C, E	FW	20.1			7.3	7.3	Kelley <i>et al.</i> (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 <i>et al.</i> (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 <i>et al.</i> (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 <i>et al.</i> (1992)
Maple/gum Forested Swamp	VA	C	FW		0.5			0.5	Harriss <i>et al.</i> (1982)
Emergent Tidal Freshwater Marsh	VA	C	FW		96.2			96.2	Neubauer <i>et al.</i> (2000)
Oak Swamp (Bank Site)	VA	C	FW	117.0	43.7	0.37	42.6	43.7	Wilson <i>et al.</i> (1989)
Emergent Macrophytes (Peltandra)	VA	C	FW	155.0			56.4	56.4	Wilson <i>et al.</i> (1989)
Emergent Macrophytes (Smartweed)	VA	C	FW	83.0			30.2	30.2	Wilson <i>et al.</i> (1989)
Ash Tree Swamp	VA	C	FW	152.0			55.3	55.3	Wilson <i>et al.</i> (1989)
Bog	WA	C	FW	73.0			26.6	26.6	Lansdown <i>et al.</i> (1992)
Lowland Shrub and Forested Wetland	WI	T	FW		12.4			12.4	Werner <i>et al.</i> (2003)
Sphagnum Eriophorum (Poor Fen)	WV	C	FW	6.6			2.4	2.4	Yavitt <i>et al.</i> (1990)
Sphagnum Shrub (Fen)	WV	C	FW	0.1			0.0	0.0	Yavitt <i>et al.</i> (1990)
Polytrichum Shrub (Fen)	WV	C	FW	-0.1			0.0	0.0	Yavitt <i>et al.</i> (1990)
Sphagnum Forest	WV	C	FW	9.6			3.5	3.5	Yavitt <i>et al.</i> (1990)
Sedge Meadow	WV	C	FW	1.5			0.5	0.5	Yavitt <i>et al.</i> (1990)
Beaver Pond	WV	C	FW	250.0			91.0	91.0	Yavitt <i>et al.</i> (1990)
Low Gradient Headwater Stream	WV	C	FW	300.0			109.2	109.2	Yavitt <i>et al.</i> (1990)
Sphagnum-Eriophorum	WV	C	FW	52.1	19.0	0.37	18.9	19.0	Yavitt <i>et al.</i> (1993)
Polytrichum	WV	C	FW	41.1	15.0	0.37	15.0	15.0	Yavitt <i>et al.</i> (1993)
Sphagnum-Shrub	WV	C	FW	4.4	1.6	0.37	1.6	1.6	Yavitt <i>et al.</i> (1993)
Salt Marsh	DE	C	SW	0.5			0.2	0.2	Bartlett <i>et al.</i> (1985)
Red Mangroves	FL	C	SW	4.2			1.4	1.4	Bartlett <i>et al.</i> (1989)
Dwarf Red Mangrove	FL	C	SW	81.9			27.9	27.9	Bartlett <i>et al.</i> (1989)
High Marsh	FL	C	SW	3.9			1.3	1.3	Bartlett <i>et al.</i> (1985)
Salt Marsh	FL	C	SW	0.6			0.2	0.2	Bartlett <i>et al.</i> (1985)
Salt Water Mangroves	FL	C	SW	4.0			1.4	1.4	Harriss <i>et al.</i> (1988)
Salt Marsh	GA	C	SW	13.4			4.6	4.6	Bartlett <i>et al.</i> (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 <i>et al.</i> (1997)
Salt Marsh	LA	C	SW	15.7	5.7	0.36	5.4	5.7	DeLaune <i>et al.</i> (1983)
Brackish	LA	C	SW	267.0	97.0		91.1	97.0	DeLaune <i>et al.</i> (1983)
Salt Marsh	LA	C	SW	4.8	1.7	0.35	1.6	1.7	DeLaune <i>et al.</i> (1983)
Brackish	LA	C	SW	17.0	6.4	0.38	5.8	6.4	DeLaune <i>et al.</i> (1983)
Cypress Swamp - Floodplain	SC	C	SW	1.5			0.5	0.5	Bartlett <i>et al.</i> (1985)
Salt Marsh	SC	C	SW	0.4			0.1	0.1	Bartlett <i>et al.</i> (1985)
Salt Marsh	VA	C	SW	3.0	1.3	0.43	1.0	1.3	Bartlett <i>et al.</i> (1985)
Salt Marsh	VA	C	SW	5.0	1.2	0.24	1.7	1.2	Bartlett <i>et al.</i> (1985)
Salt Meadow	VA	C	SW	2.0	0.4	0.22	0.7	0.4	Bartlett <i>et al.</i> (1985)
Salt Marsh	VA	C	SW	-0.8			-0.3	-0.3	Bartlett <i>et al.</i> (1985)
Salt Marsh	VA	C	SW	1.5			0.5	0.5	Bartlett <i>et al.</i> (1985)
Salt Meadow	VA	C	SW	-1.9			-0.6	-0.6	Bartlett <i>et al.</i> (1985)
Tidal Salt Marsh	VA	C	SW	16.0	5.6	0.35	5.5	5.6	Bartlett <i>et al.</i> (1987)
Tidal Brackish Marsh	VA	C	SW	64.6	22.4	0.35	22.0	22.4	Bartlett <i>et al.</i> (1987)
Tidal Brackish/Fresh Marsh	VA	C	SW	53.5	18.2	0.34	18.2	18.2	Bartlett <i>et al.</i> (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

1

2 ^aC = chamber, T = tower, eddy covariance, E = ebullition measured separately.

3 ^bOutlier that was removed from further analysis.

1

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Chapter 14. Human Settlements and the North American Carbon Cycle

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KEY FINDINGS

- Human settlements occupy almost 5 % of the North American land area.
- There is currently insufficient information to determine the complete carbon balance of human settlements in North America. Fossil fuel emissions, however, very likely dominate carbon fluxes from settlements.
- An estimated 410 to 1679 Mt C are currently stored in the urban tree component of North American settlements. The growth of urban trees in North America produces a sink of approximately 16 to 49 Mt C yr⁻¹, which is 1 to 3% of the fossil fuel emissions from North America in 2003.
- Estimates of historical trends of the net carbon balance of North American settlements are not available. Fossil fuel emissions have likely gone up with the growth of urban lands but the net balance of carbon loss during conversion of natural to urban or suburban land cover and subsequent sequestration in lawns and urban trees is highly uncertain.
- The density and development patterns of human settlements are drivers of fossil fuel emissions, especially in the residential and transportation sectors. Biological carbon gains and losses are influenced by type of predevelopment land cover, post-development urban design and landscaping choices, soil and landscape management practices, and the time since land conversion.
- Projections of future trends in the net carbon balance of North American settlements are not available. However, the projected expansion of urban areas in North America will strongly impact the future North American carbon cycle as human settlements affect (1) the direct emission of CO₂ 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.

- 1 • Research is needed to improve comprehensive carbon inventories for settled areas, to improve
2 understanding of how development processes relate to driving forces for the carbon cycle, and to
3 improve linkages between understandings of human and environmental systems in settled areas.
4
-

5
6 Activities in human settlements form the basis for much of North America's contribution to global
7 CO₂ emissions. Settlements such as cities, towns, and suburbs vary widely in density, form, and
8 distribution. Urban settlements, as they have been defined by the census bureaus of the United States,
9 Canada, and Mexico, make up approximately 75 to 80% of the population of the continent, and this
10 proportion is projected to continue to increase (United Nations, 2004). The density and forms of new
11 development will strongly impact the future trajectory of the North American carbon cycle as human
12 settlements affect the carbon cycle by (1) direct emission of CO₂ from fossil fuel combustion,
13 (2) alterations to plant and soil carbon cycles in conversion of wildlands to residential and urban land
14 cover, and (3) indirect effects of residential and urban land cover on energy use and ecosystem carbon
15 cycling.
16

17 **CARBON INVENTORIES OF HUMAN SETTLEMENTS**

18 Conversion of agricultural and wildlands to settlements of varying densities is occurring at a rapid
19 rate in North America, faster, in fact, than the rate of population growth. For example, according to U.S.
20 Census Bureau estimates, urban land in the coterminous United States increased by 20% in the 1990s
21 (Nowak *et al.*, 2005) while the population increased by 13%. Given these trends, it is important to
22 determine the carbon balance of different types of settlements and how future urban policy and planning
23 may impact the magnitude of CO₂ sources and sinks at regional, continental, and global scales. However,
24 unlike many other types of common land cover, complete carbon inventories including fossil fuel
25 emissions and biological sources and sinks of carbon have been conducted only rarely for settlements as a
26 whole. Assessing the carbon balance of settlements is challenging, as they are characterized by large CO₂
27 emissions from fuel combustion and decomposition of organic waste as well as transformations to
28 vegetation and soil that affect carbon sources and sinks.

29 Determining the extent of human settlements across North America also presents a challenge, as
30 definitions of "developed," "built-up," and "urban" land vary greatly, particularly among nations. The
31 U.S., Canadian, and Mexican census definitions are not consistent; in addition, several other classification
32 schemes for defining and mapping settlements have been developed, such as the U.S. Department of
33 Agriculture's National Resource Inventory categorization of developed land, which uses a variety of
34 methods based on satellite imagery. One method of classifying settled land cover that has been
35 consistently applied at a continental scale is the Global Rural-Urban Mapping Project conducted by a

1 consortium of institutions, including Columbia University and the World Bank (CIESIN *et al.*, 2004).
2 This estimate, which is based on nighttime lights satellite imagery, is 1,039,450 km², almost 5 % of the
3 total continental land area (Fig. 14-1).

4
5 **Fig. 14-1. North America urban extents.**

6
7 Currently, there is insufficient information to determine the complete current or historical carbon
8 balance of total continental land area. Fossil fuel emissions very likely dominate carbon fluxes from
9 settlements, just as settlement-related emissions likely dominate total fossil fuel consumption in North
10 America. However, specific estimates of the proportion of total fossil fuel emissions directly attributable
11 to settlements are difficult to make given current inventory methods, which are often conducted on a state
12 or province-wide basis. In addition, the biological component of the carbon balance of settlements is
13 highly uncertain, particularly with regard to the influence of urbanization on soil carbon pools and
14 biogenic greenhouse gas emissions.

15 For the urban tree component of the settlement carbon balance, carbon stocks and sequestration have
16 been estimated for urban land cover (as defined by the U.S. Census Bureau) in the coterminous United
17 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
18 Mt C yr⁻¹) (Nowak and Crane, 2002). These estimates encompass a great deal of regional variability and
19 contain some uncertainty about differences in carbon allocation between urban and natural trees, as urban
20 trees have been less studied. However, to a first approximation, these estimates can be used to infer a
21 probable range of urban tree carbon stocks and gross sequestration on a continental basis. Nowak and
22 Crane (2002) estimated that urban tree carbon storage in the Canadian border states (excluding semi-arid
23 Montana, Idaho, and North Dakota) ranged from 24 to 45 t C ha⁻¹, and carbon sequestration ranged from
24 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
25 (28,045 km² from the 1996 Canadian Census and 131,560 km² from CIESIN *et al.*, 2004), Canadian
26 urban forest carbon stocks are between 67 and 592 Mt while carbon sequestration rates are between 2.2
27 and 19.7 Mt C yr⁻¹. Similarly, for Mexico, Nowak and Crane (2002) estimated that urban carbon storage
28 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,
29 respectively, leading to estimates of 10 to 107 Mt C stored in urban trees in Mexico and 0.2 to 3.1 Mt C
30 yr⁻¹ sequestered. Estimates of historical trends are not available.

31 While complete national or continental-scale estimates of the carbon budget of settlements including
32 fossil fuels, vegetation, and soils are not available, several methods are available to assess the full carbon
33 balance of individual settlements and can be applied in the next several years toward constructing larger-
34 scale inventories. Atmospheric measurements can be used to determine the net losses of carbon from

1 settlements and urbanizing regions (Grimmond *et al.*, 2002; Grimmond *et al.*, 2004; Nemitz *et al.*, 2002;
2 Soegaard and Moller-Jensen, 2003). Specific sources of CO₂ can be determined from unique isotopic
3 signatures (Pataki *et al.*, 2003; Pataki *et al.*, 2006b) and from the relationship between CO₂ and carbon
4 monoxide (Lin *et al.*, 2004). Many of these techniques have been commonly applied to natural
5 ecosystems and may be easily adapted for settled regions. In addition, there have been several attempts to
6 quantify the “metabolism” of human settlements in terms of their inputs and outputs of energy, materials,
7 and wastes (Decker *et al.*, 2000) and the “footprint” of settlements in terms of the land area required to
8 supply their consumption of resources and to offset CO₂ emissions (Folke *et al.*, 1997). Often these
9 calculations include local flows and transformations of materials as well as upstream energy use and
10 carbon appropriation, such as remote electrical power generation and food production.

