APPENDIX A

Historical Overview of the Development of United States, Canadian, and Mexican Ecosystem Sources and Sinks for Atmospheric Carbon

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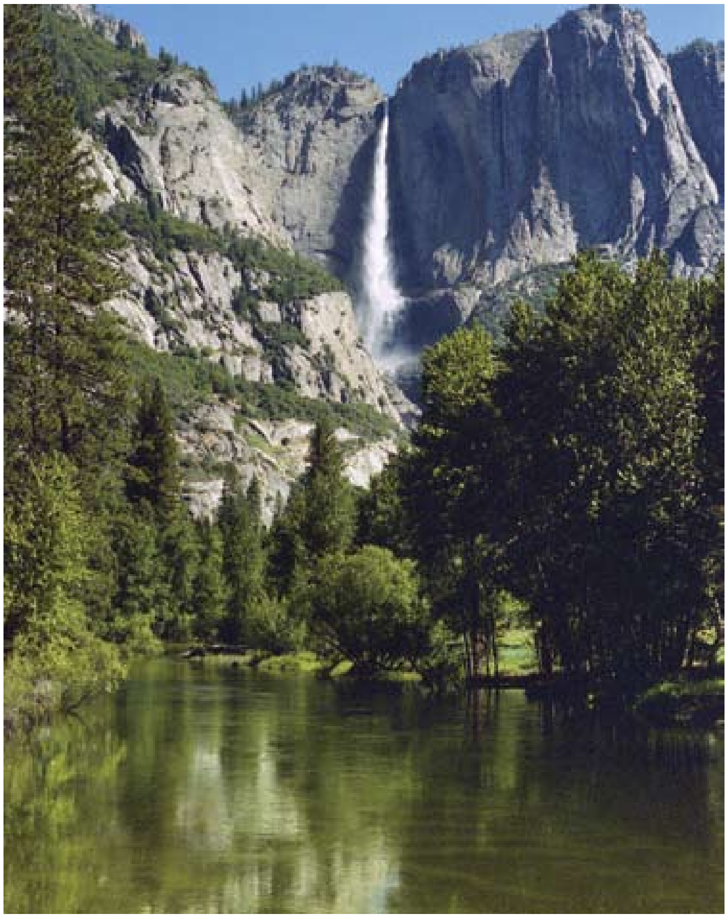
Although the lands of the New World were inhabited before the arrival of Europeans, the changes since arrival have been enormous, especially during the last two centuries. Peak United States emissions from land-use change occurred late in the nineteenth century, and the last few decades have experienced a carbon sink (Houghton *et al.*, 1999; Hurtt *et al.*, 2002). In Canada, peak emissions occurred nearly a century later than in the United States, and current data show that land-use change causes a net carbon sink (Environment Canada, 2005). In Mexico, the emissions of carbon continue to increase from net deforestation. All three countries may be in different stages of the same development pattern (Figure 3.2)



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

The total area of forests and woodlands in the United States declined as a result of agricultural expansion by

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

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

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

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

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

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

APPENDIX B

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

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

Where eddy-covariance and inventory measurements are concurrent, the rates of accumulation or loss of biomass are often consistent to within several tens of g C per m^2 per year for a one-year sample (10 g C per year is 5% of a typical net sink of two metric tons of carbon per hectare per year for an Eastern deciduous successional forest). Published intercomparisons in North America exist for western coniferous forests (Law *et al.*, 2001), agricultural sites (Verma *et al.*, 2005), and eastern deciduous forests (Barford *et al.*, 2001; Cook *et al.*, 2004; Curtis *et al.*, 2002; Ehmann *et*

Table B.1 Carbon budget for Harvard Forest from forest inventory and eddy-covariance flux measurements, 1993-2001. Source: Barford *et al.* (2001), Table 1. Numbers in parentheses give the ranges of the 95% confidence intervals. Following the sign convention in Barford *et al.* (2001), positive values represent uptake from the atmosphere (*i.e.*, a sink) and negative values a release (*i.e.*, a source).

Component	Change in carbon stock or flux (Mg C per ha per year)[a]	Totals
Change in live biomass		
A. Above-ground		
1. Growth	1.4 (±0.2)	
2. Mortality	−0.6 (±0.6)	
B. Below-ground (estimated)		
1. Growth	0.3	
2. Mortality	−0.1	
Subtotal		1.0 (±0.2)
Change in dead wood		
A. Mortality		
1. Above-ground	0.6 (±0.6)	
2. Below-ground	0.1	
B. Respiration	−0.3 (±0.3)	
Subtotal		0.4 (±0.3)
Change in soil carbon (net)		0.2 (±0.1)
Sum of carbon budget figures		1.6 (±0.4)
Sum of eddy-covariance flux measurements		2.0 (±0.4)

[a] 1 Mg C per ha per year = 100g C per m^2 per year.

al., 2002; Gough *et al*., in review). Multiyear studies at two sites (Barford *et al*., 2001; Gough *et al*., in review) show that 5- to 10-year averages converge toward inventory measurements. Table B.1 from Barford *et al.* (2001) shows the results of nearly a decade of concurrent measurements in an eastern deciduous forest.

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

APPENDIX C

Industry and Waste Management - Supplemental Material

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This appendix presents diagrams of the carbon flows in Canada, the United States, and Mexico, respectively (Figures C.1 through C.3). The numerical data in these figures are shown in thousands of metric tons of carbon, which can be converted into thousands of metric tons of carbon dioxide (CO_2) equivalents by multiplying the carbon values by 44/12 (*i.e.*, the ratio of CO_2 mass to carbon mass). The combined carbon flows for all three nations are presented in Figure 8.2 in Chapter 8 of this report.

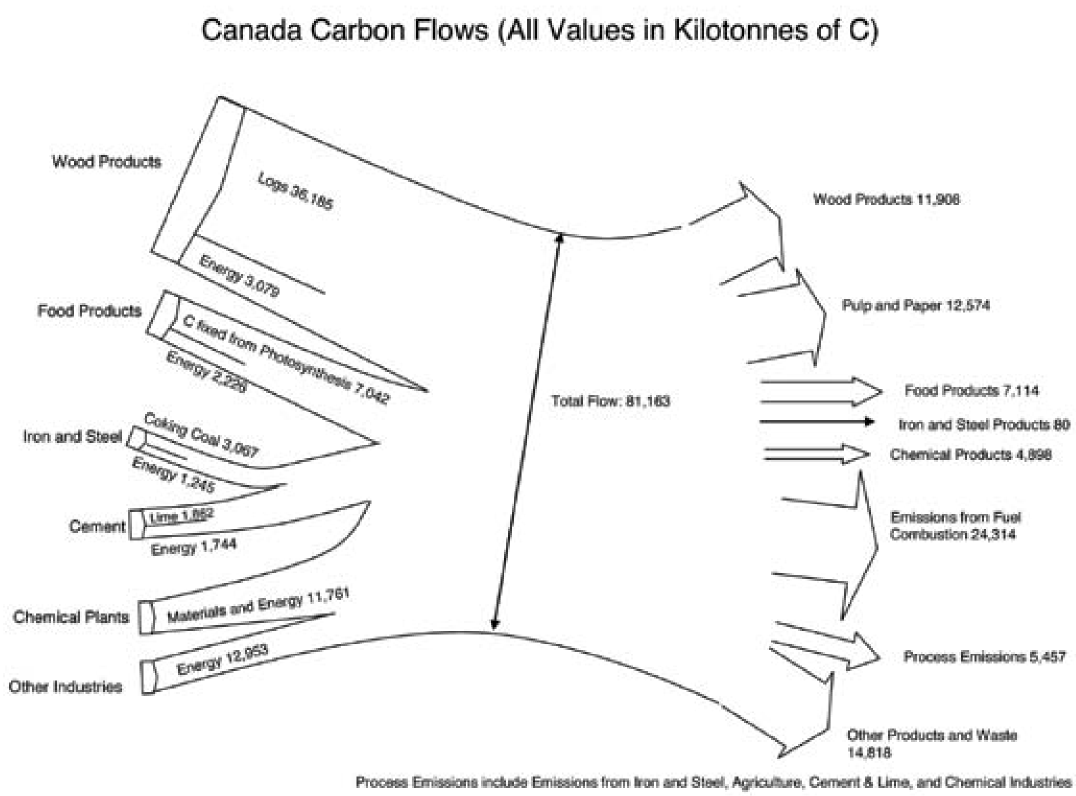

Figure C.1 Carbon flows, Canada. *Source*: Energy data from Statistics Canada Industrial Consumption of Energy survey, conversion coefficients and process emissions from Environment Canada, Canada GHG Inventory (2002). Production data from Statistics Canada, CANSIM Table 002-0010, Tables 303-0010, -0014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

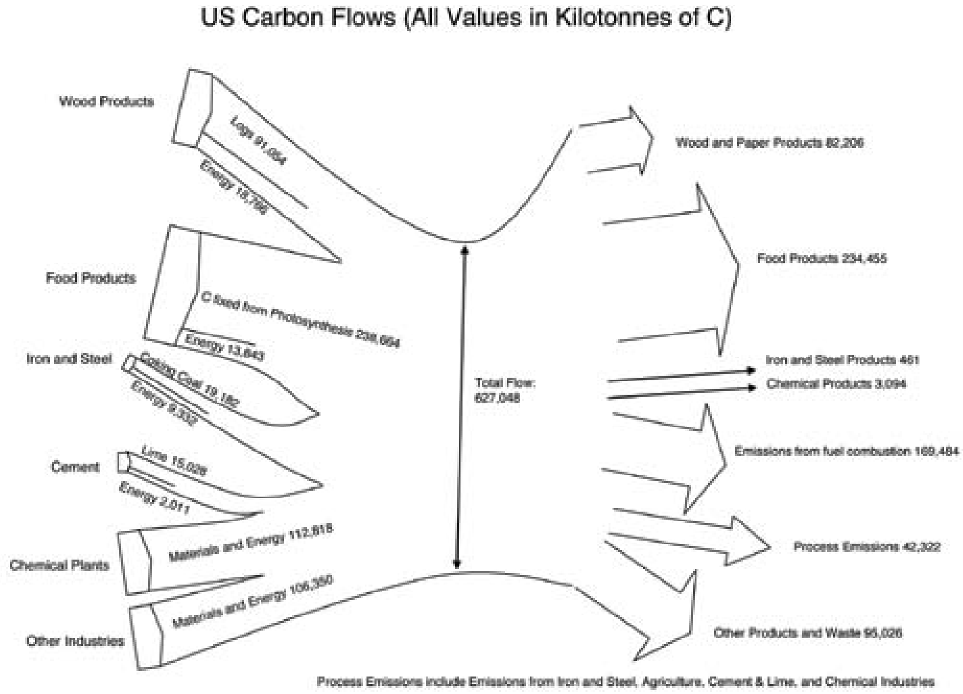

Figure C.2 Carbon flows, United States. *Source:* Energy data from IEA Oil Information (2004), IEA Coal Information (2005), IEA Natural Gas Information (2004). Process emissions: EPA, U.S. Emissions Inventory. Production of forestry products: USDA Database; FO-2471000 and -2472010, U.S. Timber Production,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

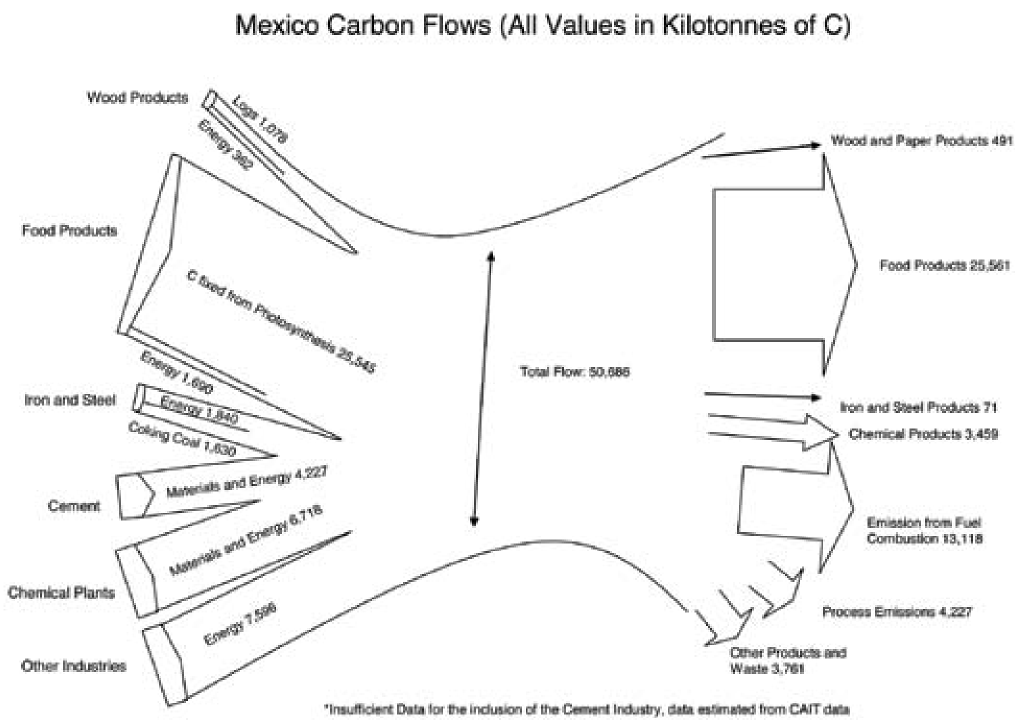

Figure C.3 Carbon flows, Mexico. *Source:* Energy data from IEA Oil Information (2004), IEA Coal Information (2005), IEA Natural Gas Information (2004). Process emissions: EPA, U.S. Emissions Inventory. Production of forestry products: USDA Database; FO-2471000, -2472010, -2482000, -2483040, -6342000,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

APPENDIX D

Ecosystem Carbon Fluxes

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The recent history of disturbance largely determines whether a forest system will be a net source or sink of carbon. For example, net ecosystem productivity (NEP, see Table D.1 for a list of definitions and acronyms used in this appendix) is being measured across a range of forest types in Canada using the eddy covariance technique. In mature forests, values range from -19.6 tons of carbon per hectare (t C per ha) per year in a white pine plantation in southern Ontario (Arain and Restrepo-Coupe, 2005) to -3.2 t C per ha per year in a jack pine forest (Amiro *et al.*, 2005; Griffis *et al.*, 2003). In recently disturbed forests, NEP ranges from +58.0 t C per ha per year in a harvested Douglas-fir forest (Humphreys *et al.*, 2005) to +5.7 t C per ha per year in a seven 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).

Table D.1 Ecosystem Productivity Terms and Definitions. (Terms and definitions apply to Appendices D and E of this report.)

Term	Acronym	Definition
Net Primary Production	NPP	Net uptake of carbon by plants in excess of respiratory loss
Heterotrophic Respiration	R_h	Respiratory loss by above- and below-ground heterotrophs (herbivores, decomposers, *etc.*)
Net Ecosystem Production	NEP	Net carbon accumulation within the ecosystem after all gains and losses are accounted for, typically measured using ground-based techniques. By convention, positive values of NEP represent accumulaitons of carbon by the ecosystem, and negative values represent carbon loss.
Net Ecosystem Exchange	NEE	The net flux of carbon between the land and the atmosphere, typically measured using eddy covariance techniques. Note: NEE and NEP are equivalent terms but are not always identical because of measurement and scaling issues, and the sign conventions are reversed. Positive values of NEE (net ecosystem exchange with the atmoshpere) usually refer to carbon released to the atmosphere (*i.e.*, a source), and negative values refer to carbon uptake (*i.e.*, a sink).

Sources: Randerson *et al.* (2002); Chapin *et al.* (2006).

Table D.2 Comparison of net ecosystem exchange (NEE) for different types and ages of temperate forests. Negative NEE means the forest is a sink for atmospheric CO_2. Eighty-one site years of data are from multiple published papers from each of the AmeriFlux network sites, and a network synthesis paper (Law *et al.*, 2002). NEE was averaged by site, then the mean was determined by forest type and age class. SD is standard deviation among sites in the forest type and age class.

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

In the United States, extensive land-based measurements of forest/atmosphere carbon exchange reveal patterns and causes of sink or source strength (Table D.2). Results show that net ecosystem exchange (NEE) of carbon in temperate forests ranges from a source of +12.7 t C per ha per year to a sink of -5.9 t C per ha per year. Forests identified as sources are primarily forests in the earliest stages of regeneration (up to about eight years) following stand-replacing disturbances such as wildfire and logging (Law *et al.*, 2002). Mature temperate deciduous broadleaf forests and mature evergreen coniferous forests were an average sink of -2.7 and -2.5 t C per ha per year, respectively (12 sites, 54 site-years of data). Values ranged from a source of +0.3 for a mixed deciduous and evergreen forest to a sink of -5.8 for an aggrading deciduous forest, averaged over multiple years. Young temperate evergreen coniferous forests (8 to 20 years) ranged from a sink of -0.6 to -5.9 t C per ha per year (mean -3.1). These forests are still rapidly growing and have not reached the capacity for carbon uptake.

Mature forests can have substantial stocks of sequestered carbon. Disturbances that damage or replace forests can result in the land being a net source of carbon dioxide (CO_2) for a few years in mild climates to 10-20 years in harsh climates while the forests are recovering (Law *et al.*, 2002; Clark *et al.*, 2004). Thus, the range of observed annual NEE of CO_2 ranges from a source of about +13 t C per ha per year in a clearcut forest to a net sink of -6 t C per ha in mature temperate forests.

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

APPENDIX E

Principles of Forest Management for Enhancing Carbon Sequestration

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The net rate of carbon accumulation has been generally understood (Woodwell and Whittaker, 1968) as the difference between gross primary production (gains) and respiration (losses), although this neglects important processes such as leaching of dissolved organic compounds (DOCs), emission of methane (CH_4), fire, harvests, or erosion that may contribute substantially to carbon loss and gain in forest ecosystems (Schulze *et al.*, 1999; Harmon, 2001; Chapin *et al.*, 2006). The net ecosystem carbon balance (NECB) in forests is, therefore, defined as net ecosystem production, or NEP, plus the non-physiological horizontal and vertical transfers into and out of the forest stand.

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

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

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

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

- Management activities that move live carbon to dead pools (such as coarse woody debris [CWD] or soil carbon) over short periods of time will often dramatically enhance decomposition (R_h), although considerable carbon can be stored in decomposing pools (Harmon and Marks, 2002). Regimes seeking to reduce the decomposition-related flows from residue following harvest may enhance overall sink capacity of these forests if these materials are used for energy generation or placed into forest products that last longer than the residue.
- Despite the importance of decomposition rates to the overall stand-level forest carbon balance, management of CWD pools is mostly impacted by recruitment of new CWD rather than by changing decomposition rates (Janisch and Harmon, 2002; Pregitzer and Euskirchen, 2004). Decreasing the interval between harvests can significantly decrease the store in this pool.
- Live coarse root biomass accounts for approximately

20-25% of aboveground forest biomass (Jenkins *et al.*, 2003), and there is additional biomass in fine roots. Following harvest, this pool of live root biomass is transferred to the dead biomass pool, which can form a significant carbon store. Note that roots of various size classes and existing under varying environmental conditions decompose at different rates.

- Some carbon can be sequestered in wood products from harvested wood, though, due to manufacturing losses, only about 60% of the carbon harvested is stored in products (Harmon, 1996). Clearly, longer-lived products will sequester carbon for longer periods of time.
- According to international convention, the replacement of fossil fuel by biomass fuel can be counted as an emissions offset if the wood is produced from sustainably managed forests (Schoene and Netto, 2005)

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

- Practices aimed at increasing NPP: fertilization; genetically improved trees that grow faster (Peterson *et al.*, 1999); any management activity that enhances growth rate without causing a concomitant increase in decomposition (Stanturf *et al.*, 2003; Stainback and Alavalapati, 2005).
- Practices aimed at reducing R_h (*i.e.*, minimizing the time forests are a source to the atmosphere following disturbance): low impact harvesting (that does not promote soil respiration); utilization of logging residues (biomass energy and fuels); incorporation of logging residue into soil during site prep (but note that this could also speed up decomposition); thinning to capture mortality; fertilization.