11 To conduct metabolic and footprint analyses of specific settlements, energy and fuel use statistics are
12 needed for individual municipalities, and these data are seldom made available at that scale.
13 Consequently, metabolic and footprint analyses of carbon flows and conversions associated with
14 metropolitan regions have been conducted for a relatively small number of cities. A metabolic analysis of
15 the Toronto metropolitan region showed per capita net CO₂ emissions of 14 t CO₂ yr⁻¹ (Sahely *et al.*,
16 2003), higher than analyses of other large metropolitan areas in developed countries (Newman, 1999;
17 Pataki *et al.*, 2006a; Warren-Rhodes and Koenig, 2001). In contrast, an analysis of Mexico City estimated
18 per capita CO₂ emissions of 3.4 t CO₂ yr⁻¹ (Romero Lankao *et al.*, 2004). Local emissions inventories can
19 provide useful supplements to national and global inventories in order to ensure that emissions reductions
20 policies are applied effectively and equitably (Easterling *et al.*, 2003).

21 Current projections for urban land development in North America highlight the importance of
22 improving carbon inventories of settlements and assessing patterns and impacts of future urban and rural
23 development. Projections for increases in the extent of developed land cover in the United States in the
24 next 25 years are as high as 79%, which would increase the proportion of developed land from 5.2% to
25 9.2% of total land cover (Alig *et al.*, 2004). The potential consequences of this increase for the carbon
26 cycle are significant in terms of CO₂ emissions from an expanded housing stock and transportation
27 network as well as from conversion of agricultural land, forest, rangeland, and other ecosystems to urban
28 land cover. Because the dynamics of carbon cycling in settled areas encompass a range of physical,
29 biological, social, and economic processes, studies of the potential impacts of future development on the
30 carbon cycle must be interdisciplinary. Large-scale research on what has been called the study “of cities
31 as ecosystems” (Pickett *et al.*, 2001) has begun only relatively recently, pioneered by interdisciplinary
32 studies such as the National Science Foundation’s Long-Term Ecological Research sites in the central
33 Arizona-Phoenix area and in Baltimore (Grimm *et al.*, 2000). Although there is not yet sufficient data to
34 construct a complete carbon inventory of settlements across North America, it is a feasible research goal

1 to do so in the next several years if additional studies in individual municipalities are conducted in a
2 variety of urbanizing regions.

3 4 **TRENDS AND DRIVERS**

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

12 13 **Fossil Fuel Emissions**

14 The density and patterns of development of human settlements (i.e., their “form”) are drivers of the
15 magnitude of the fossil fuel emissions component of the carbon cycle. The size and number of residences
16 and households influence CO₂ emissions from the residential sector, and the spatial distribution of
17 residences, commercial districts, and transportation networks is a key influence in the vehicular and
18 transportation sectors. Many of the attributes of urban form that influence the magnitude of fossil fuel
19 emissions are linked to historical patterns of economic development, which have differed in Canada, the
20 United States, and Mexico. The future trajectory of development and associated levels of affluence and
21 technological and social change will strongly influence key aspects of urban form such as residence size,
22 vehicle miles traveled, and investment in urban infrastructure, along with associated fossil fuel emissions.
23 Whereas emissions from the transportation and residential sectors are discussed in detail in Chapters 7
24 and 9, respectively, this chapter discusses specific aspects of the form of human settlements that affect the
25 current continental carbon balance and its possible future trajectories.

26 Household size in terms of the number of occupants per household has been declining in North
27 America (Table 14-1) while the average size of new residences has been increasing. For example, the
28 average size of new, single family homes in the United States increased from 139 m² (1500 ft²) to more
29 than 214 m² (2300 ft²) between 1970 and 2004 (NAHB, 2005). These trends have contributed to increases
30 in per capita CO₂ emissions from the residential sector as well as increases in the consumption of land for
31 residential and urban development (Alig *et al.*, 2003; Ironmonger *et al.*, 1995; Liu *et al.*, 2003; MacKellar
32 *et al.*, 1995). In addition, when considering total emissions from settlements, the trajectory of the
33 transportation and residential sectors may be linked. There have been a number of qualitative discussions
34 of the role of “urban sprawl” in influencing fossil fuel and pollutant emissions from cities (CEC, 2001;

1 Gonzalez, 2005), although definitions of urban sprawl vary (Ewing *et al.*, 2003). Quantitative linkages
2 between urban form and energy use have been attempted by comparing datasets for a variety of cities, but
3 the results have been difficult to interpret due to the large number of factors that may affect transportation
4 patterns and energy consumption (Anderson *et al.*, 1996). For example, in a seminal analysis of data from
5 a variety of cities, Kenworthy and Newman (1990) found a negative correlation between population
6 density and per capita energy use in the transportation sector. However, their data have been reanalyzed
7 and reinterpreted in a number of subsequent studies that have highlighted other important driving
8 variables, such as income levels, employment density, and transit choice (Gomez-Ibanez, 1991; Gordon
9 and Richardson, 1989; Mindali *et al.*, 2004).

10
11 **Table 14-1. Increases in number of households and the total population of the United States, Canada,**
12 **and Mexico between 1985 and 2000.** (United Nations, 2002; United Nations Habitat, 2003).

13
14 Quantifying the nature and extent of the linkage between development patterns of human settlements
15 and greenhouse gas emissions is critical from the perspective of evaluating the potential impacts of land
16 use policy. One way forward is to further the application of integrated land use and transportation models
17 that have been developed to analyze future patterns of urban development in a variety of cities (Agarwal
18 *et al.*, 2000; EPA, 2000; Hunt *et al.*, 2005). Only a handful have been applied to date for generating fossil
19 fuel emissions scenarios from individual metropolitan areas (Jaccard *et al.*, 1997; Pataki *et al.*, 2006a),
20 such that larger-scale national or continental projections for human settlements are not currently available.
21 However, there is potential to add a carbon cycle component to these models that would assess the
22 linkages between land use and land cover change, residential and commercial energy use and emissions,
23 emissions from the transportation sector, and net carbon gains and losses in biological sinks following
24 land conversion. A critical feature of these models is that they may be used to evaluate future scenarios
25 and the potential impacts of policies to influence land use patterns and transportation networks in
26 individual settlements and developing regions.

27 28 **Vegetation and Soils in Human Settlements**

29 Human settlements contain vegetation and soils that are often overlooked in national inventories, as
30 they fall outside common classification schemes. Nevertheless, patterns of development affect the carbon
31 balance of biological systems, both in the replacement of natural ecosystems with rural, residential, or
32 urban land cover and in processes within settlements that affect constructed and managed land cover. In
33 the United States, satellite data and ecosystem modeling for the mid-1990s suggested that urbanization

1 occurred largely on productive agricultural land and therefore caused a net loss of carbon fixed by
2 photosynthesis of 40 Mt C yr⁻¹ (Imhoff *et al.*, 2004).

3 Urban forests and vegetation sequester carbon directly as described under carbon inventories. In
4 addition, urban trees influence the carbon balance of municipalities indirectly through their effects on
5 energy use. Depending on their placement relative to buildings, trees may cause shading and windbreak
6 effects, as well as evaporative cooling due to transpiration (Akbari, 2002; Oke, 1989; Taha, 1997). These
7 effects have been estimated in a variety of studies, mostly involving model calculations that suggest that
8 urban trees generally result in net reductions in energy use (Akbari, 2002; Akbari and Konopacki, 2005;
9 Akbari *et al.*, 1997; Akbari and Taha, 1992; Huang *et al.*, 1987). Taking into account CO₂ emissions
10 resulting from tree maintenance and decomposition of removed trees, “avoided” emissions from energy
11 savings were responsible for approximately half of the total net reduction in CO₂ emissions from seven
12 municipal urban forests, with the remainder attributable to direct sequestration of CO₂ (McPherson *et al.*,
13 2005). Direct measurements of the components of urban energy balance that quantify the contribution of
14 vegetation are needed to validate these estimates.

15 Like natural ecosystems, soils in human settlements contain carbon, although rates of sequestration
16 are much more uncertain in urban soils than in natural soils. In general, soil carbon is generally lost
17 following disturbances associated with conversion from natural to urban or suburban land cover (Pouyat
18 *et al.*, 2002). Soil carbon pools may subsequently increase at varying rates, depending on the soil and land
19 cover type, local climate, and management intensity (Golubiewski, 2006; Pouyat *et al.*, 2002; Qian and
20 Follet, 2002). In ecosystems with low rates of carbon sequestration in native soil such as arid and
21 semiarid ecosystems, conversion to highly managed, settled land cover can result in higher rates of carbon
22 sequestration and storage than pre-settlement due to large inputs of water, fertilizer, and organic matter
23 (Golubiewski, 2006). Pouyat *et al.* (2006) used urban soil organic carbon measurements to estimate the
24 total above- and below-ground carbon storage, including soil carbon, in U.S. urban land cover to be 2,640
25 Mt (1,890 to 3,300 Mt). This range does not include the uncertainty in classifying urban land cover, but
26 applies the range of uncertainty in aboveground urban carbon stocks reported in Nowak and Crane (2002)
27 and the standard deviation of urban soil carbon densities reported in Pouyat *et al.* (2006). In addition,
28 irrigated and fertilized urban soils have been associated with higher emissions of CO₂ and the potent
29 greenhouse gas N₂O relative to natural soils, offsetting some potential gains of sequestering carbon in
30 urban soils (Kaye *et al.*, 2004; Kaye *et al.*, 2005; Koerner and Klopatek, 2002). Finally, full carbon
31 accounting that incorporates fossil fuel emissions associated with soil management (e.g., irrigation and
32 fertilizer production and transport) has not yet been conducted. In general, additional data on soil carbon
33 balance in human settlements are required to assess the potential for managing urban and residential soils
34 for carbon sequestration.

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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 CO₂ 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 health 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 CO₂ and air pollutants

1 in Mexico City, West *et al.* (2004) found that in practice, if electrical energy is primarily generated in
2 remote locations relative to the urban area, cost-effective energy efficiency programs may have a
3 relatively small effect on air quality. In that case, options for reducing greenhouse gas emissions would
4 have to be implemented primarily in the transportation sector to appreciably affect air quality.

6 RESEARCH NEEDS

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

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

24
25 In general, there has been a focus in carbon cycle science on measuring carbon stocks and fluxes in
26 natural ecosystems, and consequently highly managed and human-dominated systems such as settlements
27 have been underrepresented in many regional and national inventories. To assess the full carbon balance
28 of settlements ranging from rural developments to large cities, a wide range of measurement techniques
29 and scientific, economic, and social science disciplines are required to understand the dynamics of urban
30 expansion, transportation, economic development, and biological sources and sinks. An advantage to an
31 interdisciplinary focus on the study of human settlements from a carbon cycle perspective is that human
32 activities and biological impacts in and surrounding settled areas encompass many aspects of
33 perturbations to atmospheric CO₂, including a large proportion of national CO₂ emissions and changes in
34 carbon sinks resulting from land use change.

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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 CO₂ 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.

- 1 EPA (2000) Projecting land-use change: A summary of models for assessing the effects of community growth and
2 change on land-use patterns. U.S. Environmental Protection Agency, Washington, D.C. EPA/600/R-00/098.
- 3 Ewing R, Pendall R, Chen D (2003) Measuring sprawl and its transportation impacts. *Transportation Research*
4 *Record* 1831:175–183.
- 5 Folke C, Jansson A, Larsson J, Costanza R (1997) Ecosystem appropriation by cities. *Ambio* 26:167–172.
- 6 Golubiewski NE (2006) Urbanization transforms prairie carbon pools: Effects of landscaping in Colorado's Front
7 Range. *Ecological Applications* 16(2): 555-51.
- 8 Gomez-Ibanez JA (1991) A global view of automobile dependence: review of "Cities and automobile dependence:
9 An international sourcebook" by P.W.G. Newman and J.R. Kenworthy". *Journal of the American Planning*
10 *Association* 57:376–379.
- 11 Gonzalez GA (2005) Urban sprawl, global warming and the limits of ecological modernisation. *Environmental*
12 *Politics* 14:344–362.
- 13 Gordon P, Richardson HW (1989) Gasoline consumption and cities: A reply. *Journal of the American Planning*
14 *Association* 55:342–346.
- 15 Grimm NB, Grove JM, Pickett STA, Redman CL (2000) Integrated approaches to long-term studies of urban
16 ecological systems. *Bioscience* 50:571–584.
- 17 Grimmond CSB, King TS, Cropley FD, Nowak DJ, Souch C (2002) Local-scale fluxes of carbon dioxide in urban
18 environments: methodological challenges and results from Chicago. *Environmental Pollution* 116:S243–S254.
- 19 Grimmond CSB, Salmond JA, Oke TR, Offerle B, Lemonsu A (2004) Flux and turbulence measurements at a
20 densely built-up site in Marseille: heat, mass (water and carbon dioxide), and momentum. *Journal of*
21 *Geophysical Research–Atmospheres* 109:DOI: 10.1029/2004JD004936.
- 22 Huang YJ, Akbari H, Taha H, Rosenfeld H (1987) The potential of vegetation in reducing summer cooling loads in
23 residential buildings. *Journal of Climate and Applied Meteorology* 26:1103–1116.
- 24 Hunt JD, Kriger DS, Miller EJ (2005) Current operation urban land-use-transport modelling frameworks: A review.
25 *Transport Reviews* 25:329–376.
- 26 ICLEI (1993) Cities for climate protection: An international campaign to reduce urban emissions of greenhouse
27 gases. <http://www.iclei.org/index.php?id=1651>, last accessed 30 Mar 2006.
- 28 ICLEI (2000) Best practices for climate protection: A local government guide. ICLEI, Berkeley, CA.
- 29 Imhoff ML, Bounoua L, DeFries RS, Lawrence WT, Stutzer D, Compton JT, Ricketts T (2004) The consequences
30 of urban land transformations on net primary productivity in the United States. *Remote Sensing of the*
31 *Environment* 89:434–443.
- 32 Ironmonger DS, Aitken CK, Erbas B (1995) Economies of scale in energy use in adult-only households. *Energy*
33 *Economics* 17:301–310.
- 34 Jaccard M, Failing L, Berry T (1997) From equipment to infrastructure: community energy management and
35 greenhouse gas emission reduction. *Energy Policy* 25:1065–1074.
- 36 Kaye JP, Burke IC, Mosier AR, Guerschman JP (2004) Methane and nitrous oxide fluxes from urban soils to the
37 atmosphere. *Ecological Applications* 14:975–981.