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

APPENDIX F

Wetlands - Supplemental Material

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F.1 INVENTORIES

F.1.1 Current Wetland Area and Rates of Loss

The ability to estimate soil carbon pools and fluxes in North American wetlands is constrained by the national inventories (or lack thereof) for Canada, the United States, and Mexico (Davidson *et al.*, 1999). The National Wetland Inventory (NWI) program of the United States has repeatedly sampled several thousand wetland sites using aerial photographs and more limited field verification. The data are summarized in a series of reports detailing changes in wetland area in the conterminous United States for the periods of the mid-1950s to mid-1970s (Frayer *et al.*, 1983), mid-1970s to mid-1980s (Dahl and Johnson, 1991), and 1986 to 1997 (Dahl, 2000). We used these relatively high-quality data sets extensively for estimating wetland area and loss rates in the conterminous United States, including mud flats. However, the usefulness of the NWI inventory reports for carbon budgeting is limited by the level of classification used to define wetland categories within the Cowardin *et al.* (1979) wetland classification system. At the level used in the national status and trend reports, vegetated freshwater wetlands are classified by dominant physiognomic vegetation type, and it is impossible to make the important distinction between wetlands with deep organic soils (*i.e.*, peatlands) and wetlands with mineral soils. The data are not at an adequate spatial resolution to combine with U.S. Department of Agriculture (USDA) Natural 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 original peatland area (Bridgham *et al.*, 2000) and combined these data with regional estimates of loss (Armentano and Menges, 1986) to estimate current peatland area in the conterminous United States. We calculated the current area of freshwater mineral-soil (FWMS) wetlands in the conterminous United States by subtracting peatland area from total wetland area (Dahl, 2000). This approach was limited by the Armentano and Menges peatland area data being current only up to the early 1980s, although large losses of peatlands since then are unlikely due to the institution of wetland protection laws.

We used a similar approach for Alaskan peatlands: peatland area was determined by the NRCS soil inventory (N. Bliss, query of the NRCS State Soil Geographic [STATSGO] database, February 2006) and overall wetland inventory was determined by standard NWI methods (Hall *et al.*, 1994). However, our peatland estimate of 132,000 km^2 (Table F.1) is 22% of the often cited value by Kivinen and Pakarinen (1981) of 596,000 km^2.

Kivinen and Pakarinen also used NRCS soils data (Rieger *et al.*, 1979) for their peatland estimates, but they defined a peatland as having a minimum organic layer thickness of 30 cm, whereas the current United States and Canadian soil taxonomies require a 40-cm thickness. The original 1979 Alaska soil inventory has been reclassified with current United States soil taxonomy (J. Moore, Alaska State Soil Scientist, personal comm.). Using the reclassified soil inventory, Alaska has 417,000 km^2 of wetlands with a histic modifier (*i.e.*, a surface organic layer between 20 and 60 cm thick) that are not Histosols or Histels, indicating significant carbon accumulation in the surface horizons of FWMS wetlands. Thus, we conclude that Kivinen and Pakarinen's Alaska peatland area estimate is higher because many Alaskan wetlands have a thin organic horizon that is not deep enough to qualify as a peatland under current soil taxonomy. Our smaller peatland area significantly lowers our estimate of carbon pools and fluxes in Alaskan peatlands compared to earlier studies (see *Carbon Pools* below).

The area of salt marsh in the conterminous United States, Canada, and Alaska were taken from Mendels-

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

	Permafrost peatlands	Non-permafrost peatlands	Mineral-soil freshwater	Salt marsh	Mangrove	Mudflat	Total
Canada							
Current	422[a]	714[a]	159[b]	0.4[c]	0	6[d]	1301
Historical	424[e]	726[f]	359[g]	1.3[b]	0	7[h]	1517
Alaska							
Current	89[i]	43[i]	556[j]	1.4[c]	0	7[k]	696
Historical	89	43	556	1.4	0	7	696
Conterminous United States							
Current	0	93[l]	312[m]	20[c]	3[c]	2[n]	431
Historical	0	111[i]	762[o]	22[n]	4[n]	3[n]	901
Mexico							
Current	0	10[p]	21[p]	0	5[c]	ND[q]	36
Historical	0	45[p]		0	8[h]	ND	53
North America							
Current	511	861	1,047	22	8	15	2,463
Historical	513	894[r]	1,706[r]	25	12	17	3,167
Global							
Current	3,443[s]		2,315[t]	22[u]	181[v]	ND	5,961
Historical	4,000[w]		5,000[x]	29[y]	278[y]	ND	9,307

[a] Tarnocai *et al.* (2005).

[b] National Wetlands Working Group (1988).

[c] Brackish and salt marsh areas from Mendelssohn and McKee (2000); freshwater tidal wetlands for the conterminous United States only from Odum *et al.* (1984) and Field *et al.* (1991).

[d] Estimated from the area of Canadian salt marshes and the ratio of mudflat to salt marsh area reported by Hanson and Calkins (1996).

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

[f] 9000 km^2 lost to reservoir flooding (Rubec, 1996), 250 km^2 to forestry drainage (Rubec, 1996), 124 km^2 to peat harvesting for horticulture (Cleary *et al*., 2005), and 16 km^2 to oil sands mining (Turetsky *et al*., 2002). See note e for permafrost melting estimate.

[g] Rubec (1996).

[h] Estimated loss rate for the Americas from Valiela *et al.* (2001) for approximately 1980 to 1990.

[i] Historical area from NRCS soil inventory (Bridgham *et al*., 2000), except Alaska inventory updated by N. Bliss from a February 2006 query of the STATSGO database. Less than 1% wetland losses have occurred in Alaska (Dahl, 1990).

[j] Total freshwater wetland area from Hall *et al.* (1994) minus peatland area.

[k] Hall *et al.* (1994).

[l] Historical area from Bridgham *et al.* (2000) minus losses in Armentano and Menges (1986).

[m] Overall freshwater wetland area from Dahl (2000) minus peatland area.

[n] Dahl (2000). Historical area estimates are only from the 1950s.

[o] Total historical wetland area from Dahl (1990) minus historical peatland area minus historical estuarine area.

[p] Spiers (1999) and Davidson (1999).

[q] ND indicates that no data are available.

[r] Assuming that historical proportion of peatlands to total wetlands in Mexico was the same as today.

[s] Bridgham *et al.* (2000) for the United States, Tarnocai *et al.* (2005) for Canada, Joosten, and Clarke (2002) for the rest of world. Recent range in literature 2,974,000–3,985,000 km^2 (Matthews and Fung, 1987; Aselmann and Crutzen, 1989; Maltby and Immirzi, 1993; Bridgham *et al*., 2000; Joosten and Clarke, 2002).

[t] Average of 2,289,000 km^2 from Matthews and Fung (1987) and 2,341,000 km^2 Aselmann and Crutzen (1989).

[u] Chmura *et al.* (2003). Underestimated because no inventories were available for the continents Asia, South America, and Australia which are mangrove-dominated but also support salt marsh.

[v] Spalding (1997).

[w] Range from 3,880 to 4,086 in Maltby and Immirzi (1993).

[x] Approximately 50% loss from Moser *et al.* (1996).

[y] Assumed a 25% loss rate outside North America for tidal marshes; a loss rate of 35% was used for mangroves (Valiela *et al.,* 2001).

sohn and McKee (2000). Because these estimates include brackish tidal marshes, they cannot be compared directly to the area of Canadian salt marshes. Compilations of freshwater, tidal wetland area are difficult to find, but there is approximately 1,640 km^2 on the east coast of the United States (Odum *et al*., 1984) and 470 km^2 on the United States' Gulf Coast (Field *et al*., 1991). Although some freshwater tidal wetlands are forested, this total was added to the tidal marsh area for the conterminous United States. Mangrove area was taken from Mendelssohn and McKee (2000), and is similar to an estimate by Lugo and Snedaker (1974).

The original area of tidal wetlands in the conterminous United States was based on the NWI (Dahl, 2000), which we considered to be the most defensible estimate available. However, "original" here only refers to the 1950s, when the first national wetland inventory was conducted in the conterminous United States to provide a historic baseline area. It is almost certain that the actual loss of tidal wetland area in the conterminous United States over a longer time frame was larger than the 7.7% figure used in our calculation. Valiela *et al.* (2001) estimated a loss of 31% of mangrove area in the United States from 1958 to 1982, but acknowledged a high level of uncertainty in this figure. We assumed that the original area of Alaskan tidal wetlands was similar to the current area because there has been relatively little development pressure in Alaska. To estimate loss of global tidal wetlands, we arbitrarily used a figure of 35% loss for tidal marshes outside of the United States and Mexico.

A regular national inventory of Canada's wetlands has not been undertaken, although wetland area has been mapped by ecoregion (National Wetlands Working Group, 1988). Extensive recent effort has gone into mapping Canadian peatlands (Tarnocai, 1998; Tarnocai *et al*., 2005). We calculated the current area of mineral-soil wetlands as the difference between total wetland area and peatland area in National Wetland Working Group (1988). The original area of FWMS wetland area was obtained from Rubec (1996). Canadian salt marsh estimates were taken from a compilation by Mendelssohn and McKee (2000). The compilation does not include brackish or freshwater tidal marshes, and we were unable to locate other estimates of Canadian brackish marsh area. The original area of salt marshes was estimated from the National Wetland Working Group (1988), but it is highly uncertain. There are no reliable country-wide estimates of mud flat area for Canada, but a highly uncertain extrapolation from a limited number of regional estimates was possible based upon the ratio of mudflat to salt marsh area reported by Hanson and Calkins (1996).

No national wetland inventories have been done for Mexico. Current freshwater wetland estimates for Mexico were taken from Davidson *et al.* (1999) and Spiers (1999), who used inventories of discrete wetland regions performed by a variety of organizations. Thus, freshwater wetland area estimates for Mexico are highly unreliable and are possibly a large underestimate. For mangrove area in Mexico, we used the estimates compiled by Mendelssohn and McKee (2000), which are similar to estimates reported in Davidson *et al.* (1999) and Spalding *et al.* (1997). We could find no estimates of tidal marsh or mud flat area for Mexico. Since most vegetated Mexican tidal wetlands are dominated by mangroves (Olmsted, 1993; Mendelssohn and McKee, 2000), the omission of Mexican tidal marshes should not significantly affect our carbon budget. However, there may be large areas of mud flat that would significantly increase our estimate of carbon pools and sequestration in this country. We used the Valiela *et al.* (2001) estimate of 38% for mangrove loss in the Americas, which roughly covers the period 1980 to 1990. This is less than the rough worldwide estimate of 50% wetland loss since the 1880s that is often cited (see Zedler and Kercher, 2005) and is probably conservative. A global loss rate of 35% was used for mangrove area globally based on the analysis of Valiela *et al.* (2001).

F.2 CARBON POOLS

F.2.1 Freshwater Mineral-Soil (Gleysol) Carbon Pools

Gleysol is a soil classification used by the Food and Agriculture Organization (FAO) and many countries that denotes mineral soils formed under waterlogged conditions (FAO-UNESCO, 1974). Tarnocai (1998) reported a soil carbon density of 200 Mg C per hectare (ha) for Canadian Gleysols to 1-m depth. Batjes (1996) determined soil carbon content globally from the Soil Map of the World (FAO, 1991) and a large database of soil pedons. He estimated an average value for soil carbon density of 199 Mg C per ha (CV[1] = 212%, n = 14 pedons) for Gleysols of the world to 2-m depth; to 1-m depth, he reported a soil carbon density of 131 Mg C per ha (CV = 109%, n =142 pedons).

Gleysols are not part of the United States' soil taxonomy scheme, and mineral soils with attributes reflecting waterlogged conditions are distributed among numerous soil groups. We queried the NRCS State Soil Geographic (STATSGO) soils database for soil carbon density in "wet" mineral soils of the conterminous United States (all soils that had a surface texture described as peat, muck, or mucky peat, or appeared on the 1993 list of hydric soils, which were not classified as Histosols) (N. Bliss, query of NRCS STATSGO database, December 2005). We used the average soil carbon densities of 162 Mg C per ha from this query for FWMS wetlands in the conterminous United States and Mexico.

[1] CV is the "coefficient of variation," or 100 times the standard deviation divided by the mean.

Table F.2 Soil carbon pools (Gt) and fluxes (Mt per year) of wetlands in North America and the world. "Sequestration in current wetlands" refers to carbon sequestration in extant wetlands; "oxidation in former wetlands" refers to emissions from wetlands that have been converted to non-wetland uses or conversion among wetland types due to human influence; "historical loss in sequestration capacity" refers to the loss in the carbon sequestration function of wetlands that have been converted to non-wetland uses; "change in flux from wetland conversions" is the sum of the two previous fluxes. Positive flux numbers indicate a net flux into the atmosphere, whereas negative numbers indicate a net flux into the ecosystem.

	Permafrost peatlands	Non-permafrost peatlands	Mineral-soil freshwater	Salt marsh	Mangrove	Mudflat	Total
Canada							
Pool Size in Current Wetlands	47.4[a]	102.9[b]	4.6[a]	0.0[c]	0.0[c]	0.1[d]	155.0
Sequestration in Current Wetlands	-5.5[e]	-13.6[e]	-2.7[f]	-0.1	0.0[c]	-1.2[d]	-23.0
Oxidation in Former Wetlands	0.2[g]		0.0[h]	0.0[i]	0.0	0.0	0.2
Historical Loss in Sequestration Capacity	0.0[e]	0.2[e]	3.4[f]	0.2	0.0	0.3	4.2
Change in Flux From Wetland Conversions	0.4		3.4	0.2	0.0	0.3	4.3
Alaska							
Pool Size in Current Wetlands	9.3[j]	6.2[j]	26.0[k]	0.0	0.0	0.1	41.7
Sequestration in Current Wetlands	-1.2[e]	-0.8[e]	-9.4[f]	-0.3	0.0	-1.6	-13.3
Oxidation in Former Wetlands	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Historical Loss in Sequestration Capacity	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Change in Flux From Wetland Conversions	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Conterminous United States							
Pool Size in Current Wetlands	0	14.0[l]	5.1[k]	0.4	0.1	0.0	19.6
Sequestration in Current Wetlands	0	-6.6[m]	-5.3[f]	-4.4	-0.5	-0.5	-17.3
Oxidation in Former Wetlands	0	18.0[n]	0.0[h]	0.0	0.0	0.0	18.0
Historical Loss in Sequestration Capacity	0	1.2[m]	7.6[f]	0.4	0.0	0.1	9.4
Change in Flux from Wetland Conversions	0	19.2	7.6	0.4	0.0	0.1	27.4
Mexico							
Pool Size in Current Wetlands	0	1.5[l]	0.3[k]	0.0	0.1	ND	1.9
Sequestration in Current Wetlands	0	-1.6[o]	-0.4[f]	0.0	-1.6	ND	-3.6
Oxidation in Former Wetlands	0	ND	ND	0.0	0.0	0.0	ND
Historical Loss in Sequestration Capacity	0	ND	ND	0.0	1.0	ND	ND
Change in Flux from Wetland Conversions	0	ND	ND	0.0	1.0	ND	ND
North America							
Pool Size in Current Wetlands	56.7	124.6	36.0	0.4	0.2	0.3	218.2
Sequestration in Current Wetlands	-6.6	-22.6	-17.7	-4.8	-2.1	-3.3	-57.2
Oxidation in Former Wetlands	18.2		0.0	0.0	0.0	0.0	18.2
Historical Loss in Sequestration Capacity	0	1.4	11.0	0.5	1.0	0.5	14.5
Change in Flux from Wetland Conversions	19.6		11.0	0.5	1.0	0.5	32.7
Global							
Pool Size in Current Wetlands	462[p]		46[q]	0.4[r]	4.9[r]	ND	513
Sequestration in Current Wetlands	-55[s]		-39[f]	-4.6[r]	-38.0[r]	ND	-137
Oxidation in Former Wetlands	205[t]		ND	0	0	0	205
Historical Loss in Sequestration Capacity	16[t]		45[f]	0.7[u]	20[v]	ND	82
Change in Flux From Wetland Conversions	221[t]		> 45	0.7	20	ND	287

[a] Tarnocai (1998); mineral soil to 1-m depth.
[b] Tarnocai *et al.* (2005).
[c] Rates and pools calculated from Chmura *et al.* (2003) using country-specific data (sedimentation accumulation rates in Mg C per ha per year: Mexican mangroves = 3.3, conterminous United States mangroves = 1.8, conterminous United States tidal marshes = 2.2, tidal marshes in Canada and Alaska = 2.1); areas from Table 13F.1.
[d] Assumed the same carbon density and accumulation rates as the adjacent vegetated wetland ecosystem (mangrove data for Mexico and salt marsh data elsewhere).
[e] Assumed carbon accumulation rate of 0.13 Mg C per ha per year for permafrost peatlands and 0.19 Mg C per ha per year for non-permafrost peatlands. Reported range of long-term apparent accumulation rates from 0.05-0.35 (Ovenden, 1990; Maltby and Immirzi, 1993; Trumbore and Harden, 1997; Vitt *et al*., 2000; Turunen *et al*., 2004).
[f] Rate calculated as the geometric mean sediment accumulation rate of 2.2 Mg sediment per ha per year (range 0-80) from Johnston (1991) and Craft and Casey (2000) times 7.7 % C (CV = 109) (Batjes 1996).
[g] Sum of 0.24 Mt C per year from horticulture removal of peat (Cleary *et al*., 2005) and 0.10 Mt C per year from increased peat sequestration due to permafrost melting (Turetsky *et al*., 2002).
[h] Assumed that the net oxidation of 8.6% of the soil carbon pool (Euliss *et al*., 2006) over 50 years after conversion to non-wetland use.
[i] Assumed that conversion of tidal systems is caused by fill and results in burial and preservation of SOM, Sedimentary Organic Matter, rather than oxidation.
[j] Soil carbon densities of 1,441 Mg C per ha for Histosols and 1,048 Mg C per ha for Histels (Tarnocai *et al*., 2005).
[k] Soil carbon density of 162 Mg C per ha for the conterminous United States and Mexico and 468 Mg C per ha for Alaska based upon NRCS STATSGO database and soil pedon information.
[l] Assumed soil carbon density of 1,500 Mg C per ha.
[m] Webb and Webb (1988).
[n] Estimated loss rate as of early 1980s (Armentano and Menges, 1986). Overall, wetlands losses in the United States have declined dramatically since then (Dahl, 2000) and probably even more so for Histosols, so this number may still be representative.
[o] Using peat accumulation rate of 1.6 Mg C per ha (range 1.0–2.25) (Maltby and Immirzi, 1993).
[p] From Maltby and Immirzi (1993). Range of 234 to 679 GtC (Gorham, 1991; Maltby and Immirzi, 1993; Eswaran *et al*., 1995; Batjes, 1996; Lappalainen, 1996; Joosten and Clarke, 2002).
[q] Soil carbon density of 199 Mg C per ha (Batjes, 1996).
[r] Chmura *et al.* (2003).
[s] Joosten and Clarke (2002) reported range of -40 to -70 Mt C per year. Using the peatland estimate in Table F.1 and a C accumulation rate of 0.19 Mg C per ha per year, we calculate a global flux of -65 Mt C per year in peatlands.
[t] Current oxidative flux is the difference between the change in flux and the historical loss in sequestration capacity from this table. The change in flux is from Maltby and Immirzi (1993) (reported range 176 to 266 Mt C per year) and the historical loss in sequestration capacity is from this table for North America, from Armentano and Menges (1986) for other northern peatlands, and from Maltby and Immirzi (1993) for tropical peatlands.
[u] Assumed that global rates approximate the North America rate because most salt marshes inventoried are in North America.
[v] Assumed 25% loss globally since the late 1800s.
ND indicates that no data are available.