- 1 Kaye JP, McCulley RL, Burke IC (2005) Carbon fluxes, nitrogen cycling, and soil microbial communities in
2 adjacent urban, native and agricultural ecosystems. *Global Change Biology* 11:575–587.
- 3 Kenworthy JR, Newman PWG (1990) Cities and transport energy: lessons from a global survey. *Ekistics* 34:258–
4 268.
- 5 Koerner B, Klopatek J (2002) Anthropogenic and natural CO₂ emission sources in an arid urban environment.
6 *Environmental Pollution* 116:S45–S51.
- 7 Kousky C, Schneider SH (2003) Global climate policy: will cities lead the way? *Climate Policy* 3:359–372.
- 8 Lin JC, Gerbig C, Wofsy SC, Andrews AE, Daube BC, Grainger BC, Stephens BB, Bakwin PS, Hollinger DY
9 (2004) Measuring fluxes of trace gases at regional scales by Lagrangian observations: Application to the CO₂
10 budget and rectification airborne (COBRA study). *Journal of Geophysical Research–Atmospheres*
11 109:DOI:10.1029/2004JD004754.
- 12 Liu J, Daily GC, Ehrlich PR, Luck GW (2003) Effects of household dynamics on resource consumption and
13 biodiversity. *Nature* 421:530–533.
- 14 MacKellar FL, Lutz W, Prinz C, Goujon A (1995) Population, households, and CO₂ emissions. *Population and*
15 *Development Review* 21:849–865.
- 16 McPherson EG, Simpson JR, Peper PF, Maco SE, Xiao Q (2005) Municipal forest benefits and costs in five U.S.
17 cities. *Journal of Forestry* In press.
- 18 Mindali O, Raveh A, Saloman I (2004) Urban density and energy consumption: A new look at old statistics.
19 *Transportation Research Record* 38A:143–162.
- 20 NAHB (2005) Housing facts, figures and trends. National Association of Home Builders, Washington, DC.
- 21 Nemitz E, Hargreaves K, McDonald AG, Dorsey JR, Fowler D (2002) Micrometeorological measurements of the
22 urban heat budget and CO₂ emissions on a city scale. *Environmental Science and Technology* 36:3139–3146.
- 23 Newman PWG (1999) Sustainability and cities: extending the metabolism model. *Landscape and Urban Planning*
24 44:219–226.
- 25 Nowak DJ, Crane DE (2002) Carbon storage and sequestration by urban trees in the USA. *Environmental Pollution*
26 116:381–389.
- 27 Nowak DJ, Walton JT, Dwyer JF, Kaya LG, Myeong S (2005) The increasing influence of urban environments on
28 U.S. forest management. *Journal of Forestry* In press.
- 29 Oke TR (1989) The micrometeorology of the urban forest. *Phil. Trans. R. Soc. Lond. B* 324:335–349.
- 30 Pataki DE, Alig RJ, Fung AS, Golubiewski NE, Kennedy CA, McPherson EG, Nowak DJ, Pouyat RV, Romero
31 Lankao P (2006a) Urban ecosystems and the North American carbon cycle. *Global Change Biology* In press.
- 32 Pataki DE, Bowling DR, Ehleringer JR (2003) The seasonal cycle of carbon dioxide and its isotopic composition in
33 an urban atmosphere: anthropogenic and biogenic effects. *Journal of Geophysical Research* 108:4735.
- 34 Pataki DE, Bowling DR, Ehleringer JR, Zobitz JM (2006b) High resolution monitoring of urban carbon dioxide
35 sources. *Geophysical Research Letters* 33, L03813, doi:10.1029/2005GL024822.

- 1 Pickett STA, Cadenasso ML, Grove JM, Nilon CH, Pouyat RV, Zipperer WC, Costanza R (2001) Urban ecological
2 systems: linking terrestrial ecological, physical, and socioeconomic components of metropolitan areas. *Annual*
3 *Review of Ecology and Systematics* 32:127–157.
- 4 Pouyat R, Groffman P, Yesilonis I, Hernandez L (2002) Soil carbon pools and fluxes in urban ecosystems.
5 *Environmental Pollution* 116:S107–S118.
- 6 Pouyat RV, Yesilonis I, Nowak DJ (2006) Carbon storage by urban soils in the USA. *Journal of Environmental*
7 *Quality* In press.
- 8 Qian Y, Follet RF (2002) Assessing soil carbon sequestration in turfgrass systems using long-term soil testing data.
9 *Agronomy Journal* 94:930–935.
- 10 Romero Lankao P, Lopez H, Rosas A, Gunther G, Correa Z (2004) Can cities reduce global warming? Urban
11 development and the carbon cycle in Latin America. IAI, UAM-X, IHDP, GCP, Mexico.
- 12 Sahely HR, Dudding S, Kennedy CA (2003) Estimating the urban metabolism of Canadian cities: Greater Toronto
13 Area case study. *Canadian Journal of Civil Engineering* 30:468–483.
- 14 Soegaard H, Moller-Jensen L (2003) Toward a spatial CO₂ budget of metropolitan region based on textural image
15 classification and flux measurements. *Remote Sensing of the Environment* 87:283–294.
- 16 Taha H (1997) Urban climates and heat islands: albedo, evapotranspiration, and anthropogenic heat. *Energy and*
17 *Buildings* 25:99–103.
- 18 United Nations (2002) *Demographic Yearbook*, <http://unstats.un.org/unsd/demographic/products/dyb/default.htm>.
- 19 United Nations (2004) *World urbanization prospects: The 2003 Revision*. U.N. Dept. of Economic and Social
20 Affairs, Population Division, New York. E.04.XIII.6.
- 21 United Nations Habitat (2003) *Global Observatory Database*, <http://www.unchs.org/programmes/guo>, last accessed
22 10 Nov 2005.
- 23 Warren-Rhodes K, Koenig A (2001) Ecosystem appropriation by Hong Kong and its implications for sustainable
24 development. *Ecological Economics* 39:347–359.
- 25 West JJ, Osnaya P, Laguna I, Martinez J, Fernandez A (2004) Co-control of urban air pollutants and greenhouse
26 gases in Mexico City. *Environmental Science and Technology* 38:3474–3481.

1 **Table 14-1. Increases in number of households and the total population of the United States, Canada, and**
 2 **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

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1 Figure 14-1. North America urban extents.



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Chapter 15. Coastal Oceans, Lakes and Rivers

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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 CO₂ 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 ± 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 CO₂ 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 CO₂ in the future or become a sink.
- Options and measures for sequestration of carbon in the ocean include deep-sea injection of CO₂ and iron fertilization, although it is unresolved how important, feasible or acceptable any of these options might be for the North American region.

- 1 • Highly variable sea-air CO₂ fluxes in coastal areas may introduce errors in North American CO₂ fluxes
2 calculated by atmospheric inversion methods. Reducing these errors will require ocean observatories
3 utilizing fixed and mobile platforms with instrumentation to measure critical stocks and fluxes as part
4 of coordinated national and international research programs. Ocean carbon sequestration studies
5 should also be continued.
-

9 **INVENTORIES (STOCKS AND FLUXES, QUANTIFICATION)**

10 This chapter focuses on the role that aquatic systems play in modulating atmospheric carbon dioxide
11 (CO₂). The chapter quantifies water-atmosphere CO₂ fluxes and considers how the underlying stocks and
12 rate processes affect them. Aquatic stocks of living carbon are small relative to stocks in the terrestrial
13 environments, but turnover rates are very high. In addition aquatic stocks are not well characterized
14 because of their spatial and temporal variability. The complexity of transformations in aquatic systems,
15 the limited data on the transformations, and space considerations have led to the focus on water-
16 atmosphere CO₂ fluxes. Aquatic systems, primarily the oceans, act as a huge reservoir for inorganic
17 carbon, containing about 50 times as much CO₂ as the atmosphere, and atmospheric concentration of CO₂
18 would be much higher in the absence of current ocean processes.

19 The ocean's biological pump converts inorganic carbon in the upper ocean to organic particulate
20 carbon by photosynthesis, transports the organic carbon from the surface by sinking, and therefore plays a
21 critical role in removing atmospheric CO₂ (Gruber and Sarmiento, 2002) in combination with physical
22 and chemical processes. The net sea-air CO₂ flux over the global ocean appears to be well constrained to
23 be about $1,800 \pm 500$ Mt C [1 Mt = one million (10^6) metric tons] or 1.8 ± 0.5 Gt C yr⁻¹ [1 Gt = one
24 billion (10^9) metric tons] from the atmosphere into the ocean (Figure 15-1 and Table 15-1). (See Chapter
25 2 for a description of how ocean carbon fluxes relate to the global carbon cycle.) The atmosphere is well
26 mixed and nearly homogenous. The large spatial variability in sea-air CO₂ fluxes shown in Figure 15-1 is
27 driven by a combination of physical, chemical, and biological processes in the ocean. The flux over the
28 coastal margins has neither been well characterized (Liu *et al.*, 2000) nor integrated into global
29 calculations because there are large variations over small spatial and temporal scales, and observations
30 have been limited. The need for higher spatial resolution to resolve the coastal variability has hampered
31 modeling efforts. In the following sections we review existing information on the coastal ocean carbon
32 cycle and its relationship to the global ocean, and we present the results of a new analysis of about a half
33 million observations of sea-air flux of CO₂ in coastal waters surrounding the North American continent.

1 **Table 15-1. Climatological mean distribution of the net sea-air CO₂ flux (in Gt C yr⁻¹) over the global**
2 **ocean regions (excluding coastal areas) in reference year 1995.** The fluxes are based on about 1.75
3 million partial pressure measurements for CO₂ in surface ocean waters, excluding the measurements made
4 in the equatorial Pacific (10°N- 10°S) during El Niño periods (see Takahashi *et al.*, 2002). The
5 NCAR/NCEP 42-year mean wind speeds and the (wind speed)² dependence for sea-air gas transfer rate are
6 used (Wanninkhof, 1992). Plus signs indicate that the ocean is a source for atmospheric CO₂, and negative
7 signs indicate that ocean is a sink. The ocean uptake has also been estimated on the basis of the following
8 methods: temporal changes in atmospheric oxygen and CO₂ concentrations (Keeling and Garcia, 2002;
9 Bender *et al.*, 2005), ¹³C/¹²C ratios in sea and air (Battle *et al.*, 2000; Quay *et al.*, 2003), ocean CO₂
10 inventories (Sabine *et al.*, 2004), and coupled carbon cycle and ocean general circulation models
11 (Sarmiento *et al.*, 2000; Gruber and Sarmiento, 2002). The consensus is that the oceans take up 1.3 to 2.3
12 Gt C yr⁻¹

13
14 **Figure 15-1. Global distribution of air-sea CO₂ flux.** The white line represents zero flux and separates
15 sources and sinks. The sources are primarily in the tropics (yellow and red) with a few areas of deep mixing
16 at high latitudes. Updated from Takahashi *et al.* (2002).