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

For Alaska, many soil investigations have been conducted since the STATSGO soil data was coded. We updated STATSGO by calculating soil carbon densities from data obtained from the NRCS on 479 pedons collected in Alaska, and then we used this data for both FWMS wetlands and peatlands. For some of the Histosols, missing bulk densities were calculated using averages of measured bulk densities for the closest matching class in the USDA Soil Taxonomy (NRCS, 1999). A matching procedure was developed for relating sets of pedons to sets of STATSGO components. If there were multiple components for each map unit in STATSGO, the percentage of the component was used to scale area and carbon data. We compared matching sets of pedons to sets of components at the four top levels of the United States' soil taxonomy: Orders, Suborders, Great Groups, and Subgroups. For example, the soil carbon for all pedons having the same soil order were averaged, and the carbon content was applied to all of the soil components of the same order (*e.g.*, Histosol pedons are used to characterize Histosol components). At the Order level, all components were matched with pedon data. At the suborder level, pedon data were not available to match approximately 20,000 km^2 (compared to the nearly 1,500,000-km^2 area of soil in the state), but the soil characteristics were more closely associated with the appropriate land areas than at the Order level. At the Great Group and Subgroup levels, pedon data were unavailable for much larger areas, even though the quality of the data when available became better. For this study, we used the Suborder-level matching. The resulting soil carbon density for Alaskan FWMS wetlands was 469 Mg C per ha, reflecting large areas of wetlands with a histic epipedon as noted above.

F.2.2 Peatland Soil Carbon Pools

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

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

The peatlands of the conterminous United States are different in texture, and probably depth, from those in Canada and Alaska, so it is probably inappropriate to use the soil carbon densities for Canadian peatlands for those in the conterminous United States. For example, we compared the relative percentage of the Histosol suborders (excluding the small area of Folists, as they are predominantly upland soils) for Canada (Tarnocai, 1998), Alaska (updated STATSGO data, J. Moore, personal comm.), and the conterminous United States (NRCS, 1999). The relative percentage of Fibrists, Hemists, and Saprists, respectively, in Canada are 37%, 62%, and 1%, in Alaska are 53%, 27%, and 20%, and in the conterminous United States are 1%, 19%, and 80%. Using the STATSGO database (N. Bliss, query of NRCS STATSGO database, December 2005), the average soil carbon density for Histosols in the conterminous United States is 1,089 Mg C per ha, but this is an underestimate as many peatlands were not sampled to their maximum depth. Armentano and Menges (1986) reported average carbon density of conterminous United States' peatlands to 1-m depth of 1,147 to 1,125 Mg C per ha. Malterer (1996) gave soil carbon densities of conterminous United States' peatlands of 2,902 Mg C per ha for Fibrist, 1,874 Mg C per ha for Hemists, and 2,740 Mg C per ha for Saprists, but it is unclear how he derived these estimates. Batjes (1996) and Eswaran *et al.* (1995) gave average soil carbon densities to 1-m depth for global peatlands of 776 and 2,235 Mg C per ha, respectively. We chose to use an average carbon density of 1,500 Mg C per ha, which is in the middle of the reported range, for peatlands in the conterminous United States and Mexico.

F.2.3 Estuarine Soil Carbon Pools

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

F.2.4 Plant Carbon Pools

While extensive data on plant biomass in individual wetlands have been published, no systematic inventory of wetland plant biomass has been undertaken in North America. Nationally, the forest carbon biomass pool (including above-ground and below-ground biomass) has been estimated to be 54.9 Mg C per ha (Birdsey, 1992), which we used for forested wetlands in the United States and Canada. This approach assumes that wetland forests do not have substantially different biomass carbon densities from upland forests. There is one

Table F.3 Plant carbon pools (Gt) and fluxes (Mt per year) of wetlands in North America and the world. Positive flux numbers indicate a net flux into the atmosphere, whereas negative numbers indicate a net flux into the ecosystem.

	Permafrost peatlands	Non-permafrost peatlands	Mineral-soil freshwater	Salt marsh	Mangrove	Total
Canada						
Pool Size in Current Wetlands	1.4[a]		0.3[b]	0.0[c]	0.0	1.7
Sequestration in Current Wetlands	0.0	ND		0.0	0.0	0.0
Alaska						
Pool Size in Current Wetlands	0.4[a]		1.1[d]	0.0	0.0	1.5
Sequestration in Current Wetlands	0.0	0.0	0.0	0.0	0.0	0.0
Conterminous United States						
Pool Size in Current Wetlands	0.0	1.5[d]		0.0	0.0	1.5
Sequestration in Current Wetlands	0.0	-10.3[e]		0.0	0.0	-10.3
Mexico						
Pool Size in Current Wetlands	0.0	0.0[b]	0.0[b]	0.0	0.1	0.1
Sequestration in Current Wetlands	0.0	ND	ND	0.0	ND	0.0
North America						
Pool Size in Current Wetlands	4.8			0.0	0.1	4.9
Sequestration in Current Wetlands	0.0	-10.3		0.0	ND	-10.3
Global						
Pool Size in Current Wetlands	6.9[b]		4.6[b]	0.0[f]	4.0[g]	15.5
Sequestration in Current Wetlands	0.0	ND	ND	0.0	ND	ND

[a] Biomass for non-forested peatlands from Vitt *et al.* (2000), assuming 50% of biomass is below-ground. Forest biomass density from Birdsey (1992) and forested area from Tarnocai *et al.* (2005) for Canada and from Hall *et al.* (1994) for Alaska.

[b] Assumed 2000 g C per m^2 in above-ground and below-ground plant biomass (Gorham, 1991).

[c] Biomass data from Mitsch and Gosselink (1993).

[d] Biomass for non-forested wetlands from Gorham (1991). Forest biomass density from Birdsey (1992), and forested area from Hall *et al.* (1994) for Alaska and Dahl (2000) for the conterminous United States.

[e] 50 g C per m^2 per yr sequestration from forest growth from a southeastern United States regional assessment of wetland forest growth (Brown *et al.*, 2001).

[f] Assumed that global pools approximate those from North America because most salt marshes inventoried are in North America.

[g] Twilley *et al.* (1992).

ND indicates that no data are available.

regional assessment of forested wetlands in the southeastern United States, which comprise approximately 35% of the total forested wetland area in the conterminous United States. We utilized the southeastern United States regional inventory to evaluate this assumption; above-ground tree biomass averaged 125.2 m^3 per ha for softwood stands and 116.1 m^3 per ha for hardwood stands. Using an average wood density and carbon content, the carbon density for these forests would be 33 Mg C per ha for softwood stands and 42 Mg C per ha for hardwood stands. However, these estimates do not include understory vegetation, below-ground biomass, or dead trees, which account for 49% of the total forest biomass (Birdsey, 1992). Using that factor to make an adjustment for total forest biomass, the range would be 49 to 66 Mg C per ha for the softwood and hardwood stands, respectively. Accordingly, the assumption of using 54.9 Mg C per ha seems reasonable for a national-level estimate.

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

Since Tarnocai *et al.* (2005) divided Canadian peatland area into bog and fen, we used above-ground biomass for each community type from Vitt *et al.* (2000), and assumed that 50% of biomass is below-ground. We used the average bog and fen plant biomass from Vitt *et al.* (2000) for Alaskan peatlands. For other wetland areas, we used an average value of 20.0 Mg C per ha for non-forested wetland biomass carbon density (Gorham, 1991).

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

Plant biomass carbon data are presented in Table F.3.

F.3 CARBON FLUXES

F.3.1 Peatland Soil Carbon Accumulation Rates

Most studies report the long-term apparent rate of carbon accumulation (LORCA) in peatlands based upon basal peat dates, but this assumes a linear accumulation rate through time. However, due to the slow decay of the accumulated peat, the true rate of carbon accumulation will always be less than the LORCA (Clymo *et al.*, 1998), so most reported rates are inherently biased upwards. Tolonen and Turunen (1996) found that the true rate of peat accumulation was about 67% of the LORCA.

For estimates of soil carbon sequestration in conterminous United States' peatlands, we used the LORCA data from 82 sites and 215 cores throughout eastern North America (Webb and Webb III, 1988). They reported a median accumulation rate of 0.066 cm per year (mean = 0.092, sd = 0.085). We converted this value into a carbon accumulation rate of -0.71 Mg C per ha per year by assuming 58% C (see NRCS Soil Survey Laboratory Information Manual, available on-line at http://soils.usda.gov/survey/nscd/lim/), a bulk density of 0.28 g per cm^3, and an organic matter content of 69%. (Positive carbon fluxes indicate net fluxes to the atmosphere, whereas negative carbon fluxes indicate net fluxes into an ecosystem.) The bulk density and organic matter content were the area-weighted and depth-weighted average from all Histosol soil map units greater than 202.5 ha (n = 3,884) in the conterminous United States from the National Soil Information System (NASIS) data base provided by S. Campbell (USDA NRCS, Portland, Oreg.). For comparison, Armentano and Menges (1986) used soil carbon accumulation rates that ranged from -0.48 Mg C per ha per year in northern conterminous United States peatlands to -2.25 Mg C per ha per year in Florida peatlands.

Peatlands accumulate lesser amounts of soil carbon at higher latitudes, with especially low accumulation rates in permafrost peatlands (Ovenden, 1990; Robinson and Moore, 1999). The rates used in this report reflect this gradient, going from -0.13 to -0.19 to -0.71 Mg C per ha per year in permafrost peatlands, non-permafrost Canadian and Alaskan peatlands, and peatlands in the conterminous United States and Mexico, respectively (Table F.2).

F.3.2 Freshwater Mineral-Soil Wetland Carbon Accumulation Rates

Many studies have estimated sediment deposition rates in FWMS wetlands, with a geometric mean rate of 2.2 Mg sediment per ha per year (n = 26, arithmetic mean = 16.3, range 0 to 80.0) in a compilation by Johnston (1991), along with those reported more recently in Craft and Casey (2000). As can be seen by the difference between the geometric and arithmetic means, this dataset is log-normally distributed with several large outliers. Assuming 7.7% carbon for FWMS wetlands (Batjes, 1996), this gives a geometric mean accumulation rate of 0.17 Mg C per ha per year. Johnston (1991) and Craft and Casey (2000) reported more studies with only vertical

sediment accumulation rates, with a geometric mean of 0.23 cm per year (n = 34, arithmetic mean = 0.63 cm per year, range -0.6 to 2.6). If we assume a bulk density of 1.00 g per cm^3 for FWMS wetlands (Batjes, 1996; Smith *et al.*, 2001), this converts into an unrealistically large accumulation rate of 1.85 Mg C per ha per year.

We suggest that caution is necessary in interpretation of these data for a number of reasons. There is large variability in sedimentation rates among studies, and even within a site, sedimentation rates are highly variable depending on the local deposition environment (Johnston *et al.*, 2001). Researchers may have preferentially chosen wetlands with high sedimentation rates to study this process, providing a bias towards greater carbon sequestration. Rates of erosion and resultant deposition have substantially decreased during the last century in the conterminous United States (Craft and Casey, 2000; Trimble and Crosson, 2000). More fundamentally, it is important to distinguish between autochthonous carbon (derived from onsite plant production) and allochthonous carbon (imported from outside the wetland) in soil carbon storage. The soil carbon stored in peatlands is of autochthonous origin and represents sequestration of atmospheric carbon dioxide at the landscape scale. In contrast, a unknown portion of the soil carbon that is stored in FWMS wetlands is of allochthonous origin. However, conterminous United States' soils average between 0.9 and 1.3% soil carbon, which is much less than the average carbon content of FWMS wetlands (7.7%) (Batjes, 1996), suggesting a substantial autochthonous input to FWMS wetlands.

At a landscape scale, redistribution of sediments from uplands to wetlands represents net carbon sequestration only to the extent that the soil carbon is replaced in the terrestrial source area and/or decomposition rates are substantially lower in the receiving wetland (Stallard, 1998; Harden *et al.*, 1999). Agricultural lands are a major source of erosion (Meade *et al.*, 1990, as cited in Stallard, 1998), but it appears that, after large initial losses, soil carbon is relatively stable (Stallard, 1998; Smith *et al.*, 2001) or even increases (Harden *et al.*, 1999) under modern agricultural techniques. It is also generally assumed that sediment carbon deposited in anaerobic environments, such as occur in many wetlands, is relatively recalcitrant (Stallard, 1998; Smith *et al.*, 2001). For example, in a variety of Minnesota wetland soils, carbon mineralization was approximately six times slower anaerobically than aerobically (Bridgham *et al.*, 1998). However, time since initial deposition and organic quality of sediments appears to be an important constraint on its relative reactivity. Kristensen *et al.* (1995) found that relatively fresh, labile organic matter had similar decomposition rates aerobically and anaerobically, whereas "aged," recalcitrant organic matter decomposed ten times slower anaerobically. Gunnison *et al.* (1983) found that freshly flooded soils had twice as rapid carbon mineralization rates as sediments. In newly constructed reservoirs, sediments maintained these rapid mineralization rates even 6-10 years after initial flooding. Overall, these latter two studies suggest that there may be substantial carbon mineralization in freshly deposited allochthonous sediments in wetlands, but we feel that the data are not adequate to account for this effect quantitatively.

We use a landscape-level sediment sequestration rate of 0.17 Mg C per ha per year for FWMS wetlands in North America, while acknowledging the low level of confidence in this estimate. Johnston (1991) and Craft and Casey (2000) only gave sedimentation rates in FWMS wetlands in the conterminous United States. Since most FWMS wetlands in Canada are in more developed and agricultural regions, we felt that it was reasonable to use the sedimentation estimates from these studies. However, most Alaskan FWMS wetlands are relatively pristine, with little anthropogenic sediment input, but as described above, most have an extensive histic epipedon, so at least historically, they have actively accumulated soil carbon. Given that our soil carbon accumulation rate for Alaskan peatlands is 0.19 Mg C per ha per year, our sediment sequestration rate of 0.17 Mg C per ha per year for Alaskan FWMS wetlands does not seem unreasonable.

Table F.4 Methane fluxes (Mt per year) from wetlands in North America and the world.

	Permafrost peatlands	Non-perma-frost peatlands	Mineral-soil freshwater	Salt marsh	Mangrove	Mudflat	Total
Canada							
CH_4 Flux in Current Wetlands	1.1[a]	2.1[b]	1.2	0.0	0.0	0.0[c]	4.4
Historical change in CH_4 Flux	0.0	0.3	-1.5	0.0	0.0	0.0	-1.2
Alaska							
CH_4 Flux in Current Wetlands	0.2	0.1	1.4	0.0	0.0	0.0	1.7
Historical change in CH_4 Flux	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Conterminous United States							
CH_4 Flux in Current Wetlands	0.0	0.7	2.4	0.0	0.0	0.0	3.1
Historical change in CH_4 Flux	0.0	-0.1	-3.4	0.0	0.0	0.0	-3.5
Mexico							
CH_4 Flux in Current Wetlands	0.0	0.1	0.2	0.0	0.0	ND	0.2
Historical change in CH_4 Flux	0.0	-0.1		0.0	0.0	ND	-0.1
North America							
CH_4 Flux in Current Wetlands	1.3	3.0	5.1	0.0	0.0	0.0	9.4
Historical change in CH_4 Flux	0.0	-4.9		0.0	0.0	0.0	-4.9
Global							
CH_4 Flux in Current Wetlands	14.1[d]	22.5[d]	68.0[d]	0.0[e]	0.2	ND	105[f]
Historical change in CH_4 Flux	-3.6[g]		-79[g]	0.0[e]	-0.1	ND	-83

[a] Used CH_4 flux of 2.5 g per m^2 per yr (range 0 to 130, likely mean 2 to 3) (Moore and Roulet, 1995) for Canadian peatlands and all Alaskan freshwater wetlands. Used CH_4 flux of 7.6 g per m^2 per yr for Canadian freshwater mineral-soil wetlands and all United States and Mexican freshwater wetlands and 1.3 g per m^2 per yr for estuarine wetlands—from synthesis of published CH_4 fluxes for the United States (see Table F.5).

[b] Includes a 17-fold increase in CH_4 flux (Kelly *et al.*, 1997) in the 9000 km^2 of reservoirs that have been formed on peatlands (Rubec, 1996) and an estimated CH_4 flux of 15 g per m^2 per yr (Moore *et al.*, 1998) from 2,630 km^2 of melted permafrost peatlands (Vitt *et al.*, 1994).

[c] Assumed trace gas fluxes from unvegetated estuarine wetlands (*i.e.*, mudflats) was the same as adjacent wetlands.

[d] Bartlett and Harriss (1993).

[e] Assumed that global rates approximate the North America rate because most salt marsh area is in North America.

[f] Ehhalt *et al.* (2001), range of 92 to 237 Mt per yr.

[g] Using rates from Bartlett and Harriss (1993) and historical loss of area in Table 1.

ND indicates that no data are available.

F.3.3 Estuarine Carbon Accumulation Rates

Carbon accumulation in tidal wetlands was assumed to be entirely in the soil pool. This should provide a reasonable estimate because marshes are primarily herbaceous, and mangrove biomass should be in steady state unless the site was converted to another use. An important difference between soil carbon sequestration in tidal and non-tidal systems is that tidal sequestration occurs primarily through burial driven by sea level rise. For this reason, carbon accumulation rates can be estimated well with data on changes in soil surface elevation and carbon density. Rates of soil carbon accumulation were calculated from Chmura *et al.* (2003) as described above for the soil carbon pool (rates in Mg C per ha per year are 3.3 for Mexican mangroves; 1.8 and 2.2 for mangroves and tidal marshes, respectively, in the conterminous United States; 2.1 for tidal marshes in Canada and Alaska). These estimates are based on a variety of methods, such as ^{210}Pb dating and soil elevation tables, which integrate vertical soil accumulation rates over periods of time ranging from 1–100 years. The soil carbon sequestered in estuarine wetland sediments is likely to be a mixture of both allochthonous and autochthonous sources. However, without better information, we assumed that *in situ* rates of soil carbon sequestration in estuarine wetlands is representative of the true landscape-level rate.

F.3.4 Extractive Uses of Peat

Use of peat for energy production is, and always has been, negligible in North America, as opposed to other parts of the world (WEC, 2001). However, Canada produces a greater volume of horticultural and agricultural peat than any other country in the world (WEC, 2001). Currently, 124 km^2 of Canadian peatlands have been under extraction now or in the past (Cleary *et al*., 2005). A life-cycle analysis by these authors estimated that as of 1990 Canada emitted 0.2 Mt per year of CO_2-C equivalents through peat extraction. The United States' production of horticultural peat is about 19% of Canada's (Joosten and Clarke, 2002), which assuming a similar life-cycle as for Canada, suggests that the United States produces 0.05 Mt of CO_2-C equivalents through peat extraction.

F.3.5 Methane Fluxes

Moore and Roulet (1995) reported a range of methane fluxes from 0 to 130 g CH_4 per m^2 per year from 120 peatland sites in Canada, with the majority <10 g CH_4 per m^2 per year. They estimated a low average flux rate of 2 to 3 g CH_4 per m^2 per year, which equaled an emission of 2–3 Mt CH_4 per year from Canadian peatlands. We used an estimate of 2.5 g CH_4 per m^2 per year for Canadian peatlands and Alaskan freshwater wetlands (Table F.4).