17 18 **Global Coastal Ocean Carbon Fluxes**

19 The carbon cycle in coastal oceans involves a series of processes, including runoff from terrestrial
20 environments, upwelling and mixing of high CO₂ water from below, photosynthesis at the sea surface,
21 sinking of organic particles, respiration, production and consumption of dissolved organic carbon, and
22 sea-air CO₂ fluxes (Figure 15-2). Although fluxes in the coastal oceans are large relative to surface area,
23 there is disagreement as to whether these regions are a net sink or a net source of CO₂ to the atmosphere
24 (Tsunogai *et al.*, 1999; Cai and Dai, 2004; Thomas *et al.*, 2004). Great uncertainties remain in coastal
25 carbon fluxes, which are complex and dynamic, varying rapidly over short distances and at high
26 frequencies. Only recently have new technologies allowed for the measurement of these rapidly changing
27 fluxes (Friederich *et al.*, 1995 and 2002; Hales and Takahashi, 2004).

28
29 **Figure 15-2. In the top panel, mean air/sea CO₂ flux is calculated from shipboard measurements on a**
30 **line perpendicular to the central California coast.** Flux within Monterey Bay (~0–20 km offshore) is
31 into the ocean, flux across the active upwelling region (~20–75 km offshore) is from the ocean, and flux in
32 the California Current (75–300 km) is on average into the ocean. These fluxes result from the processes
33 shown in the bottom panel. California Undercurrent water, which has a high CO₂ partial pressure, upwells
34 near shore, and is advected offshore into the California Current and into Monterey Bay. Phytoplankton
35 growing in the upwelled water use CO₂ as a carbon source, and CO₂ is drawn to low levels in those areas.
36 Phytoplankton carbon eventually sinks or is subducted below the euphotic zone, where it decays, elevating

1 the CO₂ levels of subsurface waters. Where the level of surface CO₂ is higher than the level of atmospheric
2 CO₂, diffusion drives CO₂ into the atmosphere. Conversely, where the level of surface CO₂ is lower than
3 that of atmospheric CO₂, diffusion drives CO₂ into the ocean. The net sea/air flux on this spatial scale is
4 near zero. DIC = dissolved inorganic carbon; POC = particulate organic carbon. Updated from Pennington
5 *et al.* (in press).

6
7 Carbon is transported from land to sea mostly by rivers in four components: CO₂ dissolved in water,
8 organic carbon dissolved in water, particulate inorganic carbon (e. g. calcium carbonate, CaCO₃), and
9 particulate organic carbon. The global rate of river input has been estimated to be 1,000 Mt C yr⁻¹, about
10 38% of it as dissolved CO₂ (or 384 Mt C yr⁻¹), 25% as dissolved organic matter, 21% as organic particles
11 and 17% as CaCO₃ particles (Gattuso *et al.*, 1998). Estimates for the riverine dissolved CO₂ flux vary
12 from 385 to 429 Mt C yr⁻¹ (Sarmiento and Sundquist, 1992). The Mississippi River, the seventh-largest in
13 freshwater discharge in the world, delivers about 13 Mt C yr⁻¹ as dissolved CO₂ (Cai, 2003). Organic
14 matter in continental shelf sediments exhibits only weak isotope and chemical signatures of terrestrial
15 origin, suggesting that riverine organic matter is reprocessed in coastal environments on a time scale of 20
16 to 130 years (Hedges *et al.*, 1997; Benner and Opsahl, 2001). Of the organic carbon, about 30% is
17 accumulating in estuaries, marshes, and deltas, and a large portion (20% to 60%) of the remaining 70% is
18 readily and rapidly oxidized in coastal waters (Smith and Hollibaugh, 1997). Only about 10% is estimated
19 to be contributed by human activities, such as agriculture and forest clearing (Gattuso *et al.*, 1998), and
20 the rest is a part of the natural carbon cycle.

21 One of the major differences between coastal and open ocean systems is the activity of the biological
22 pump. In coastal environments, the pump operates much more efficiently, leading to rapid reduction of
23 surface CO₂ and thus complicating the accurate quantification of sea-air CO₂ fluxes. For example,
24 Ducklow and McCallister (2004) constructed a carbon balance for the coastal oceans using the framework
25 of the ocean carbon cycle of Gruber and Sarmiento (2002) and estimated a net CO₂ removal by primary
26 productivity of 1,200 Mt C yr⁻¹ and a large CO₂ sink of 900 Mt C yr⁻¹ for the atmosphere. In contrast,
27 Smith and Hollibaugh (1993) estimated a biological pump of about 200 Mt C yr⁻¹ and concluded that the
28 coastal oceans are a weak CO₂ sink of 100 Mt C yr⁻¹, about one-ninth of the estimate by Ducklow and
29 McCallister (2004). Since the estimated sea-air CO₂ flux depends on quantities that are not well
30 constrained, the mass balance provides widely varying results.

31 32 **North American Coastal Carbon**

33 Two important types of North American coastal ocean environments can be identified: (1) river-
34 dominated coastal margins with large inputs of fresh water, organic matter, and nutrients from land (e.g.,

1 Mid- and South-Atlantic Bights) and (2) coastal upwelling zones (e.g., the California-Oregon-Washington
2 coasts, along the eastern boundary of the Pacific) where physical processes bring cool, high-nutrient and
3 high-CO₂ waters to the surface (Cai *et al.*, 2003). In both environments, the biological uptake of CO₂
4 plays an important role in determining whether an area becomes a sink or a source for the atmosphere.

5 High biological productivity fueled by nutrients added to coastal waters can lead to seawater
6 becoming a CO₂ sink during the summer growing season, as observed in the Bering Sea Shelf (Codispoti
7 and Friederich, 1986) and the northwest waters off Oregon and Washington (van Geen *et al.*, 2000; Hales
8 *et al.*, 2005). Similar CO₂ draw-downs may occur in the coastal waters of the Gulf of Alaska and in the
9 Gulf of Mexico near the Mississippi River outflow. Coastal upwelling results in a very high concentration
10 of CO₂ for the surface water (as high as 1,000 µatm), and hence the surface water becomes a strong CO₂
11 source. This is followed by rapid biological uptake of CO₂, which causes the water to become a strong
12 CO₂ sink (Friederich *et al.*, 2002; Hales *et al.*, 2005).

13 A review of North American coastal carbon fluxes has been carried out by Doney *et al.* (2004) (Table
14 15-2). The information reviewed was very limited in space (only 13 locations) and time, leading Doney *et*
15 *al.* to conclude that it was unrealistic to reliably estimate an annual flux for North American coastal
16 waters. Measurement programs have increased recently, and we have used the newly available data to
17 calculate annual North American coastal fluxes for the first time.

18
19 **Table 15-2. Variability of CO₂ distributions and fluxes in U.S. coastal waters from regional surveys**
20 **and moored measurements (from Doney *et al.* 2004).**

21 22 **Synthesis of Available North American Sea-Air Coastal CO₂ Fluxes**

23 A large data set consisting of 550,000 measurements of the partial pressure of CO₂ (pCO₂) in surface
24 waters has been assembled and analyzed (Figure 15-3; see Appendix 15A for details). pCO₂ is measured
25 in a carrier gas equilibrated with seawater and, as such, it is a measure of the outflux/influx tendency of
26 CO₂ from the atmosphere. pCO₂ is affected by physical and biological processes increasing with
27 temperature and decreasing with photosynthesis. The data were obtained by the authors and collaborators,
28 quality-controlled, and assembled in a uniform electronic format for analysis (available at
29 www.ldeo.columbia.edu/res/pi/CO2). Observations in each 1° × 1° pixel area were compiled into a single
30 year and were analyzed for time-space variability. Seasonal and interannual variations were not well
31 characterized except in a few locations (Friederich *et al.*, 2002). The annual mean sea-air pCO₂ difference
32 (delta pCO₂) was computed for 5°-wide zones along the North American continent and was plotted as a
33 function of latitude for four regions (Figure 15-4): North Atlantic, Gulf of Mexico/Caribbean, North
34 Pacific, and Bering/Chukchi Seas. Figure 15-4A shows the fluxes in the first nearshore band, and Figure

1 15-4B shows the fluxes for a band that is several hundred kilometers from shore. The average fluxes for
2 them and for the intermediate bands are given in Table 15-3. The flux and area data are listed in Table 15-
3 4. A full complement of seasonal observations are lacking in the Arctic Sea, including Hudson Bay, the
4 northern Labrador Sea, and the Gulf of St. Lawrence; the northern Bering Sea; the Gulf of Alaska; the
5 Gulf of California; and the Gulf of Mexico and the Caribbean Sea.

6
7 **Figure 15-3. (A). Distribution of coastal CO₂ partial pressure measurements made between 1979 and**
8 **2004. (B). The distribution of the net sea-air CO₂ flux over 1° × 1° pixel areas (N-S 100 km, E-W 80**
9 **km) around North America.** The flux (grams of carbon per square meter per year) represents the
10 climatological mean over the 25-year period. The magenta-blue colors indicate that the ocean water is a
11 sink for atmospheric CO₂, and the green-yellow-orange colors indicate that the sea is a CO₂ sink. The data
12 were obtained by the authors and collaborators of this chapter and are archived at the Lamont-Doherty
13 Earth Observatory (www.ldeo.columbia.edu/res/pi/CO2).

14
15 **Figure 15-4. Estimated sea-air CO₂ fluxes (grams of carbon per square meter per year) from 550,000**
16 **seawater CO₂ partial pressure (pCO₂) observations made from 1979 to 2004 in ocean waters**
17 **surrounding the North American continent.** (A) Waters within one degree (about 80 km) of the coast
18 and (B) open ocean waters between 300 and 900 km from the shore (see Figure 15-3B). The annual mean
19 sea-air pCO₂ difference (delta pCO₂) values were calculated from the weekly mean atmospheric CO₂
20 concentrations in the GLOBALVIEW-CO₂ database (2004) over the same pixel area in the same week and
21 year as the seawater pCO₂ was measured. The monthly net sea-air CO₂ flux was computed from the mean
22 monthly wind speeds in the National Centers for Environmental Prediction/National Center for
23 Atmospheric Research (NCEP/NCAR) database in the (wind speed)² formulation for the sea-air gas
24 transfer rate by Wanninkhof (1992). The ± uncertainties represent one standard deviation.

25
26 **Table 15-3. Climatological mean annual sea-air CO₂ flux (g C m⁻² yr⁻¹) over the oceans surrounding**
27 **North America.** Negative values indicate that the ocean is a CO₂ sink for the atmosphere. N is the number
28 of seawater pCO₂ measurements. The ± uncertainty is given by one standard deviation of measurements
29 used for analysis and represents primarily the seasonal variability.

30
31 The offshore patterns follow the same general trend found in the global data set shown in Figure 15-1.
32 On an annual basis the lower latitudes tend to be a source of CO₂ to the atmosphere, whereas the higher
33 latitudes tend to be sinks (Figures 15-3B and 15-4B). The major difference in the coastal waters is that the
34 latitude where CO₂ starts to enter the ocean is further north than it is in the open ocean, particularly in the
35 Atlantic. A more detailed region-by-region description follows.

1 Pacific Ocean

2 Observations made in waters along the Pacific coast of North America illustrate how widely coastal
3 waters vary in space and time, in this case driven by upwelling and relaxation (Friederich *et al.*, 2002).
4 Figure 15-5A shows a summertime quasi-synoptic distributions of temperature, salinity, and pCO₂ in
5 surface waters based on measurements made in for July through September 2005. The effects of the
6 Columbia River plume emanating from ~46°N are clearly seen (colder temperature, low salinity, and low
7 pCO₂), as are coastal upwelling effects off Cape Mendocino (~40°N) (colder, high salinity, and very high
8 pCO₂). These coastal features are confined to within 300 km from the coast. The 1997–2005 time-series
9 data for surface water pCO₂ observed off Monterey Bay (Figure 15-5B) show the large, rapidly
10 fluctuating sea-air CO₂ fluxes during the summer upwelling season in each year as well as the low-pCO₂
11 periods during the 1997–1998 and 2002–2003 El Niño events. In spite of the large seasonal variability,
12 ranging from 200 to 750 μatm, the annual mean sea-air pCO₂ difference and the net CO₂ flux over the
13 waters off Monterey Bay areas (~37°N) are close to zero (Pennington *et al.*, in press). The seasonal
14 amplitude decreases away from the shore and in the open ocean bands, where the sea-air CO₂ flux
15 changes seasonally in response to seawater temperature (out of the ocean in summer and into the ocean in
16 winter).