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

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

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

Using the conversion factors for freshwater and estuarine wetlands, we estimated average annual fluxes from the average daily fluxes. The data were highly log-normally distributed, so we used geometric means. For freshwater wetlands, the geometric

mean estimated annual flux rate was 7.1 g CH_4 per m^2 per year (n = 74, 1 SE = 0.8, arithmetic mean = 38.6), which is very similar to the geometric mean measured rate of 8.1 g CH_4 per m^2 per year (n = 32, arithmetic mean = 32.1). For estuarine wetlands, the geometric mean estimated annual flux rate was 1.3 g CH_4 per m^2 per year (n = 25, 1 SE = 0.2, arithmetic mean = 9.8), which is smaller than the geometric mean measured rate of 5.0 g CH_4 per m^2 per year (n = 13, arithmetic mean = 16.9).

Finally, we combined both approaches. In cases where a paper presented an annual value, we used that number. In cases where only an average daily number was presented, we used that value corrected with the appropriate conversion factor. For conterminous United States wetlands, FWMS Canadian wetlands, and Mexican wetlands, we used a geometric mean flux of 7.6 g CH_4 per m^2 per year, and for estuarine wetlands, we used a geometric mean flux of 1.3 g CH_4 per m^2 per year.

F.3.6 Plant Carbon Fluxes

For ecosystems at approximately steady state, plant biomass should be reasonably constant on average because plant production is roughly balanced by mortality and subsequent decomposition. We assumed insignificant plant biomass accumulation in freshwater and estuarine marshes because they are dominated by herbaceous plants that do not accumulate carbon in wood. Sequestration in plants in relatively undisturbed forested wetlands in Alaska and many parts of Canada is probably small, although there may be substantial logging of Canadian forested wetlands for which we do not have data. Similarly, no data was available to evaluate the effect of harvesting of woody biomass in Mexican mangroves on carbon fluxes.

Tree biomass carbon sequestration averages -1.40 Mg C per ha per year in United States' forests across all forest types (Birdsey, 1992). Using the tree growth estimates from the southeastern United States regional assessment of wetland forests (Brown *et al.*, 2001) yields an even lower estimate of sequestration in above-ground tree biomass (approx. -0.50 Mg C per ha per year). We used this lower value and area estimates from Dahl (2000) to estimate that forested wetlands in the conterminous United States currently sequester -10.3 Mt C per year.

Table F.5 Methane fluxes measured in the conterminous United States. The conversion factor is the ratio of the daily average flux to the measured annual flux × 10[3]. The calculated annual flux was determined based upon the average conversion factor for freshwater (FW) and saltwater wetlands (SW). The measured annual flux was used if that was available; otherwise, the calculated annual flux was used.

Habitat	State	Method[a]	Salt/ Fresh	Daily Average Flux (mg CH_4 per m² per day)	Measured Annual Flux (g CH_4 per m² per year)	Conversion Factor	Estimated Annual Flux (g CH_4 per m² per year)	Used Annual Flux (g CH_4 per m² per year)	Reference
Fens	CO	C	FW		40.7			40.7	Chimner and Cooper (2003)
Wet Alpine Meadow	CO	C	FW	0.1			0.0	0.0	Neff *et al.* (1994)
Lake - Average	CO	C	FW	25.4			9.2	9.2	Smith and Lewis (1992)
Wetland - Average	CO	C	FW	28.3			10.3	10.3	Smith and Lewis (1992)
Nuphar Bed	CO	C	FW	202.1			73.6	73.6	Smith and Lewis (1992)
Tundra - Carex Meadow	CO	C	FW	2.8			1.0	1.0	West et al. (1999)
Tundra - Acomastylis Meadow	CO	C	FW	-0.5			-0.2	-0.2	West *et al.* (1999)
Tundra - Kobresia Meadow	CO	C	FW	-0.8			-0.3	-0.3	West *et al.* (1999)
Moist Grassy	CO	C	FW	6.1	1.9	0.32	2.2	1.9	Wickland *et al.* (1999)
Moist Mossy	CO	C	FW	1.5	0.5	0.33	0.5	0.5	Wickland *et al.* (1999)
Wetland	CO	C	FW		41.7			41.7	Wickland *et al.* (1999)
Hardwood Hammock	FL	C	FW	0.0			0.0	0.0	Bartlett *et al.* (1989)
Dwarf Cypress / Sawgrass	FL	C	FW	7.5			2.7	2.7	Bartlett *et al.* (1989)
Spikerush	FL	C	FW	29.4			10.7	10.7	Bartlett *et al.* (1989)
Sawgrass < 1m	FL	C	FW	38.8			14.1	14.1	Bartlett *et al.* (1989)
Sawgrass/Spikerush/ Periphyton	FL	C	FW	45.1			16.4	16.4	Bartlett *et al.* (1989)
Swamp Forest	FL	C	FW	68.9			25.1	25.1	Bartlett *et al.* (1989)
Sawgrass > 1m	FL	C	FW	71.9			26.2	26.2	Bartlett *et al.* (1989)
Sawgrass	FL	C	FW	107.0			38.9	38.9	Burke *et al.* (1988)
Pond Open Water	FL	C	FW	624.0			227.1	227.1	Burke *et al.* (1988)
Everglades - Cladium	FL	C	FW	45.4			16.5	16.5	Chanton *et al.* (1993)
Everglades - Typha	FL	C	FW	142.9			52.0	52.0	Chanton *et al.* (1993)
Wet Prairie (Marl)	FL	C	FW	87.0			31.6	31.6	Happell *et al.* (1993)
Wet Prairie (Marl)	FL	C	FW	27.4			10.0	10.0	Happell *et al.* (1993)
Marsh (Marl)	FL	C	FW	30.0			10.9	10.9	Happell *et al.* (1993)
Marsh (Marl)	FL	C	FW	49.6			18.0	18.0	Happell *et al.* (1993)
Marsh (Peat)	FL	C	FW	45.4			16.5	16.5	Happell *et al.* (1993)
Marsh (Peat)	FL	C	FW	13.0			4.7	4.7	Happell *et al.* (1993)
Marsh (Peat)	FL	C	FW	163.6			59.6	59.6	Happell *et al.* (1993)
Marsh (Peat)	FL	C	FW	20.4			7.4	7.4	Happell *et al.* (1993)
Wet Prairie / Sawgrass	FL	C	FW	61.0			22.2	22.2	Harriss *et al.* (1988)
Wetland Forest	FL	C	FW	59.0			21.5	21.5	Harriss *et al.* (1988)
Cypress Swamp - Flowing Water	FL	C	FW	67.0			24.4	24.4	Harriss and Sebacher (1981)
Open Water Swamp	FL	C	FW	480.0			174.7	174.7	Schipper and Reddy (1994)
Waterlily Slough	FL	C	FW	91.0			33.1	33.1	Schipper and Reddy (1994)

[a] C = chamber, T = tower, eddy covariance, E = ebulition measured separately.

Habitat	State	Method[a]	Salt/ Fresh	Daily Average Flux (mg CH_4 per m² per day)	Measured Annual Flux (g CH_4 per m² per year)	Conversion Factor	Estimated Annual Flux (g CH_4 per m² per year)	Used Annual Flux (g CH_4 per m² per year)	Reference
Cypress Swamp - Deep Water	GA	C	FW	92.3			33.6	33.6	Harriss and Sebacher (1981)
Bottomland Hard-woods/ Swamps	GA	C	FW		23.0			23.0	Pulliam (1993)
Swamp Forest	LA	C	FW	146.0			53.1	53.1	Alford *et al.* (1997)
Freshwater Marsh	LA	C	FW	251.0			91.4	91.4	Alford *et al.* (1997)
Fresh	LA	C	FW	587.0	213.0	0.36	213.6	213.0	DeLaune *et al.* (1983)
Fresh	LA	C	FW	49.0	18.7	0.38	17.8	18.7	DeLaune *et al.* (1983)
Sphagnum Bog	MD	C	FW	-1.1			-0.4	-0.4	Yavitt *et al.* (1990)
Bog	MI	C	FW	193.0			70.2	70.2	Shannon and White (1994)
Bog	MI	C	FW	28.0			10.2	10.2	Shannon and White (1994)
Beaver Meadow	MN	C	FW		2.3			2.3	Bridgham *et al.* (1995)
Open Bogs	MN	C	FW		0.0			0.0	Bridgham *et al.* (1995)
Bog (Forested Hum-mock)	MN	C	FW	10.0	3.5	0.35	3.6	3.5	Dise (1993)
Bog (Forested Hol-low)	MN	C	FW	38.0	13.8	0.36	13.8	13.8	Dise (1993)
Fen Lagg	MN	C	FW	35.0	12.6	0.36	12.7	12.6	Dise (1993)
Bog (Open Bog)	MN	C	FW	118.0	43.1	0.37	42.9	43.1	Dise (1993)
Fen (Open Poor Fen)	MN	C	FW	180.0	65.7	0.37	65.5	65.7	Dise (1993)
Poor Fen	MN	C	FW	242.0			88.1	88.1	Dise and Verry (2001)
Sedge Meadow	MN	C	FW		11.7			11.7	Naiman *et al.* (1991)
Submergent	MN	C	FW		14.4			14.4	Naiman *et al.* (1991)
Deep Water	MN	C	FW		0.5			0.5	Naiman *et al.* (1991)
Poor Fen	MN	T	FW		14.6			14.6	Shurpali and Verma (1998)
Submerged Tidal	NC	C, E	FW	144.8			52.7	52.7	Kelly *et al.* (1995)
Banks Tidal	NC	C, E	FW	20.1			7.3	7.3	Kelly *et al.* (1995)
Tidal Marsh	NC	C	FW	3.0	1.0	0.34	1.1	1.0	Megonigal and Schlesinger (2002)
Tidal Marsh	NC	C	FW	3.5	2.3	0.65	1.3	2.3	Megonigal and Schlesinger (2002)
Prairie Marsh	NE	T	FW		64.0			64.0	Kim *et al.* (1999)
Poor Fen	NH	C	FW	503.3	110.6	0.22	183.2	110.6	Carroll and Crill (1997)
Poor Fen	NH	C	FW		69.3			69.3	Frolking and Crill (1994)
Forested Peatland	NY	C	FW	0.6	0.2	0.37	0.2	0.2	Coles and Yavitt (2004)
Pools Forested Swamp	NY	C	FW	224.6	69.0	0.31	81.7	69.0	Miller *et al.* (1999)
Typha Marsh - Min-eral 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)