17
18 **Figure 15-5. Time-space variability of coastal waters off the west coast of North America.** (A) Quasi-
19 synoptic distribution of the temperature, salinity, and pCO₂ in surface waters during July–September 2005.
20 The Columbia River plume (~46°N) and the upwelling of deep waters off the Cape Mendocino (~40°N) are
21 clearly seen. (B) 1997–2005 time-series data for sea-air CO₂ flux from a mooring off Monterey Bay,
22 California. Seawater is a CO₂ source for the atmosphere during the summer upwelling events, but
23 biological uptake reduces levels very rapidly. These rapid fluctuations can affect atmospheric CO₂ levels.
24 For example, if CO₂ from the sea is mixed into a static column, a 500-m-thick planetary boundary layer
25 over the course of one day, atmospheric CO₂ concentration would change by 2.5 μatm. If the column of air
26 is mixed vertically through the troposphere to 500 mbar, a change of about 0.5 μatm would occur. The
27 effects would be diluted as the column of air mixes laterally. However, this demonstrates that the large
28 fluctuations of sea-air CO₂ flux observed over coastal waters could affect the concentration of CO₂
29 significantly enough to affect estimates of air-land flux based on the inversion of atmospheric CO₂ data.
30 Sea-air CO₂ flux was low during the 1997–1998 and 2002–2003 El Niño periods.

31
32 The open ocean Pacific waters south of 30°N are on the annual average a CO₂ source to the
33 atmosphere, whereas the area north of 40°N is a sink, and the zone between 30° and 40°N is neutral
34 (Takahashi *et al.*, 2002). Coastal waters in the 40°N through 45°N zone (northern California-Oregon
35 coasts) are even a stronger CO₂ sink, associated with nutrient input and stratification by the Columbia

1 River (Hales *et al.*, 2005). On the other hand, coastal pCO₂ values in the 15°N through 40°N zones have
2 pCO₂ values similar to open ocean values and to the atmosphere. In the zones 15°N through 40°N, the
3 annual mean values for the net sea-air CO₂ flux are nearly zero, consistent with the finding by Pennington
4 *et al.* (in press).

6 **Atlantic Ocean**

7 With the exception of the 5°N–10°N zone, the open ocean areas are an annual net sink for
8 atmospheric CO₂. The open oceans become more intense CO₂ sinks toward higher latitudes, especially
9 north of 35°N (Figure 15-3B). Between 15°N and 45°N, the open ocean waters are a CO₂ sink (Takahashi
10 *et al.*, 2002), whereas the nearshore waters are a CO₂ source. Accordingly, in contrast to the Pacific coast,
11 the latitude where Atlantic coastal waters become a CO₂ sink is located further north than that for the
12 open ocean fluxes. In the areas north of 45°N, the open ocean waters are a strong CO₂ sink due primarily
13 to the cold Labrador Sea waters.

14 In the coastal zone very high pCO₂ values (up to 2,600 µatm) are observed occasionally in areas
15 within 10 km offshore of the barrier islands. These waters have salinities around 20 and appear to
16 represent outflow of estuarine/marsh waters rich in carbon (Cai *et al.*, 2003). The large contribution of
17 fresh water that is rich in organic matter relative to the Pacific contributes to the coastal Atlantic source.
18 Offshore fluxes are in phase with the seasonal cycle of warming and cooling; fluxes are out of the ocean
19 in summer and fall and are the inverse in winter and spring.

21 **Bering and Chukchi Seas**

22 Although measurements in these high-latitude waters are limited, the relevant data for the Bering Sea
23 (south of 65°N) and Chukchi Sea (north of 65°N) are plotted as a function of the latitude in Figure 15-4.
24 The values for the areas north of 55°N are for the summer months only; CO₂ observations are not
25 available during winter seasons. Although data scatter widely, the coastal and open ocean waters are a
26 strong CO₂ sink during the summer months due to photosynthetic draw-down of CO₂. The data in the
27 70°–75°N zone are from the shallow shelf areas in the Chukchi Sea. These waters are a very strong CO₂
28 sink (sea-air pCO₂ differences ranging from –80 to –180 µatm) with little changes between the coastal
29 and open ocean areas. The sea-air CO₂ flux during winter months is not known.

31 **Gulf of Mexico and Caribbean Sea**

32 Although observations are limited, available data suggest that these waters are a strong CO₂ source
33 (Figure 15-4 and Table 15-3). A subsurface anoxic zone has been formed in the Texas-Louisiana coast as
34 a result of the increased addition of anthropogenic nutrients and organic carbon by the Mississippi River

1 (e.g., Lohrenz *et al.*, 1999). The carbon-nutrient cycle in the northern Gulf of Mexico is also being
2 investigated (e.g., Cai, 2003), and the studies suggest that at times those waters are locally a strong CO₂
3 sink due to high biological production.

4 5 **SYNTHESIS**

6 An analysis of half a million measurements of sea-air flux of CO₂ shows that the nearshore
7 (< 100 km) coastal waters surrounding North America are a net CO₂ source for the atmosphere on an
8 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
9 Gulf of Mexico and Caribbean Sea. The open oceans are a net CO₂ sink on an annual average (Table 15-
10 4; Takahashi *et al.*, 2004). The results do not include some portions of the Arctic Sea, Bering Sea, Gulf of
11 Alaska, Gulf of Mexico, or Caribbean Sea because of insufficient data. Observations in these areas will be
12 needed to improve estimates. These results are consistent with recent global estimates that suggest that
13 nearshore areas receiving terrestrial organic carbon input are sources of CO₂ to the atmosphere and that
14 marginal seas are sinks (Borges, 2005; Borges *et al.*, in press). Hence, the net contribution from North
15 American ocean margins is small and difficult to distinguish from zero. It is not clear how much of the
16 open ocean sink results from photosynthesis driven by nutrients of coastal origin.

17
18 **Table 15-4. Areas (km²) and mean annual sea-air CO₂ flux (Mt C yr⁻¹) over four ocean regions**
19 **surrounding North America.** Since the observations in the areas north of 60°N in the Chukchi Sea were
20 made only during the summer months, the fluxes from that area are not included. The ± uncertainty is given
21 by one standard deviation of measurements used for analysis and represents primarily the seasonal
22 variability.

23 24 **TRENDS AND DRIVERS**

25 The sea-to-air CO₂ flux from the coastal zone is small (about 1%) compared with the global ocean
26 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-
27 sea CO₂ budget. However, coastal waters undergo large variations in sea-air CO₂ flux on daily to seasonal
28 time scales and on small spatial scales (Figure 15-5). Fluxes can change on the order of 250 g C m⁻² yr⁻¹
29 or 0.7 g C m⁻² day⁻¹ on a day to day basis (Figure 15-5). These large fluctuations can significantly
30 modulate atmospheric CO₂ concentrations over the adjacent continent and need to be considered when
31 using the distribution of CO₂ in calculations of continental fluxes.

32 Freshwater bodies have not been treated in this analysis except to note the large surface pCO₂
33 resulting from estuaries along the east coast. The Great Lakes and rivers also represent net sources of CO₂
34 as, in the same manner as the estuaries, organic material from the terrestrial environment is oxidized so

1 that respiration exceeds photosynthesis. Interestingly, the effect of fresh water is opposite along the coast
2 of the Pacific northwest, where increased stratification and iron inputs enhance photosynthetic activity
3 (Ware and Thomson, 2005), resulting in a large sink for atmospheric CO₂ (Figure 15-3). A similar
4 process may be at work at the mouth of the Amazon (Körtzinger, 2003). This emphasizes once again the
5 important role of biological processes in controlling the sea-air fluxes of CO₂.

6 The sea-air fluxes and the underlying carbon cycle processes that determine them (Figure 15-2) vary
7 seasonally, interannually, and on longer time scales. The eastern Pacific, including the U.S. west coast, is
8 subject to changes associated with large-scale climate oscillations such as El Niño (Chavez *et al.*, 1999;
9 Feely *et al.*, 2002; Feely *et al.*, in press) and the Pacific Decadal Oscillation (PDO) (Chavez *et al.*, 2003;
10 Hare and Mantua, 2000; Takahashi *et al.*, 2003). These climate patterns, and others like the North
11 Atlantic Oscillation (NAO), alter the oceanic CO₂ sink/source conditions directly through seawater
12 temperature changes as well as ecosystem variations that occur via complex physical-biological
13 interactions (Hare and Mantua, 2000; Chavez *et al.*, 2003; Patra *et al.*, 2005). For example, during El
14 Niño, upwelling of high CO₂ waters is dramatically reduced along central California (Figure 15-5) but so
15 is photosynthetic uptake of CO₂ (Chavez *et al.* 2002) so the net effect of climate variability and change on
16 sea-air fluxes remains uncertain. What is certain is that the biological, chemical and physical processes
17 controlling the sea-air fluxes of CO₂ are strongly affected by natural and anthropogenic change and that
18 efforts to track them need to be considered in global carbon management plans.

19 20 **OPTIONS AND MEASURES**

21 Two options for ocean carbon sequestration have been considered: (1) deep-sea injection of CO₂
22 (Brewer, 2003) and (2) ocean iron fertilization (Martin, 1990). The first might be viable in North
23 American coastal waters, although cost and potential biological side effects are unresolved issues. The
24 largest potential for iron fertilization resides in the equatorial Pacific and the Southern Ocean, although it
25 could be considered for the open ocean waters of the Gulf of Alaska and offshore waters of coastal
26 upwelling systems. Iron fertilization would be an economical alternative, but there is still disagreement
27 over how much carbon would be sequestered (Bakker *et al.*, 2001; Boyd *et al.*, 2000; Coale *et al.*, 2004;
28 Gervais *et al.*, 2002) and what the potential side effects would be (Chisholm *et al.*, 2001).

29 30 **R&D NEEDS VIS A VIS OPTIONS**

31 Waters with highly variable sea-air CO₂ fluxes are located primarily within 100 km of the coast
32 (Figure 15-5). With the exception of a few areas, the available observations are grossly inadequate to
33 resolve the high-frequency, small-spatial-scale variations. These high intensity sea-air CO₂ flux events
34 may introduce errors in continental CO₂ fluxes calculated by atmospheric inversion methods. Achieving a

1 comprehensive understanding of the carbon cycle in waters surrounding the North American continent
2 will require development of advanced technologies and sustained research efforts. Both of these seem to
3 be on the horizon with (1) the advent of ocean observatories that include novel fixed and mobile
4 platforms together with developing instrumentation to measure critical stocks and fluxes and (2) national
5 and international research programs that include the Integrated Ocean Observing System (IOOS) and
6 Ocean Carbon and Climate Change (OC³). Given the importance of aquatic systems to atmospheric CO₂
7 concentrations, these developing efforts must be strongly encouraged. Ocean carbon sequestration studies
8 should also be continued.

10 REFERENCES

- 11 Bakker, D. C. E., Watson, A. J. and Law, C. S. (2001). Southern Ocean iron enrichment promotes inorganic carbon
12 drawdown. *Deep-Sea Res. II*, 48, 2483–2507.
- 13 Battle, M., Bender, M. L., Tans, P. P., White, J. W. C., Ellis, J. T., Conway, T. and Francey, R. J. (2000). Global
14 carbon sinks and their variability inferred from atmospheric O₂ and δ¹³C. *Science*, 287, 2467–2470.
- 15 Bender, M. L., Ho, D. T., Hendricks, M. B., Mika, R., Bazttle, M. O., Tans, P. P., Conway, T. J., Sturtevant, B. and
16 Cassar, N. (2005). Atmospheric O₂/N₂ changes, 1993–2002: Implications for the partitioning of fossil fuel CO₂
17 sequestration. *Glob. Biogeochem. Cycles*, 19, GB4017, doi:10.1029/2004GB002410.
- 18 Benner, R. and Opsahl, S. (2001). Molecular indicators of the sources and transformations of dissolved organic
19 matter in the Mississippi River plume. *Organic Geochem.*, 32, 597–611.
- 20 Boehme, S.E., Sabine, C.L. and Reimers, C.E. (1998) CO₂ fluxes from a coastal transect: a time-series approach.
21 *Marine Chemistry*, 63, 49–67.
- 22 Borges, A.V. (2005) Do we have enough pieces of the jigsaw to integrate CO₂ fluxes in the Coastal Ocean?,
23 *Estuaries*, 28, 3–27.
- 24 Borges, A.V., B. Delille, and M. Frankignoulle (in press) Budgeting sinks and sources of CO₂ in the coastal ocean:
25 Diversity of ecosystems counts.
- 26 Boyd, P. W., *et al.* (2000). A mesoscale phytoplankton bloom in the polar Southern Ocean stimulated by iron
27 fertilization. *Nature*, 407, 695–702.
- 28 Brewer, P.G. (2003) Direct Injection of Carbon Dioxide into the Oceans. In: “The Carbon Dioxide Dilemma:
29 Promising Technologies and Policies.” National Academies Press, pp. 43–51
- 30 Cai, W.-J. (2003). Riverine inorganic carbon flux and rate of biological uptake in the Mississippi River plume.
31 *Geophys. Res. Lett.*, 30, 1032.
- 32 Cai, W.-J., Wang, Z. A. and Wang, Y.-C. (2003). The role of marsh-dominated heterotrophic continental margins in
33 transport of CO₂ between the atmosphere, the land-sea interface and the oceans. *Geophys. Res. Letters*, 30, No.
34 16, 1849, doi”10.1029/2003GL017633, 2003.
- 35 Cai, W. J., and M. Dai (2004), Comment on "Enhanced open ocean storage of CO₂ from shelf sea pumping",
36 *Science*, 306, 1477c.

- 1 Chavez, F. P., Strutton, P. G., Friederich, G. E., Feely, R. A., Feldman, G. C., Foley, D. G. and McPhaden, M. J.
2 (1999). Biological and chemical response of the equatorial Pacific Ocean to 1997–98 El Niño. *Science*, 286,
3 2126–2131.
- 4 Chavez, F. P., Pennington, J. T., Castro, C. G., Ryan, J. P., Michisaki, R. M., Schlining, B., Walz, P., Buck, K. R.,
5 McFayden, A., and Collins, C. A. (2002). Biological and chemical consequences of the 1997–98 El Niño in
6 central California waters. *Progress in Oceanography*, 54, 205–232.
- 7 Chavez, F. P., Ryan, J., Lluch-Cota, S. and Miguel, N. C. (2003). From anchovies to sardines and back:
8 Multidecadal change in the Pacific Ocean. *Science*, 299, 217–221.
- 9 Chisholm, S.W., P.G. Falkowski and J. Cullen. (2001). Discrediting ocean fertilization, *Science*, 294, 309–310.
- 10 Codispoti, L. A. and Friederich, G. E. (1986) Variability in the inorganic carbon system over the southeastern
11 Bering Sea shelf during the spring of 1980 and spring-summer 1981. *Continental Shelf Research*, 5, 133–160.
- 12 Coale, K. H. *et al.* (2004). Southern Ocean iron enrichment experiment: Carbon cycling in high- and low-Si waters.
13 *Science*, 304, 408–414.
- 14 DaSilva, A., Young, C., and Levitus, S. (1994). *Atlas of Marine Surface Data 1994*, NOAA Atlas NESDIS 6, U.S.
15 Department of Commerce, Washington, D.C.
- 16 DeGrandpre, M.D., Hammar, T.R., Wallace, D.W.R. and Wirick, C.D. (1997) Simultaneous mooring-based
17 measurements of seawater CO₂ and O₂ off Cape Hatteras, North Carolina. *Limnology and Oceanography*, 42,
18 21–28.
- 19 DeGrandpre, M.D., Olbu., G. J., Beatty, C.M., Hammar, T. R. (2002) Air-sea CO₂ fluxes on the U.S. Middle
20 Atlantic Bight. *Deep-Sea Research II*, 49, 4355–4367.
- 21 Doney, S. C., Anderson, R., Bishop, J., Caldeira, K., Carlson, C., Carr, M.-E., Feely, R., Hood, M., Hopkinson, C.,
22 Jahnke, R., Karl, D., Kleypas, J., Lee, C., Letelier, R., McClain, C., Sabine, C., Sarmiento, J., Stephens, B., and
23 Weller, R. (2004). *Ocean Carbon and Climate Change (OCCC): An Implementation Strategy for U.S. Ocean*
24 *Carbon Cycle Science*, UCAR, Boulder, CO, 108 pp.
- 25 Ducklow, H. W., McCallister, S. L. (2004). The biogeochemistry of carbon dioxide in the coastal oceans. In “THE
26 SEA” Vol. 13, A. R. Robinson, and K. H. Brink editors, vol. 13, 269–315, John Wiley & Sons, New York.
- 27 Feely, R. A., Boutin, J., Coasca, C. E., Dandonneau, Y., Etcheto, J., Inoue, H., Ishii, M., LeQuere, C., Mackey, D. J.,
28 McPhaden, M., Metzl, N., Poisson, A. and Wanninkhof, R. (2002). Seasonal and interannual variability of CO₂
29 in the equatorial Pacific. *Deep-Sea Res., II* 49, 2443–2469.
- 30 Feely, R. A., Takahashi, T., Wanninkhof, R., McPhaden, M. J., Cosca, C. E., Sutherland, S. C. and Carr, M-E. (in
31 press). Decadal variability of the air-sea CO₂ fluxes in the equatorial Pacific Ocean. *Journal of Geophysical*
32 *Research*.
- 33 Friederich, G.E., P.G. Brewer, R. Herlein, and F.P. Chavez (1995). Measurement of sea surface partial pressure of
34 CO₂ from a moored buoy. *Deep-Sea Research*, 42, 1175–1186.
- 35 Friederich, G., P. Walz, M. Burczynski and F.P. Chavez (2002) Inorganic Carbon in the Central California
36 Upwelling System During the 1997–1999 El Niño -La Niña Event. *Progress in Oceanography*, 54, 185–204.

- 1 Gattuso, J. M., Frankignoulle, M. and Wollast, R. (1998). Carbon and carbonate metabolism in coastal aquatic
2 ecosystem. *Annual Review of Ecology and Systematics*, 29, 405–434.
- 3 Gervais, F., U. Riebesell and M.Y. Gorbunov (2002) Changes in primary productivity and chlorophyll a in response
4 to iron fertilization in the Southern Polar Frontal Zone. *Limnol. Oceanogr.*, 47, 1324.
- 5 Gruber, N. and Sarmiento, J. L. (2002). Large –scale biogeochemical-physical interactions in elemental cycles. In
6 *The Sea*, Vol. 12, A. R. Robinson, J. McCarthy and B. J. Rothschild editors, 337–399. John Wiley & Sons, New
7 York.
- 8 Hales, B. and Takahashi, T. (2004). High-resolution biogeochemical investigation of the Ross Sea, Antarctica,
9 during the AESOPS (U. S.JGOFS) Program . *Global Biogeochem. Cycles*, Vol. 18, No. 3, GB3006, doi.
10 10.1029/2003GB002165
- 11 Hales, B., Takahashi, T. and Bandstra, L. (2005). Atmospheric CO₂ uptake by a coastal upwelling system. *Global*
12 *Biogeochem. Cycles*, 19. doi.10.1029/2004GB002295.sinks. *Global Biogeochem. Cycles*, 18, GB1010,
13 doi:10.1029/2003GB002111.
- 14 Hare, S. R. and Mantua, N. J. (2000). Empirical evidence for North Pacific regime shifts in 1977 and 1989. *Progress*
15 *in Oceanogr.*, 47, 103–145.
- 16 Hedges, J. I., R. G. Keil, and R. Benner (1997) What happens to terrestrial organic matter in the ocean?, *Organic*
17 *Geochemistry*, 27, 195–212.
- 18 Keeling, R. F. and Garcia, H. The change in oceanic O₂ inventory associated with recent global warming. *Proc. U.*
19 *S. National Acad. Sci.*, 99, 7848–7853, 2002.
- 20 Körtzinger, A. (2003) A significant CO₂ sink in the tropical Atlantic Ocean associated with the Amazon river
21 plume, *Geophysical Research Letters*, 30, 2287, doi:10.1029/2003GL018841.
- 22 Liu, K. K., Iseki, K., Chao, S.-Y. (2000). Continental margin carbon fluxes. In “The Changing Ocean Carbon
23 cycle”. R. Hansen, H. W. Ducklow and J. G. field editors, Cambridge University Press, Cambridge, UK, pp.
24 187–239.
- 25 Lohrenz, S. E., Dagg, M. J. and Whitley, T. E. (1999). Nutrients, irradiance, and mixing as factors regulating
26 primary production in coastal water impacted by the Mississippi River plume. *Continental Shelf Res.*, 19, 1113–
27 1141.
- 28 Martin, J.H. (1990) Glacial-interglacial CO₂ change: The iron hypothesis. *Paleoceanography*, 5, 1–13.
- 29 Millero, F. J., Hiscock, W.T., Huang, F., Roche, M., and Zhang, J-Z. (2001) Seasonal variation of the carbonate
30 system in Florida Bay. *Bulletin of Marine Science* 68, 101–123.
- 31 Park, P.K., Gordon, L.I. and Alvarez-Borrego, S. (1974) The carbon dioxide system of the Bering Sea, In
32 *Oceanography of the Bering Sea*, D.W. Hood editor, Occasional Publication No. 2, Institute of Marine Science,
33 University of Alaska, Fairbanks, AK.
- 34 Patra, P. K., Maksyutov, S., Ishizawa, M., Nakazawa, T., Takahashi, T. and Ukita, J. (2005). Interannual and
35 decadal changes in the sea-air CO₂ flux from atmospheric CO₂ inverse modeling. *Glob. Biogeochem. Cycles*,
36 19, GB4013, doi:10.1029/2004GB002257, 2005

- 1 Pennington, J.T., Castro, C.G., Collins, C.A., Evans, W.W. IV, Friederich, G.E., Michisaki, R.P., Chavez, F.P. (in
2 press) A carbon budget for the northern and central California coastal upwelling system. Continental Margins
3 Task Team, The Synthesis Book. Chapter 2.2, California Current System. 32 mss. pp.
- 4 Quay, P., Sommerup, R., Westby, T., Sutsman, J. and McNichol, A. (2003). Changes in the $^{13}\text{C}/^{12}\text{C}$ of dissolved
5 inorganic carbon in the ocean as a tracer of anthropogenic CO_2 uptake. *Glob. Biogeochem. Cycles*, 17, No. 1,
6 DOI 10.1029/2001GB001817.
- 7 Sabine, C. L., Feely R. A., Gruber, N., Key, R. M., Lee, K., Bullister, J. L., Wanninkhof, R., Wong, C. S., Wallace,
8 D. W. R., Tilbrook, B., Peng, T-H, Kozyr, A., Ono, T. and Rios, A. F. (2004). The oceanic sink for
9 anthropogenic CO_2 . *Science*, 305, 367–371.
- 10 Sarmiento, J. L. and Sundquist, E. T. (1992). Revised budget for the oceanic uptake of anthropogenic carbon
11 dioxide. *Nature*, 356, 589–593.
- 12 Sarmiento, J. L., Monfray, P., Maier-Reimer, E., Aumont, O., Murnane, R. J. and Orr, J. C. (2000). Sea-air CO_2
13 fluxes and carbon transport: A comparison of three ocean general circulation models. *Glob. Biogeochem.*
14 *Cycles*, 14, 1267–1281.
- 15 Simpson, J.J. (1985) Air-sea exchange of carbon dioxide and oxygen induced by phytoplankton: Methods and
16 interpretation, in *Mapping Strategies in Chemical Oceanography*, A. Zirino, editor, American Chemical Society,
17 Washington, D.C., 409–450.
- 18 Smith, S. V. and Hollinbaugh (1993). Coastal metabolism and the oceanic organic carbon balance. *Review of*
19 *Geophys.*, 31, 75–89.
- 20 Takahashi, T., Sutherland, S. C., Sweeney, C., Poisson, A., Metzl, N., Tillbrook, B., Bates, N., Wanninkhof, R.,
21 Feely, R. A., Sabine, C., Olafsson, J. and Nojiri, Y. (2002). Global sea-air CO_2 flux based on climatological
22 surface ocean pCO_2 , and seasonal biological and temperature effects, *Deep-Sea Res. II*, 49, 1601–1622.
- 23 Takahashi, T., Sutherland, S. C., Feely, R. A. and Cosca, C. (2003). Decadal variation of the surface water pCO_2 in
24 the western and central Equatorial Pacific. *Science*, 302, 852–856.
- 25 Thomas, H., Y. Bozec, K. Elkalay, and H. J. W. De Baar (2004), Enhanced open ocean storage of CO_2 from shelf
26 sea pumping, *Science*, 304, 1005–1008.
- 27 Tsunogai, S., S. Watanabe, and T. Sato (1999) Is there a "continental shelf pump" for the absorption of atmospheric
28 CO_2 ?, *Tellus B*, 5, 701–712.
- 29 van Geen, A. Takesue, R. K., Goddard, J., Takahashi, T., Barth, J.A., and Smith, R.L. (2000) Carbon and nutrient
30 dynamics during upwelling off Cape, Blanco, Oregon. *Deep-Sea Research II*, 49, 4369–4385.
- 31 Wanninkhof, R. (1992). Relationship between wind speed and gas exchange. *Jour. Geophys. Res.*, 97, 7373–7382.
- 32 Ware, D. M. and Thomson, R. D. (2005) Bottom-Up Ecosystem Trophic Dynamics Determine Fish Production in
33 the Northeast Pacific. *Science*, 308, 1280 – 1284.