Habitat	State	Method[a]	Salt/ Fresh	Daily Average Flux (mg CH_4 per m² per day)	Measured Annual Flux (g CH_4 per m² per year)	Conversion Factor	Estimated Annual Flux (g CH_4 per m² per year)	Used Annual Flux (g CH_4 per m² per year)	Reference
Swamp	VA	C	FW	470.3			171.2	171.2	Chanton *et al.* (1992)
Maple/Gum Forested Swamp	VA	C	FW		0.5			0.5	Harriss *et al.* (1982)
Emergent Tidal Freshwater Marsh	VA	C	FW		96.2			96.2	Neubauer et al. (2000)
Oak Swamp (Bank Site)	VA	C	FW	117.0	43.7	0.37	42.6	43.7	Wilson *et al.* (1989)
Emergent Macro-phytes (Peltandra)	VA	C	FW	155.0			56.4	56.4	Wilson *et al.* (1989)
Emergent Macro-phytes (Smartweed)	VA	C	FW	83.0			30.2	30.2	Wilson *et al.* (1989)
Ash Tree Swamp	VA	C	FW	152.0			55.3	55.3	Wilson *et al.* (1989)
Bog	WA	C	FW	73.0			26.6	26.6	Lansdown *et al.* (1992)
Lowland Shrub and Forested Wetland	WI	T	FW		12.4			12.4	Werner *et al.* (2003)
Sphagnum/Eriopho-rum (Poor Fen)	WV	C	FW	6.6			2.4	2.4	Yavitt *et al.* (1990)
Sphagnum/Shrub (Fen)	WV	C	FW	0.1			0.0	0.0	Yavitt *et al.* (1990)
Polytrichum/Shrub (Fen)	WV	C	FW	-0.1			0.0	0.0	Yavitt *et al.* (1990)
Sphagnum/Forest	WV	C	FW	9.6			3.5	3.5	Yavitt *et al.* (1990)
Sedge Meadow	WV	C	FW	1.5			0.5	0.5	Yavitt *et al.* (1990)
Beaver Pond	WV	C	FW	250.0			91.0	91.0	Yavitt *et al.* (1990)
Low Gradient Head-water Stream	WV	C	FW	300.0			109.2	109.2	Yavitt *et al.* (1990)
Sphagnum/Eriopho-rum	WV	C	FW	52.1	19.0	0.37	18.9	19.0	Yavitt *et al.* (1993)
Polytrichum	WV	C	FW	41.1	15.0	0.37	15.0	15.0	Yavitt *et al.* (1993)
Sphagnum/Shrub	WV	C	FW	4.4	1.6	0.37	1.6	1.6	Yavitt *et al.* (1993)
Salt Marsh	DE	C	SW	0.5			0.2	0.2	Bartlett *et al.* (1985)
Red Mangroves	FL	C	SW	4.2			1.4	1.4	Bartlett *et al.* (1989)
Dwarf Red Man-grove	FL	C	SW	81.9			27.9	27.9	Bartlett *et al.* (1989)
High Marsh	FL	C	SW	3.9			1.3	1.3	Bartlett *et al.* (1985)
Salt Marsh	FL	C	SW	0.6			0.2	0.2	Bartlett *et al.* (1985)
Salt Water Man-groves	FL	C	SW	4.0			1.4	1.4	Harriss *et al.* (1988)
Salt Marsh	GA	C	SW	13.4			4.6	4.6	Bartlett *et al.* (1985)
Short Spartina Marsh - High Marsh	GA	C	SW	145.2	53.1	0.37	49.5	53.1	King and Wiebe (1978
Mid Marsh	GA	C	SW	15.8	5.8	0.37	5.4	5.8	King and Wiebe (1978)
Tall Spartina Marsh - Low Marsh	GA	C	SW	1.2	0.4	0.34	0.4	0.4	King and Wiebe (1978)
Intermediate Marsh	LA	C	SW	912[b]					Alford *et al.* (1997)
Salt Marsh	LA	C	SW	15.7	5.7	0.36	5.4	5.7	DeLaune *et al.* (1983)
Brackish	LA	C	SW	267.0	97.0		91.1	97.0	DeLaune *et al.* (1983)
Salt Marsh	LA	C	SW	4.8	1.7	0.35	1.6	1.7	DeLaune *et al.* (1983)
Brackish	LA	C	SW	17.0	6.4	0.38	5.8	6.4	DeLaune *et al.* (1983)

[b] Outlier that was removed from further analysis.

Habitat	State	Method[a]	Salt/ Fresh	Daily Average Flux (mg CH_4 per m² per day)	Measured Annual Flux (g CH_4 per m² per year)	Conversion Factor	Estimated Annual Flux (g CH_4 per m² per year)	Used Annual Flux (g CH_4 per m² per year)	Reference
Cypress Swamp - Floodplain	SC	C	SW	1.5			0.5	0.5	Bartlett *et al.* (1985)
Salt Marsh	SC	C	SW	0.4			0.1	0.1	Bartlett *et al.* (1985)
Salt Marsh	VA	C	SW	3.0	1.3	0.43	1.0	1.3	Bartlett *et al.* (1985)
Salt Marsh	VA	C	SW	5.0	1.2	0.24	1.7	1.2	Bartlett *et al.* (1985)
Salt Meadow	VA	C	SW	2.0	0.4	0.22	0.7	0.4	Bartlett *et al.* (1985)
Salt Marsh	VA	C	SW	-0.8			-0.3	-0.3	Bartlett *et al.* (1985)
Salt Marsh	VA	C	SW	1.5			0.5	0.5	Bartlett *et al.* (1985)
Salt Meadow	VA	C	SW	-1.9			-0.6	-0.6	Bartlett *et al.* (1985)
Tidal Salt Marsh	VA	C	SW	16.0	5.6	0.35	5.5	5.6	Bartlett *et al.* (1987)
Tidal Brackish Marsh	VA	C	SW	64.6	22.4	0.35	22.0	22.4	Bartlett *et al.* (1987)
Tidal Brackish/Fresh Marsh	VA	C	SW	53.5	18.2	0.34	18.2	18.2	Bartlett *et al.* (1987)
Freshwater									
n					32	18	74	88	
Arithmetic Mean					32.1	0.36	38.6	36.0	
Arithmetic Standard Error					7.9	0.02	6.0	5.0	
Geometric Mean					8.1		7.1	7.6	
Geometric Standard Error					2.1		0.82	2.2	
Saltwater									
n					13	12	25	25	
Arithmetic Mean					16.9	0.34	9.8	10.3	
Arithmetic Standard Error					7.8	0.02	4.1	4.4	
Geometric Mean					5.0		1.3	1.3	
Geometric Standard Error					2.0		0.2	3.3	

APPENDIX G

New pCO_2 Database for Coastal Ocean Waters Surrounding North America

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A database for the partial pressure of carbon dioxide (pCO_2), temperature, and salinity in surface waters within about 1,000 km from the shore of the North American continent has been assembled. About 550,000 seawater pCO_2 observations were made from 1979 to 2004 by the authors and collaborators of Chapter 15. The pCO_2 data have been obtained by a method using an infrared gas analyzer or gas-chromatograph for the determination of CO_2 concentrations in a carrier gas equilibrated with seawater at a known temperature and total pressure. The precision of pCO_2 measurements has been estimated to be about ± 0.7% on average. The quality-controlled data are archived at http://www.ldeo.columbia.edu/res/pi/CO2.

The zonal distribution of the surface water pCO_2, sea surface temperature (SST), and salinity data shows that the greatest variability is confined within 300 km from the shores of both the Atlantic and Pacific. Observations made in various years were combined into a single year and were averaged into 1° × 1° pixels (approximately N-S 100 km by E-W 80 km) for the analysis. Accordingly, the results represent a climatological mean condition over the past 25 years. Finer resolutions (10 × 10 km) may be desirable for some areas close to shore because of outflow of estuarine and river waters and upwelling. However, for this study, which is aimed at a broad picture of waters surrounding the continent, the fine scale measurements have been incorporated into the 1° × 1° pixels. In addition, data with salinities of less than 16.0 are considered to be inland waters and have been excluded from the analysis.

Climatological monthly and annual mean values for pCO_2 in each zone were computed first. Then, the air-sea pCO_2 difference, which represents the thermodynamic driving potential for air-sea CO_2 gas transfer, was estimated using the atmospheric CO_2 concentration data. Finally, the net air-sea CO_2 flux was computed using transfer coefficients estimated on the basis of climatological mean monthly wind speeds using the (wind speed)^2 formulation of Wanninkhof (1992). The transfer coefficient depends on the state of turbulence above and below the air-sea interface and is commonly parameterized as a function of wind speeds (corrected to 10 m above the sea surface). However, selection of wind data is problematic because wind speeds vary with the time scale (hourly, diurnal, or seasonal). For example, fluxes calculated for the South Atlantic Bight from 6-h mean wind speeds in the NCEP/NCAR version 2 file (1° × 1° mean) were lower than those estimated using the monthly mean. This discrepancy suggests that ships used commonly for coastal carbon studies tend to be small and, hence, are rarely at sea under high wind conditions, so observations are biased toward lower winds. Taking into account that the observations have been made infrequently over multiple years, the gas transfer coefficients estimated from climatological mean monthly wind speeds may be more representative. The Schmidt number is computed using measured SST and climatological mean salinity (DaSilva *et al*., 1994). The flux values in a given month are then averaged to yield a climatological mean flux (and standard deviation) for each month. This procedure assumes implicitly that the seawater pCO_2 changes at much slower rates in space and time than the wind speed and that the seawater pCO_2 does not correlate with the wind speed.