Table 15-1. Climatological mean distribution of the net sea-air CO₂ 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 CO₂ 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 CO₂, 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 CO₂ concentrations (Keeling and Garcia, 2002; Bender *et al.*, 2005), ¹³C/¹²C ratios in sea and air (Battle *et al.*, 2000; Quay *et al.*, 2003), ocean CO₂ 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 (10⁶ km²)	152.0	74.6	53.0	41.1	320.7
% of area	47	23	17	13	100

17
18

1 **Table 15-2. Variability of CO₂ distributions and fluxes in U.S. coastal waters from regional surveys and**
 2 **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 <i>et al.</i> (1998)
Cape Hatteras, North Carolina	ND*	-1.0 to +1.2	ND	Moored meas.	DeGrandpre <i>et al.</i> (1997)
Middle Atlantic Bight, inner shelf	150–620	ND	-0.9	Regional survey	DeGrandpre <i>et al.</i> (2002)
Middle Atlantic Bight, middle shelf	220–480	ND	-1.6	Regional survey	DeGrandpre <i>et al.</i> (2002)
Middle Atlantic Bight, outer shelf	300–430	ND	-0.7	Regional survey	DeGrandpre <i>et al.</i> (2002)
Florida Bay, Florida	325–725	ND	ND	Regional survey	Millero <i>et al.</i> (2001)
Southern California Coastal Fronts	130–580	ND	ND	Regional survey	Simpson (1985)
Coastal Calif. (M-1; Monterey Bay)	245–550	-8 to +50	1997–98: -1.0 1998–99: +1.1	Moored meas.	Friederich <i>et al.</i> (2002)
Oregon Coast	250–640	ND	ND	Regional survey	van Geen <i>et al.</i> (2000)
Bering Sea Shelf in spring (April–June)	130–400	-8 to -12	-8	Regional survey	Codispoti <i>et al.</i> (1986)
South Atlantic Bight	300–1200	ND	2.5	Regional survey	Cai <i>et al.</i> (2003)
Miss. River Plume (summer)	80–800	ND	ND	Regional survey	Cai <i>et al.</i> (2003)
Bering Sea (Aug–Sep.)	192–400	ND	ND	Regional survey	Park <i>et al.</i> (1974)

3 * ND = no data available

1
 2 **Table 15-3. Climatological mean annual sea-air CO₂ flux (g C m⁻² yr⁻¹) over the oceans surrounding North**
 3 **America.** Negative values indicate that the ocean is a CO₂ sink for the atmosphere. N is the number of seawater
 4 pCO₂ measurements. The ± uncertainty is given by one standard deviation of measurements used for analysis and
 5 represents primarily the seasonal variability.

6

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

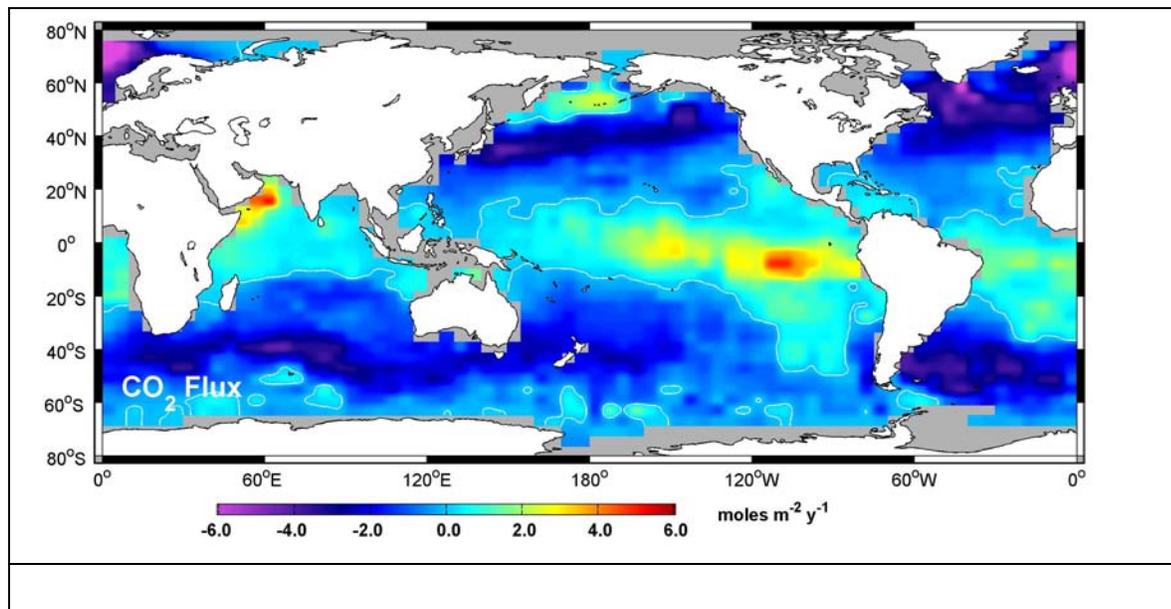
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1
 2 **Table 15-4. Areas (km²) and mean annual sea-air CO₂ flux (Mt C yr⁻¹) over four ocean regions surrounding**
 3 **North America.** Since the observations in the areas north of 60°N in the Chukchi Sea were made only during the
 4 summer months, the fluxes from that area are not included. The ± uncertainty is given by one standard deviation of
 5 measurements used for analysis and represents primarily the seasonal variability.

Ocean areas (km ²)					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

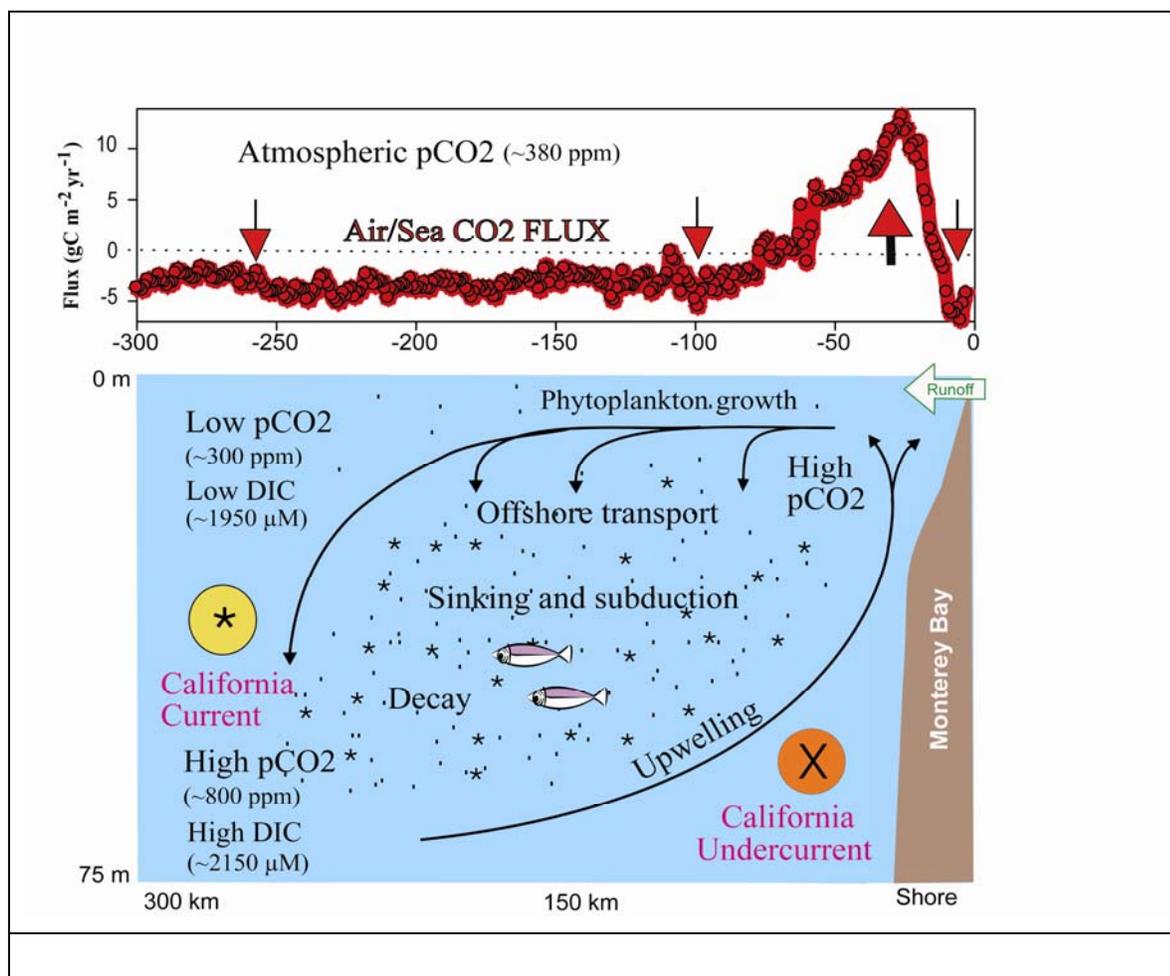
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- 1 **Figure 15-1. Global distribution of air-sea CO₂ flux. The white line represents zero flux and separates sources**
- 2 **and sinks.** The sources are primarily in the tropics (yellow and red) with a few areas of deep mixing at high
- 3 latitudes. Updated from Takahashi *et al.* (2002).

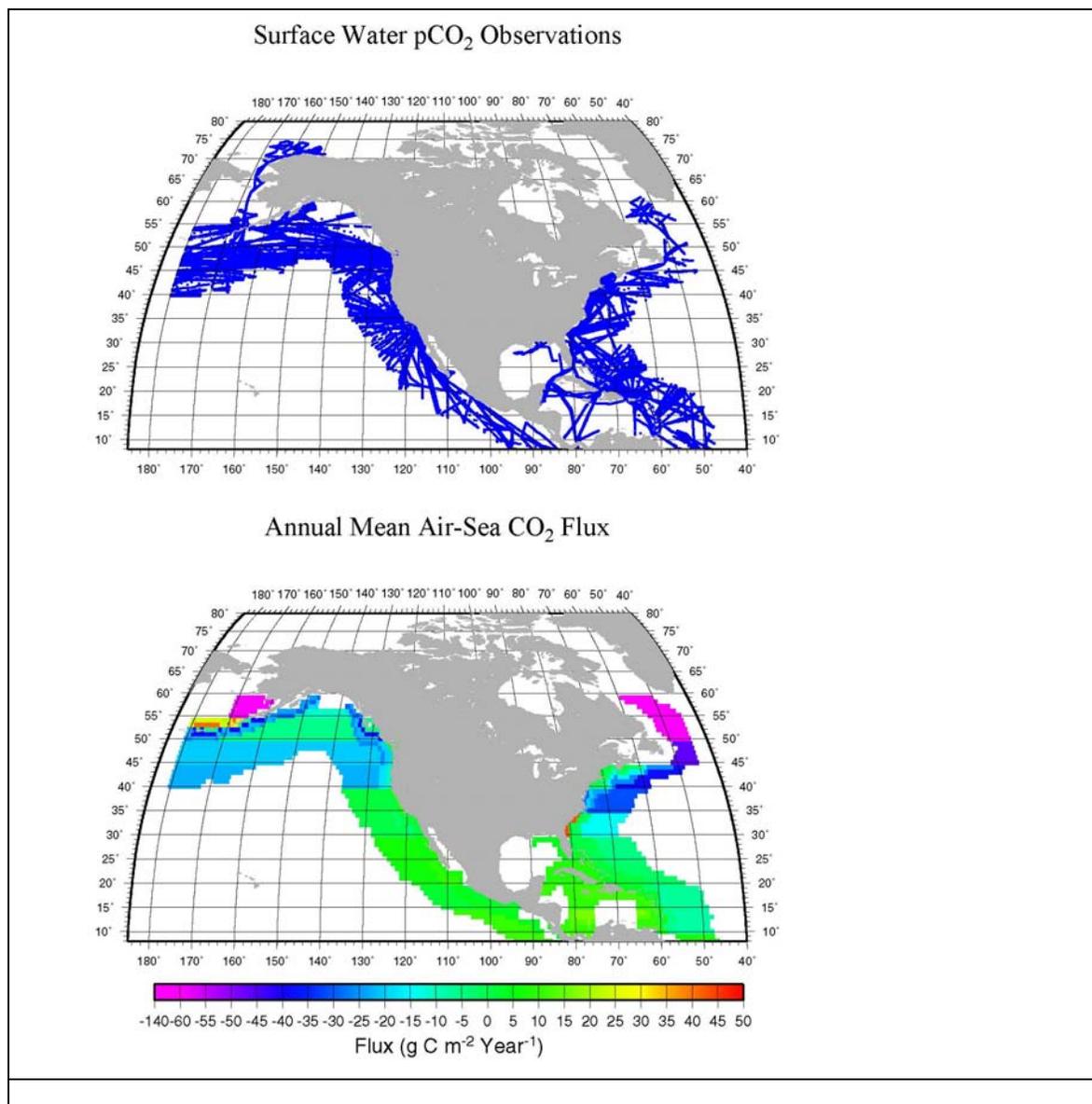


1 **Figure 15-2. In the top panel, mean air/sea CO₂ flux is calculated from shipboard measurements on a line**
 2 **perpendicular to the central California coast.** Flux within Monterey Bay (~0–20 km offshore) is into the ocean,
 3 flux across the active upwelling region (~20–75 km offshore) is from the ocean, and flux in the California Current
 4 (75–300 km) is on average into the ocean. These fluxes result from the processes shown in the bottom panel.
 5 California Undercurrent water, which has a high CO₂ partial pressure, upwells near shore, and is advected offshore
 6 into the California Current and into Monterey Bay. Phytoplankton growing in the upwelled water use CO₂ as a
 7 carbon source, and CO₂ is drawn to low levels in those areas. Phytoplankton carbon eventually sinks or is subducted
 8 below the euphotic zone, where it decays, elevating the CO₂ levels of subsurface waters. Where the level of surface
 9 CO₂ is higher than the level of atmospheric CO₂, diffusion drives CO₂ into the atmosphere. Conversely, where the
 10 level of surface CO₂ is lower than that of atmospheric CO₂, diffusion drives CO₂ into the ocean. The net sea/air flux
 11 on this spatial scale is near zero. DIC = dissolved inorganic carbon; POC = particulate organic carbon. Updated from
 12 Pennington *et al.* (in press).

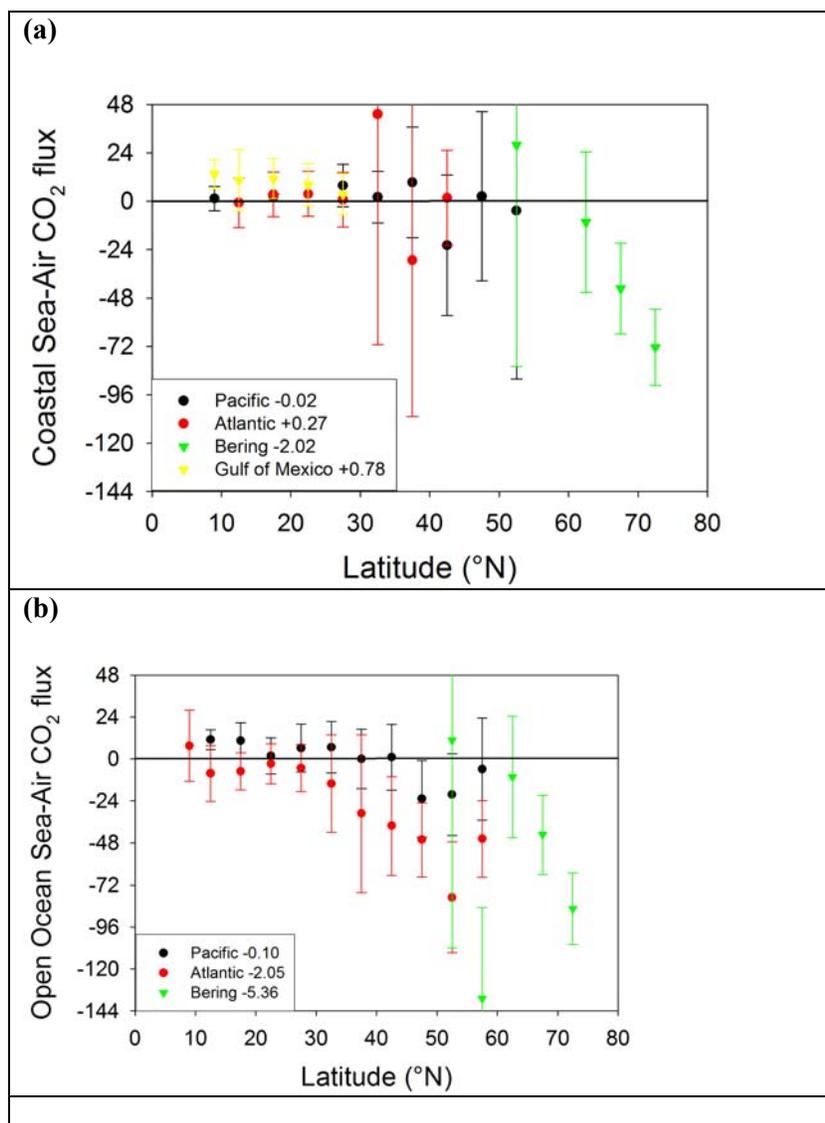
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1 **Figure 15-3. (A).** Distribution of coastal CO₂ partial pressure measurements made between 1979 and 2004.
 2 **(B).** The distribution of the net sea-air CO₂ flux over 1° × 1° pixel areas (N-S 100 km, E-W 80 km) around
 3 **North America.** The flux (grams of carbon per square meter per year) represents the climatological mean over the
 4 25-year period. The magenta-blue colors indicate that the ocean water is a sink for atmospheric CO₂, and the green-
 5 yellow-orange colors indicate that the sea is a CO₂ sink. The data were obtained by the authors and collaborators of
 6 this chapter and are archived at the Lamont-Doherty Earth Observatory (www.ldeo.columbia.edu/res/pi/CO2).
 7

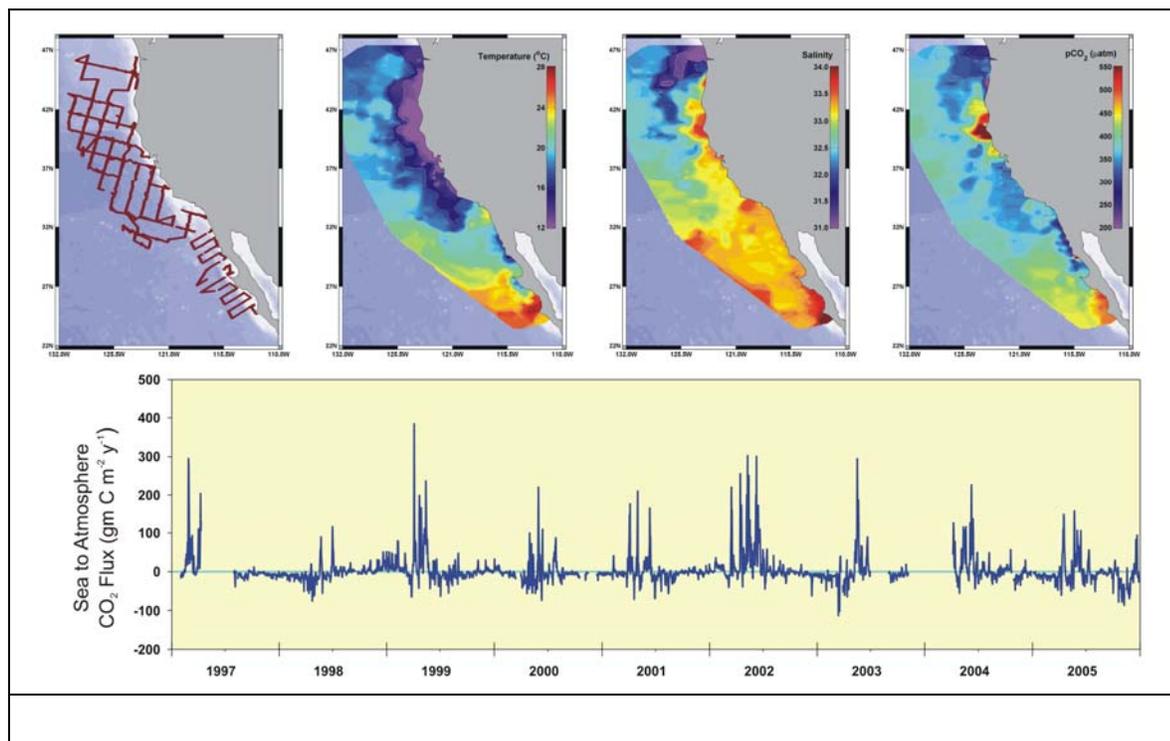


1 **Figure 15-4. Estimated sea-air CO₂ fluxes (grams of carbon per square meter per year) from 550,000**
 2 **seawater CO₂ partial pressure (pCO₂) observations made from 1979 to 2004 in ocean waters surrounding the**
 3 **North American continent. (A) Waters within one degree (about 80 km) of the coast and (B) open ocean waters**
 4 **between 300 and 900 km from the shore (see Figure 15-3B). The annual mean sea-air pCO₂ difference (delta pCO₂)**
 5 **values were calculated from the weekly mean atmospheric CO₂ concentrations in the GLOBALVIEW-CO₂ database**
 6 **(2004) over the same pixel area in the same week and year as the seawater pCO₂ was measured. The monthly net**
 7 **sea-air CO₂ flux was computed from the mean monthly wind speeds in the National Centers for Environmental**
 8 **Prediction/National Center for Atmospheric Research (NCEP/NCAR) database in the (wind speed)² formulation for**
 9 **the sea-air gas transfer rate by Wanninkhof (1992). The ± uncertainties represent one standard deviation.**



1 **Figure 15-5. Time-space variability of coastal waters off the west coast of North America.** (A) Quasi-synoptic
 2 distribution of the temperature, salinity, and pCO₂ in surface waters during July–September 2005. The Columbia
 3 River plume (~46°N) and the upwelling of deep waters off the Cape Mendocino (~40°N) are clearly seen. (B) 1997–
 4 2005 time-series data for sea-air CO₂ flux from a mooring off Monterey Bay, California. Seawater is a CO₂ source
 5 for the atmosphere during the summer upwelling events, but biological uptake reduces levels very rapidly. These
 6 rapid fluctuations can affect atmospheric CO₂ levels. For example, if CO₂ from the sea is mixed into a static column,
 7 a 500-m-thick planetary boundary layer over the course of one day, atmospheric CO₂ concentration would change
 8 by 2.5 μatm. If the column of air is mixed vertically through the troposphere to 500 mbar, a change of about 0.5
 9 μatm would occur. The effects would be diluted as the column of air mixes laterally. However, this demonstrates
 10 that the large fluctuations of sea-air CO₂ flux observed over coastal waters could affect the concentration of CO₂
 11 significantly enough to affect estimates of air-land flux based on the inversion of atmospheric CO₂ data. Sea-air CO₂
 12 flux was low during the 1997–1998 and 2002–2003 El Niño periods.

13



14

Appendix 15A

Database and Methods

1
2
3
4 A database for pCO₂, temperature and salinity in surface waters within about 1,000 km from the shore
5 of the North American continent has been assembled. About 550,000 seawater pCO₂ observations were
6 made from 1979 to 2004 by the authors and collaborators of Chapter 15. The pCO₂ data have been
7 obtained by a method using an infrared gas analyzer or gas-chromatograph for the determination of CO₂
8 concentrations in a carrier gas equilibrated with seawater at a known temperature and total pressure. The
9 precision of pCO₂ measurements has been estimated to be about ± 0.7% on average. The quality-
10 controlled data are archived at www.ldeo.columbia.edu/res/pi/CO2.

11 The zonal distribution of the surface water pCO₂, sea surface temperature (SST), and salinity data
12 shows that the greatest variability is confined within 300 km from the shores of both the Atlantic and
13 Pacific. Observations made in various years were combined into a single year and were averaged into 1°
14 × 1° pixels (approximately N-S 100 km by E-W 80 km) for the analysis. Accordingly, the results
15 represent a climatological mean condition over the past 25 years. Finer resolutions (10 × 10 km) may be
16 desirable for some areas close to shore because of outflow of estuarine and river waters and upwelling.
17 However, for this study, which is aimed at a broad picture of waters surrounding the continent, the fine
18 scale measurements have been incorporated into the 1° × 1° pixels. In addition, data with salinities of less
19 than 16.0 are considered to be inland waters and have been excluded from the analysis.

20 Climatological monthly and annual mean values for pCO₂ in each zone were computed first. Then
21 the sea-air pCO₂ difference, which represents the thermodynamic driving potential for sea-air CO₂ gas
22 transfer, was estimated using the atmospheric CO₂ concentration data. Finally, the net sea-air CO₂ flux
23 was computed using transfer coefficients estimated on the basis of climatological mean monthly wind
24 speeds using the (wind speed)² formulation of Wanninkhof (1992). The transfer coefficient depends on
25 the state of turbulence above and below the sea-air interface and is commonly parameterized as a function
26 of wind speeds (corrected to 10 m above the sea surface). However, selection of wind data is problematic
27 because wind speeds vary with the time scale (hourly, diurnal, or seasonal). For example, fluxes
28 calculated for the South Atlantic Bight from 6-h mean wind speeds in the NCEP/NCAR version 2 file (1°
29 × 1° mean) were lower than those estimated using the monthly mean. This discrepancy suggests that ships
30 used commonly for coastal carbon studies tend to be small and hence are rarely at sea under high wind
31 conditions, so observations are biased toward lower winds. Taking into account that the observations have
32 been made infrequently over multiple years, the gas transfer coefficients estimated from climatological
33 mean monthly wind speeds may be more representative. The Schmidt number is computed using

1 measured SST and climatological mean salinity (Da Silva *et al.* 1994). The flux values in a given month
2 are then averaged to yield a climatological mean flux (and standard deviation) for each month. This
3 procedure assumes implicitly that the seawater pCO₂ changes at much slower rates in space and time than
4 the wind speed and that the seawater pCO₂ does not correlate with the wind speed.

1

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